# Wildlife Densities and Habitat Use Across Temporal and Spatial Scales on the Mid-Atlantic Outer Continental Shelf (2012-2014)

Final Report to the
Department of Energy
Office of Energy Efficiency
and Renewable Energy
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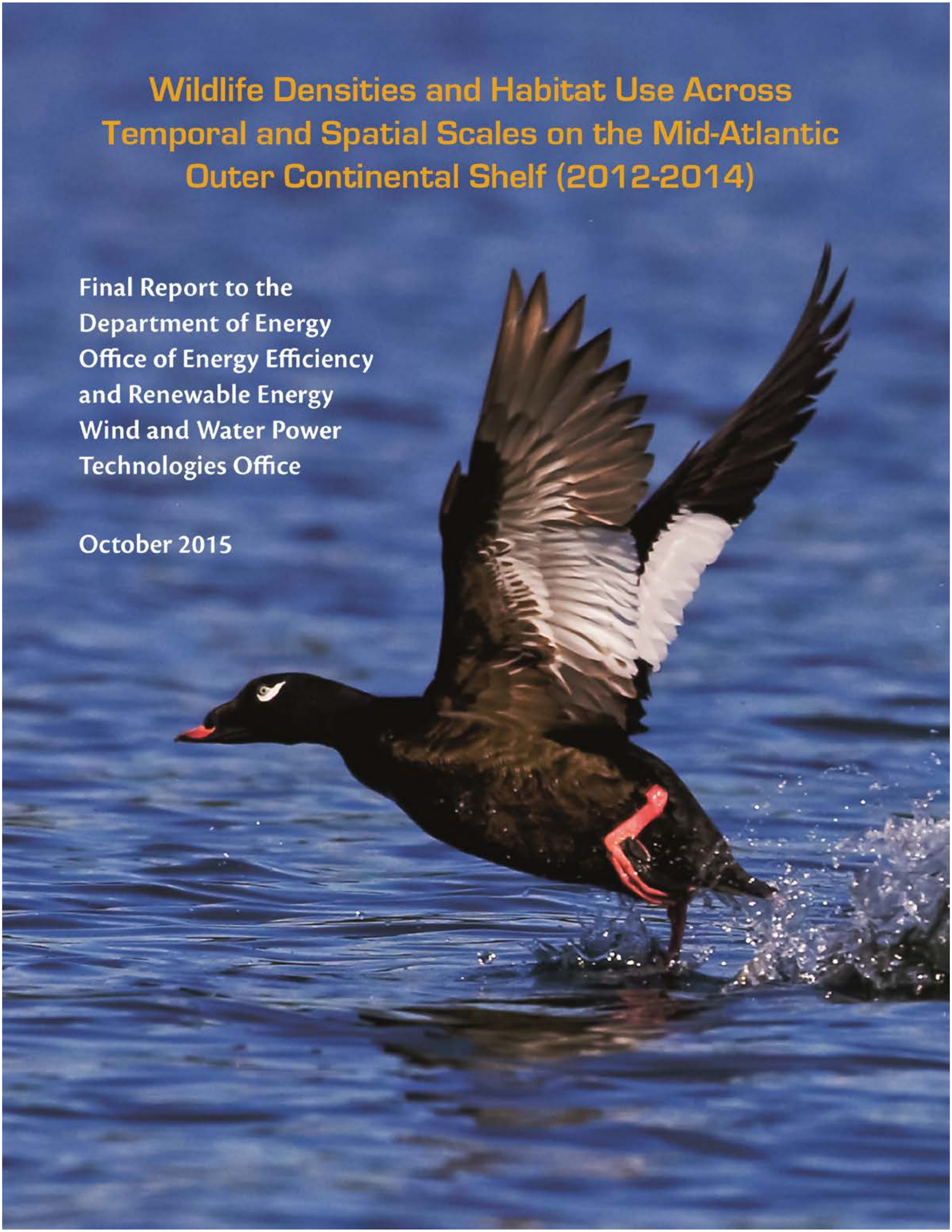

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Funders, authors, collaborators, and additional acknowledgements for each specific report chapter are included in subsequent chapters.

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# Executive Summary

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The Mid-Atlantic Baseline Studies Project was funded by the Department of Energy's (DOE) Wind and Water Power Technologies Office in 2011, with additional support from a wide range of partners. The study was intended to help address environmental barriers to offshore wind energy development in the mid-Atlantic region and promote the incorporation of environmental data into siting and permitting processes. The study goal was to provide regulators, developers, and other stakeholders with comprehensive baseline ecological data and analyses that could help address environmental permitting requirements for current and future projects, and would serve as a starting point for more site-specific studies. In particular, we produced information that could be used to identify: 1) important wildlife areas, 2) data gaps, and 3) approaches for collecting and incorporating natural resource data into decision making. To address this goal, project funders and collaborators from a range of academic institutions, non-governmental organizations, federal agencies, foundations, and private companies came together to study bird, sea turtle, and marine mammal distributions, densities, and movements on the mid-Atlantic Outer Continental Shelf between 2012 and 2014. The specific study area in the mid-Atlantic was chosen because it was viewed as a likely region for near-term wind energy development offshore of Delaware, Maryland, and Virginia, particularly within three federally designated Wind Energy Areas (WEAs).

Specific project activities and goals included the following:

- Conduct standardized surveys to quantify bird, sea turtle, and marine mammal densities seasonally and annually throughout the study region, identify important habitat use or aggregation areas, and examine temporal variation in these patterns. Several survey approaches were employed to reach this goal.
- Develop statistical models to help understand the drivers of wildlife distribution and abundance patterns, and predict the combinations of environmental conditions likely to support large densities of birds, sea turtles, and marine mammals.
- Use individual tracking data for several focal bird species to provide information on population connectivity, individual movements, and seasonal site fidelity that is complementary to survey data.
- Identify species that are likely to be exposed to offshore wind energy development activities in the mid-Atlantic study area.
- Compare high resolution digital video aerial surveys to boat-based surveys, and publish results on the validity of high resolution digital video aerial surveys as a survey method for offshore development in U.S. waters.

- Develop U.S.-based technological resources for future monitoring efforts, and explore technological advancements and assessment methods aimed at simplifying and minimizing the cost of environmental risk assessments.
- Help meet data needs associated with National Environmental Policy Act (NEPA), Marine Mammal Protection Act, and Endangered Species Act requirements, by contributing several years of data and analysis towards future Environmental Impact Statements.
- Disseminate results to stakeholders and regulators through publicly accessible technical and summary reports, geospatial map layers, scientific manuscripts, and in-person briefings.

Funding by DOE was leveraged by the Biodiversity Research Institute (BRI) and other collaborators to conduct additional wildlife research in several topic areas. During the second year of survey efforts, the Maryland Department of Natural Resources (MDNR) and the Maryland Energy Administration (MEA) funded the expansion of boat and aerial surveys to cover a larger extent in Maryland’s state and federal waters (the Maryland Project; Figure I). Unless noted otherwise, data from the two projects were fully integrated, and survey data presented throughout this report include the Maryland extension transects along with data funded through DOE for the Mid-Atlantic Baseline Studies Project. Seabird tracking studies for this project were also jointly funded, and conducted as part of longer-term research into seabird movements and habitat use developed and initiated by the Bureau of Ocean Energy Management (BOEM), U.S. Fish and Wildlife Service (USFWS), and Sea Duck Joint Venture (SDJV), and carried out in coordination with BRI, U.S. Geological Survey (USGS), Memorial University of Newfoundland, and other partners.

## Offshore wind and wildlife

Offshore wind energy development has progressed rapidly in Europe since the first facility became operational in 1991 (Breton and Moe 2009), and it is now being pursued in other regions of the world, including the U.S. This renewable resource has the potential to reduce the impacts of climate change and ocean acidification by lowering global carbon emissions (IPCC 2014), and thus to positively affect many species. Offshore wind energy developments may also affect local wildlife more directly. Researchers are still learning about how offshore wind energy facilities affect marine ecosystems, but it seems clear that effects vary during different development phases, and that species respond in a variety of ways. Some species are negatively affected, while others show no net effect, or may even be affected positively. Possible effects to fish, marine mammals, sea turtles, birds, and bats include: mortality or injury from collisions with turbines or vessels; displacement from, or attraction to, habitat use areas; avoidance of facilities during migration or daily movements, which may necessitate increased energetic expenditures; and changes to habitat or prey populations, including artificial reef effects (Fox et al. 2006, Kunz et al. 2007, Boehlert and Gill 2010, Langston 2013, Bailey et al. 2014, Bergström et al. 2014). The scale of development is likely to be important in determining the significance of these effects. Overall, the cumulative effects to wildlife will be dependent on the size and number of wind facilities that are built, as well as local topography, climate, species ranges, and other oceanographic and biological factors. Effects from offshore wind may also be combined with other natural and anthropogenic stressors. As a result, ecological context is essential for understanding and minimizing effects of offshore development on wildlife.

## Project components

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure II). Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure II. An additional study effort, which explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

This report consists of six parts:

I. *Project overview*, which includes the executive summary (this section), a background chapter on the study area and methods (Chapter 1), and a longer synthesis that integrates findings from all project components to identify larger patterns and trends (Chapter 2);
II. *Examining wildlife distributions and relative abundance from a digital video aerial survey platform* (Chapters 3-6);
III. *Examining wildlife distributions and abundance using boat-based surveys* (Chapters 7-12);
IV. *Integrating data across survey platforms* (Chapters 13-19);
V. *Individual movements and habitat use for focal bird species* (Chapters 20-25); and
VI. *Nocturnal avian migration monitoring* (Chapters 26-27).

### *Project overview*

The mid-Atlantic region is used by a broad suite of wide-ranging marine wildlife species across the annual cycle. This, along with the high levels of productivity in the region, mean that it is essential to understand the dynamics of this ecosystem in order to manage it effectively, particularly with regard to anthropogenic stressors, such as offshore development. In Chapter 1, we briefly discuss the ecosystem of the Mid-Atlantic Bight and describe the methods employed in the Mid-Atlantic Baseline Studies Project and Maryland Project. We discuss the relative strengths of digital video aerial surveys and other methods employed in this study, with a particular focus on comparing boat-based surveys and digital video aerial surveys. We also briefly discuss the various approaches used to present results in this report.

In Chapter 2, we summarize persistent and seasonal patterns in wildlife distributions that were observed during the two years of this study, including offshore surveys, individual tracking, and methods of studying nocturnal avian migration in the offshore environment. We present a series of case studies on specific taxa or phenomena that integrate data gained from these different methods to examine in detail the abundance and distributions of potentially vulnerable taxa. In addition, we discuss a number of similar baseline studies that have recently been conducted along the eastern seaboard. Observed community composition, distribution patterns, phenology, and behaviors in this study all varied somewhat from other recent baseline studies, as might be expected based on these studies' different latitudes, bathymetry, and other characteristics. However, at a broad scale, geographic and temporal patterns in the mid-Atlantic were consistent with findings from other recent baseline studies on the eastern seaboard (e.g., Geo-Marine Inc. 2010, Paton et al. 2010). In particular, overall abundance and species diversity were driven in large part by bathymetry, and tended to be highest in shallow water areas (which in many cases are coincident with areas closer to shore, though not always). In some cases, results from other baseline studies have been used to identify areas of high biodiversity and priorities for conservation, ultimately influencing the choice of lease sites for offshore wind development (Rhode Island Coastal Resources Management Council 2013)[1]. The data developed in this study of the mid-Atlantic could be used to identify important areas for conservation as well as areas that would minimize exposure of wildlife during future development.

### *Examining wildlife distributions and relative abundance from a digital aerial survey platform*

Fifteen aerial surveys were conducted over two years by HiDef Aerial Surveying, Ltd., using high resolution digital video. Digital aerial survey approaches have largely replaced visual aerial surveys for offshore wind energy research in Europe, as their higher flight speeds and much higher flight altitudes make them safer to conduct than visual aerial surveys, and reduce or eliminate disturbance to wildlife compared to visual aerial or boat-based survey approaches. They also produce archivable data, which allow for a robust quality assurance and audit process. There are still limitations to this method, however, including difficulties identifying some species, and a lack of defined statistical approaches for using the data for some purposes, due to the relative novelty of the survey method.

This study includes the first application of this technology on a large spatial scale in the United States. Surveys were conducted along transects with a dense spatial coverage (20% ground coverage) within WEAs, as well as a broader sawtooth transect throughout the remainder of the study area (Figure I). Four belly-mounted cameras recorded video footage during surveys, which was later analyzed to locate and identify animals (Chapter 3). Detailed video data analysis and management protocols were developed by BRI, in consultation with HiDef, including the Quality Assurance and Quality Control (QA/QC) protocol used to audit survey results (Chapter 4). Twenty percent of all video was included in blind re-reviews to ensure consistency in locating and identifying objects. Identification of animals to species proved difficult for some taxa due to variations in image quality and other factors (Chapter 5). Newer generations of camera systems currently used in Europe have greatly improved upon the identification rates obtained in this study (HiDef Aerial Surveying Ltd. unpubl. data). We make several recommendations in this report for the future use of digital aerial surveys in the Western Hemisphere, including the explicit examination of variables affecting species identifications and detection (Chapter 6).

Completed analysis provided data on the number of target organisms in the video, the species or other identification category of organisms (Chapter 5), the approximate flight height for flying birds and bats (Hatch et al. 2013), and geospatial data for all objects that may be used in modeling efforts. Over 100,000 animals were observed within the study area over two years of digital video aerial surveys, including over 46,000 birds and 60,000 aquatic animals. Digital video aerial surveys proved to be particularly good at detecting aquatic animals located near the water's surface, such as sea turtles and large migratory schools of rays. In fact, rays (Batoidea) were the most abundant animal observed in aerial surveys, representing over 44% of all observations. Scoters (*Melanitta* spp.) were the most abundant

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[1] www.boem.gov/BOEM-Newsroom/Press-Releases/2012/press05302012.aspx

avian species observed in the aerial surveys (20% of all observations; Chapter 5). Flight heights were estimated for 5,299 birds and bats (of the >7,000 animals observed in flight). Roughly 60% of flying animals were estimated to be in the lowest flight altitude category (0-20 m above the water's surface). Another 37% were estimated to be at altitude ranges between 20 and 200 m, which are in or near the potential rotor-sweep zone for future offshore wind energy development along the Eastern Seaboard (Willmott et al. 2013), depending on the size and type of turbines (Chapter 5).

### *Examining wildlife distributions and abundance using boat surveys*

To accompany (and compare with) data from digital aerial surveys, 16 boat surveys were conducted over two years (Figure I). Standardized boat-based surveys with distance estimation are a widely used method of obtaining density data for birds, sea turtles, and marine mammals (Chapter 7); the study design was particularly optimized for avian species, and detected a wide variety of seabird species as well as raptors, passerines, and other taxa (Chapter 8). A total of 64,642 animals were observed on the survey, including over 62,000 birds and 1,500 aquatic animals, with the greatest numbers observed in December and January, when large flocks of wintering birds were present in the study area (Chapter 8). Scoters were the most abundant animal observed in boat surveys, constituting 34% of all observations.

While conducting these surveys, we also collected environmental covariate data in order to assess fine-scale patterns of these environmental variables in relation to wildlife densities. In particular, fisheries sonar (a scientific echo sounder) was used to estimate relative prey biomass in the same areas as boat survey observations (Chapter 9). We observed high levels of spatial and temporal variation in prey biomass across the study area and between surveys, although mean depth of biomass in the water column did not vary significantly between seasons. Total biomass, summed across all water depths, was higher in nearshore areas in the summer and fall and in the southern end of the study area during winter surveys.

These data were used to examine spatial associations between feeding seabirds and acoustically detected prey (Chapter 10). There were statistically significant associations between seabirds and patches of prey biomass for bird species that feed largely or entirely near the water's surface (Northern Gannets [*Morus bassanus*], Laughing Gulls [*Larus atricilla*], Common Terns [*Sterna hirundo*], and Royal Terns [*Thalasseus elegans*]), but not for deep-diving species such as loons and sea ducks. Identifying the spatial and temporal associations and lags between aquatic biomass and seabird behavior is helpful for understanding how these birds make decisions in the marine environment, and in turn may help managers to determine the behaviors or environmental conditions that present the highest risk of interactions between seabirds and offshore wind energy development.

A broader geographic and temporal scale of analysis is required to develop products appropriate for use in siting future development projects, and to fully assess exposure to wildlife from proposed projects. These goals also require correction of certain biases associated with boat survey data, such as distance bias, in which observers are less likely to see animals located farther from the survey vessel. Hierarchical Bayesian statistical approaches, as applied to survey data in Chapters 11-12, allow distribution models to be chosen to fit the observed data (Gardner et al. 2008, Zipkin et al. 2010), and incorporate distance estimation and environmental covariates into the model structure in order to predict animal distributions and abundance on a broad geographic scale. Project collaborators first focused on the development of a community distance sampling (CDS) model for seabirds using data from the first boat survey in April 2012 (Chapter 11). This novel multi-species approach for estimating seabird abundance and distributions explicitly estimated detection as well as abundance parameters. By sharing information across species, this community model allowed us to make inferences about abundance, distribution, and response to environmental variables of rare species for which there would not be enough data to run individual models.

Building on the CDS model in Chapter 11, Chapter 12 examined survey data from 15 boat surveys and incorporated remotely collected environmental covariate data into the hierarchical modeling structure. This approach accounted for imperfect detection to estimate “true” abundance, and predicted seabird distributions by season to help identify important habitat use areas and patterns. Seabird distributions were spatially, seasonally, and taxonomically variable; winter was the period of highest predicted abundance and species diversity, particularly during the second year of surveys. High species density and diversity was also predicted to occur in spring and fall, suggesting that migratory and overwintering species dominate the region’s species composition. Distributions for some species, such as Common Terns and Red-throated Loons (*Gavia stellata*), were concentrated farther offshore in spring (during the pre-breeding migratory period). While summer was the period of lowest overall predicted abundance, several federally- and state-listed *Threatened* or *Endangered* species were present in the region during that time of year, including include Roseate Terns (*Sterna dougallii*), Least Terns (*Sternula antillarum*), Common Terns, Forster’s Terns (*Sterna forsteri*), and Royal Terns. The community distance sampling model enabled us to accommodate these relatively rare species and estimate their relationships with habitat features, improving our understanding of their distributions.

### *Integrating data across survey platforms*

Several chapters in this report focus on contrasting boat-based and digital video aerial survey approaches (Chapters 13-14 and 18). In some cases, data from one survey approach were used independently to analyze wildlife distributions and relative abundance (e.g., in the case of sea turtles, Chapters 15 and 17, or Bottlenose Dolphins [*Tursiops truncatus*], Chapter 15). In other cases, digital video aerial survey data and boat survey data were used jointly (Chapters 16-17 and 19) to describe distributions and abundance of animals across the study area.

In order to test the utility of high resolution digital video aerial surveys on the Atlantic coast, and to integrate new aerial survey data with historical data, we compared the digital aerial data to boat-based surveys using experimentally controlled methods (Chapter 13). This comparison indicated largely complementary strengths of the two survey approaches, though it also highlighted their respective weaknesses (namely, the need for additional analytical development for digital survey data, and the issue of disturbance to wildlife populations caused by the vessel during boat-based surveys). The two survey methods found similar distribution patterns for scoters, but were poorly correlated for highly mobile Northern Gannets, which at the density of transects in the comparison study were not adequately surveyed by the plane’s relatively narrow transect strip width. In addition to this formal comparison of methods, project collaborators also pursued other approaches for comparing and contrasting the two full survey datasets. Species identification rates, as well as detection rates, varied considerably between methods for some taxa (Chapter 14). More birds per unit effort, and more bird species, were observed in the boat surveys, and birds made up a higher proportion of boat observations (98%) compared to digital video aerial surveys (43%). In contrast, much higher counts and species diversity of sea turtles and other aquatic animals (sharks, rays, fish, etc.), were detected on the aerial surveys than on the boat surveys (Chapter 14). Gulls and terns (Laridae), loons (Gaviidae), and auks (Alcidae) all had much higher identification rates to the species level from the boat surveys than in aerial video (Chapter 14). The limitation of many aerial identifications to the family or genus level was likely due in part to video image quality, but was also a result of the exhaustive quality assurance and audit protocol followed by aerial video reviewers, and characteristics inherent to the video review process itself (such as the use of multiple levels of "certainty" criteria in identifications). However, aerial video observers were better at identifying the most common avian family, Anatidae (scoters, ducks, and geese), to species than were boat observers, perhaps due in part to disturbance to this taxon from the survey vessel (Chapter 13). Identification rates of toothed whales (Odontoceti) were higher on boat surveys, but baleen whales (Mysticeti) had higher rates of identification from aerial surveys.

In a preliminary analysis of data for four seabird groups (terns, gannets, loons, and alcids), remotely-collected environmental data were incorporated into boat and aerial models (Chapter 18). Results were compared to determine if the two sampling methods detected similar patterns in seabird abundance, with the goal of determining how best to combine boat and digital aerial survey data for an integrated analysis. Accounting for detection resulted in higher abundance for the boat-based than the aerial models. Similar species-habitat relationships were estimated between the two survey types for gannets, terns, and loons, but alcids were less consistent between the survey types and years. These results suggested that a model combining both data types could be powerful for understanding seabird distributions, but that caution may be required for species like alcids where different patterns were observed between surveys, possibly due to differences in the sampling domain, detectability, or temporal variation.

Thus, the integration and combined analysis of the two survey datasets provided an opportunity to create higher-quality end products by incorporating complementary data streams. On a small scale, this led to the publication of a scientific paper on Eastern Red Bat (*Lasiurus borealis*) migration in the offshore environment of the mid-Atlantic (Hatch et al. 2013). Surveys were not designed to detect bats and other terrestrial species (due to their small body size, and because many of these species are thought to migrate exclusively at night). Despite this limitation, 17 bats were observed altogether, including two during boat surveys and 15 in video aerial surveys. Weather conditions were good at the time of these observations, suggesting that these bats were deliberately migrating offshore rather than driven offshore by high winds or other severe weather (Hatch et al. 2013). Despite their generally nocturnal behavioral patterns, bats were observed during morning daylight hours. Observations occurred between approximately 16 and 70 km from shore, and all bats with estimable flight heights in the aerial survey video were estimated to be >200 m above sea level, above the rotor swept zone of current offshore wind turbine models. Little is known about the migration and movements of tree bat species in North America, but anecdotal observations of bats migrating over the Atlantic Ocean (particularly during fall migration periods) have been reported since at least the 1890s (Hatch et al. 2013). Despite the relatively small sample size, the observations from this study provide further evidence of bat movements well offshore, and offer insight into their flight heights above sea level and the times of day at which such migrations may occur.

Collaborators also used the two datasets to identify temporal and spatial patterns of species presence and relative abundance in the study area, including the identification of “hotspots,” or geographic areas with consistently high numbers of animals through time (adapted from Santora and Veit 2013). Persistent hotspots likely provided important habitat for foraging, roosting, or other activities (Santora and Veit 2013). The presence and relative abundance of different species varied widely by time of year, with late fall to early spring identified as a time with high effort-corrected counts of animals in the study area, though many aquatic animals peaked in abundance in the summer. For many taxa, hotspots were most consistently observed in areas within 30-40 km from shore, particularly offshore of the mouths of Chesapeake Bay and Delaware Bay (Chapter 17). These areas consistently showed high species diversity and abundance of animals across all taxa observed in this study, and may have been attractive to many animals due to environmental gradients in salinity, water temperature, and other factors that created reliable foraging habitat in these locations. Areas offshore of northern Maryland also showed high diversity and abundance, although this may have been partially due to the high survey effort in nearshore waters in this region. Species that were consistently observed farther offshore on the Outer Continental Shelf included sea turtles, Common Dolphins (*Delphinus delphis*), Common Loons (*Gavia immer*), and alcids.

The incorporation of environmental covariates into modeling efforts allowed for the prediction of relative densities across the study area for many taxa (Chapters 12, 15, and 18-19), with one or both survey datasets used to describe populations of interest. In some cases, one survey method was significantly better than the other for surveying a particular taxon. For example, sea turtles were much more frequently observed in digital aerial surveys than in boat surveys, likely in large part because the turtles could be detected even when they were fully submerged. Because of these high detection rates, we used only the aerial survey results to develop predictive models of sea turtle distributions (Chapter 15). Sea turtles were most abundant from May to October, and their densities were correlated with warmer water temperatures and greater distances from shore. There was substantial overlap between sea turtle distributions and WEAs, particularly in the southern part of the study area. Bottlenose Dolphin distributions were modeled using boat data, and they were predicted to use primarily more nearshore areas with high levels of primary productivity and higher sea surface temperatures in spring, summer, and fall. There were few observations of the species during cooler months.

In other cases, boat and digital aerial survey datasets could be combined using recently developed integrated modeling frameworks (as with several seabird groups; Chapter 19). Common Loons and Red-throated Loons, which proved difficult to distinguish in aerial video, provided a test case for using boat-based species identifications to inform aerial models and develop spatially explicit species-specific estimates of relative abundance (Chapters 16-17). In Chapter 19, project collaborators developed an integrated modeling approach in which predictions of marine bird abundance and distribution were jointly informed by aerial surveys (which encompassed a large geographic area), and boat surveys (which allowed for estimation of detection probability). Integrated models were developed for the same four taxa examined in Chapter 18 (terns, alcids, loons, and gannets). The combined predictions of this chapter generally supported the conclusions of Chapters 12 and 17-18, which found that the distribution of marine birds was often patchy, species- and survey-specific, and correlated with habitat covariates. The integrated models had noticeable improvements in predicting local hotspots and marine bird distributions relative to models that only included boat-based data. The greater spatial span of aerial surveys may have assisted in the detection of latitudinal gradients and hotspots, especially those occurring outside of areas surveyed by the boat. The integrated models, however, often had lower predictive power than boat-only models for describing observations from other surveys conducted in the same season, which was likely a consequence of dynamic relationships between boat and aerial surveys and changing habitat covariates (Winiarski et al. 2013, 2014). While additional exploration and model development is needed, these results indicate that joint modeling approaches may be a fruitful avenue of continued research.

### *Individual movements and habitat use for focal bird species*

We investigated the spatial and temporal patterns of offshore bird migration and winter habitat use through the use of satellite telemetry, and incorporated remotely collected covariate information into models to determine how birds' use of space covaried with environmental conditions. We tracked the movements of 149 individuals from three focal avian taxa: seabirds (Red-throated Loon and Northern Gannet); sea ducks (Surf Scoter, *Melanitta perspicillata*); and raptors (Peregrine Falcon, *Falco peregrinus*). Wintering movements and habitat use in the mid-Atlantic study area were the main focus of telemetry studies for seabirds. Kernel-based utilization distributions and resource selection functions, used to examine scoter habitat use in a previous study (Loring et al. 2014), allowed for the identification of important habitat use areas for Surf Scoters (Chapter 20). Scoters are likely to use more geographically stable prey resources than are Red-throated Loons or Northern Gannets (Chapter 18, Appendix A), and modifications to this approach using different resource selection methods and temporally variable environmental covariates were applied to Red-throated Loons (Chapter 21) and Northern Gannets (Chapter 22). Surf Scoters in core-use areas used shallow (<40 m) areas within 4.5 km from shore. Red-throated Loons also tended to use nearshore areas, and in our sample the greatest chance for interaction between Red-throated Loons and WEAs generally occurred during the spring migration period (late March to early May). In contrast, Northern Gannets ranged widely over the Outer Continental Shelf during winter (Chapter 22). Though core habitat of tagged Northern Gannets within the study area included the protected inshore waters of the major bays and bay mouths, individual birds displayed extensive movements up and down the eastern seaboard between the core use areas, increasing the likelihood that they would encounter offshore wind developments in the region repeatedly throughout the winter.

Chapters 20-22 explored spatial patterns and movements of three target species using fairly traditional methods, which collapse the temporal component of movement data into a single period for analysis. Time-variant kernel density analysis allowed for a more explicit understanding of habitat use through time (Keating and Cherry 2009; Chapter 23). This is an effective tool for showing fine-scale temporal variation in use of the study area, especially in and around WEAs (Chapter 23). The tracking results are preliminary, however, as data were drawn from the first two years of an extended (four year) satellite tracking project.

As well as movements and general habitat use, satellite telemetry can also provide detailed information on specific behaviors. Northern Gannet interactions with offshore wind energy development are hypothesized to largely occur as a result of their foraging behaviors, which include a large proportion of time spent soaring at or near the altitude of the rotor-swept zone for offshore wind turbines (Garthe et al. 2000, Langston 2010). Being able to differentiate between foraging and other behaviors in telemetry data allows us to better determine areas of potential conflict between offshore wind energy development and gannet habitat use. In Chapter 24, we used Northern Gannet positional data in a behavioral state switching state-space model (SSSM) in a Bayesian modeling framework (Jonsen et al. 2007) to analyze telemetry data from the nonbreeding period and examine the habitat characteristics at locations that were used by Northern Gannets for foraging. Weekly sea surface temperature front density was a very strong predictor of foraging activity, indicating that Northern Gannets responded dynamically to either the change in water temperature itself, or to the increase in prey availability that is likely occurring in areas with high front density (Chapter 24).

Unlike these seabird species, Peregrine Falcons migrate through the project study area but seldom winter in or near the mid-Atlantic. Peregrines are probably the most commonly encountered non-piscivorous raptor in marine settings, and they are commonly observed foraging or perching far from shore at offshore islands, oil drilling platforms, and large offshore vessels (Voous 1961, Cochran 1975, 1985; Russell 2005, Johnson et al. 2011, DeSorbo et al. 2012). We used satellite telemetry to document falcon movements and use of space within the project study area during fall migration (Chapter 25). Data were analyzed using a dynamic Brownian Bridge Movement Model, which improves upon traditional (i.e., fixed kernel) approaches used in Chapters 20-22 by accounting for the order in which locations were fixed, the time interval between them, and location error, and thus generating space use estimations that more accurately depict high and low use areas and movement corridors. During this study, Peregrine Falcons regularly used habitat hundreds of kilometers offshore along the Atlantic coast, including within and east of the mid-Atlantic study area. Twelve of the 13 tracked falcons that continued their fall migration beyond the mid-Atlantic coast initiated a significant transoceanic flight from coastal North Carolina to the Caribbean or South America. Birds tracked in this study were all captured on offshore islands, and it remains unclear what proportion of the Peregrine Falcon population ventures offshore. However, findings from this study are consistent with observations elsewhere (Cochran 1975, Fuller et al. 1998, Desorbo et al. 2012) and suggest that this species commonly uses offshore habitats along the Atlantic flyway.

### *Nocturnal migration monitoring*

Limited information is available on nocturnal avian migrants in the offshore environment. The project team investigated the species composition, general spatial patterns, and weather-dependent and seasonal variation in offshore bird migration through a combination of acoustic and radar data collection. Both the nocturnal passive acoustic avian monitoring from the boat (Chapter 26) and the analysis of WSR-88 radar data, also known as NEXt generation RADar (NEXRAD, Chapter 27) were undertaken to determine the utility of these approaches for examining avian migration in the offshore environment, and to improve our understanding of migratory patterns in the offshore environment on the Atlantic coast of the U.S. Many bird species can be identified by their vocalizations, so nocturnal acoustic monitoring stations can provide species-specific presence-absence data and indices of activity for birds that vocalize during migration ("migratory flight calls"). When the boat stayed overnight on the water (seven total occasions over two years, located 25-46 km from shore), we detected migratory flight calls from at least 15 species, including both passerines and shorebirds. Migrant passerines were not detected on the majority of the seven survey nights, but during one survey occasion 40 km off the coast, we detected 123 individual calls in one night. This is consistent with other studies of nocturnal avian migration over the ocean (as well as over land), which have found that migratory activity is highly episodic and appear to be largely driven by variations in weather (Hill et al. 2014). This pilot study suggests that a diverse range of landbirds may migrate over the mid-Atlantic Outer Continental Shelf at low altitudes, though a more extensive effort is warranted before drawing broader conclusions about the frequency of such occurrences.

Developed as a tool to monitor meteorological phenomena, weather surveillance radars regularly detect flying animals in the atmosphere at night (Bridge et al. 2011, Chilson et al. 2012), including passerines, shorebirds, waterfowl, bats, insects, and other nocturnal migrants. We used WSR-88D (NEXRAD) weather radar to study nocturnal migration off the mid-Atlantic coast of the U.S. from New York to North Carolina (Chapter 27). We compared migratory activity at sites over land and up to 80 km out to sea, controlling for variables that could affect measured levels of migratory activity, and we identified the environmental variables correlated with offshore activity. The high level of altitudinal overlap between our measurements and turbine heights suggests that our predictions of bioscatter levels in the offshore environment are highly relevant to migration occurring at rotor-sweep heights. After controlling for biases in measured levels of migratory activity (including varying distances of sampling sites from the radar units), we found that in fall, there was no significant difference in migratory activity at offshore vs. terrestrial sites across the mid-Atlantic region. This suggests that migration over open water areas may be quite common during this season. There is a strong weather-related component to offshore fall migration, however, as discussed above; there were high levels of daily variation in activity at our study sites, and offshore activity was particularly high under west winds. Responses also varied by topographic location along the coast, and some offshore areas had consistently higher predicted activity levels, most notably the New York Bight (south of Long Island) and offshore of North Carolina. In spring, there was still substantial offshore activity around North Carolina, but predicted levels of nocturnal offshore migration were fairly minimal in other locations.

These data suggest that while birds are less likely to migrate offshore in spring, during the fall there appear to be multiple "jumping off points" along the coast for tailwind-aided overwater migrations. Trans-oceanic migrations, once thought to be extreme events only undertaken by few individuals or species with extreme physiological adaptations (DeLuca et al. 2015), are perhaps more commonplace than previously thought in this region. While additional research will be required to determine the degree to which certain taxa use the offshore environment, the levels of migratory activity predicted in offshore locations suggest that regulators for offshore wind energy development may want to consider potential impacts to migrants, including terrestrial species (passerines, shorebirds, bats, etc.), in offshore wind development scenarios. This may be particularly important in locations with consistently higher levels of migratory activity, such as the New York Bight, and areas offshore of North Carolina. Predicted levels of activity in many other parts of the study area were also intermittently high, however, suggesting that offshore migration is a widespread phenomenon, and should be regarded as such during planning activities.

## Synthesis: Advancements in the state of our knowledge

### *The mid-Atlantic ecosystem*

The mid-Atlantic region is used by a broad range of marine wildlife species across the entire annual cycle, due in part to a relatively high level of productivity, as compared to many other areas in the western North Atlantic (Yoder et al. 2001). The importance of the region to wildlife is also partially due to the region's central location on the eastern edge of the continent (a major migratory corridor for many species). As a result, the mid-Atlantic supports large populations of marine wildlife during breeding, nonbreeding, and migratory periods, and this results in a complex ecosystem where the community composition is constantly shifting, and temporal and geographic patterns are highly variable.

The Mid-Atlantic Baseline Studies Project and Maryland Project have filled a significant information gap for wildlife in a large swath of the mid-Atlantic region between New Jersey and North Carolina. In part, this area was a focus due to its ecological significance and relative lack of data on wildlife distributions. Additionally, this region has great economic importance, including commercial fisheries, shipping, and the potential for offshore renewable energy development. The mid-Atlantic region has a relatively high wind energy potential, and is also located near large energy markets on the U.S. Atlantic coast (Baker 2011). Thus, the region has been a focus for offshore wind developers and regulators in recent years, and several of the first federally designated WEAs are located off the mid-Atlantic coast. To minimize the effects of development activities on wildlife populations, however, the complexities of this ecosystem require that a range of study methods be used to obtain a comprehensive view of ecosystem structure and configuration.

### *Study methods and comparisons*

Field study methods have a substantial influence on resulting analysis and presentation of wildlife distribution data. Often, study methods involve tradeoffs between geographic vs. temporal coverage, information on animal abundance (or relative abundance) vs. accurate species identifications, and detailed behavior or movement data vs. information on population distributions (Chapter 1). Each of the methods that we used to examine marine wildlife distributions and movements in the mid-Atlantic had inherent strengths and weaknesses. Our evaluation of the utility of each survey method in documenting different types of data is necessarily subjective in many cases, and is dependent upon the specific study design implemented for this project (i.e., the study area, available technology, sample size, and other factors).

Compared to the other study methods used in this project, boat and aerial surveys provided relatively comprehensive information on wildlife populations in the offshore environment (Chapter 1). Each showed distinct benefits in detecting different taxa. High resolution digital video aerial surveys provided better detection rates for aquatic animals, likely due to a combination of reduced disturbance, reduced glare, and a unique field of view compared to boat-based and visual aerial surveys, allowing for submerged animals to more easily be detected in the upper reaches of the water column (Chapters 5 and 14; Normandeau Associates Inc., 2012). Boat surveys provided better detection rates for many birds, however, which is probably due to a combination of availability bias, detection bias, and identification issues in digital video aerial surveys (Chapters 5 and 13-14). Digital aerial surveys have the advantage of being auditable and archivable, and include an extensive quality assurance process, which may lead to a greater degree of reliability in species identifications. The safety and speed with which digital aerial surveys can be conducted also make this approach attractive in the offshore environment, and the capabilities of digital aerial surveys will likely continue to improve with technological advances in the field. Boat-based surveys can provide detailed behavioral data, however, and had generally better rates of identification of animals to species. The analytical approaches for boat survey data are also well established, while additional technological advances and analytical developments for digital aerial surveys would strengthen this approach for understanding wildlife distributions in the offshore environment of North America (Chapter 6).

### *Patterns of wildlife distribution and abundance*

Distribution and relative abundance of wildlife in the mid-Atlantic is largely driven by environmental variables, including weather, habitat characteristics, prey distributions, and the topography of the coastline. Important environmental factors influencing species distributions included distance to shore, sea surface temperature, primary productivity levels (e.g., chlorophyll *a*), salinity, seafloor slope, and sediment type, though wildlife responses to these factors varied widely by species and time of year (Chapters 12, 15, and 20-22). There are strong seasonal variations in community composition and wildlife distributions. The breadth of the region is used during spring and fall migration by seabirds, landbirds, sea turtles, cetaceans, rays, and other taxa. Many of these taxa are also part-time or year-round residents of the study area, using it for foraging during the breeding season, or for foraging, roosting, and other activities during non-breeding periods.

#### *Spring*

During the spring (March-May), sea surface temperatures begin to rise, and salinity across surface waters begins to decrease. As the season progresses, primary productivity begins to increase within and adjacent to the bays as nutrient rich spring runoff flows into the bays and mixes with coastal waters (Smith and Kemp 1995). High species diversity was observed in the spring, suggesting that migratory and overwintering species dominate the region’s species composition (Chapter 12). During this time, wintering seabirds departed the region to begin their migrations towards breeding grounds inland or to the north. In our study, Surf Scoters departed the area between January and May, Red-throated Loons between March and May, and Northern Gannets between February and May (Chapters 20-23). During spring, songbirds and shorebirds migrated through the region both along the coast and over open waters (Chapters 17 and 27). Summer resident seabirds, such as terns, shearwaters (Procellaridae), and storm-petrels (Hydrobatidae), arrived after migrating from wintering grounds in the south or breeding grounds in the Southern Hemisphere (Chapters 5, 8, and 17). Spring also marked the arrival of

Bottlenose Dolphins and a variety of sea turtle species, which were predicted to occur in high densities offshore of Virginia during this period (Chapter 15).

### *Summer*

During summer (June-August), overall primary productivity is generally low across the Outer Continental Shelf, but chlorophyll concentrations increase in shallow nearshore areas where upwelling can occur (Xu et al. 2011). Additionally, primary production within the bays is at its peak, contributing to higher productivity at the bay mouths where coastal and estuarine waters mix (Smith and Kemp 1995; Flemer 1970). Through hydroacoustic surveys, we generally observed higher levels of aquatic biomass in these regions during the summer (Chapter 9). Seabirds were generally more associated with nearshore habitat in summer than they were in the spring (Chapter 12). Breeding seabirds were found foraging near the shore and near the mouths of the bays (Chapter 12 and 17); specifically, terns (including Common Terns, and others), were predicted to be associated with nearshore habitat (Chapters 18 and 19). Non-breeding species from the southern hemisphere, such as Great Shearwaters (*Puffinus gravis*) and Wilson's Storm-Petrels (*Oceanites oceanicus*), generally occupied a wider swath of the study area (Chapter 17). In early summer, large numbers of Cownose Rays (*Rhinoptera bonasus*) migrated through the study area on their way to feeding grounds in the Chesapeake Bay and Delaware Bay (Chapter 5; Blaylock 1993). Sea turtles and Bottlenose Dolphins were most abundant across the study area in the summer, with the more inshore coastal ecotype of Bottlenose Dolphins more heavily represented than the offshore population of this species (Chapter 15; Kenney, 1990). In the summer, both Bottlenose Dolphin and sea turtle distributions were influenced by sea surface temperatures and primary productivity (Chapter 15), with Bottlenose Dolphins predicted to occur primarily in nearshore areas, and sea turtles still predicted to occur primarily in the southern end of the study area (Chapter 15).

### *Fall*

In the fall (September-November), the mixing of stratified water re-oxygenates the water column, setting the stage for a significant phytoplankton bloom that occurs across shallow waters in the region between late fall and early spring (Schofield et al. 2008, Xu et al. 2011). Decreased flow of fresh water from the Delaware Bay and Chesapeake Bay during the summer and fall causes salinity to rise over the course of the season, as saltier water is pushed closer to shore. In the early fall, Cownose Rays moved out of the bays and aggregated in dense groups within the study area as they migrated south (Chapter 5). Seabird species composition changed over the course of the fall, as summer residents migrated south to more suitable climes and winter residents migrated into the region from breeding grounds farther north or inland (Chapter 17). Seabirds continued to be more associated with nearshore habitats as compared to winter and spring (Chapter 12). In our telemetry studies, tagged Surf Scoters arrived in the wintering area between October and December, while Red-throated Loons arrived between November and December, and Northern Gannets between August and December (Chapters 20-23). As in the spring, songbirds and shorebirds were recorded flying over open waters as they migrated through the study area (Chapters 17 and 26-27). Nocturnal migratory activity was higher at many offshore locations than along the coast, particularly south of Long Island and offshore of the Carolinas (Chapter 27). Additionally, Peregrine Falcons and Eastern Red Bats migrated over open water through the study area (Chapters 17 and 25; Hatch et al., 2013). Large schools of forage fish were observed in the study area, particularly in nearshore regions (Chapters 9 and 17). Sea turtles remained widespread across the study area through October (Chapter 15). Bottlenose Dolphins also remained until late fall, while Common Dolphins arrived in the study area in November (Chapter 15 and 17).

### *Winter*

During winter (December-February), sea surface temperatures are at their lowest and least variable across the study area (Schofield et al. 2008). Wintering seabirds occupied habitat throughout the study area, with variation in distribution patterns among species (Chapters 12, 17 and 19) and individuals. Northern Gannets were the most ubiquitous seabird in the study area during this period, and were often observed in the bays as well as relatively far out on the shelf in search of prey (Chapters 10, 12, 17, and 22). For Northern Gannets, we found that the chances of foraging increased with the number of sea surface temperature fronts in an area, as the temperature fronts likely aggregated prey (Chapter 24). Scoters were observed in large aggregations at the mouths of the Chesapeake Bay and Delaware Bay (Chapter 17). Common Loons, in contrast, were most often observed individually and were widely dispersed throughout the study area, generally more associated with lower sea surface temperatures (Chapters 16 and 17). Many Bonaparte's Gulls (*Chroicocephalus philadelphia*) were observed in the study area on both survey platforms in winter (Chapters 5 and 8). Alcids were predicted to occur in small numbers throughout the study area (Chapter 19). Baleen whales were most commonly observed during this season. Of the 51 large whales observed in this study, 31 were observed between December and February (Chapter 17). Common Dolphins occupied habitat throughout the study area during the winter, predominantly in offshore areas (Chapters 15 and 17).

### *Persistent patterns*

Primary productivity forms the base of the pelagic food chain on which nearly all species observed during this study rely. In general, primary productivity in the mid-Atlantic is higher in nearshore areas, although patterns vary seasonally (see above). Schools of forage fishes were most commonly observed in nearshore waters, particularly offshore of northern Delaware and Maryland, around the mouth of Delaware Bay (Chapters 5 and 17). In turn, despite seasonal variation in habitat characteristics, areas within about 30-40 km of shore appeared to provide important foraging habitat for a wide range of species year-round. In particular, analyses of survey data indicated that areas near the mouths of the Chesapeake Bay and Delaware Bay were consistent hotspots of species diversity and abundance during this study (Chapter 17). Telemetry studies also highlighted these as high use areas for migratory seabirds in winter, even for species that were highly mobile and used a broad range of habitats, such as the Northern Gannet (Chapter 22). These areas were likely attractive to a wide variety of high trophic-level species, such as seabirds and marine mammals, due to foraging opportunities arising from gradients in salinity, water temperature, and other factors offshore of the mouths of the bays, and the consistently higher primary productivity relative to the broader study area. Areas in northern Maryland within roughly 20-30 km of shore were also consistent hotspots for biodiversity and abundance for many taxa, although this may have been partially driven by the more inshore study design implemented in the region as compared to the remainder of the study area.

Inter-annual variation was substantial, and with only two years of data included in this study, the distribution, abundance, and movement patterns that we present in this report may not be representative of longer-term (e.g., inter-decadal) patterns. In particular, the importance of certain environmental variables, such as sea surface temperature, in predicting animal distributions indicates that these species may well respond to future environmental shifts brought about by anthropogenic effects and climatic change (Griffies 2004, Tallis et al. 2010). This study is an important first step, however, towards understanding how bird, marine mammal, and sea turtle populations in the mid-Atlantic may be exposed to offshore wind energy development and other anthropogenic activities. The results of this study provide insight to help address environmental permitting requirements for current and future offshore development projects, and serve as a starting point for more site-specific studies, risk analyses, and evaluation of potential measures to avoid and minimize those risks.

## Next steps

Several project activities will continue after the project end date, including development of additional joint modeling approaches for integrating boat and digital video aerial survey datasets. All data generated from this project will also be made publically available in late 2015 via the Northwest Atlantic Seabird Catalog (formerly the Compendium of Avian Information)[2], a relational database hosted by the U.S. Fish and Wildlife Service that contains decades of survey data on seabirds, marine mammals, sea turtles, and other wildlife across a broad spatial scale in the northwest Atlantic (O'Connell et al. 2009). Data are also hosted and available for download on the project web page (www.briloon.org/MABS), and certain analytical products are also expected to be incorporated into other public databases, such as the Mid-Atlantic Regional Ocean Council's (MARCO) Data Portal (http://midatlanticocean.org/data-portal/).

Effects to wildlife from offshore development can be thought of as a combination of exposure to development and operation activities; hazards posed to individuals that are exposed; and the implications of individual-level impacts for population vulnerability (Crichton 1999, Fox et al. 2006). In this baseline study of wildlife distributions and movements, we focused on developing a better understanding of wildlife exposure to future offshore development in the mid-Atlantic. While exposure to offshore development does not necessarily indicate that exposed animals will suffer deleterious effects, or that any impacts that do occur will translate to population-level impacts, this study is an important first step towards understanding the implications of offshore wind energy development for bird, marine mammal, and sea turtle populations in the mid-Atlantic United States. The siting and permitting of future projects, as well as efforts to minimize potential effects by considering the timing of construction activities and other approaches, will rely on baseline data such as these. As planning and development move forward, however, it will be important to take steps beyond this baseline assessment in order to focus on species most likely to be impacted due to their conservation status or other factors.

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## Figures

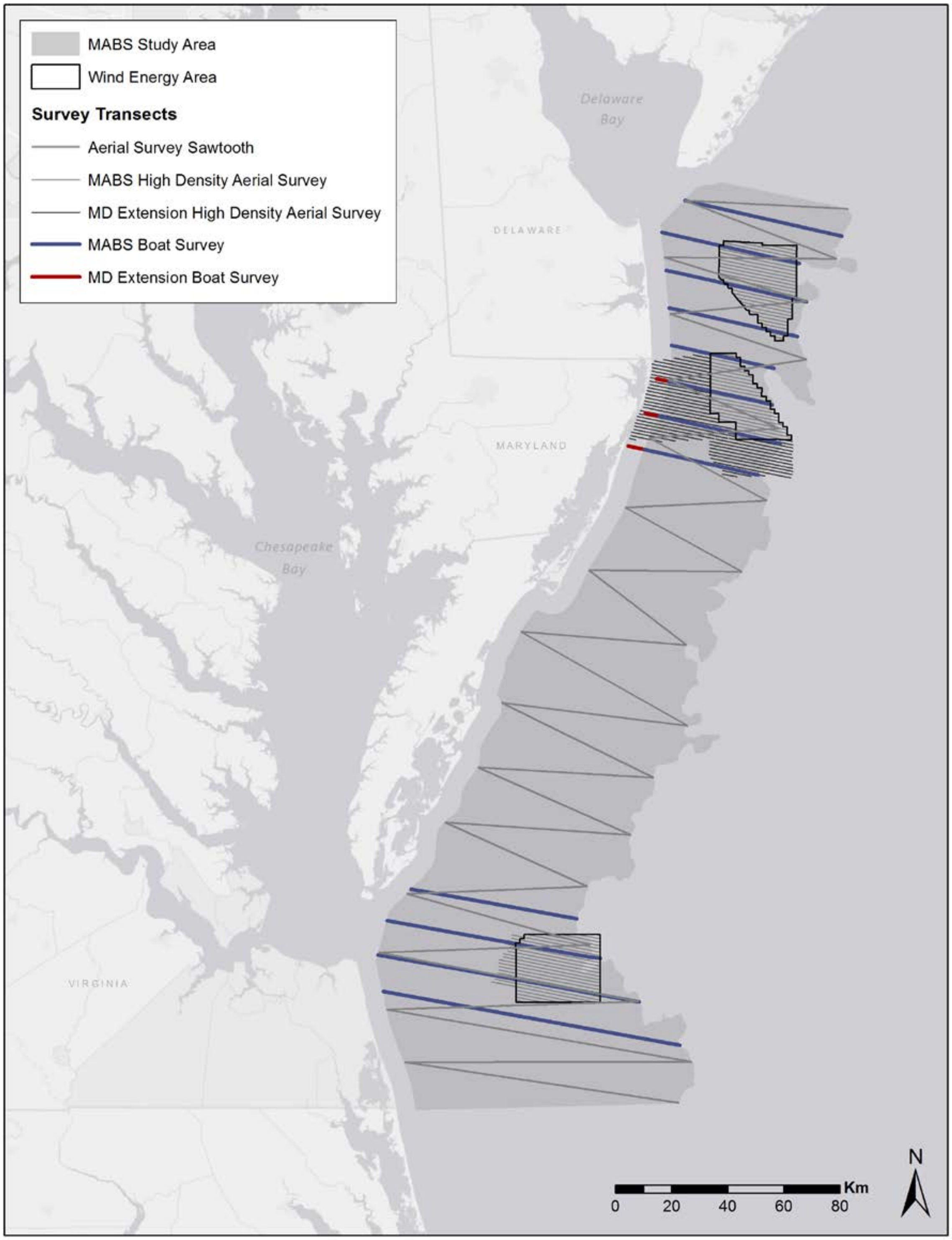

**Figure I. Map of aerial and boat survey transects for the Mid-Atlantic Baseline Studies and Maryland Projects.** High resolution digital video aerial survey transects are shown in gray and black and boat based survey transects are shown in red and blue.

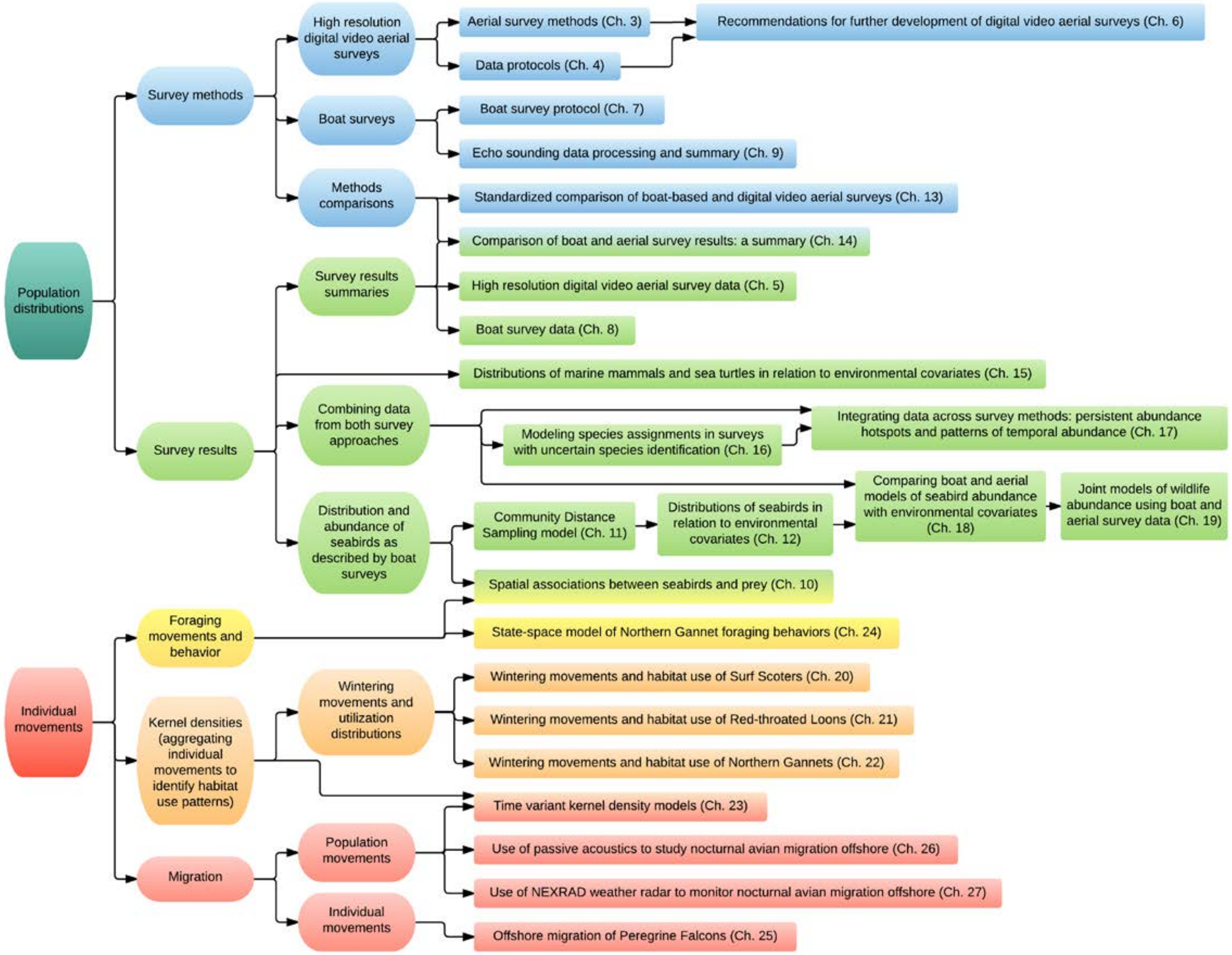

**Figure II. Organization of chapters within this final report.**

# Chapter 1: Ecosystem background and project activities

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** www.briloon.org/mabs

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## Abstract

The mid-Atlantic region is used by a broad suite of wide-ranging marine wildlife species across the annual cycle. This, along with the high levels of productivity in the region, mean that it is essential to understand the dynamics of this ecosystem in order to manage it effectively, particularly with regard to anthropogenic stressors, such as offshore development. The Mid-Atlantic Baseline Studies Project and Maryland Project, described here, provide two years of intensive survey data and other information (2012-2014) to improve our understanding of this ecosystem.

The study areas include waters on the Outer Continental Shelf off the coasts of Delaware, Maryland, and Virginia, and extend from the state-federal boundary (5.6 km from shore) east to the 30 m isobath (with the exception of some waters offshore of Maryland, where the study area extends westward to the shoreline). Methods employed in this study included boat surveys, high resolution digital video aerial surveys, satellite telemetry for focal species, and several approaches for examining nocturnal avian migration patterns. This is the first study to use high resolution digital video aerial surveys on a large scale in North America, as it is a relatively new method for collecting distribution and abundance data on animals in the marine ecosystem. We discuss the relative strengths of digital video aerial surveys and other methods employed in this study, with a particular focus on comparing boat-based surveys and digital video aerial surveys. We also briefly discuss the various approaches used to present results in this report; understanding each analytical method and its limitations is essential to appropriately interpret maps, figures, and other analyses.

Boat-based and digital video aerial surveys each showed distinct benefits in detecting different taxa. Digital aerial surveys have the added advantage of being auditable and archivable, and include an extensive quality assurance process, which may lead to a greater degree of reliability in species identifications. The safety and speed with which digital aerial surveys can be conducted also make this approach attractive in the offshore environment, and the capabilities of digital aerial surveys will likely continue to improve with technological advances in the field. Boat surveys can provide detailed behavioral data, had generally better rates of identification of animals to species, and the analytical approaches for boat survey data are well established.

This study also provides the first comprehensive assessment of taxa that are likely to become exposed to future offshore wind energy development in the mid-Atlantic region. This information may be used during permitting processes for future development, as well as for siting projects and designing development plans that minimize wildlife impacts.

## Ecosystem background

The interactions among biota (e.g., organisms, populations, and communities) and abiota (i.e., the physical environment) comprise an ecosystem. The study of ecology attempts to identify these critical connections between organisms and their environment, and explain how those relationships affect, or are impacted by, the physical attributes of their habitats. Establishing baseline ecosystem function, to identify areas of important habitat and high species biodiversity, is crucial to wildlife management.

For the last few decades there has been wide recognition that traditional methods of resource management, where management actions or environmental assessments target a single population, species, or issue, are extremely limiting or potentially misleading (Ehler and Douvere, 2009). Since the 1990s, management and regulatory agencies have increasingly recognized the importance of addressing research, conservation, and planning at the ecosystem scale (Christensen et al., 1996; Grumbine, 1994). Despite this fundamental shift in our collective thinking, however, few research studies are conducted at broad enough geographic or temporal scales to provide the data necessary to fully understand the complex relationships between species and their dynamic physical environments (Arkema et al., 2006; Leslie and McLeod, 2011; Ruckelshaus et al., 2008). In general, our narrow understanding of these relationships hinders the development and implementation of large-scale, ecosystem-wide management strategies, as well as the prediction of responses of species to broad environmental shifts brought about by anthropogenic effects and climatic change (Griffies, 2004; Tallis et al., 2010).

Marine ecosystems are particularly complex and dynamic assemblages that involve multitudes of co-evolved species. Thus, research studies integrated across taxonomic groups and among trophic levels are critical to understanding marine ecosystem processes and mechanisms (Wiebe et al., 2009). To date, marine studies at the ecosystem scale have largely focused on the assessment and management of commercial fish stocks (Pikitch et al., 2004; Smith et al., 2007). In this study, however, we not only analyze the distributions and movements of prominent marine wildlife species across a large swath of the mid-Atlantic coastal region, but also examine the influence of biotic and abiotic factors, such as productivity, depth, and salinity, on these distributions and movements. This ecosystem-based approach establishes a broad baseline from which we may be able to detect and understand the impacts of future activities in this ecologically and economically important region.

### *Importance of the mid-Atlantic study area to wildlife*

Politically, the coastal mid-Atlantic region includes the states of Virginia, Maryland, Delaware, New Jersey, and New York. Oceanographically, however, the waters off the East Coast of the U.S. are divided into three large geographic zones (the Gulf of Maine/Bay of Fundy, Mid-Atlantic Bight, and the South Atlantic Bight). The central sector, the Mid-Atlantic Bight, spans an area from Cape Cod south to Cape Hatteras. This central region of the Outer Continental Shelf is characterized by a broad expanse of gently-sloping, sandy-bottomed continental shelf that extends up to 150 km to the shelf edge, where the waters reach about 200 m deep. Beyond the shelf edge, the continental slope descends rapidly to around 3,000 m. Much of this mid-Atlantic coastal region is bathed in cool Arctic waters, brought south by the Labrador Current as it travels down the east coast. At the southern end of this region, around Cape Hatteras, these cool waters collide with the warmer Gulf Stream current (Townsend et al., 2006). The region also exhibits a strong seasonal cycle in sea surface temperatures (spanning approximately 5-

30 °C), and in salinity, with large volumes of fresh water emptying onto the shelf via the Hudson Estuary, Delaware Bay, and Chesapeake Bay.

Seasonal stratification on the shelf drives overall annual primary productivity across the broader study area, with the largest and most persistent phytoplankton blooms in the late fall and winter (Schofield et al., 2008; Yoder et al., 2001). However, areas near the mouths of the Delaware Bay and Chesapeake Bay typically have the highest levels of chlorophyll *a* in the study area, due to their proximity to highly productive estuarine ecosystems. The influxes of fresh water from the bays deliver nutrients such as nitrogen and phosphorous and year-round mixing of saline and fresh waters through estuarine circulation, in combination with strong tidal currents, boost primary productivity in these areas. As water flows from the bays into the study area, nutrient- and phytoplankton-rich waters are swept southwards by the Labrador Current into other nearshore areas. In these shallow coastal waters, sunlight is able to penetrate a relatively high proportion of the water column (Schofield et al., 2008; Xu et al., 2011), further fueling photosynthetic activity and growth of phytoplankton where nutrients are available.

Phytoplankton blooms are followed by a pulse in secondary productivity–zooplankton species foraging on the phytoplankton–which in turn become food for larger predators, such as small fishes. The Mid-Atlantic Bight is generally rich in these small, schooling epipelagic fishes, known as 'forage fish' due to their critical importance for many piscivorous predators, and their pivotal role in driving ecosystems worldwide (Pikitch et al. 2014). In the mid-Atlantic region, key forage fish species include Atlantic menhaden (*Brevoortia tyrranus*), Atlantic mackerel (*Scomber scombrus*), butterfish (*Peprilus triacanthus*), sand lance (*Ammodytes americanus* and *A. dubius*), anchovies (including *Anchoa mitchelli*, *A. hepsetus*, and *Engraulis eurystole*), and 'river herring', including the alewife (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*; Clay et al., 2014; Kenney et al., 1997; Safina et al., 1990). Two large invertebrate species, the longfin inshore squid (*Loligo paeleii*) and the northern shortfin squid (*Illex illecebrosus*), are also important prey items for a broad range of predators in the Mid-Atlantic Bight (Dawe et al., 2007; Hendrickson, 2004). In this study, we observed numerous shoals of small fish across the study area, most commonly from May to October (Chapter 17). The presence of these forage fish populations indicate the exceptional productivity of the area, and are likely responsible, in part, for the relatively high density of predators that use the area (Chapter 10).

Thus, the mid-Atlantic region is used by a broad range of marine wildlife species across the entire annual cycle, due in part to a relatively high level of productivity, as compared to many other areas in the western North Atlantic (Yoder et al., 2001). The importance of the region to wildlife is also partially due to the region's central location on the eastern edge of the continent (a major migratory corridor for many species). This results in a complex ecosystem where the community composition is constantly shifting, and temporal and geographic patterns are highly variable. The mid-Atlantic supports large populations of marine wildlife in the summer, some of which breed in the area, such as coastal birds and some sea turtles. Other summer residents visit from the Southern Hemisphere (where they breed during the austral summer), such as shearwaters (Procellaridae) and storm-petrels (Hydrobatidae). In the fall, many of the summer residents leave the area and migrate south to warmer climes, but are replaced by species that breed further north and winter in the mid-Atlantic, such as Northern Gannets (*Morus* *bassanus*). Many marine species also make annual migrations up and down the eastern seaboard, taking them directly through the mid-Atlantic region in spring and fall. Many migrant terrestrial species, such as landbirds and bats, may follow the coastline on their annual trips, or choose more direct flight routes over expanses of open water.

The Mid-Atlantic Baseline Studies and Maryland Projects fill a significant information gap for wildlife in a large swath of the mid-Atlantic region between New Jersey and North Carolina. In part, this area is a focus due to its ecological significance and relative lack of data on wildlife distributions. Additionally, this region has great economic importance, including commercial fisheries, shipping, and the potential for offshore renewable energy development. Areas with an annual average wind speed of 7 m/s (15.7 mph) or greater at 90 m in height are considered suitable for offshore wind energy generation (Schwartz et al., 2010). The mid-Atlantic region has a relatively high wind energy potential, with an annual average predicted offshore wind speed of 7-9 m/s (16-20 mph), and is also located near large energy markets on the U.S. Atlantic coast (Baker, 2011). Thus, the region has been a focus for offshore wind developers and regulators in recent years, and several of the first federally designated Wind Energy Areas (WEAs) are located off the mid-Atlantic coast. To minimize the effects of development activities on wildlife populations, however, the complexities of this ecosystem require that a range of study methods be used to obtain a comprehensive view of ecosystem structure and configuration.

In this overview of project methods, we discuss the range of study approaches used to examine the diurnal and nocturnal distributions, abundance, habitat use, and movements of sea turtles, marine mammals, birds, and other wildlife. Within this report, we present survey results in a variety of ways, and a brief overview of the advantages and disadvantages of each analytical approach are also discussed.

## Methods used in the Mid-Atlantic Baseline Studies and Maryland Projects

The Mid-Atlantic Baseline Studies project area extends from three nautical miles off the coastline (the interface of state and federal waters) east to the 30 m isobath (roughly 40-90 km from shore), and includes waters on the Outer Continental Shelf off the coasts of Delaware, Maryland, and Virginia (Figure 1-1). The Maryland Project, added to the scope of the Baseline Studies Project during the second year of surveys, includes more intensive coverage of waters offshore of Maryland, including state waters (within 5.6 km of shore). Study methods included boat surveys, high resolution digital video aerial surveys, satellite telemetry for focal species, and several approaches for examining nocturnal avian migration patterns. This study includes the first use of digital aerial surveys on a large scale in North America, and the combination of different survey approaches allowed for a comparison of digital aerial vs. boat survey results.

Each of the methods that we used to examine marine wildlife distributions and movements in the mid-Atlantic had inherent strengths and weaknesses. By using a complimentary suite of methods, we aimed to minimize knowledge gaps and develop a comprehensive understanding of the mid-Atlantic marine ecosystem.

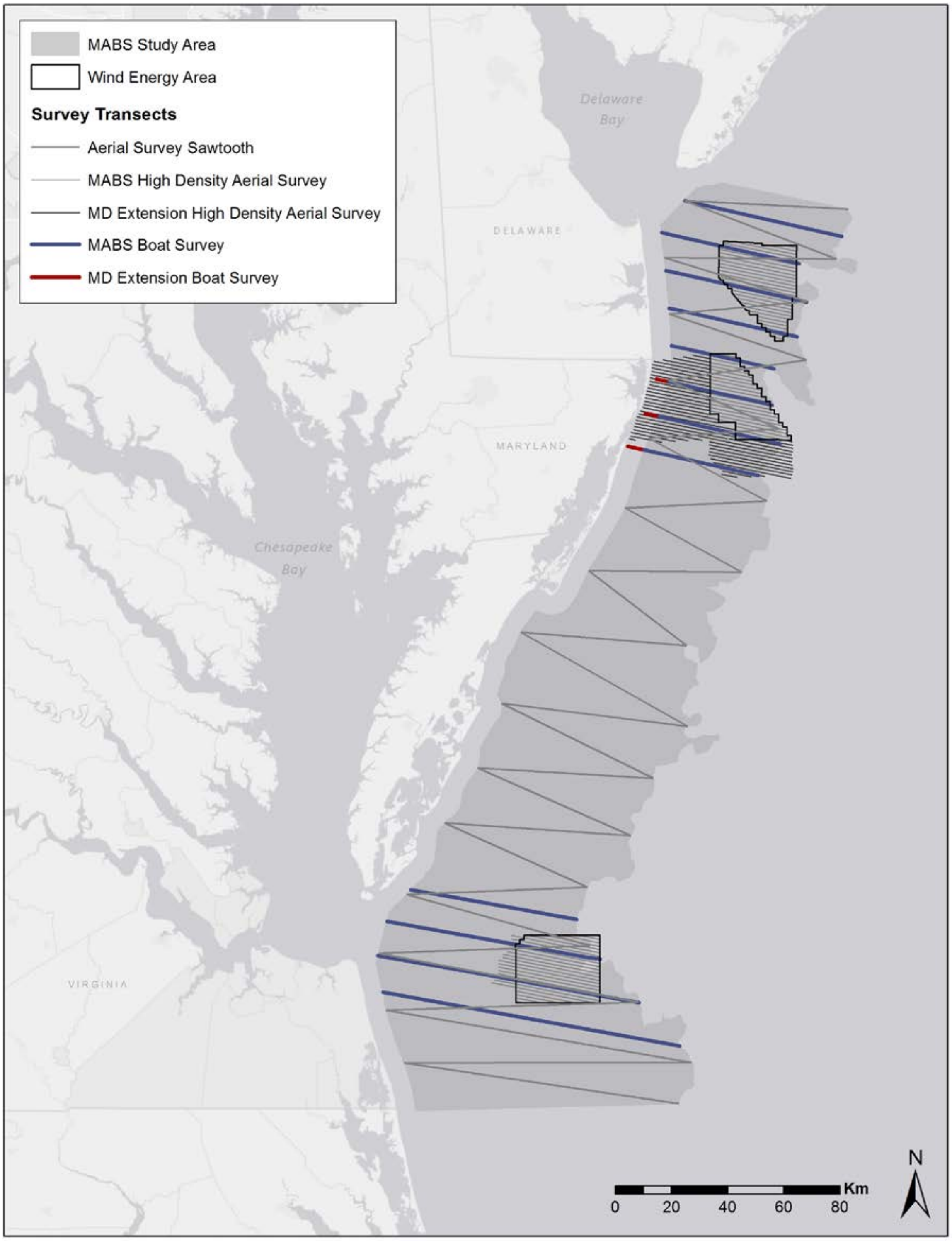

**Figure 1-1. Map of aerial and boat survey transects for the Mid-Atlantic Baseline Studies and Maryland Projects (2012-2014).** High resolution digital video aerial survey transects are shown in light and dark gray; boat-based survey transects are shown in blue and red. Maryland extension transects (initiated in 2013) are shown in red (boat) and dark gray (aerial).

### Boat surveys

Boat-based surveys are a widely-used method to monitor offshore wildlife. They provide a great deal of information about marine ecosystems, and are a key element in this study of the mid-Atlantic. Due to the relatively slow speed of survey vessels, observers have considerable time to collect data on species presence and abundance, and often record information on observed behaviors, such as an animal's interactions with conspecifics, or other marine fauna (e.g., while in multi-species feeding aggregations; Chapters 7, 10, 18A). Observers also collect *in-situ* environmental and biological data, such as wind speed, wave height, sea surface temperature, salinity, and biomass densities (Chapters 7 and 9). Relating these directly to sightings can help explain animals' distributions and the drivers of those distributions, as well as variations in detection rates for different species (Ainley et al., 2005).

Detection of animals is not perfect, although there are methods to account for missed animals on the survey transects (Hedley and Buckland, 2004; Royle et al., 2004; Spear et al., 2004). An observer's ability to detect an animal correctly decreases with increased distance between the observer and the animal, and can be further limited by deteriorating weather conditions (Royle et al., 2004). Depending on the size of the survey vessel and target taxon, boat-based surveys are limited in their ability to collect data under certain weather conditions; mammal-focused surveys have particularly strict limitations on the wave height in which accurate data can be collected (Evans and Hammond 2004). The quality of the data collected, including species identifications and distance data used for developing abundance estimates, is also dependent on the skills of the observer, which can be variable (Spear et al., 2004). When observers are unable to identify individuals to species, they are trained to record the genus or family, so as to avoid misidentification. Uncertainty in these species identifications is difficult to measure, however, and is generally under-recognized or ignored in boat-based surveys, with potential implications for abundance estimation (Conn et al., 2013; Hobbs and Waite, 2010). Employing two independent observers can be used to assess observer biases (Nichols et al., 2000; Ronconi and Burger, 2009), but without any permanent record of observations, it is difficult to verify identifications on boat surveys.

The movement of the survey vessel through the environment can alter animal behaviors as well, whether through disturbance or attraction (Chapter 13; Bodey et al., 2014; Schwemmer et al., 2014; Spear et al., 2004). Some marine birds, such as scoters (*Melanitta* spp.), auks (Alcidae), and loons (*Gavia* spp.), will flush or dive when approached by a boat, even from several hundred meters away (Henkel et al., 2007; Schwemmer et al., 2014). Other seabirds that scavenge from fishing boats, such as gannets, are attracted to slow-moving vessels from several kilometers away (Spear et al., 2004; Votier et al., 2013). Marine mammals and sea turtles also react to the presence of vessels, with responses varying depending on the size/type of vessel, vessel speed, and the species involved (Hazel et al., 2007; Mattson et al., 2005; Normandeau Associates Inc., 2013; Richardson et al., 1995).

There is a tradeoff between maximizing survey coverage and minimizing sampling time. Surveying more of the study area provides greater statistical power, as more information on species distributions can be collected over a broader range of environmental features; however, increased time to cover the study area risks greater turnover of animals in the study region, resulting in the potential for double counting of individuals or groups as they move around the area (Spear et al., 2004). In addition, boat surveys are conducted during daylight hours in fair weather conditions, which limit our understanding of nocturnal behaviors and animal behaviors in harsher weather conditions.

Lastly, data collected from boat-based surveys present "snapshots" at given points in time. Although boat-based surveys provide an excellent opportunity for collecting behavioral and population-level data across a broad spatial extent (e.g., seasonally), they do not easily allow for understanding individual movements and use of the study area. We were able to compensate for some of these limitations in this study through the use of weather radar and satellite telemetry.

### *High resolution video aerial surveys*

High resolution video aerial surveys are a relatively new method for collecting distribution and abundance data on animals in the marine ecosystem (Thaxter and Burton, 2009). Though digital video aerial surveys have become common practice for offshore wind energy planning and monitoring in Europe (Buckland et al., 2012), this study is the first to use these methods on a broad spatial scale in the United States. Digital aerial surveys have a high cost efficiency on broad spatial scales, and are expected to largely replace traditional visual surveys, by boat or aircraft, in the offshore environment in Europe (Buckland et al., 2012). High resolution video surveys collect information on abundance for most species, and the width of the survey transect is predetermined by the camera's field of view, allowing for easy calculation of the size of the surveyed area. Given the altitude at which surveys can be flown (>600 m), there is minimal disturbance to marine wildlife, unlike with survey vessels (Chapter 13; Buckland et al., 2012). This high altitude is considerably safer than low-level visual surveys, which are flown at 60-180 m, and also allows for the collection of survey data pre- and post-construction at offshore wind facilities. High-resolution digital video aerial surveys also allow for the estimation of flight heights for flying animals using parallax, or the movement of animals relative to the ocean background (Chapter 5; Hatch et al., 2013), data which are sometimes used to try to understand potential collision risk for animals flying through a project site. Digital aerial surveys are also excellent for collecting data on aquatic animals such as marine mammals and sea turtles (Chapters 14-15; Normandeau Associates Inc., 2013). As with boat surveys, digital aerial surveys are only flown in daylight hours under fair weather, which limits our understanding of animal behaviors at night and in harsh weather conditions. As with boat surveys, digital aerial surveys provide a "snapshot" of animal distributions at a given point in time, rather than data on movements or behaviors within the ecosystem.

Importantly, the data collected using digital surveys are recorded, allowing for species identification verifications, the application of rigorous audit protocols, and archived footage for later review (Chapters 3-4). This is a distinct advantage over visual survey approaches. The survey transects are relatively narrow, however, which in our study may have led to problems of availability for highly mobile animals (Chapter 13). Researchers continue to develop solutions to correct for many of the detection biases described above for boat-based surveys (Chapter 11). Digital aerial surveys avoid the distance bias common to visual methods, but to date, other forms of detection bias have not been addressed for digital aerial surveys (Chapter 13).

In this study, identification to species of most taxa in digital video aerial surveys was lower than identification rates for boat surveys (Chapters 13 and 14). Recent technological advancements in camera designs and image quality have improved identification rates beyond what occurred in this study (HiDef Aerial Surveying, unpubl. data), but it is likely that some taxonomic groups may remain easier to identify from a vessel (Chapter 14). The high speed of digital aerial survey aircraft, while beneficial for cost-effective completion of surveys in large or remote study areas, means that digital surveys provide only basic information on behavior, such as “flying”, “sitting”, and “moving”, because the footage recorded of a specific target animal is brief (<1 second), and more complex behaviors can rarely be discerned.

### *Satellite telemetry*

Satellite telemetry allows us to track the movements of individual animals within their environment, and potentially identify marine biodiversity hotspots and ecologically important areas (Montevecchi et al., 2012). In this study, we deployed satellite transmitters on four different avian taxa (Chapters 20-25). With this method temporal coverage is limited only by battery power and tag longevity, making it possible to track movements of individual birds at a seasonal or annual temporal scale and to collect data regardless of weather or time of day (though such tracking is seldom continuous, due to power limitations). There is a distinct tradeoff between this level of individual temporal and spatial coverage and sample size, however, and it can be difficult to extrapolate population-wide distributions and behaviors from a few individuals (Lindberg and Walker, 2014). Moreover, tracking does not allow for development of relative abundance estimates. Instead, kernel density estimates are often used to characterize and visualize home ranges, and utilization distributions can be extended to quantify the relative frequency distribution of an animal’s occurrence in space and time (Keating and Cherry, 2009; Loring et al., 2014; Worton, 1989).

The mid-winter deployment of satellite tags in the study area, dictated by the study design, proved to be problematic for studying detailed winter movements of individuals because it split the winter season over two years. Tag longevity for implants was also disappointing in some cases, also limiting our ability to track individuals through an entire second winter season.

By definition, telemetry studies are species-specific—they do not provide data on the broader marine community (at least not directly). Remote tracking by satellite remains the best approach available for studying animal movements, however, including both diurnal and nocturnal movements. Although transmitters were only deployed on birds in this study, there is analogous technology available for turtles and mammals (which is being deployed on turtles and pinnipeds in other studies, including the ongoing Atlantic Marine Assessment Program for Protected Species [AMAPPS[1]] study). Devices can only be deployed on species and individuals robust enough to carry them, but increasingly smaller and lighter units are in development (Guilford et al., 2011).

### *Nocturnal avian passive acoustics*

Oceans and other large bodies of water can act as barriers to migrating landbirds, including passerines and raptors, but many species make long transoceanic flights (Delingat et al., 2008). Cape May and Delaware Bay are both known as areas where large numbers of migrants stop over during migration (Clark et al., 1993; Moore et al., 1995), but there is less known about migrant use of offshore regions of

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[1] www.nefsc.noaa.gov/psb/AMAPPS/

the mid-Atlantic. Many landbird species migrate at night and emit short species-specific vocalizations during flight (Evans 2012). Nocturnal passive acoustic monitoring stations can record these flight calls and provide data on species presence, as well as an index of migratory activity. In this study, we deployed an avian passive acoustic monitoring system on the survey vessel to test the effectiveness of this method from such a platform, and to obtain preliminary data about the species composition of nocturnal migrants in the offshore environment of the mid-Atlantic (Chapter 26).

Passive acoustic monitoring is useful for obtaining information on the species composition of nocturnal migrant populations offshore, which is not currently possible via other methods explored in our study. It also provides extensive temporal coverage, as recorders run continuously. There are limitations to this method, however, including poor geographic coverage, intensive analytical requirements, difficulties with differentiating some species acoustically, variation in acoustic activity among target species, and issues associated with attempting to use call frequency as an index of abundance. But our options for studying nocturnal migration over water are currently limited, and even presence information (unaccompanied by information on behavior or abundance) can be useful data. Our study was focused on avian migration, but equivalent studies exist for other acoustically active taxa, including marine mammals, bats, and some fishes. The Bureau of Ocean Energy Management (BOEM), the Maryland Department of Natural Resources[2], and other agencies are currently funding mammal acoustic studies in the study area (e.g., Bailey and Rice, 2015; Muirhead et al., 2014).

### *WSR-88 weather radar*

Weather surveillance radars regularly detect "bioscatter", reflectivity caused by biological entities in the atmosphere, and are increasingly being incorporated into studies of avian and bat nocturnal migratory activity (Chilson et al., 2012). Our study incorporated WSR-88 (NEXRAD) weather radar to identify potential offshore migration pathways and timing, as well as environmental and temporal variables correlated with these patterns (Chapter 27). Though they lack the fine scale resolution of traditional marine radar, NEXRAD data allow for efficient monitoring of geographical and temporal patterns in migration on a broad scale (Gauthreaux and Belser, 2003), at any time of day or night, and have proved useful for developing a better understanding of patterns of offshore migratory activity.

Geographic coverage is poor in offshore areas along the eastern seaboard, compared with terrestrial locations, and characteristics of the radar beam make it increasingly difficult to detect low altitude and low density bioscatter with increasing range from the radar, though we present an analytical approach for addressing this issue (Chapter 27). NEXRAD data also does not allow for identification to species, nor direct translation of migratory activity (measured as radar reflectivity) to actual abundance of animals using the radar technology in this study, since the sizes of individuals being detected are unknown. However, weather radars provide information on the nocturnal distributions and migratory patterns of animals at a scale which is impossible to achieve via other methods. Innovations developed during this study allowed for targeted exclusion of meteorological phenomena, greatly improving the sample size of available data and allowing for examination of migratory activity even during nights with precipitation, which had previously been impossible.

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[2] www.boem.gov/Determining-Offshore-Use-by-Marine-Mammals-Maryland-PAM/

### *Comparing and integrating methods*

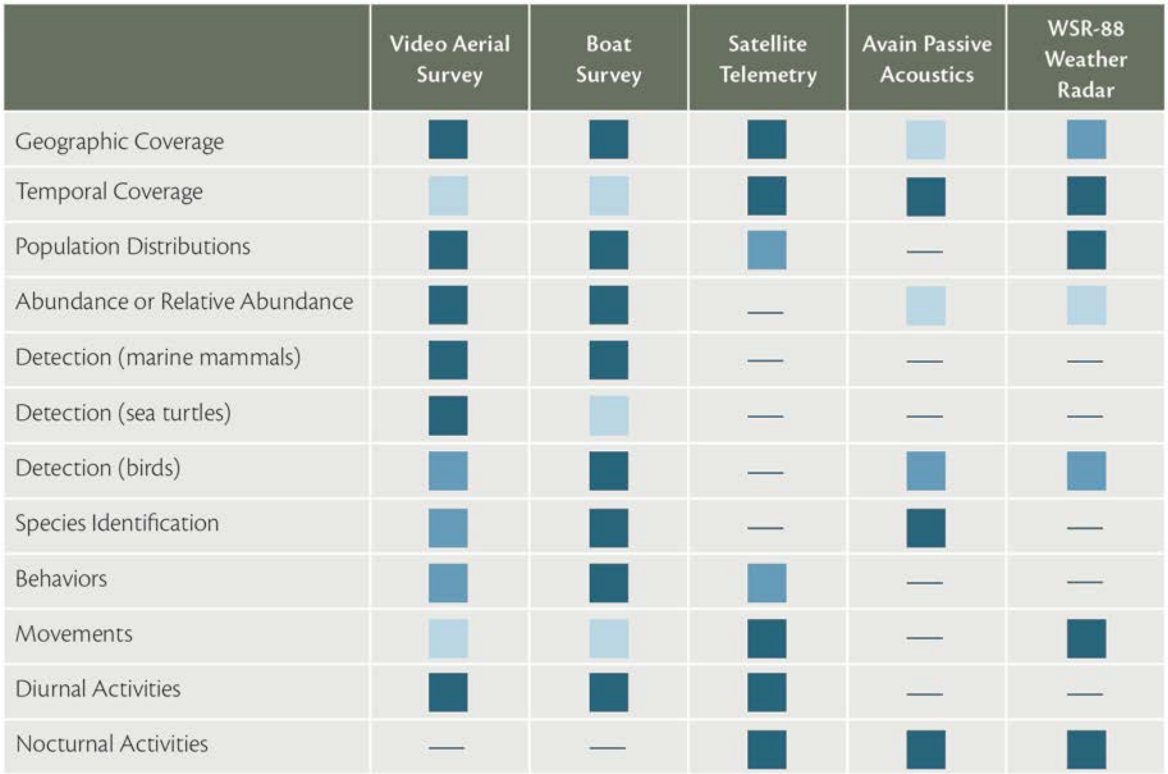
By using the five research methods outlined above to collect a broad range of data, we aimed to develop a more complete picture of the mid-Atlantic study region. For example, the inclusion of satellite telemetry provided information on broad-scale movements of specific species in the environment, including nocturnal movements and habitat use, which was missing from our survey data; but the survey data allowed for population-level analyses of abundance and distributions that were not possible with tracking data alone.

Each of the methods that we used to examine marine wildlife distributions and movements in the mid-Atlantic had inherent strengths and weaknesses (Table 1-1). Our evaluation of the utility of each survey method in documenting different types of data is necessarily subjective in many cases, and is dependent upon the specific study design implemented for this project (i.e., the study area, available technology, sample size, and other factors). Seabird telemetry efforts in this study, for example, provided useful information on population distributions in the mid-Atlantic, due to the large sample sizes made possible via collaborations among multiple organizations and funding agencies (Chapters 20-24), but many telemetry studies are limited by sample size, and are constrained in their population inference as a result.

Compared to the other study methods used in this project, boat and aerial surveys provided relatively comprehensive information on wildlife populations in the offshore environment (Table 1-1). Each showed distinct benefits in detecting different taxa (Figure 1-2). High resolution digital video aerial surveys provided better detection rates for aquatic animals, likely due to a combination of reduced disturbance, reduced glare, and a better field of view than is provided by either boat or visual aerial surveys, allowing for submerged animals to more easily be detected in the upper reaches of the water column (Chapters 5 and 14; Normandeau Associates Inc., 2013). Boat surveys provided better detection rates for many birds, however (Figure 1-2), which is probably due to a combination of availability bias, detection bias, and identification issues in digital video aerial surveys (Chapters 5 and 13-14). Digital aerial surveys have the advantage of being auditable and archivable, however, and include an extensive quality assurance process, which may lead to a greater degree of reliability in species identifications. The safety and speed with which digital aerial surveys can be conducted also make this approach attractive in the offshore environment, and the capabilities of digital aerial surveys will likely continue to improve with technological advances in the field. Boat surveys can provide detailed behavioral data, however, and had generally better rates of identification of animals to species. The analytical approaches for boat survey data are also well established.

Though each methodology has clear limitations, survey data were a main focus for determining distributions and relative abundance of taxa of interest throughout the study area, and for developing analytical products that are useful for marine spatial planning and decision making regarding offshore development activities. By using a complementary suite of methods, we aimed to minimize knowledge gaps and develop a more comprehensive understanding of the mid-Atlantic marine ecosystem.

**Table 1-1. Methods for studying offshore wildlife that were incorporated into this study.** Relative strengths and weaknesses of each approach are indicated by depth of color (dark blue = good; medium blue = fair; pale blue = poor). A dash indicates that data were not available from this survey method. Values are subjective; for example, detection of avian species in our boat surveys was probably better than detection in our digital video aerial surveys in many cases, at least after correction for distance bias in aerial data (Chapters 13-14 and 18), so boat surveys were categorized as “good” for this type of data, while digital video aerial surveys were “fair.” Avian passive acoustics were also rated “fair” for this data category, as some species and individuals emit flight calls infrequently, or at different time periods throughout the night, limiting species-specific detections.



| | Video Aerial Survey | Boat Survey | Satellite Telemetry | Avain Passive Acoustics | WSR-88 Weather Radar |
|---|---|---|---|---|---|
| Geographic Coverage | ■ | ■ | ■ | □ | ▨ |
| Temporal Coverage | □ | □ | ■ | ■ | ■ |
| Population Distributions | ■ | ■ | ▨ | — | ■ |
| Abundance or Relative Abundance | ■ | ■ | — | □ | □ |
| Detection (marine mammals) | ■ | ■ | — | — | — |
| Detection (sea turtles) | ■ | □ | — | — | — |
| Detection (birds) | ▨ | ■ | — | ▨ | ▨ |
| Species Identification | ▨ | ■ | — | ■ | — |
| Behaviors | ▨ | ■ | ▨ | — | — |
| Movements | □ | □ | ■ | — | ■ |
| Diurnal Activities | ■ | ■ | ■ | — | — |
| Nocturnal Activities | — | — | ■ | ■ | ■ |

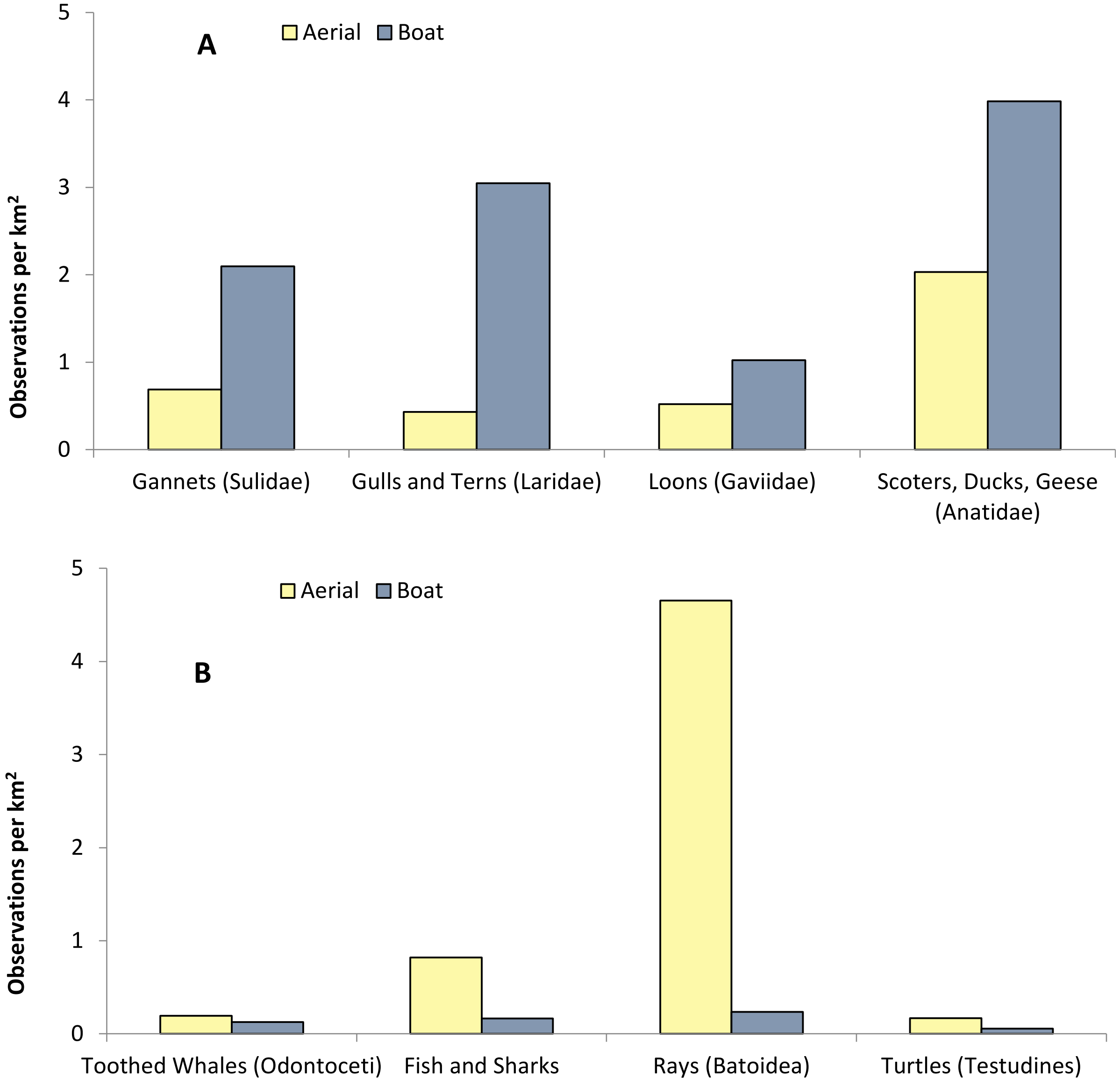


**Figure 1-2. Comparison of total effort-corrected boat and aerial survey counts by taxon for all surveys.** Aerial densities were calculated using transect strip widths (either 200 or 300 m). A) Effective boat transect strip widths were estimated using distance data for each avian family (Chapter 14). B) There were insufficient data from boat surveys to develop reliable distance curves for many aquatic taxa, so estimated boat transect widths for this figure were based on the median distance of observations from the boat across all surveys (Odontoceti = 300m; Fish/Sharks = 50m; Batoidea = 7.5m; Testudines = 100 m). Observations of groups that were not individually counted or identified (e.g., bait balls, ray schools) are excluded from this figure (see Chapter 5).

## Interpretation and analysis of survey data

Analysis and presentation of data collected via the above study methods can take many forms. In this report, we adopted a variety of approaches for presenting spatially and temporally explicit results, and it is essential to understand their limitations in order to use resulting products appropriately. Simply mapping raw survey or tracking data, while intuitively straightforward, has several severe limitations; for example, mapping raw data precludes prediction of animal distributions, so that the only locations where estimations about animal distribution or abundance can be made are directly where surveys were conducted (Table 1-2). In addition, there are several known sources of bias associated with survey data that prevent consistency across a spatial extent, making it hard to compare values between different locations without first controlling for those biases (Burnham and Anderson, 1984; Spear et al., 2004; Wintle et al., 2004).

Analysis of survey data often includes a variety of analytical corrections to account for bias and more accurately estimate how many (and which) animals are present. For example, sea state and the distance of animals from the boat transect are common factors that affect detection of animals (Chapters 11-12; Spear et al. 2004; Royle, Dawson, and Bates 2004; Hedley and Buckland 2004; Evans and Hammond 2004). We would expect lower detectability of animals that are far away, especially during high sea states, so by including these two factors in a model of animal abundance, we estimated the proportion of animals that observers may have missed (Chapter 12). Survey effort is another factor that greatly influences observations made in a given location. If survey effort varies across a region (as in our boat and aerial surveys across the mid-Atlantic study area), then areas surveyed more intensely are going to appear to support more animals. Thus, in addition to correcting for sources of bias in survey datasets, it is essential to correct for the amount of survey effort expended in different areas in order to develop maps of distribution or abundance that show real biological patterns (Table 1-2).

Biotic and abiotic factors, including weather, habitat characteristics, prey distributions, and hydrography, drive the distribution and abundance of marine wildlife (Ainley et al., 2005). Environmental factors, or covariates, that we believe could be important for predicting animal distributions or abundance can be incorporated into a single modeling framework with effort and bias corrections (Chapters 12, 15, and 18). This allows us to identify correlations between these covariates and to understand the factors influencing animal distribution. These relationships can also be used to predict animal distributions to locations or time periods where/when surveys were not conducted, given available environmental covariate data. Maps showing a continuous prediction surface across a large spatial scale are generally based on model predictions, rather than observed data; the data are used to determine relationships with environmental factors, and those relationships are mapped across the scale of the environmental factors.

**Table 1-2. General approaches for presenting spatial data from offshore surveys.** The distance between animals and the transect line, observer abilities, and environmental conditions all can affect detection of animals, causing biases in observed data that must be corrected in order to use survey data to estimate wildlife densities or abundance.

| Data Presentation | Advantages | Disadvantages | Example (map from this report) |
|---|---|---|---|
| Raw observation data | No assumptions—presents what was observed and where. | Does not incorporate known sources of observer bias.<br>Does not allow for predictions of wildlife distribution/abundance in areas that were not surveyed, or to predict future distributions in surveyed areas. | Figure 17-28 (large whales observed during boat and aerial surveys) |
| Bias-corrected and effort-corrected data | Uses known sources of observation bias to correct raw data and improve estimates of where animals are present, and in what numbers.<br>Uses information about where animals were not seen during surveys, in order to correct counts for variation in survey effort between locations. | Does not allow for predictions of wildlife distribution/abundance in areas that were not surveyed, or to predict future distributions in surveyed areas. | Figure 5-12 (maps of relative ray densities, corrected for effort, across areas surveyed by plane) |
| Predictive model | Uses other environmental or habitat data to find correlations with effort- and bias-corrected observation data. Allows researchers to attempt to identify WHY animals are there, not just where they are.<br>Allows for predictions of wildlife distribution/abundance in areas that were not surveyed, or to predict future distributions in surveyed areas. | Predictions include several implicit assumptions (e.g., consistency of species-habitat relationships across unsampled time/space) and require habitat data from unsampled locations that have similar levels of variation as the sampled habitat. | Figure 12-3 (predicted abundance of scoter flocks during the nonbreeding season, throughout the study area) |

There are several types of modeling frameworks that can incorporate these different objectives; in this study we have focused specifically on generalized additive models (GAMs) and on generalized linear models (GLMs) using a hierarchical Bayesian framework (for a review of the use of GAMs and GLMs in ecology, see Guisan et al. 2002). Hierarchical approaches in a Bayesian framework (Chapters 11-12, 16, and 18) can be useful for situations where distribution patterns or resource use vary with scale, and where species of interest are highly mobile and may be periodically unavailable for detection (Mordecai et al., 2011). They can provide an easily interpretable measure of uncertainty in predicted results, and allow for better fit of the model to observed data (Gardner et al., 2008; Zipkin et al., 2010). Generalized additive models (Chapter 15) are semi-parametric extensions of GLMs that use smoothing functions for predictor variables to improve model fit, and can be particularly useful for situations with highly non-linear and non-monotonic relationships between predictor and response variables (Guisan et al., 2002; Hastie and Tibshirani, 1990). This highly tailored model fit, however, can make it somewhat more difficult to interpret or generalize results to other locations or time periods (Guisan et al., 2002). Both modeling frameworks discussed in this report incorporate environmental covariates, effort corrections, and observation biases into their structure for the purposes of estimating absolute abundance (as opposed to relative abundance).

Due to limitations inherent in raw data (e.g., detection bias), we generally avoided mapping raw counts, except in cases where we had insufficient data to conduct more reliable analyses (for example, with large cetaceans; Chapter 15). The ray distribution maps presented in Chapter 5 (Figure 5-12) are an example of effort-corrected data; all observations and survey effort were aggregated into 4.8x4.8 km lease blocks, so that we could compare the number of observations made per lease block area (regardless of how much surveying was actually conducted in each block). This correction did not include the incorporation of observation biases or environmental covariates, however, and resulting estimates of ray observations per unit area are presented as relative estimates of ray abundance for each lease block. Fully effort-corrected and bias-corrected predictive models, which allow for an understanding of the mechanisms driving animal distributions, are presented in several other chapters in this report (Chapters 11-12, 16, and 18).

Environmental conditions are not static, and developing the capability to predict where animals will be (both in the future, and in areas that were not surveyed) based on environmental factors is essential to understanding potential changes in future distributions and abundance (Guisan and Thuiller, 2005; Zipkin et al., 2010). Due to the inherent variability in marine systems, however, it is unclear how useful descriptions of past distributions (particularly with relatively few years of data, as with this study) will be for predicting future distributions, especially over the longer term. Predictive models involve several implicit assumptions, such as consistent species-habitat relationships across unsampled time and space (Guisan et al., 2002), and it is important to understand the limitations of any analytical approach so that results can be correctly interpreted.

### *Combining data from different sources: survey data*

Regulators and resource managers are often required to make decisions using imperfect information on wildlife resources. Wildlife data are collected in a variety of approaches and circumstances, which makes them difficult to use collectively in decision-making. As the survey data for this study were collected from both boat-based and digital video aerial platforms, there were analytical challenges involved in combining those data to develop joint products that can aid in assessing and managing wildlife resources.

Data gathered using boat and digital video aerial methods may not be directly comparable, due to differences in transect design and study area coverage, as well as detection and availability of taxa of interest. Boat survey data require distance correction, where effective strip widths vary by taxon, making it more difficult to calculate effort data; digital video aerial data have a defined strip width and are not distance-biased, but lack a defined analytical framework for incorporating other potential sources of detection and availability bias. Each method appears to be more efficient at surveying some taxa than others (Chapter 14). We also identified several different species-habitat relationships from boat survey data than from digital aerial data (Chapter 18). As a result of this variability, our approaches for combining datasets to develop the best possible distribution and abundance data varied by taxon and analytical goal. In some cases (sea turtles in Chapter 15, for example), one survey dataset alone provided the best available picture of animal distributions, and combining datasets was not effective using approaches developed to date. In other cases, we evaluated potential exposure of the marine bird community to offshore development by developing a preliminary model to integrate data from the two survey platforms, and producing a single prediction of abundance and distribution to identify ecological drivers of distribution, abundance, and local hotspots (Chapter 19). Joint modeling approaches that more formally integrate the two datasets will be published in an addendum to this final report.

Initial efforts at integrating data included the following approaches:

- Using species identifications from the boat survey to inform species proportions in the video aerial dataset (Chapter 16).
- Using effort-corrected relative abundance ratios of taxa in boat vs. video aerial surveys to weight each dataset in combined maps of persistent hotspots of relative abundance (Chapter 17).
- Comparing datasets, particularly in relation to environmental covariates, to understand when and how integration is warranted (Chapter 18).
- Developing predictions of marine bird abundance and distribution that are jointly informed by aerial surveys, which encompass a large geographic area, and boat surveys, which allow for estimation of detection probability (Chapter 19).

The results of these efforts are summarized in Chapter 2 of this report, *Synthesis of Project Findings*.

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# Chapter 2: Synthesis of project findings

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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## Abstract

This study provides baseline data on the distributions, movements, habitat use, and abundance of wildlife on the mid-Atlantic Outer Continental Shelf offshore of Delaware, Maryland, and Virginia. Despite focused studies along the Atlantic coast in recent years, the Mid-Atlantic Baseline Studies Project and Maryland Project, described here, fill a significant information gap for a large swath of the mid-Atlantic region between New Jersey and North Carolina. The mid-Atlantic study area is a complex ecosystem with highly variable temporal and geographic patterns, driven in part by the influence of the Gulf Stream to the east, and the Chesapeake Bay and Delaware Bay to the west. This area provides important habitat for a wide variety of marine wildlife over the course of the year.

In this chapter, we summarize persistent and seasonal patterns in wildlife distributions that were observed during the two years of this study, including offshore surveys, individual tracking, and methods of studying nocturnal avian migration in the offshore environment. We also present a series of case studies on specific taxa or phenomena that integrate data gained from these different methods, to examine in detail the abundance and distributions of potentially vulnerable taxa, and discuss other recent baseline studies along the eastern seaboard to provide context for this study's results. The breadth of the region is used during spring and fall migration by seabirds, landbirds, sea turtles, cetaceans, rays, and other taxa. Many of these taxa are also part-time or year-round residents of the study area, using it for foraging during the breeding season, or for foraging or roosting during non-breeding periods. Despite seasonal variation in habitat characteristics, areas near the mouths of the Chesapeake Bay and Delaware Bay remain important for many different taxa throughout the year. Boat and aerial surveys and satellite telemetry data consistently showed high species diversity, abundance, and habitat use patterns in nearshore waters adjacent to and directly south of the bay mouths (roughly within 30 km of shore). These areas are likely attractive to a wide variety of high trophic-level species, due to their consistently higher primary productivity relative to the broader study area. Areas in northern Maryland within roughly 20-30 km of shore were also consistent hotspots for biodiversity and abundance for many taxa, although this may have been partially driven by the more inshore study design implemented in the region as compared to the remainder of the study area.

Exposure to offshore development activities comprises one component of identifying risk, where risk is defined as a combination of exposure to a stressor, the hazard posed to individuals by that stressor, and the vulnerability of the population to those individual effects. Seasonal data on wildlife species composition, distributions, and relative abundance are essential for providing a baseline understanding of when and where animals have the potential to be affected by anthropogenic activities, and for identifying species or taxa of particular interest for future study. Thus, this study is an important first step towards understanding how bird, marine mammal, and sea turtle populations in the mid-Atlantic may be exposed to offshore wind energy construction and operations, as well as other anthropogenic activities. The results of this study provide insight to help address environmental permitting requirements for current and future offshore development projects, and serve as a starting point for more site-specific studies, risk analyses, and evaluation of potential measures to avoid and minimize those risks.

## Background

Marine spatial planning, a priority of both international agencies (Ehler and Douvere, 2009) and the U.S. federal government (White House Council on Environmental Quality, 2010), is designed to examine the spatial and temporal distribution of activities in the marine environment and develop effective plans for the use of marine resources based on a framework of sound science. Ultimately, by improving collaboration and coordination among all coastal and ocean users and stakeholders, Marine Spatial Planning is designed to address the demand for economic development while maintaining marine ecosystem resilience (National Ocean Council, 2013).

A number of databases have been specifically designed to compile existing marine wildlife data for the western North Atlantic for use in marine spatial planning, as well as other conservation and resource management efforts. The more prominent of these include: (1) the Ocean Biogeographic Information System Spatial Ecological Analysis of Megavertebrate Populations (OBIS-SEAMAP; Halpin et al., 2009); (2) the Northwest Atlantic Seabird Catalog, formerly known as the Avian Compendium, currently managed by the U.S. Fish and Wildlife Service (USFWS; O'Connell et al., 2009); (3) the Marine Cadastre[1], a joint initiative of the Bureau of Ocean Energy Management (BOEM) and National Oceanic and Atmospheric Administration (NOAA); and (4) the data portals of the regional ocean planning councils along the east coast (Northeast Regional Ocean Council, NROC[2], Mid-Atlantic Regional Council on the Ocean, MARCO[3], and the Governors' South Atlantic Alliance, GSAA[4]). These databases have been used to assess existing data coverage and identify geographic, temporal, and taxon-specific gaps in our knowledge of wildlife along the east coast of North America (Kot et al., 2010; O'Connell et al., 2009).

A number of recent studies have been designed to address these gaps, by collecting new survey data to identify patterns in the distribution and abundance of marine wildlife in specific areas. The broadest of these is the Atlantic Marine Assessment Program for Protected Species (AMAPPS). This joint NOAA, BOEM, USFWS, and U.S. Navy project uses traditional visual aerial and boat-based surveys to collect broad-scale data on the seasonal distribution and abundance of marine wildlife across the Atlantic Outer Continental Shelf to enhance spatial modeling exercises (Northeast Fisheries Science Center and Southeast Fisheries Science Center, 2013). Several other baseline studies have occurred at the state level. The State of New Jersey carried out a two-year (2008-2009), broad scale study – the Ocean/Wind Power Ecological Baseline Studies – to determine the distribution of wildlife species and their use of offshore waters, and potential areas for offshore wind power development (Geo-Marine Inc., 2010a). The study included the marine waters of the southern half of the state out to 37 km offshore, employing a combination of traditional visual aerial and boat-based surveys, as well as radar and acoustic techniques, to inform ecological and predictive modeling exercises. Likewise, in recent years the State of Rhode Island has developed a management plan for marine waters immediately off its coast – a roughly 3,800 km$^2$ area, including Rhode Island Sound and Block Island Sound – known as the Ocean Special Area Management Plan (OSAMP). This is a comprehensive strategy for zoning Rhode Island's offshore waters

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[1] www.marinecadastre.gov
[2] www.northeastoceancouncil.org
[3] www.midatlanticocean.org
[4] www.gsaaportal.org

using an ecosystem-based approach, and is designed to help develop policy through both scientific research and public input (Winiarski et al., 2012).

Recently, the State of Maryland[5] has been working with resource experts and user groups to compile data and information on habitats, human uses, and resources in Maryland waters. Using existing data and information, marine spatial planning tools have helped identify areas most suitable for various types of activities in order to reduce conflict among uses, facilitate compatible uses, and reduce environmental impacts to preserve crucial ecosystem services.

Despite these and other focused studies along the Atlantic coast in recent years, several geographic holes still remain in recent survey activities and data collection, which must be filled for effective marine spatial planning efforts in those areas. The Mid-Atlantic Baseline Studies Project and Maryland Project, described here, fill a significant information gap for a large swath of the mid-Atlantic region between New Jersey and North Carolina (see methods described in Chapter 1). This area includes three major wind planning areas, the federally-designated Wind Energy Areas (WEAs), for which there were limited data on the distribution and relative abundance of wildlife prior to this study. These studies provided new data for these locations, and perhaps more importantly, provided data of sufficient geographic and temporal resolution to allow for a rigorous examination of seasonal wildlife distribution patterns. The high levels of productivity in the mid-Atlantic region, and its year-round importance to a broad suite of species, mean that it is essential to understand this ecosystem in order to manage it effectively, particularly with regard to anthropogenic stressors such as offshore development.

## Patterns of wildlife distributions and habitat use in the mid-Atlantic study area

### *Seasonal patterns*

The mid-Atlantic region provides important habitat for marine wildlife over the course of the year. With each season comes a unique shift in habitat characteristics, and with it a different array of species reliant on the specific resources available (Table 2-1).

#### *Spring*

During the spring (March-May), sea surface temperatures begin to rise, and salinity across surface waters begins to decrease. As the season progresses, primary productivity begins to increase within and adjacent to the bays as nutrient rich spring runoff flows into the bays and mixes with coastal waters (Smith and Kemp 1995). Across the broader shelf within the study area, however, primary production decreases as waters begin to warm and stratify (Xu et al., 2011).

High species diversity was observed in the spring, suggesting that migratory and overwintering species dominate the region's species composition (Chapter 11). During this time, wintering seabirds departed the region to begin their migrations towards breeding grounds inland or to the north. In our study, Surf Scoters (*Melanitta perspicillata*) departed the area between January and May, Red-throated Loons (*Gavia stellata*) between March and May, and Northern Gannets (*Morus bassanus*) in between February

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[5] www.dnr.state.md.us/ccs/coastal_resources/oceanplanning

and May (Chapters 20-23). During spring, songbirds and shorebirds migrated through the region both along the coast and over open waters (Chapters 17 and 27). Summer resident seabirds, such as terns, shearwaters, and storm-petrels, arrived after migrating from wintering grounds in the south or breeding grounds in the Southern Hemisphere (Chapters 5, 8, and 17). Spring also marked the arrival of Bottlenose Dolphins (*Tursiops truncatus*) and a variety of sea turtle species, which were predicted to occur in high densities offshore of Virginia (Chapter 15).

### *Summer*

During summer (June-August), the sea surface warms to peak temperatures (generally ranging from 20-30˚C, Chapter 12), forming a strong thermocline (Castelao et al., 2010). In shallow waters close to shore, high temperatures may persist throughout the water column (Castelao et al., 2010). Average salinity values are at their lowest in summer, with lowest salinity values at the top of the water column extending across the shelf (Castelao et al., 2010). While overall primary productivity is generally low across the shelf during summer, chlorophyll concentrations increase in shallow nearshore areas where upwelling can occur (Xu et al., 2011). Additionally, primary production within the bays is at its peak, contributing to higher productivity at the bay mouths where coastal and estuarine waters mix (Smith and Kemp 1995; Flemer 1970). Through hydroacoustic surveys, we generally observed higher levels of aquatic biomass in these regions during the summer months (Chapter 9).

In the summer, seabirds were generally more associated with nearshore habitat than they are in the spring (Chapter 12). Breeding seabirds were found foraging near the shore and near the mouths of the bays (Chapter 12 and 17); specifically, terns (including Common Terns, *Sterna hirundo*, and others), were predicted to be associated with nearshore habitat (Chapters 18-19). Non-breeding species from the southern hemisphere, such as Great Shearwaters (*Puffinus gravis*) and Wilson's Storm-Petrels (*Oceanites oceanicus*), generally occupied a wider swath of the study area (Chapter 17). In early summer, large numbers of Cownose Rays (*Rhinoptera bonasus*) migrated through the study area on their way to feeding grounds in the Chesapeake Bay and Delaware Bay (Chapter 5; Blaylock 1993). Sea turtles and Bottlenose Dolphins were most abundant across the study area in the summer, with the more inshore coastal ecotype of Bottlenose Dolphins more heavily represented than the offshore population of this species (Chapter 15; Kenney, 1990). In the summer, both Bottlenose Dolphin and sea turtle distributions were influenced by sea surface temperatures and primary productivity (Chapter 15), with Bottlenose predicted to occur primarily in nearshore areas, and sea turtles still predicted to occur primarily in the southern end of the study area (Chapter 15).

### *Fall*

In the fall (September-November), stronger winds help initiate mixing of stratified water, leading to cooler and less variable sea surface temperatures across the region; temperatures continue to decrease as the season progresses and days become shorter (Schofield et al., 2008). The mixing of stratified water re-oxygenates the water column, setting the stage for a significant phytoplankton bloom that occurs across shallow waters in the region between late fall and early spring (Schofield et al., 2008; Xu et al., 2011). Decreased flow of fresh water from the Delaware Bay and Chesapeake Bay during the summer and fall causes salinity to rise over the course of the season, as saltier water is pushed closer to shore.

In the early fall, Cownose Rays moved out of the bays and aggregated in dense groups within the study area as they migrated south, likely prompted by changing water temperatures (Chapter 5; Goodman et al., 2011). Seabird species composition changed over the course of the fall, as summer residents migrated south to warmer climes and winter residents migrated into the region from breeding grounds farther north or inland (Chapter 17). Seabirds continued to be more associated with nearshore habitats as compared to winter and spring (Chapter 12). In our telemetry studies, tagged Surf Scoters migrated south from the breeding grounds and arrived in the wintering area between October and December, while Red-throated Loons arrived between November and December, and Northern Gannets between August and December (Chapters 20-23). As in the spring, songbirds and shorebirds were recorded flying over open waters as they migrated through the study area (Chapters 17 and 26-27). Peregrine Falcons (*Falco peregrinus anatum*) migrated over open water through the study area (Chapter 25), as did Eastern Red Bats (Chapter 17; Hatch et al., 2013). Alcids moved into the study region in the fall. Large schools of baitfish were observed in the study area in the fall, particularly offshore of Maryland where high density aerial surveys were conducted in nearshore regions, though they were found on the inshore transects all along the coast (Chapters 9 and 17). Although uncommon due to their small population sizes, baleen whales such as the Common Minke Whale (*Balaenoptera acutorostrata*) and Northern Right Whale (*Eubalaena glacialis*) were observed within the study area in the fall. Sea turtles remained widespread across the study area through October (Chapter 15). Bottlenose Dolphins also remained until late fall, while Common Dolphins (*Delphinus delphis*) arrived in the study area in November (Chapters 15 and 17).

### *Winter*

During winter (December-February), sea surface temperatures are at their lowest and least variable across the study area, generally ranging from 5-15˚C, with the coolest temperatures found close to shore (Schofield et al., 2008). Salinity follows a similar pattern, generally increasing with distance from shore (Castelao et al., 2010). Primary productivity peaks within shallow waters (roughly to the 40 m isobath, well past the spatial extent of our study area; Xu et al. 2011; Schofield et al. 2008).

Wintering seabirds occupied habitat throughout the study area, with variation in distribution patterns among species (Chapters 12, 17, and 19) and individuals. Northern Gannets were the most ubiquitous seabird in the study area during this period, and were often observed in the bays as well as relatively far out on the shelf in search of prey (Chapters 10, 12, 17, and 22). For Northern Gannets, we found that the chances of foraging increased with the number of sea surface temperature fronts in an area, as the temperature fronts likely aggregated prey (Chapter 24). Scoters (*Melanitta* spp.) were observed in large aggregations at the mouths of the Chesapeake Bay and Delaware Bay (Chapter 17). Common Loons (*Gavia immer*), in contrast, were most often observed individually and were widely dispersed throughout the study area, generally more associated with lower sea surface temperatures (Chapters 16-17). Many Bonaparte's Gulls (*Chroicocephalus philadelphia*) were observed in the study area on both survey platforms in winter (Chapters 5 and 8). Alcids were predicted to occur in small numbers throughout the study area (Chapter 19). Baleen whales were most commonly observed during this season; of the 51 large whales observed in this study, 31 were observed between December and

February (Chapter 17). Common Dolphins occupied habitat throughout the study area during the winter, predominantly in offshore areas (Chapters 15 and 17).

### *Persistent patterns*

Results from the weather radar study showed persistent patterns in the use of the region by nocturnal migrants, including shorebirds and songbirds, particularly during fall migration (Chapter 27). While offshore migration was most likely to occur under certain environmental conditions at different locations along the east coast, during this season there was no statistical difference in predicted levels of migratory activity in terrestrial vs. offshore locations, including locations up to 80 km offshore. In particular, the data suggested that there may be substantial offshore migration pathways that begin with “jumping off points” at certain locations along the coast (including Long Island, New York, and the Carolinas).

Primary productivity forms the base of the pelagic food chain on which nearly all species observed during this study rely. In general, primary productivity in the mid-Atlantic is higher in nearshore areas, although patterns vary seasonally (see above). Digital aerial surveys captured large numbers of schools of forage fishes in nearshore waters, with most of these recorded on the Maryland Project transects (Chapter 5), the most heavily surveyed section inshore. Bait balls were most persistently observed in high numbers in this region, in addition to nearshore regions offshore of Delaware, around the mouth of Delaware Bay (Chapter 17). In turn, despite seasonal variation in habitat characteristics, areas within about 30-40 km of shore appeared to provide important foraging habitat for a multitude of species year-round. In particular, areas near the mouths of the Chesapeake Bay and Delaware Bay consistently showed high species diversity and abundance of animals across all taxa observed in surveys during this study (Figure 2-1). Telemetry studies also highlighted these same areas around the mouths of Chesapeake Bay and Delaware Bay as high use areas for migratory seabirds in winter, even for species that were highly mobile and used a broad range of habitats, such as the Northern Gannet (Chapter 22). These areas were likely attractive to a wide variety of high trophic-level species, such as seabirds and marine mammals, due to foraging opportunities arising from consistently higher primary productivity relative to the broader study area.

**Table 2-1. Seasonal habitat use within the mid-Atlantic study area for major taxonomic groups.** There is no single definition for each season, as the life history periods of specific species vary, but generally speaking, for this table we consider that spring = Mar.-May, summer = Jun.-Aug., fall = Sep.-Nov., and winter = Dec.-Feb. Dashes indicate that we obtained no data for that taxon and time period. It should be noted that this table is not comprehensive; individuals of many seabird species, for example, migrate through the study area without taking up residence in summer or winter.

| Species Group | Spring | Summer | Fall | Winter | Report chapters with additional information |
|---|---|---|---|---|---|
| Wintering seabirds | Depart from or migrate through study area | Few individuals observed | Arrive in or migrate through study area | Abundant; utilize habitat throughout study area, though many species concentrated in the western parts of the study area and at the bay mouths | 5<br>8<br>10<br>12<br>16-24 |
| Breeding and non-breeding summer resident seabirds | Arrive in or migrate through study area | Local breeders nest on shore and forage across the study area, concentrated near bay mouths; non-breeders are more ubiquitous across the study area | Depart from or migrate through study area | Few individuals observed | 5<br>8<br>12<br>17-19 |
| Songbirds and other landbirds | Migrate through study area | Small flocks of swallows (Hirundinidae) and individuals of other species observed across study area | Migrate through study area | Few individuals observed | 8<br>17<br>26-27 |
| Shorebirds | Migrate through study area. | Generally not present; few individuals observed throughout study area | Migrate through study area | Few individuals observed | 8<br>17<br>26-27 |
| Bats | -- | -- | Migrate through study area | -- | 17 |
| Baleen whales | Migrate through study area | -- | Migrate through study area | Observed throughout study area | 5<br>8<br>15<br>17 |
| Toothed whales (dolphins and porpoises) | Bottlenose Dolphins arrive in or migrate through study area; Common Dolphins depart from or migrate through study area | Season of highest overall abundance; Bottlenose Dolphin most commonly observed | Present across study area; Bottlenose Dolphin commonly observed; Common Dolphin arriving in or migrating through study area | Season of lowest overall abundance; Common Dolphin observed across study area | 5<br>8<br>15<br>17 |
| Turtles | Arrive in or migrate through study area; observed across study area, most densely in the southeast | Commonly observed across entire study area; higher densities offshore and in the southern part of the study area | All species distributed across study area as they migrate south to wintering or nesting grounds; higher densities offshore | -- | 5<br>14-15<br>17 |
| Rays | Few individuals observed | Present in large numbers and broadly distributed across study area | Present in large numbers and dense aggregations during migration | Few individuals observed | 5<br>14<br>17 |
| Forage Fishes | Moderately abundant; occur throughout study area | Abundant; occur throughout study area; generally more dense closer to shore | Abundant; higher densities close to shore | Few groups visually observed, but high acoustic detection; highest densities near the mouth of Chesapeake Bay | 9<br>14<br>17 |

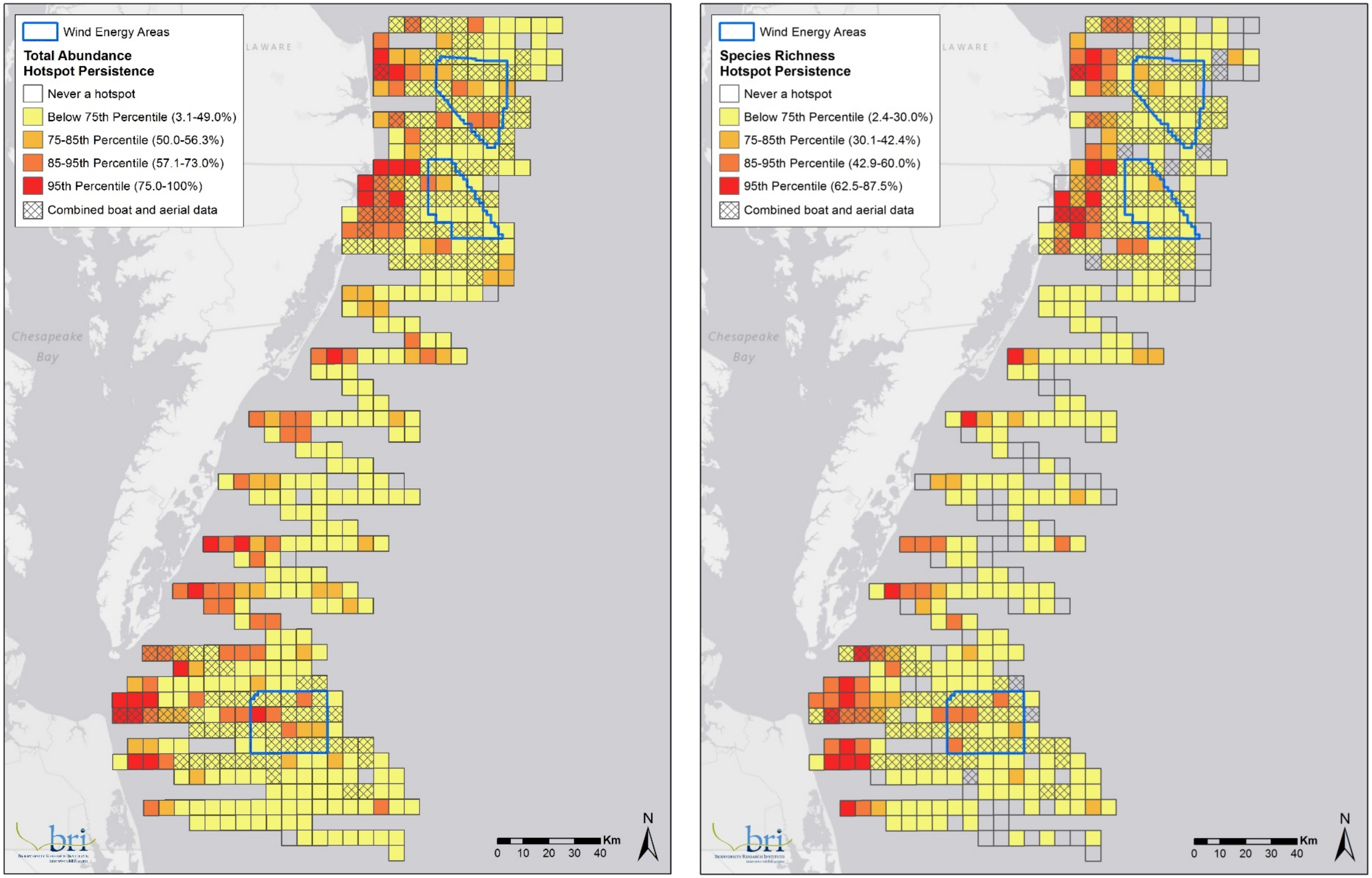

**Figure 2-1. Classified persistent abundance hotspots identified across all taxa (left) and classified persistent species richness hotspots (right).** These maps highlight areas where the greatest numbers of individuals across all taxa (left) and the greatest numbers of species (right) were consistently observed over the course of the study (Chapter 17). For each percentile category shown in the legends, the corresponding percentage of time a cell was a hotspot is shown parenthetically. Crosshatched cells were surveyed by and integrate data from both boat and aerial survey methods and integrate data from both approaches.

### *Interannual variation*

The marine ecosystem is a dynamic environment, with annual changes that can influence the distributions of wildlife (Gaston et al., 2009; Schneider and Heinemann, 1996). Interannual variation is driven primarily by changes in abiotic variables, such as sea surface temperature and currents (Ballance et al., 2006). Temperature and salinity in the mid-Atlantic have changed over the past several decades (Mountain, 2003), and there have been declines in primary productivity with an increase in winter storms (Schofield et al., 2008). The Bureau of Ocean Energy Management (BOEM) suggests a minimum of two full annual cycles for offshore surveys for wind energy development (BOEM, 2013), based on a recent analysis of interannual variation in wildlife distributions that indicates that 2-3 years of surveys may be sufficient to capture shorter-term (e.g., intra-decadal) levels of variation for some taxa (Kinlan et al., 2012b).

Between the two years of data collected in this study, we found substantial variation in the community composition, distribution, and abundance of species observed (Chapters 12, 14 and 18), as well as notable differences in environmental conditions. For example, we observed warmer waters in the second year of the study, possibly due to eddies from the Gulf Stream (warm core rings that meander north off of the main Gulf Stream over the Atlantic Outer Continental Shelf; Chapter 12). Although digital video aerial surveys for this study were conducted in June and September of 2012 and July and September of 2013, large numbers of Cownose Rays were only observed in 2013. Some variation in water temperatures, ray populations, or other factors meant that very few rays were seen in 2012 (Chapter 5). Similarly, scoters were observed in high numbers each winter on the boat survey, but more than twice as many scoters were seen in January of 2013 as in January of 2014 (Chapter 8). Seabirds are generally patchily distributed in their environment (Fauchald, 2009), leading to some level of variation in observations between survey platforms and year. Scoters, however, also responded to their environment differently between the two years, perhaps due to the increase in water temperatures in 2013 (Chapter 12), or dynamic movements in response to prey. Many other seabirds also responded differently to environmental conditions in the first year vs. the second year of surveys (Chapters 12 and 18). Particularly for rarer and more patchily distributed species, more than two years of data may be required to describe the interannual variability in their distribution patterns, and conducting surveys over a longer time frame would allow for a more complete characterization of the expected levels of variability in these patterns.

## Determining and interpreting risk

The seasonal baseline data on wildlife species composition, distributions, and relative abundance provided by this study are essential for understanding when and where animals have the potential to be affected by anthropogenic activities in the mid-Atlantic region. In the sections above, we have discussed the potential exposure of animals to offshore wind development in different seasons. Exposure itself, however, does not necessarily indicate that animals will suffer deleterious effects; the vulnerability of different species to development activities will also play a role. Risk to wildlife from offshore development can be thought of as an interaction of three factors (Crichton, 1999; Fox et al., 2006):

- *Exposure* of individuals to development and operation activities that have the potential to cause impacts. Species may be exposed if they are present in a potential development area during the times at which impact-producing activities occur. Specific behavioral traits may increase or decrease exposure of animals that are present.

- *Hazards* posed to individuals that are exposed. Hazards can be direct (for example, collision mortality) or indirect (displacement, or effects on habitat or prey populations).
- *Vulnerability* of populations to individual-level effects, or the potential for impacts to individuals to substantially affect the status of the population. This potential is related to a species' life history as well as its conservation status.

Published risk assessments for birds and offshore wind energy development have generally considered some combination of these factors (e.g., Desholm, 2009; Furness et al., 2013; Garthe and Hüppop, 2004; Willmott et al., 2013). For aquatic animals, risk assessments have focused primarily on acoustic disturbance (with potential for mortality/sublethal impacts as well as displacement) and habitat impacts (Bailey et al., 2014; Bergström et al., 2014). It is still unclear in most cases, however, what life history characteristics most influence risk, or how to translate some types of risk (such as displacement) to a biologically meaningful metric (e.g., reproductive or survival impacts). Nevertheless, site-specific pre- and post-construction monitoring will most likely be focused on particular species or topics, in order to prioritize limited funding and direct research towards taxa most likely to be affected (Rein et al., 2013). Assessments of relative risk, while imperfect, will be essential for directing efforts towards the taxa of greatest need.

In this baseline study of wildlife distributions and movements, we focused on developing a better understanding of exposure of wildlife to future offshore development in the mid-Atlantic. This study is a crucial first step towards understanding the implications of offshore wind energy development for bird, marine mammal, and sea turtle populations in the mid-Atlantic US. Future research to fill data gaps on hazards and vulnerability can be targeted towards habitat that supports high or low species abundance and diversity, as well as towards species with high levels of exposure, or species most likely to be impacted due to their behaviors, life history, or conservation status.

## Case studies: integrating results from different project components

Here, using results from multiple study efforts, we examine certain taxa and phenomena in more depth. Taxa were chosen for inclusion because they are of likely regulatory concern due to their conservation status in the U.S., or because they are known or suspected to interact with offshore wind energy development, based on the European experience to date. As discussed above, there are several types of potential effects of offshore wind energy development on wildlife, including direct mortality or injury, behavioral effects, and indirect effects to habitat or prey populations. We reference the European literature where appropriate, and briefly discuss the most likely potential effects to each taxon based on the distribution data presented in this study. Migration is also included since it is a critical stage in the life cycle for many animals, a period when they are more mobile and physiologically stressed, making them potentially more vulnerable to additional threats.

### *Red-throated Loon*

Loons are long-lived species with high adult survival and low annual productivity (Barr et al., 2000; Schmutz, 2014). Therefore, the loss of adult individuals or the chronic reduction of individual fitness has the potential to adversely affect populations. Fisheries are a major source of adult mortality, via bycatch of birds in nets (Barr et al., 2000). The Red-throated Loon has a global conservation status of Least Concern due to the species' broad global range and large population size, despite a population trend indicating a decline (BirdLife International, 2015). In the U.S., however, the US Fish and Wildlife Service has identified the Red-throated Loon as the highest priority open-water species for conservation in the mid-Atlantic US (USFWS 2008), where they are abundant during non-breeding periods (Chapters 5, 8, and 12).

In Europe, Red-throated Loons have exhibited long-term and possibly permanent displacement from offshore wind energy development areas, making effective habitat loss the primary concern for this species in relation to offshore development (Leonhard et al., 2013; Lindeboom et al., 2011; Percival, 2010). Thus, the Red-throated Loon has been ranked as the most vulnerable species to displacement in European studies (Furness et al., 2013; Garthe and Hüppop, 2004) and is considered to be at high risk of adverse effects from offshore wind energy development (Langston, 2010). BOEM and the USFWS have recognized the need for additional data on populations and movements of this species in the mid-Atlantic in relation to future offshore wind energy development, and have funded ongoing (2012-2016) satellite telemetry studies (of which the telemetry study in this report is a part; Chapters 21 and 23).

During boat and aerial surveys, 1,770 Red-throated Loons were observed (1% of all wildlife observations from surveys) and they were most common in the study area between November and May (Chapters 5 and 8). In many cases, however, Red-throated Loons and Common Loons could not be distinguished in video aerial surveys, due to a greater overlap in body size among North American loon populations than occurs in Europe. Red-throated Loons were most consistently observed within approximately 20 km of shore during surveys, unlike Common Loons, which were more widely distributed across the study area in winter (Chapters 16-17). Telemetry data indicated that Red-throated Loons preferentially used shallow nearshore waters with flat sandy substrates while wintering in the mid-Atlantic region, particularly around the mouth of Chesapeake Bay and south along the coast of Virginia, close to original capture locations (Chapter 21). Modeled boat survey data also indicated that proximity to shore was the strongest predictor of Red-throated Loon abundance, followed by relatively cold sea surface temperatures and primary productivity (though the predicted relationship with primary productivity varied by season, with loons associated with areas of lower productivity in spring and high productivity in winter). In the digital aerial survey video, 28% of flying loons (all species) were flying between 20 m and 200 m in altitude; the rotor-swept zone of offshore wind turbines depends on the turbine size and type, but will likely include altitudes within this range (Chapter 5; Willmott et al. 2013). Seventy percent of flying loons were estimated to be below this range (Chapter 5).

In the spring, satellite tagged Red-throated Loons left the study area between late March and early May, and largely followed the coast north to breeding grounds. Greatest offshore movements occurred during this departure from the study area. During fall migration, most individuals stopped over in Hudson Bay, and then moved either to the Gulf of St. Lawrence or to the Great Lakes before arriving in the study area between mid-November and late December.

*Context*

- European studies indicate that Red-throated Loons experience long-term, localized disturbance and displacement from wind energy facilities, as well as related activities such as vessel traffic.

*Take home messages*

- The greatest overlap between Red-throated Loon distributions and mid-Atlantic WEAs occurred during migration periods, when movements tended to be located farther offshore.
- In winter, Red-throated Loons were most commonly located west of the WEAs.

### *Northern Gannets*

The Northern Gannet is the largest seabird to breed in the North Atlantic Ocean. In the Western Hemisphere, they breed at six colonies in southeastern Canada—three in the Gulf of St. Lawrence, Québec, and three off the eastern and southern coasts of Newfoundland (Mowbray, 2002; Nelson, 1978). On migration, Northern Gannets move widely down the east coast of North America to winter in the shelf waters of the mid-Atlantic region, the South Atlantic Bight, and the northern Gulf of Mexico (Fifield et al., 2014; Nelson, 1978), and they were one of the most commonly observed species in surveys for this study (Chapters 5, 8, and 12). The Northern Gannet has a global Conservation Status of Least Concern due to its relatively large population size and its exceptionally large range (BirdLife International 2015). The North American breeding population, which represents 27 percent of the global population, has experienced a healthy rate of growth since 1984 (4.4% per year), although that appears to have slowed in recent years (Chardine et al., 2013). The species is vulnerable to mortality from oil spills and fisheries bycatch, however, and the Northern Gannet has been identified as a possible species at risk of collision mortality from offshore wind energy development, due to its relatively poor in-air maneuverability and foraging behaviors (which include spending a large proportion of time soaring at or near an altitude that potentially places it within the rotor-sweep zone of offshore turbines; S. Garthe, Benvenuti, and Montevecchi 2000; Langston 2010). Several recent vulnerability assessments have estimated Northern Gannets to be one of the seabirds most vulnerable to collision mortality (Furness et al., 2013; Willmott et al., 2013). There is also evidence of displacement of Northern Gannets from offshore wind facilities in Europe, however (Lindeboom et al., 2011; Vanermen et al., 2015), and a further examination of Northern Gannet responses to offshore wind facilities may improve our understanding of the scope of likely hazards for this species.

In the U.S., the USFWS has identified the Northern Gannet as a high priority species for Bird Conservation Region (BCR) 30, which includes most of the mid-Atlantic study area, and has also specifically identified the importance of understanding their movements and distributions in relation to future offshore wind energy development (Atlantic Coast Joint Venture 2008); as a result, BOEM and the USFWS have funded ongoing satellite telemetry studies of the species in the mid-Atlantic (of which the telemetry study in this report is a part; Chapters 22-23).

Northern Gannet migration was highly asynchronous and widely dispersed across the continental shelf. During the boat and aerial surveys in this study, 21,345 Northern Gannets were observed (17% of all wildlife observations), most commonly between October and April (Chapters 5 and 8). Individual Northern Gannets roamed widely across the region in winter; satellite data showed that they could range up to 50 km out onto the Outer Continental Shelf (Chapter 22), and 70% of the study area was categorized as a hotspot of gannet abundance in at least one survey (Chapter 17). The general locations used by wintering Northern Gannets seemed to be somewhat consistent, however, as during surveys they were most often observed in large numbers in nearshore waters along the length of the study area (Chapter 17). Combined, telemetry and survey data showed that Northern Gannets in the mid-Atlantic generally used habitats characterized by highly productive, shallower waters, with lower sea surface salinities, especially areas closer to shore and over fine sandy substrate. Their behavioral patterns indicated that they foraged roughly 67% of the time during winter, in relatively deeper waters, and in areas with high densities of sea surface temperature fronts (e.g., boundary areas between water masses of different temperatures). The rotor-swept zone of offshore wind turbines depends on the turbine size and type, but may include altitudes between 20 m and 200 m (Willmott et al., 2013). In the digital aerial survey video, 55% of flying gannets were below this range, with 43% between 20 m and 200 m (Chapter 5).

*Context*

- European studies indicate a range of possible effects of offshore wind development on Northern Gannets, including collision mortality and displacement.

*Take home messages*

- The broad-scale distribution movements of Northern Gannets during winter may increase the likelihood that individuals would be in the vicinity of offshore wind developments repeatedly throughout the season.
- Important foraging and habitat use areas appear to be defined by a wide variety of habitat characteristics. Construction and operations of offshore wind energy facilities, including associated vessel traffic, could potentially cause localized displacement anywhere in the study area, but this is most likely within about 30-40 km of shore where Northern Gannets were most abundant.

### *Scoters*

Scoters are medium-sized sea ducks that breed near lakes or slow-moving rivers on the Arctic tundra from Labrador to Alaska. The Surf Scoter and White-winged Scoter (*Melanitta fusca*) both have a global Conservation Status of Least Concern, due to their large population sizes and broad ranges, despite the fact that the population trends for both species indicate a decline (BirdLife International 2015). The Black Scoter (*M. americana*) is listed as Near Threatened due to suspected recent population declines (BirdLife International 2015). Threats include hunting, particularly along the east coast of North America, as well as possible habitat degradation and increased harvest of mussels for human consumption (Bordage and Savard, 2011; Savard et al., 1998). All three species use the mid-Atlantic study area in large numbers during their nonbreeding period (Chapters 5 and 8), and they are listed in several state wildlife action plans in the region (Atlantic Coast Joint Venture 2008). The USFWS has identified them as high priority species, and specifically identified the importance of understanding their movements and distributions in relation to future offshore wind energy development (Atlantic Coast Joint Venture 2008). Common Scoters (*M. nigra*) in Europe have been displaced from feeding or roosting grounds for several kilometers surrounding offshore wind energy development, resulting in short-term effective habitat loss

(Langston 2013; Leonhard et al. 2013). The species returned to a facility footprint at a project in Denmark three years after construction, although whether this was a result of habituation or changes in prey distributions, or both, remains unclear (Petersen and Fox, 2007). Vessel traffic is also known to disturb scoters, though the degree of this disturbance varies by species (Chapter 13; Schwemmer et al. 2014).

Scoters were the most abundant avian genus observed over the course of the study, with 43,339 individuals observed (25% of all wildlife observations) and were most abundant in the mid-Atlantic between October and May (Chapters 5, 8 and 17). The majority of scoter observations were not identified to species, but observations included at least 30% Black Scoters, 9% Surf Scoters, and 0.001% White-winged Scoters. Satellite tagged Surf Scoters spent an average of 133 days in the region during winter, generally arriving in the study area between mid-October and mid-December. They departed the study area between early January and mid-May, and followed the coastline north to breeding and molting areas in northern Canada. This route was reversed during fall migration as birds returned to wintering areas in or near the mid-Atlantic. In the digital aerial survey video, 77% of flying scoters (all species) were flying below 20 m in altitude; 19% were between 20 m and 200 m.

Satellite tagged Surf Scoters spent >50% of their time in the study area within or at the mouths of the bays (Chapter 20). Core use areas of Surf Scoters identified by satellite telemetry may have been heavily influenced by capture locations, but survey and telemetry data both showed that scoters use habitat characterized by shallow nearshore waters with high primary productivity (Chapters 12, 17, and 20). Large aggregations of scoters were most consistently observed during surveys at the mouth of Chesapeake Bay and just south of the mouth of Delaware Bay, within roughly 20 km of shore (Chapter 17). In the mid-Atlantic, scoter distributions appear to be mainly located closer to shore than most proposed offshore wind energy development (Chapters 12, 17, and 20). They could experience considerable disturbance from development activities in nearshore areas, however, as well as vessel activity related to projects located in WEAs or other offshore areas (particularly if vessel activity occurred near the mouths of Chesapeake Bay and Delaware Bay).

*Context*

- Based on European studies, scoters may be displaced from areas around offshore wind facilities for some period of years following construction.

*Take home messages*

- Telemetry and survey data for scoters indicated strong nearshore distribution patterns, which held true across species and were largely driven by water depth and food resources.
- In the mid-Atlantic, construction and operation of offshore wind energy facilities (and associated vessel traffic) are most likely to cause localized displacement of scoters from high-quality feeding areas if these activities occur within about 20 km from shore.

### Endangered birds

Three federally endangered bird species could interact with offshore wind energy facilities in the mid-Atlantic, based on their respective ranges: the Piping Plover (*Charadrius melodus*), Roseate Tern (*Sterna dougallii*), and the American subspecies of the Red Knot (*Calidris canutus rufa*). Due to their conservation status and protection under the Endangered Species Act, all three species are likely to be priorities for regulators during the offshore wind permitting process in the mid-Atlantic, as indeed has been the case for the Cape Wind project off the coast of Massachusetts (Normandeau Associates Inc., 2011). The primary hazard posed to terns and shorebirds from offshore wind energy development would appear to be collision mortality (Everaert and Stienen, 2007; Furness et al., 2013; Willmott et al., 2013), although impacts of construction activities on terns' prey base have also been noted at one wind facility in the UK (Perrow et al., 2011). Except in the case of a wind facility constructed on a jetty directly adjacent to a tern colony in Belgium (e.g., Everaert and Stienen 2007), however, limited evidence exists for mortalities. Development of wind facilities in locations between tern colonies and major offshore foraging grounds could pose a potential hazard, as adults would have to navigate past turbines multiple times daily (Henderson et al., 1996), and there may also be some limited exposure of Red Knots during migration; however, for wind energy facilities located farther offshore, there is likely to be limited or no interactions with Piping Plovers, which are thought to mainly migrate along the coast (Burger et al., 2011). We can provide little evidence of exposure in this study; three Roseate Terns were observed during boat surveys off of Delaware and Maryland (all observed in May or June, within about 20 m of shore), but no other confirmed observations of these species were made, likely due in part to these species' rarity. It should be noted that species identification rates for terns and shorebirds were relatively poor in the digital aerial surveys, so it is possible that additional individuals of these listed species were observed and were not able to be identified.

In addition to federally endangered bird species, state-listed species in Delaware, Maryland, and Virginia include Least Tern (*Sterna antillarum*), Peregrine Falcon, Common Tern, Forster's Tern (*S. forsteri*), Royal Tern (*Thalasseus maximus*), and Wilson's Plover (*Charadrius wilsonia*). Each of the states also lists additional bird species in various ways (as state threatened, included in state Wildlife Action Plans, in various conservation "tiers," etc.[6]). Due to their conservation status in the region, these species are also likely to be higher priority for regulators considering proposed development in the mid-Atlantic. With the exception of Common Terns, all of these species were rare, if they were seen at all, on the boat and digital aerial surveys (Chapters 5 and 8). However, telemetry data for Peregrine Falcons indicates considerable use of offshore areas during fall migration (Chapter 25; see 'Migration' section below).

*Take home messages*

- Several state- and federally-listed bird species were observed during offshore surveys, including Roseate Terns, Least Terns, Common Terns, Forster's Terns, and Royal Terns, all of which were observed most commonly in the spring, summer, and fall within roughly 20 km of shore.
- Telemetry data indicate that a large number of Peregrine Falcons may also use the mid-Atlantic study area during fall migration along the Atlantic Flyway.
- We had no confirmed sightings of Piping Plovers or Red Knots in the study area.

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[6] http://www.dgif.virginia.gov/wildlife/virginiatescspecies.pdf, http://www.dnrec.delaware.gov/fw/dwap/Documents/B%20and%20C%20-%20SGCN%20and%20Species-Habitat%20Associations.pdf, http://www.dnr.state.md.us/wildlife/Plants_Wildlife/rte/pdfs/rte_Animal_List.pdf

### ***Sea Turtles***

Sea turtles are long-lived animals with a world-wide oceanic distribution. Five species occur in our study area: the Loggerhead Sea Turtle (*Caretta caretta*), Leatherback Sea Turtle (*Dermochelys coriacea*), Kemp's Ridley Sea Turtle (Lepidochelys kempii), Hawksbill Sea Turtle (*Eretmochelys imbricata*), and Green Sea Turtle (*Chelonia mydas*). All are listed as threatened or endangered under the Endangered Species Act. As such, they are likely to be priority species for regulators during the environmental permitting process for offshore wind energy development. The mid-Atlantic region has large populations of a high diversity of turtles, but existing threats that could cause population declines (Wallace et al., 2011) include mortality from bycatch in fishing nets (Murray and Orphanides, 2013), collisions with vessels, especially those traveling at high speeds (Hazel et al., 2007), loss of nesting habitat to coastal development, and disturbance or destruction of nests by humans or other animals (Wallace et al., 2011).

Sea turtles are uncommon in European waters, so no information is available about their interactions with offshore wind facilities. Construction of offshore wind facilities has been identified as the period with the most potential risks for sea turtles, due to noise from pile driving and other activities, though the potential for injury or behavioral impacts remains largely unknown (Chapter 15; Michel, 2013). Green Turtles and Kemp's Ridley Turtles (Bartol and Ketten, 2006), Loggerhead Turtles (Martin et al., 2012), and Leatherback Turtles (Dow Piniak et al., 2012) all hear a relatively narrow range of low frequencies, with a maximum sensitivity in the range of ~100-500 Hz, which overlaps with the sounds produced by many human activities, including seismic studies, drilling, low-frequency sonar, shipping, pile driving, and operating wind turbines.

There were 1,862 sea turtles observed in total in boat and aerial surveys (1.5% of all wildlife observations). Digital video aerial surveys proved to be more effective than boat surveys at surveying sea turtle populations (Chapters 14-15; see also Normandeau Associates Inc. 2013), likely in large part because turtles could be detected even when they were fully submerged. Sea turtles were most abundant from May to October, with very few individuals present in the study area in winter (Chapters 15 and 17). Models predicted highest turtle densities in areas far from shore off of Virginia in spring, in areas with warmer sea surface temperatures; in summer, sea turtles were predicted to be distributed across a broader range, as females moved to shore to lay eggs on sandy beaches. Sea turtles were most widely distributed across the study area in fall, predominantly in offshore areas. In addition to water temperature, primary productivity and distance from shore were important influences on sea turtle densities (Chapter 15). There was substantial overlap between sea turtle distributions and areas of planned offshore wind energy development, particularly in the southern parts of the study area.

#### *Context*

- The effects of offshore wind development on sea turtles remain poorly understood, most notably in relation to noise and the potential for collisions with vessels.

#### *Take home messages*

- There may be species-specific differences in habitat use or movements that were not distinguishable in this study.

- Digital aerial surveys seem to have higher detection rates of sea turtles than other survey approaches, but application of newer technologies with improved species differentiation is needed.
- Construction of offshore wind energy facilities in mid-Atlantic WEAs is likely to occur in warmer months and sea turtles will be present during these periods.

### *Cetaceans*

All cetaceans are protected under the Marine Mammal Protection Act, and most are also protected under the Endangered Species Act. The conservation status of marine mammals, and particularly baleen whale populations, has the potential to make them a priority regardless of their exposure or the risk of individual hazards. Acoustic disturbance from a variety of human activities is viewed as a high potential risk for all marine mammals (Bergström et al., 2014), and is known to increase physiological stress (Rolland et al., 2012), disrupt communications (Dilorio and Clark, 2010; Parks et al., 2007), cause significant avoidance behavior (Tougaard et al., 2009), and is associated with mass strandings (Frantzis, 1998). European studies have indicated that Harbor Porpoises (*Phocoena phocoena*) can hear pile driving noise from offshore wind construction over 80 km from the source, and the species showed displacement up to 20 km away during construction (Thomsen et al. 2006; Teilmann and Carstensen 2012). Results of operational displacement studies in Denmark and the Netherlands have varied (Scheidat et al. 2011; Teilmann and Carstensen 2012). There has been little or no detectable avoidance during operations at some facilities, while in at least one instance, porpoise acoustic activity levels were at only 29% of pre-construction levels nine years after construction had been completed (Teilmann and Carstensen 2012). Prey availability may be an important factor affecting porpoise behavior around operational wind facilities (Teilmann and Carstensen 2012), but more information is needed. Data on disturbance to large whales by other types of anthropogenic activities have also been examined (e.g., Mccauley et al. 2000; Tyack et al. 2011), but large whales are not common in European waters where development has occurred, so no information is available about their interactions with offshore wind facilities.

We observed 3,289 marine mammals in boat and aerial surveys. The majority (99%) were dolphins and porpoises, from at least five species. Bottlenose Dolphins were the most abundant delphinid in surveys, and were observed primarily in spring, summer, and fall (Chapters 15 and 17). Cold-tolerant Common Dolphins were most frequently observed in offshore areas in winter and early spring (Chapters 15 and 17). Distance from shore, primary productivity, and sea surface temperature were important predictors of Bottlenose Dolphin distributions. This is possibly because of their use of areas of high productivity for feeding, particularly in and around the mouths of the Chesapeake Bay and Delaware Bay, and their temperature-related migratory behaviors. Many of the Bottlenose Dolphins observed in this study may have been residents from coastal stocks, leading to the nearshore distribution patterns we observed. A more robust density gradient from west to east was observed in summer, possibly due to an influx of transient populations during the warmer period.

Migratory routes for many large whale species are poorly defined, though several are known to migrate through the mid-Atlantic between their wintering and breeding grounds (Firestone et al., 2008). North Atlantic Right Whales (*Eubalaena glacialis*), the most critically endangered of these species along the east coast of North America, have already spurred the development of additional mitigation measures to minimize the potential for adverse effects from offshore wind energy development in the mid-Atlantic[7]. We can provide limited information about potential exposure from this study, though our observations may be useful in combination with data from other studies. A total of 51 observations of large cetaceans were made between boat and digital aerial surveys, with 31 of the observations occurring in winter. We observed a total of nine North Atlantic Right Whales, all of which were observed in February and March, which is an important contribution to our knowledge for this species given their small population size and our lack of data on their movements and habitat use in the mid-Atlantic. We also observed endangered Humpback Whales and Fin Whales, as well as several other whale species.

*Context*

- Offshore wind energy facilities present significant increases in underwater noise during construction, which may affect all marine mammals. Our current lack of understanding of the hazards posed to baleen whales by offshore wind energy development make these species a particular concern for regulators in the U.S.

*Take home messages*

- Relatively little is known about migratory routes for many rare whale species in the mid-Atlantic, although data from this and other studies are beginning to fill this gap.
- Bottlenose Dolphins may be most likely to be exposed to development activities during summer and in the northern end of the study area, as well as in western areas of the mid-Atlantic WEAs in spring and fall. Common Dolphins have a more offshore distribution and may be particularly and spring.

### ***Migration***

Migration is a difficult phenomenon to study, particularly in offshore areas, but our study captured a number of migratory events through the mid-Atlantic study area. Over the course of this project, we employed several methods that documented the timing and routes of animal migrations through the mid-Atlantic region, including our analysis of weather radar (NEXRAD) data, the use of avian passive acoustic recorders, satellite telemetry, and boat and aerial surveys. If we are to fully understand the potential effects of offshore activities on wildlife populations, we need to determine when and where migration occurs, and what migratory species are likely to be exposed to offshore wind energy development in the region.

*Rays*

The Cownose Ray is a species of eagle ray that primarily eats mollusks and shellfish, and has a global conservation status of Near Threatened due to overfishing in regions of Central and South America (Barker, 2006), though it is not listed in the U.S. Many elasmobranchs can detect electromagnetic fields (EMF), which are produced by offshore wind power transmission cables (Gill et al., 2009; Normandeau Associates Inc. et al., 2011). Cownose Rays use electroreception to detect their prey, however their ability to detect and tendency to react to EMFs from sub-sea cables have not yet been determined (Boehlert and Gill, 2010; Smith and Merriner, 1985). In large groups, called “fevers”, these rays migrate

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[7] http://docs.nrdc.org/oceans/files/oce_12121101a.pdf

north and into inland bays, such as Chesapeake, to breed during the summer (Goodman et al., 2011). While their breeding habits are reasonably well known, the migratory period is poorly understood. However, digital video aerial surveys recorded immense migratory schools near the water's surface in the mid-Atlantic up to 75 km from shore; we observed almost 48,000 rays in the summer and fall (Chapter 5). The unexpected detection of these massive migrations is a reminder of how little we truly know about the migratory lives of many ocean creatures.

### *Bats*

Bat fatalities have been regularly documented at terrestrial wind facilities in the U.S. (Arnett et al., 2008; Kunz et al., 2007), particularly for migratory tree-roosting species such as Eastern Red Bats (*Lasiurus borealis*). Bats are not commonly thought of as migrating offshore, but there is substantial anecdotal evidence for offshore movements in this taxon, particularly among migratory tree bats (Hatch et al., 2013), and fatalities have been documented at offshore wind facilities in Europe (European Environmental Agency, 2009). Seventeen Eastern Red Bats, were detected up to 70 km from shore in this study during both boat and aerial surveys (Chapter 17; Hatch et al., 2013). Of these, fourteen were seen on a single day of aerial surveys. They were observed flying during the day, and those that had estimable flight heights were estimated to be flying higher than 200 m above sea level (Chapter 17; Hatch et al., 2013), both unexpected behaviors for this taxon in the offshore environment.

### *Songbirds*

The movements of individual songbirds can be difficult to track because of their small body size. They also migrate at night, making the study of their migrations particularly difficult. Weather radar can detect migratory activity in the atmosphere, which allowed us to document broad-scale geographic and temporal patterns of nocturnal migrants in the offshore environment (Chapter 27). Nocturnal acoustic sensors deployed on the survey boat also allowed us to identify some of the species making these flights (Chapter 26). In this study, nocturnal migrants, including songbirds and shorebirds, regularly flew over open water, and this was particularly true in the fall, when offshore migratory activity was often higher than over land. For many songbirds, expansive areas of open water on the Outer Continental Shelf may not be the barrier to movement that we previously thought which increases the concern for effects of offshore development on these species during critical migration periods.

### *Falcons*

The Peregrine Falcon is the world's fastest animal, and their aerial dexterity allows them to catch small birds on the wing. This ability, coupled with physical stamina, allows them to migrate over large expanses of the Atlantic Ocean. Our satellite telemetry data indicated that though Peregrine Falcons often migrated relatively close to shore, individuals were capable of flying hundreds of kilometers offshore (Chapter 25) and staying in those areas for weeks. They are able to fly for several consecutive days over open water, soar and forage at night, and often roost on offshore structures and vessels (Cochran, 1975; Desorbo et al., 2012; Johnson et al., 2011; Voous, 1961). During migration, Peregrine Falcons primarily prey on other migrating birds, like songbirds and shorebirds (White et al., 2002). It is possible that falcon migratory routes in offshore areas are dictated by the migratory paths of their prey.

*Context*

- The consequences of interactions between migratory wildlife and offshore wind facilities are unclear. Some species may have increased collision risk. Others may have increased energetic expenditures from avoidance during migratory movements, although these effects will depend on the scale and number of offshore wind facilities along a migration route.

*Take home messages*

- Our research suggests that a wide variety of animals migrate through areas that have been proposed for offshore wind energy development in the mid-Atlantic region. Additional research on migrant populations may be warranted for sites proposed for development or other offshore activities.

## Discussion

This study provides a unique baseline dataset on the distributions, movements, habitat use, and relative abundance of wildlife on the mid-Atlantic Outer Continental Shelf, between about 5 and 85 km from shore. The mid-Atlantic study area is a complex ecosystem with highly variable temporal and geographic patterns, driven in part by the influence of the Gulf Stream to the east, and the Chesapeake Bay and Delaware Bay to the west. This study's boat and digital aerial surveys, individual tracking studies, and nocturnal avian migration studies provide the most comprehensive view to date of offshore wildlife populations in this region. The complexity of resulting datasets, as well as the differing and often complementary information provided by different study methodologies, have necessitated the development of a suite of analytical approaches for comparing and integrating data for use in decision making.

These varied approaches have led to several key conclusions for the mid-Atlantic study region, including:

- Boat-based surveys and digital aerial surveys each have specific advantages and disadvantages, but are largely complementary. Digital aerial surveys are particularly useful for covering offshore areas at broad scales, where general distributions of taxonomic groups are a priority; boat surveys can provide more detailed data on species identities and behaviors, but are more limited in geographic scope due to their slower survey pace (Chapters 1, 6, 13 and 19).
- Habitat gradients/fronts located in nearshore waters (near the mouths of Chesapeake Bay and Delaware Bay) are important influences on productivity and patterns of species distributions and abundance. Areas offshore of the mouths of these bays, as well as to the south of Delaware Bay along the coast, were consistent hotspots for relative abundance of many taxa, regardless of survey methodology or analytical approach.
- There is considerable variation in species composition and spatial patterns by season. As well as being a focus for wintering and breeding seabirds, the location of the study area (the central sector of the eastern seaboard) makes it a key migratory corridor. Dynamic environmental conditions also contribute to wide variation in community composition and seasonal patterns of wildlife in the region.

### ***Regional context***

Several assessments of wildlife distributions along the Atlantic coast of the United States have contributed to ecosystem-based marine spatial planning efforts in recent years, and provide context for our findings in the mid-Atlantic. In particular, baseline studies offshore of New Jersey in 2008-2009 (Geo-Marine Inc., 2010a, 2010b) and Rhode Island in 2009-2012 (Paton et al., 2010; Winiarski et al., 2012) have provided comparable datasets to the contribution that we make in this study for areas offshore of Delaware, Maryland and Virginia. Additional efforts are currently ongoing for cetaceans offshore of Maryland (S. Barco, pers. comm.) and along the entire eastern seaboard (Northeast Fisheries Science Center and Southeast Fisheries Science Center, 2013).

Assessments of historical data have also occurred in recent years; the Northwest Atlantic Seabird Catalog (formerly known as the Compendium of Avian Information) includes most of the data collected on seabird and shorebird distributions on the Atlantic Outer Continental Shelf over the past 40+ years (O'Connell et al., 2011, 2009). The Catalog includes data for other taxa as well, and similar datasets are also available for cetaceans and sea turtles (e.g., Kenney, 2011). The Ocean Biogeographic Information System-Spatial Ecological Analysis of Megavertebrate Populations (OBIS-SEAMAP) is another large compendium of data, which includes distribution, abundance, and telemetry data for marine mammals, seabirds, and sea turtles over multiple decades (Fujioka et al., 2014; Halpin et al., 2009). These databases have been used in Rhode Island (Kenney and Vigness-Raposa, 2010), New York (Kinlan et al., 2012a; Lagueux et al., 2010), and the South Atlantic Bight, offshore of the Carolinas, Georgia, and Florida (Michel, 2013), among other locations (Best et al., 2012), to assess wildlife distributions and abundance and identify data gaps.

#### *Seabirds*

Based on a subset of the Northwest Atlantic Seabird Catalog data, primarily from the 1980s, Kinlan et al. (2012) found distributions of marine birds offshore in the New York Bight to be broadly similar to this study, with some species groups showing strong nearshore distributions (e.g., sea ducks, terns, small gulls), some using the offshore environment more broadly (e.g., Northern Gannet, large gulls), and others displaying consistently offshore distributions (e.g., alcids, jaegers, and storm-petrels). Catalog data for the mid-Atlantic also indicate similar patterns to those derived from our more recent boat and aerial survey data. In Catalog datasets, Red-throated Loons and scoters were observed nearshore and primarily in the winter, for example, while Northern Gannets were seen in high densities in the fall, winter, and spring throughout much of the study area (O'Connell et al., 2009). The species of seabirds observed, along with the timing of their peak abundances and the inshore vs. offshore patterns of their distributions, were largely similar to our findings, though we saw fewer shearwaters and Wilson's Storm-Petrels than would be indicated based on the data in the Catalog. It is important to note when examining these Catalog data, however, that they cover a very broad time range, and seabird distributions could have changed since the 1970s (O'Connell et al., 2009).

Based on a review of existing data, similar species composition and distributions have also been reported for the South Atlantic Bight. Common Loons are more abundant than Red-throated Loons in the region, for example, with the latter having a more inshore distribution (Jodice et al., 2013). Data from this region include fewer alcids than the mid-Atlantic, however, and a greater variety of more southerly species, including *Pterodroma* petrels, a greater diversity of storm-petrels, tropicbirds, and boobies (Jodice et al., 2013). In general it appears that marine bird abundance may be lower in the South Atlantic Bight, likely because oceanographic features tend to not create consistent or predictable areas of increased productivity, and bathymetric features that do exist are farther offshore (Jodice et al., 2013). Regular pelagic surveys have not been conducted in this study area, which may also be a factor (Jodice et al., 2013).

Perhaps the most similar recent avian study efforts to our mid-Atlantic Baseline Studies are the New Jersey Department of Environmental Protection's boat and visual aerial surveys offshore of New Jersey in 2008-2009 (Geo-Marine Inc., 2010a) and the Rhode Island Ocean Special Area Management Plan's boat and visual aerial surveys in 2009-2012 (Paton et al., 2010; Winiarski et al., 2012). Both studies obtained some data on avian flight heights in the offshore environment, although these data were derived from visual observations during boat surveys rather than using parallax in digital video aerial surveys (Hatch et al., 2013), and thus are likely biased towards somewhat lower altitude bands than the aerial data from our study. The New Jersey study defined the potential rotor-sweep zone for offshore turbines as 31-213m (100-700 ft), and found that 4.8% of observed individuals recorded during shipboard surveys occurred in this range (Geo-Marine Inc., 2010a). Rhode Island surveys suggested 6% of observations occurred at 25-125m in altitude and <1% at >125m, although these percentages included birds on the water's surface as well (22% of all observations; Paton et al., 2010). In contrast, our aerial survey data for the mid-Atlantic suggested that 38% of flying birds occurred between 20 and 200 m in altitude, a rotor-sweep zone range that was used in one recent study to cover a variety of possible turbine types and tidal effects (Willmott et al., 2013). In all three studies, however, the highest percentage of bird observations occurred below the potential range of rotor-sweep zone heights.

The New Jersey study indicated that avian densities were highest in nearshore regions during all seasons, although the pattern was more pronounced in winter than in summer, due to differences in community composition between seasons. Winter avifauna was dominated by inshore-foraging species (e.g., scoters and Laughing Gulls, *Leucophaeus atricilla*), while the summer community included more offshore foraging species, with predictive models indicating distributions that were farther offshore and in deeper waters (Geo-Marine Inc., 2010a). This is a different pattern than observed south of New Jersey in our study, despite similar species compositions noted in both studies; Common Terns, for example, were considered to be "offshore foragers" during summer in the New Jersey study, while breeding Common Terns clearly were foraging in relatively nearshore areas in our study as compared to many other species (Chapter 17). In our mid-Atlantic study, winter was the period of highest avian abundance, and winter distributions tended to be farther offshore than summer distributions (Chapter 12), although these patterns varied substantially between years.

The Rhode Island study found that nearshore, shallow waters were important to a broad range of species (though it should be noted that in addition to offshore survey data, this dataset relied heavily on land-based seawatches, which by their nature will suggest higher abundance near the coast). Nearshore waters were important in summer for terns, gulls, shorebirds; in winter, sea ducks and loons were also commonly observed during surveys. Species that relied on the ocean for food year-round (such as shearwaters, storm-petrels, and Northern Gannets) tended to be distributed farther offshore than species that only used the ocean during part of their annual cycle (including loons, grebes, and waterfowl; Paton et al., 2010). In general, species guilds and seasonal distribution patterns were similar between Rhode Island and our mid-Atlantic study area. Fewer species were detected in Rhode Island boat surveys than in our mid-Atlantic boat surveys, however, and species composition was slightly different, as would be expected based on the two studies' different latitudes and bathymetry. For example, Black-legged Kittiwakes (*Rissa tridactyla*) were much more common in offshore areas of Rhode Island in winter than they were anywhere within our mid-Atlantic study area. This is likely in part because kittiwakes were mostly observed in >50 m water depths in Rhode Island, while our maximum water depths in the mid-Atlantic were <40 m. Fewer species and guilds were observed in Rhode Island aerial surveys as compared to our mid-Atlantic aerial surveys, as well, though species compositions were broadly similar, with the exception of Common Eiders, a common species in New England that is largely absent from the mid-Atlantic.

In Rhode Island boat surveys, 94% of avian observations were identified to species, as compared to 72% in our study (Chapter 14). Large flocks of Black Scoters and Surf Scoters greatly reduced this identification rate in the mid-Atlantic, which otherwise was 97% for boat surveys. Visual aerial surveys in 2009-2010 in Rhode Island had a species identification rate of 62% (Paton et al., 2010), as compared to 45% in digital video aerial surveys, a rate that was likely influenced by a range of factors (Chapters 1 and 14).

Winter surveys in Rhode Island detected fewer species and lower abundance than summer or fall (though Northern Gannet and Common Loon detections were highest in winter). Fall was the period of highest species diversity in the mid-Atlantic boat surveys, but winter was the period of highest abundance in our study area. Northern Gannets, while a common migrant in Rhode Island waters in spring and fall, appeared to be a much more common winter resident in mid-Atlantic waters. Sea ducks were commonly observed in Rhode Island surveys, but at nowhere near the relative abundance we observed in the mid-Atlantic, where scoters were much more abundant than any other avian taxon in both boat and aerial datasets. In both studies, however, there were large amounts of interannual variation in abundance for sea ducks, and they were consistently observed foraging in areas <25 m deep.

Both studies found Common Loons and Red-throated Loons to be common in winter; offshore of Rhode Island, most loons were observed in nearshore waters <35 m deep, but, as this was essentially the same depth range as our entire study area, we cannot determine whether loon distributions dropped off in deeper waters in the mid-Atlantic (although Red-throated Loon distributions in our study area, at least, were distinctly skewed towards nearshore and shallow waters). The same six species of alcids were observed by both studies in winter; spatial segregation between species was observed in Rhode Island, with Razorbills (*Alca torda*) specializing in shallower areas closer to land, Common Murres (*Uria aalge*) in central latitudes, and Dovekies (*Alle alle*) appearing to be offshore specialists. The alcid data in the mid-Atlantic was more difficult to parse to species, particularly the digital aerial survey data, but there was some indication that Dovekies were distributed farther offshore than Razorbills (Chapter 12).

Herring Gulls (*Larus argentatus smithsonianus*) were the most common species observed offshore of Rhode Island, particularly near summer breeding colonies and dispersed offshore in fall. Observations of this species in the mid-Atlantic were less common relative to scoters and other taxa, and seldom occurred in summer (Chapter 17), possibly because the species was located almost exclusively in state waters west of the survey area. Terns were commonly observed in summer in nearshore areas in both studies, though most terns in Rhode Island were observed by land-based observers rather than on boat or aerial surveys. Roseate Terns were almost exclusively detected in land-based point counts in Rhode Island, despite targeted boat surveys for this species in late summer, and although >100 individuals were regularly observed on Block Island in August, suggesting regular passage across Block Island Sound (Paton et al., 2010).

Bathymetry was an important driver of distributions in the Rhode Island study area. Many more sea ducks were observed in Nantucket Sound in the mid-2000's (as cited in Paton et al. 2010) than in the Rhode Island surveys in 2009-2010, and the authors suggest that this is because Nantucket Sound is mostly <20 m deep. In contrast, many species that were observed and used deeper waters in Rhode Island were not observed at all in Nantucket Sound (Paton et al., 2010). Study authors concluded that bathymetry drove patterns in water temperatures, circulation, productivity, and other variables offshore of Rhode Island, and that water depth was an important driver of distribution, abundance, and species composition of seabirds as a result. Despite the much greater numbers of sea ducks observed in the mid-Atlantic compared to Rhode Island, we suspect that bathymetry is a similarly important driver of avian distributions in our study area, with sea ducks common in shallow (nearshore) areas, and offshore specialists more common in deeper waters. Water depth and distance to shore are highly collinear in the mid-Atlantic study area, and in many cases in this report we refer to "nearshore" areas being important for many species. However, Rhode Island distribution data suggest that it is bathymetry, rather than distance to shore, that is actually driving these distributions for many species. The exception is likely to be birds breeding on the shoreline west of the study area in summer, whose foraging ranges are limited by distance from their breeding locations.

### *Avian migration patterns*

Thirty four species of landbirds and passerines were detected in land based seawatches during the Rhode Island study; many fewer species were seen from boat, with the most common being Tree Swallows (*Tachycineta bicolor*). Swallows, including Purple Martins (*Progne subis*) were also the most common of the 29 passerine, raptor, and other landbird species observed in mid-Atlantic boat surveys (Chapter 8). A slightly different species composition was detected during passive acoustic monitoring from the survey vessel at night, where migratory flight calls were mostly identified as finches, thrushes, and warblers (Chapter 26).

NEXRAD studies in New Jersey indicated that nocturnal avian migratory activity over the ocean was higher in the fall than in the spring (Geo-Marine Inc., 2010c), which was also clearly evident on our study (Chapter 27). The New Jersey study also indicated that nearshore bird densities were higher than offshore bird densities in both spring and in fall, however. Our mid-Atlantic study, which corrected for biases in measured reflectivity caused by distance from the radar unit, predicted offshore migratory activity in fall to be as high or higher than levels of migratory activity at many onshore locations.

### *Marine mammals and sea turtles*

Existing data on marine mammals and sea turtles from the Atlantic coast of the U.S. suggests largely similar patterns to what was observed during our study, although community composition differs between locations, in large part in relation to water temperature and bathymetry. Data from the South Atlantic Bight, for example, include the same five sea turtle species observed in our mid-Atlantic study area, and Loggerhead Sea Turtles were also the most abundant species in the South Atlantic (Read, 2013). Loggerheads are present in the region year-round, however, which appears not to be the case in the mid-Atlantic (Chapters 15 and 17). Sea turtles were much more abundant in the mid-Atlantic study area than in the New York Bight or southern New England, however, particularly in spring and fall, likely due to warmer ocean temperatures than in more northern latitudes (Chapters 15 and 17; Kenney and Vigness-Raposa, 2010; Lagueux et al., 2010). Species diversity may likewise be higher in the mid-Atlantic during these months, based on existing data for New England and New Jersey (Geo-Marine Inc., 2010b; Kenney and Vigness-Raposa, 2010), although none of these other recent efforts used digital aerial survey approaches, and their results for sea turtles are thus not directly comparable to those presented in this report.

As in the mid-Atlantic, the highest abundances of Bottlenose Dolphins offshore of New Jersey were predicted in spring and summer, and Common Dolphins in winter and spring (Chapters 15 and 17; Geo-Marine Inc., 2010b). Interestingly, the New Jersey study observed lower abundance of Bottlenose Dolphins during the fall months, speculating that observed coastal populations moved south of New Jersey during this time. Our study provides some corroboration for this idea, as we observed sustained abundance of Bottlenose Dolphins during this season, with highest encounter rates predicted in nearshore regions (Chapters 15 and 17). An online cetacean habitat modeling systems for the US east coast, based on ship-based and visual aerial survey data from OBIS-SEAMAP, predicted similar cetacean species in the mid-Atlantic study area to what we observed, with inshore Bottlenose Dolphin distributions being driven by water depth and specific SST ranges in the spring (Best et al., 2012).

Rare large whale species, including the North Atlantic Right Whale, Humpback Whale, and Fin Whale, are observed in southern New England primarily in spring, summer and fall, while in our study the majority of animals were seen in winter (Kenney and Vigness-Raposa, 2010). All Right Whales, for example, were observed in the mid-Atlantic study area February or March, presumably during the earlier part of their northward spring migration (Chapters 15 and 17). It should be noted, however, that studies that include passive acoustic surveys for whales (e.g., Geo-Marine Inc., 2010a; Rice et al., 2014) have found these species present year-round within their study areas, and an ongoing passive acoustic study offshore of Maryland may confirm that the same is true in the mid-Atlantic (Bailey and Rice, 2015).

As in more northerly survey locations, cetacean species that tend to occur at or beyond the continental shelf break (such as beaked whales, some types of sperm and pilot whales, and several species of dolphin) are probably most likely to be found to the east of our study area, though they may be exposed to underwater noise from development activities within the study area (Kenney and Vigness-Raposa, 2010). Cetacean abundance was predicted to be higher near the shelf break and offshore of the Outer Continental Shelf than in nearshore areas in the New York Bight (Lagueux et al., 2010), and the same may well be true in the mid-Atlantic.

### *Using data from this project in permitting and decision making*

Baseline studies along the U.S. Atlantic coast have generally found that, with the possible exception of marine mammals (above), overall abundance and species diversity tends to be highest in shallow water areas (which in many cases are coincident with areas closer to shore, though not always). Results from these studies have been used to identify areas of high biodiversity and priorities for conservation, ultimately influencing the choice of lease sites for offshore wind development. For example, the Rhode Island Coastal Resources Management Council identified *Areas Designated for Preservation*, and prohibited large-scale offshore developments and other activities (including, but not limited to, offshore wind) in areas of 20 m or less in water depth, specifically to preserve foraging habitat for sea ducks (Rhode Island Coastal Resources Management Council, 2013). In other locations along the east coast, the specific areas offered for offshore wind energy development leases (e.g., included in WEAs) have also been determined in part via the use of wildlife distribution and abundance data[8].

Results from this project represent a baseline that can be used for comparison with compatible future surveys, and to assess changes in offshore populations due to development or other causes. This study is an important first step towards understanding the implications of offshore wind energy development for bird, marine mammal, and sea turtle populations in the mid-Atlantic. These data on the geographic distributions and relative abundance of wildlife in the mid-Atlantic are expected to be useful for minimizing impacts to wildlife populations from offshore wind energy development in that they can be used to (1) help inform the siting of future projects, (2) address the environmental permitting requirements for current and future projects, and (3) inform the development of mitigation approaches aimed at minimizing potential effects. Exposure to offshore development does not necessarily indicate that exposed animals will suffer deleterious effects, however, or that effects will translate to population-level impacts. Siting and permitting future projects, as well as other efforts to minimize potential effects, will rely on the baseline data collected in this study, but must move beyond these initial steps to focus on species most likely to be impacted due to their conservation status or other factors.

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[8] www.boem.gov/BOEM-Newsroom/Press-Releases/2012/press05302012.aspx

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# Introduction to Part II
# Examining wildlife distributions and relative abundance from a digital video aerial survey platform

## Report structure

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure I).

Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure I. In Parts II (Chapters 3-6) and III (Chapters 7-12), we describe methods and results for high resolution digital video aerial surveys and boat surveys, respectively. Part IV of this report (Chapters 13-19) combines data from both survey approaches to develop a comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area. Part V (Chapters 20-25) focuses on individual movements and habitat use of focal avian species, tracked via satellite telemetry; and Part VI (Chapters 26-27) focuses on population-level migratory movements over the oceans, using several approaches for studying nocturnal avian migration. An additional study effort, which further explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

## Part II: Examining wildlife distributions and relative abundance from a digital video aerial survey platform

High resolution digital video aerial surveys are a relatively new method for collecting distribution and abundance data on animals (Thaxter and Burton 2009, Buckland et al. 2012), and ours was the first study to use this method on a broad scale in the U.S. The technology used in this study, one of several different digital aerial survey methodologies, was developed by HiDef Aerial Surveying, Ltd., in the UK. Digital aerial survey approaches have largely replaced visual aerial surveys for offshore wind energy research in Europe, as their higher flight speeds and much higher flight altitudes make them safer to conduct than visual aerial surveys, and reduces or eliminates disturbance to wildlife compared to visual aerial or boat survey approaches. They also produce archivable data, which allow for a robust quality assurance and audit process. There are still limitations to this method, however, including difficulties identifying some species, and a lack of defined statistical approaches for utilizing the data for some purposes, due to the relative novelty of the survey method.

There are four chapters in Part II of this report, focused on the use of digital video aerial surveys to examine wildlife distributions and relative abundance:

| | |
|---|---|
| Chapter 3. | High resolution digital video aerial survey methods. |
| Chapter 4. | Data management, video analysis, and audit protocols for digital video aerial surveys. |
| Chapter 5. | Summary of high resolution digital video aerial survey data. |
| Chapter 6. | Recommendations for further development of high resolution digital video aerial surveys in the U.S. |

### *Methods and protocols*

Chapter 3 briefly describes the survey methods employed for high resolution digital video aerial surveys, which are referenced throughout the following chapters. Surveys were flown in twin-engine Cessnas at 250 km/hr and an altitude of approximately 610 m, which is much higher than traditional visual aerial surveys. While analysis and management of video require substantial personnel time, the resulting data are quality-controlled and audited much more intensively than is possible with visual observation data (Chapter 4).

### *Results from mid-Atlantic digital video aerial surveys*

Surveys detected a wide variety of taxa, including marine mammals, sea turtles, rays, sharks, fish, bats, seabirds, shorebirds, and raptors (Chapter 5). Some taxa were notable for their unexpected abundance within the survey dataset (e.g., Cownose Rays, *Rhinoptera bonasus*, and sea turtles). Other taxa were not expected to be observed in surveys at all (e.g., bats; Chapter 5; Hatch et al. 2013). Flight heights of flying animals could be estimated from the aerial video using parallax, or the movement of animals relative to the ocean background (Chapter 5; Hatch et al. 2013). This information may be helpful in understanding the potential for interactions between flying animals and offshore wind turbines. For example, 59% of all birds with estimable flight heights were observed within 0 and 20 meters above sea level, which is below rotor height for most turbine designs. This type of flight height data is often used alongside information on avoidance behaviors, turbine specifications, and other data in models that attempt to estimate avian collision risk for offshore wind energy projects in Europe (e.g., Band 2012), although there is still debate in the European literature regarding the factors that best predict this risk (e.g., Cook et al. 2012, Douglas et al. 2012, Langston 2013, Furness et al. 2013).

Identification of animals to species in the video aerial survey data was variable by survey, season, and taxon (Chapter 5). In part, this is likely due to variations in image quality and other factors.

Observational data from boat or visual aerial surveys are not replicable, however, and species identifications made by observers in the moment can seldom be verified after the fact. The exhaustive quality assurance and audit protocol followed by aerial video reviewers, as well as characteristics inherent to the video review process itself (such as the use of multiple levels of “certainty” criteria in identifications), ultimately lead to fewer definitive identifications than observational approaches (Chapters 13-14). However, this also recognizes the inherent uncertainty in the identification process, which can be difficult to account for in unrecorded visual surveys. This uncertainty is generally under-recognized or ignored, as it can be difficult to measure, but in some cases species misclassification in visual surveys may actually lead to less reliable density estimates than classifying animals as “unknown” (Conn et al. 2013).

### *Recommendations for future use of digital aerial surveys in North America*

Digital aerial approaches were developed in Europe, but the application of these technologies in North America demonstrates clear avenues for additional research and development (Chapter 6). The species composition of ecological communities in the western Atlantic varies considerably in some cases from what is present in the North Sea and Baltic Sea. Early indications suggest that digital aerial surveys may have distinct advantages over visual aerial or visual boat surveys for sea turtles, for example, a taxon of considerable interest in North America but that rarely occurs in Europe (Chapters 14-15; Normandeau Associates Inc. 2012). Even pan-Atlantic species may possess different characteristics in North America than in Europe. This is clearly seen in the large range of body sizes of Common Loons that winter in the mid-Atlantic U.S. (Barr et al. 2000, Gray et al. 2014), and the resulting difficulty in differentiating Red-throated Loons and Common Loons in aerial video in this study, a difficulty largely absent from observational data collected from boats (Chapter 16). Additional exploration of species identification capabilities—for example, by conducting test flights over known-species flocks—could aid the future application of this technology in the U.S. Identification rates in digital aerial surveys have also continued to improve with technological advances in the field; the current generation of cameras being used in Europe have much higher resolution and color rendition than the cameras used in this study, with better identification rates as a result (95% for all seabirds, on average; A. Webb pers. comm.).

In addition, there is a need to further the development of analytical approaches for digital aerial surveys. Because the cameras are pointed down towards the water’s surface (Figure II), providing a more vertical field of view than that from either visual aerial of boat surveys, digital aerial surveys avoid the common problem of distance bias; but, to date, other types of detection bias have not been addressed for digital aerial surveys. Further examination of detection rates (in relation to taxon, weather, sea state, time of day, and other factors) could be a fruitful avenue for methodological development (Chapters 6 and 13). Existing audit processes for object location in aerial video could be easily modified to incorporate a double observer approach and lead to more statistically rigorous, accurate, and reliable estimates of abundance for North American populations.

### *Implications*

In addition to the four chapters in this section, the digital video aerial survey data are used in analytical efforts in Chapters 13-19. Several chapters focus on contrasting the two survey approaches (Chapters

13-14 and 18). In some cases, digital aerial survey data are used independently to analyze wildlife distributions and relative abundance (e.g., in the case of sea turtles, which were much more easily detected in video than from boat surveys; Chapters 15 and 17). In other cases, digital video aerial survey data and boat survey data are used jointly (Chapters 16-17 and 19) to describe distributions and abundance of animals across the study area.

Our application of these methods in the mid-Atlantic is expected to be useful for understanding wildlife populations and minimizing impacts to those populations from offshore wind energy development in several ways:

- First, this study has developed U.S.-based technological resources for future monitoring efforts, and explored technological advancements and assessment methods that could simplify or minimize the cost of environmental risk assessments. We also compare high resolution digital video aerial surveys to boat-based surveys, to better understand the potential uses of high resolution digital video aerial surveys in relation to offshore development in U.S. waters.
- Second, we identify species that are likely to be exposed to offshore wind energy development activities in the mid-Atlantic study area, along with their important habitat use or aggregation areas and temporal variation in distribution patterns. This information can be helpful for:
    - Informing the siting of future projects, by incorporating wildlife patterns into marine spatial planning and decision making, and by using exposure data as a first step towards defining relative risk by location;
    - Informing the permitting process for projects, by contributing data towards National Environmental Protection Act (NEPA) and other regulatory requirements, and by helping to define target taxa or research priorities on which to focus on during site-specific pre- and post-construction monitoring studies; and
    - Informing mitigation efforts and construction and operations plans, by presenting temporal data on community composition, distributions, and abundance that can be used to time certain activities to coincide with reduced potential for exposure of certain populations.
- Third, digital aerial surveys have some considerable advantages over traditional visual observation approaches, most notably in relation to survey speed and safety, but they also require some different analytical approaches than traditional surveys, which the scientific community is still in the process of developing. We explore statistical models aimed at improving our utilization of digital video aerial survey data to understand wildlife patterns.

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**Figure I. Organization of chapters within this final report.**

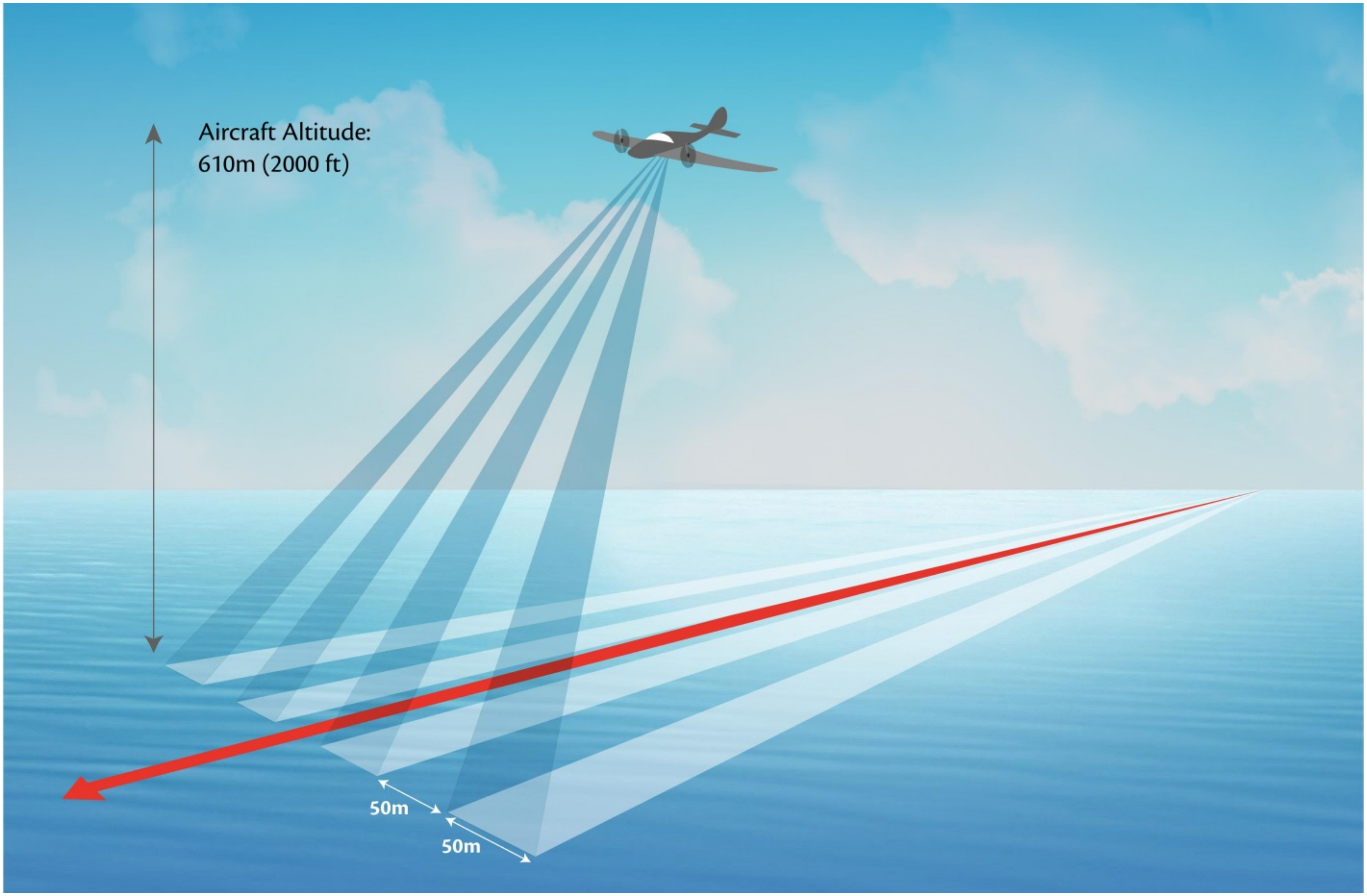

**Figure II. Digital video aerial surveys were flown at 610 meters using a twin-engine aircraft with four belly mounted cameras.** These cameras recorded non-overlapping 50 meter transect strips, for a 200 meter total transect strip width.

# Chapter 3: High resolution digital video aerial survey methods

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 3 Highlights

*Methods used to conduct high resolution digital video aerial surveys.*

### *Context*[1]

High resolution digital video aerial surveys are a new method for collecting distribution and abundance data on animals, and our study was the first to use this method on a broad scale in the U.S. The technology was developed by HiDef Aerial Surveying, Ltd., in the U.K. These methods have largely replaced visual aerial surveys for offshore wind energy research in Europe, as they are safer for the pilot and crew, reduce or eliminate disturbance to wildlife during surveys, and produce archivable and auditable data. This chapter briefly describes the methods used to collect and analyze the survey data. Chapter 4 describes the data management, object identification, and audit processes conducted for digital video aerial survey data in further detail. Basic results from the digital video aerial surveys are summarized in Chapter 5, and the data are analyzed alongside boat survey data in Part IV of this report. Recommendations for ways to improve data collection and analysis of the digital video aerial survey data are included in Chapter 6.

### *Highlights*

- Fifteen digital video aerial surveys were flown in the mid-Atlantic study area over two years (March 2012-May 2014).
- Planes flew at a speed of approximately 250 km/hr and at an altitude of 610 m. Using camera technology developed by HiDef Aerial Surveying, Ltd. of the United Kingdom, four super high-definition video cameras captured a 200 m wide transect strip.
- Video data were analyzed by two teams to locate and identify objects in the footage.
- Flight heights were estimated for flying animals using a patented extended parallax method.
- Audit processes, including blind re-review of 20% of video data, were carried out as part of both the object location and identification procedures.
- Completed datasets are available online at our website and are also included in the US Fish and Wildlife Service's Northwest Atlantic Seabird Catalog database.

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[1] For more detailed context for this chapter, please see the introduction to Part II of this report.

## Abstract

This chapter presents methods used to collect and analyze high resolution digital video aerial survey data. Fifteen high resolution digital video aerial surveys were conducted by Biodiversity Research Institute (BRI) and HiDef Aerial Surveying, Ltd. (hereafter, HiDef, the company that developed this technology in the United Kingdom) as part of a broader project to collect observations of marine birds, mammals, turtles, and other wildlife, and to inform siting and permitting processes for offshore wind energy development. Aerial transects were flown at high densities within the Delaware, Maryland, and Virginia Wind Energy Areas (WEAs); the remainder of the study area was surveyed on an efficient sawtooth transect path to provide broad-scale context for the intensive WEA surveys. High-density surveys were also conducted adjacent to the Maryland WEA in 2013-2014 as part of the Maryland Project. Precise wildlife locations, taxonomic identities, animal behaviors, and flight heights were determined from the resulting video images. Details on the analyses are found in the Video Aerial Survey Data Protocol in Chapter 4. Flight heights were calculated from video footage for flying animals using extended parallax methods developed by HiDef.

## Introduction

Digital aerial survey technologies, using either video or still photography, have been developed and successfully deployed in Europe to assess marine wildlife populations in relation to offshore wind energy development (e.g., Buckland et al., 2012; Groom et al., 2013; Thaxter and Burton, 2009). Though they have become common practice for offshore wind energy planning and monitoring in Europe (Buckland et al., 2012), this study is the first to use these methods on a broad spatial and temporal scale in the United States. Digital aerial surveys have a high cost efficiency on broad spatial scales, and it has been suggested that they may eventually largely replace traditional visual surveys, by boat or aircraft, to collect distribution and abundance data on animals in the offshore environment in Europe (Buckland et al., 2012). Importantly, the data collected using digital surveys are recorded, allowing for species identification verifications, the application of rigorous audit protocols, and archiving of footage for later review.

High resolution digital video aerial surveys (hereafter, digital video aerial surveys) were conducted on the mid-Atlantic Outer Continental Shelf offshore of Delaware, Maryland, and Virginia in 2012-2014, as part of a broader project to inform siting and permitting processes for offshore wind energy and development. In particular, aerial surveys were focused on obtaining detailed data on wildlife distributions in three federally designated Wind Energy Areas (WEAs). Wildlife locations, taxonomic identifications, animal behaviors, and flight heights were determined from the video images (discussed in additional detail in Chapter 4), and these data were used in further analyses, which are presented in later chapters.

## Data collection

Observations of marine birds, mammals, and turtles were collected in large-scale surveys across a 13,245 km$^2$ study area using super high-definition video on an aerial platform (Figure 3-1). Fourteen offshore surveys were flown by HiDef across the mid-Atlantic study area from March 2012 to May 2014. Aerial transects were flown at high densities (1 km spacing, or 20% ground coverage) within the Maryland, Delaware, and Virginia WEAs. Beginning in Year 2 of the study (March 2013), the footprint of high density surveys was extended west of the Maryland WEA to the shoreline, and 10 km south of the Maryland WEA, with funding from the state of Maryland (Figure 3-1). These Maryland extension transects were the only video aerial transects that extended into state waters (e.g., within 3 miles of shore). The remainder of the study area was surveyed using an efficient ‘sawtooth’ transect path to provide broad-scale context for the intensive WEA surveys (at about 2.1% ground coverage, beginning in September 2012; Figure 3-1). Early surveys included video footage at 2 cm GSR for the transects within the WEAs, and 3 cm GSR for the broader sawtooth survey; however, species identifications were problematic for 3 cm footage in early surveys, due to poor image clarity and color rendition, and this issue was addressed by project collaborators by discontinuing all use of 3 cm GSF for surveys beginning in September 2012 (Williams et al., 2013). The Maryland extensions added about 21% of additional transect length to the existing study design, with total combined transect length for each survey at approximately 2,866 km in Year 1, and 3,613 km in Year 2. An eighth annual survey was also added in Year 2 of the study, with funding from the state of Maryland. This survey included only the Maryland WEA and Maryland extension transects, totaling approximately 1,088 km in length, and was flown in August 2013.

In addition to the fifteen surveys described above, HiDef also flew a survey specifically designed to allow for a comparison of aerial and boat-based data collection. The flight occurred during one of the regularly scheduled boat surveys (March 2013), and followed the paths of several of the boat transects, rather than the aerial transects used in other surveys. Details regarding this comparison study can be found in Chapter 13.

HiDef worked with their video aerial survey vendor to outfit the survey aircraft and organize and schedule flights in the mid-Atlantic study area (Figure 3-1). Each survey was completed using two small commercial aircraft, allowing complete coverage of the study area in two to three days (weather permitting). The aircraft were twin-engined Cessnas, with long range fuel tanks to enhance safety when operating at sea, and had specially designed frames attached to the lower fuselage for survey cameras. Due to the height at which surveys were flown, no permits were required from the National Marine Fisheries Service (NMFS), but flights complied with all Federal Aviation Administration (FAA) regulations.

Each survey was conducted at approximately 250 km/hr and at 610 m (2,000 ft) above sea level using four super high-definition (five times HD) video cameras, angled at 30-45° from vertical and integrated with onboard navigation systems and server storage (Figure 3-2). Cameras captured up to 15 frames per second, and images were duplicated and stored onto a disk array of heavy duty disk drives or solid state recording devices within the aircraft. Video footage was shipped to the HiDef office in the UK by the video aerial survey vendor, and as a precaution video footage was also copied onto hard drives by the video aerial survey vendor and shipped to the BRI office in Gorham, Maine.

Each of the four cameras captured video images at a 50 m strip width at sea level, resulting in a 200 m wide transect strip (Figure 3-2). Surveys were flown under Visual Flight Rule (VFR) conditions and were completed in weather conditions appropriate for observations (<6 Beaufort with no low cloud cover, mist, or fog). All surveys were flown using GPS to ensure location accuracy.

## Digital video data analyses

The HiDef team reviewed each frame of the recorded footage to mark visible objects and note object categories (e.g., Bird, Buoy, Fish). These data were output to an Excel spreadsheet and marker files were generated and saved for object identifications for the BRI team (see Chapter 4 for more details). HiDef observers re-reviewed 20% of the frames in each survey to determine the rate of agreement between observers; agreement had to be at least 90% for the audit to pass. If the audit did not pass that observer's recent data were examined for consistent errors and issues were addressed. Data spreadsheets and markers for all objects that were found by the original observer and the auditor were sent to BRI staff for further analyses.

Trained BRI staff identified the objects to species, taxonomic group, or general category (e.g., flotsam and jetsam), and described animal behaviors. Identifications were based on size, shape, color, movement pattern, and clarity of the image, and confidence of identification was noted for each object. "Definite" indicated >95% certainty, "probable" indicated <95% but >50% certainty, and "possible" indicated <50% certainty in the identification. For example, if a reviewer could not substantiate that an object was a "possible Wilson's Storm-Petrel," then that object might be coded as a "definite unidentified storm-petrel," based on the specific criteria used for identifications of that species or category (size, color, shape, flight pattern, clarity of image, etc., see Chapter4 for more details). Some animals and objects were submerged underwater. Reviewers could see at some depth, but visibility of submerged objects varied based on turbidity and weather, and no formal steps were made to verify the range of depths within which animals could be accurately identified. All non-avian animals in the water column were marked as either submerged or surfacing. Completed data sheets with identification information were returned to HiDef in the UK for georeferencing and parallax calculations. Twenty percent of the identification data were audited by BRI, with at least 90% agreement required to pass. Detailed object ID, data management, and audit protocols for BRI analysis procedures are included in Chapter 4.

HiDef calculated flight altitude for moving targets using the measurement of "parallax" in the aerial video. Parallax is the apparent motion of an elevated object against a distant background due to the movement of the observer. With a known distance to the background and motion of the observer, the parallax was measured from relative positions in digital video frames and used to estimate the height of the object above the background (Hatch et al., 2013). Most objects were observed in at least eight video frames at the altitude and speed at which digital video aerial surveys were conducted. Flight height could not be accurately estimated using this approach when the animal was flying parallel to the plane and no displacement was detectable, or the animal was flying at high altitudes and was present in fewer video frames.

HiDef also georeferenced each video frame containing an animal, using GPS data from the survey flight and offset calculations to account for camera angles. Directions of movement were also translated into cardinal directions, based on the direction in which each camera was pointed during the recording time. Spreadsheets with flight height, animal direction of movement, and georeferenced data were returned to BRI to be joined with audited identification data by the data manager.

Aerial effort data were built from either the georeferenced camera reel data files or raw backup GPS data files. We preferentially built effort data from the georeferenced camera reels, which included a position for every camera frame while the survey cameras were active. This was the most accurate positional data from which to generate the effort data, as these files were only generated while the cameras were actively filming and collecting data. Early in the project, there were several partial surveys where the GPS associated with the cameras was not working properly and there were no positions associated with camera reels. However, backup GPS positioning was available, and we used these data along with planned transect lines to generate the effort for these transects. Custom scripts were written in Python for ArcGIS 10.2 (ESRI, Inc., Redlands, CA) to derive the effort lines from the camera reel geo-references and/or the backup GPS. We also generated effort polygons for the four camera stripes using another custom Python script; these stripes were derived from the transect lines, with the proper spacing between cameras (50 m) and width of the cameras' field of view (50 m each)[2]. Effort data were further associated with survey observations in post-processing.

## Additional information

The complete digital video aerial survey dataset is available for download on the project website[3]. It has also been added to the Northwest Atlantic Seabird Catalog (formerly the Compendium of Avian Information), a publicly held database housed by the USFWS that is the main repository for observations and survey data collected in Atlantic waters from Florida to Maine since 1906 (including data on marine mammals, sea turtles, and other wildlife, as well as seabirds).

This study represents the first application of high resolution digital video aerial survey technology in North America, and was also the first broad-scale application of any type of digital aerial survey in the United States. A more detailed description of video data analysis and management procedures is available in the following chapter of this report (Chapter 4). The digital video aerial survey data are summarized in Chapter 5, and used alongside boat survey data in analyses in Part IV of this report (Chapters 13-19).

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[2] On the first three surveys, the sawtooth transect was flown at 3 cm GSR, so the transect width was 75 m and the spacing between cameras was 25 m.

[3] www.briloon.org/mabs/data

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## Figures

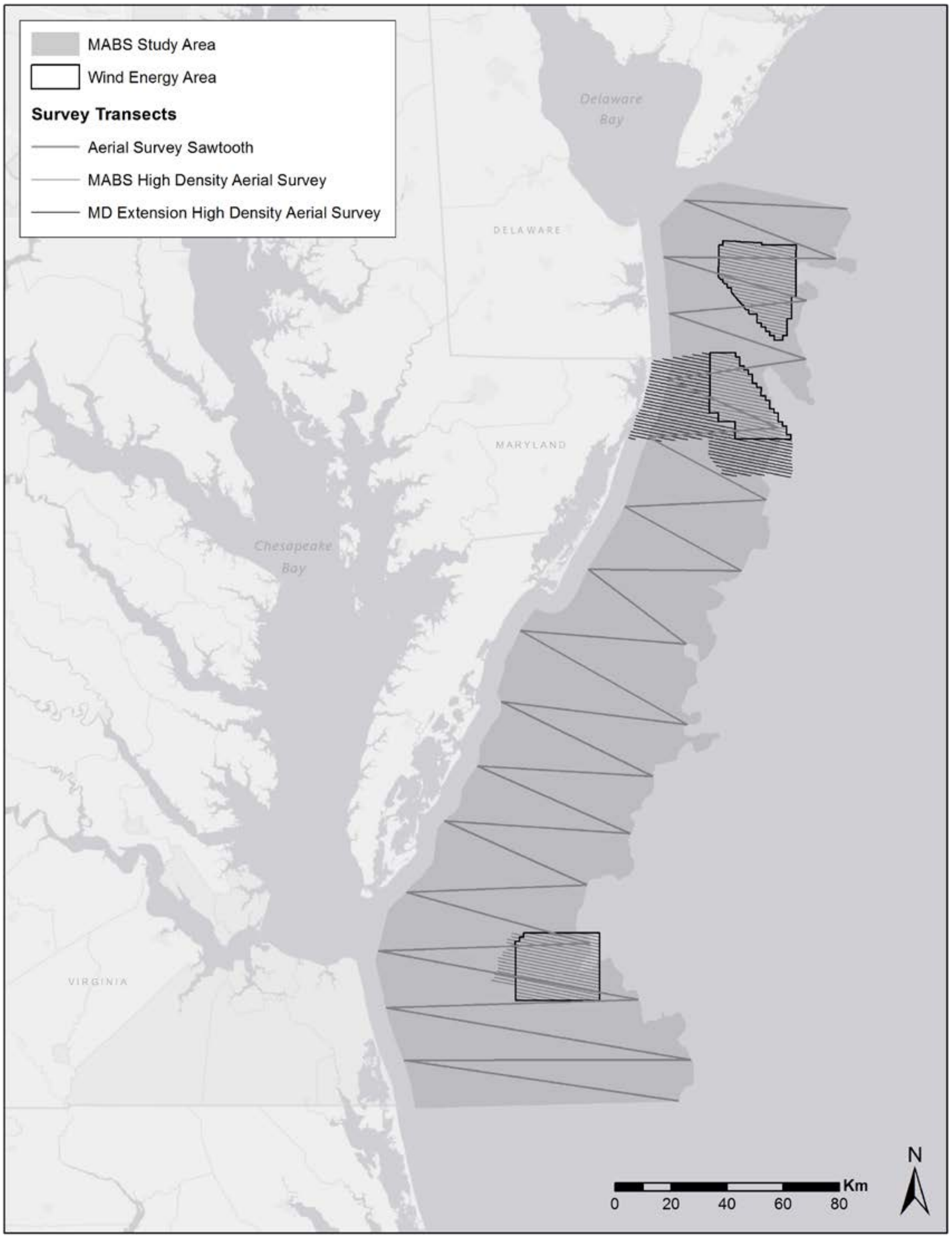

**Figure 3-1. Map of digital video aerial survey transects for the Mid-Atlantic Baseline Studies and Maryland Projects (2012-2014).** Mid-Atlantic Baseline Studies transects are shown in light gray. High-density Maryland extension transects (flown in 2013-2014) are shown in dark gray.

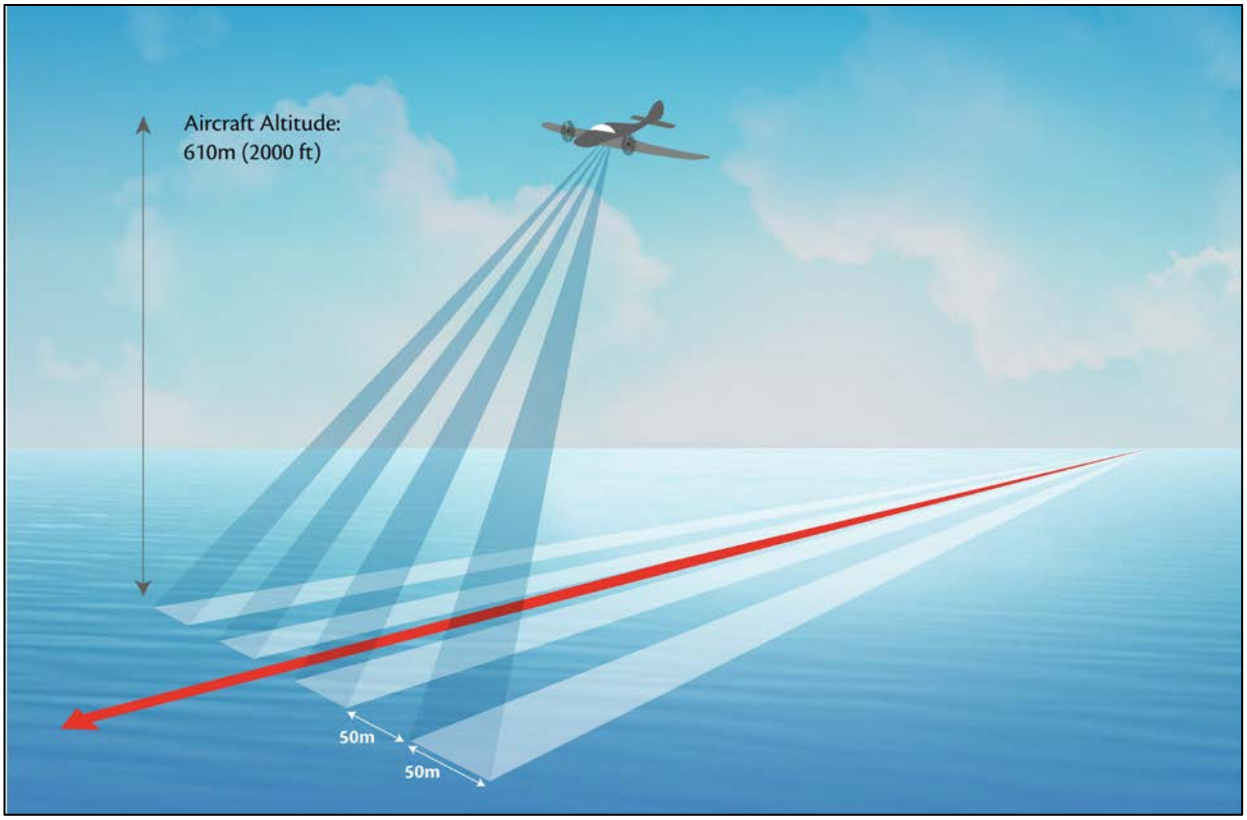

**Figure 3-2. Digital video aerial surveys were flown at 610 m using a twin-engine aircraft with four belly mounted cameras.** These cameras recorded non-overlapping 50 m transect strips, for a 200 m total transect strip width.

# Chapter 4: High resolution digital video aerial survey data protocols

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 4 Highlights

*Protocol for data analysis, data management, and audit procedures for the high resolution digital aerial surveys in the mid-Atlantic.*

### *Context*[1]

High resolution digital video aerial surveys are a relatively new method for collecting distribution and abundance data on animals in the offshore environment, and our study was the first to use this method on a broad scale in the U.S. The technology was developed by HiDef Aerial Surveying, Ltd., in the United Kingdom. This chapter describes the methods used to analyze the survey video, in particular describing the object identification and audit procedures in detail. Data collection methods and some analysis processes are described in Chapter 3. Basic results from the digital video aerial surveys are summarized in Chapter 5, and the data are analyzed alongside boat survey data in Part IV of this report. Recommendations for ways to improve data collection and analysis of the digital video aerial survey data are included in Chapter 6.

### *Highlights*

- Aerial video data were collected on abundance and behaviors of marine birds, mammals, turtles, and other wildlife within the Mid-Atlantic Baseline Studies project area.
- Locations of individual animals (or objects), taxonomic identifications, behaviors, and flight heights were determined from the video images.
- A random 20% of the objects identified were blindly audited through re-review with an extensive arbitration process in cases of disagreement.
- 100% of threatened and endangered species were audited with exact matches required.
- Example images, data collection spreadsheets, and definitions of identification categories are included.

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[1] For more detailed context for this chapter, please see the introduction to Part II of this report.

## Abstract

High resolution digital video aerial surveys were conducted in the mid-Atlantic, as a part of the Mid-Atlantic Baseline Studies Project, to produce data to inform siting and permitting processes for offshore wind energy development. Data were collected on the abundance and behaviors of marine birds, mammals, turtles, and other wildlife. Wildlife locations, taxonomic identifications, animal behaviors, and flight heights were determined from the video images. This chapter describes the protocol for the data analysis process, the procedure for identifying objects, and quality assurance and quality control procedures. Example images are included to illustrate methods used during the analysis, along with definitions of identification categories used, and tables showing the types of data collected during video analyses.

## Introduction

The goal of the Department of Energy-funded Mid-Atlantic Baseline Studies Project (2012-2015) is to produce the data required to inform siting and permitting processes for offshore wind energy development in federal waters of the mid-Atlantic region (DE-VA; Figure 4-1). Data on the abundance and movements of marine, coastal and migratory birds, marine mammals, sea turtles, and other megafauna are collected within federally designated Wind Energy Areas (WEAs) and elsewhere within the study area, and analyzed using a variety of technologies and methods.

As one component of this study, BRI and HiDef Aerial Surveying Limited (HiDef) are conducting large-scale surveys across the entire study area using high resolution video on an aerial platform. HiDef's technique uses an array of four high resolution video cameras (which can be either belly or nose mounted depending on aircraft type) on twin-engine Cessna aircraft to capture detailed footage and a consistent viewing frame of the ocean surface; survey flights are conducted at 2,000 feet above sea level (Chapter 3). Wildlife are observed in the video footage, georeferenced, and identified to species or lowest taxonomic order. This technique also allows assessment of individual behavior and estimation of animal flight height.

Some components of the Mid-Atlantic Baseline Studies Project's survey and analysis procedures are conducted by HiDef, while others are conducted by the Video Review Laboratory at BRI. As digital video aerial surveys are a relatively new technique for monitoring wildlife in the offshore environment, protocols for data management, data analysis, and quality assurance procedures have only recently been developed by practitioners in Europe and elsewhere, and these protocols are in many cases under continued development as techniques and technologies are further refined. In order to provide transparency and accountability for all video data review conducted by BRI during the project, Video Review Lab personnel have developed detailed data protocols for the components of the data analysis and data management processes for which they are responsible.

## Overview of data analysis process

The general process for recording and analyzing high definition video aerial data includes the following steps. A glossary of terms may be found in Appendix 4A.

*I. HiDef Aerial Surveying Ltd.*

a) HiDef works with their digital video aerial survey vendor to outfit the survey aircraft and undertake survey flights in the mid-Atlantic region.

b) The digital video aerial survey vendor ships the video footage to HiDef in the UK, and also makes hard drive copies of the video and ships them to the BRI office in Maine.

c) The HiDef review team views each frame to mark visible objects (or *targets*) using proprietary video processing software, and notes object categories (e.g., Bird, Buoy, Fish). These data are outputted to an Excel spreadsheet, and the markers generated through this process are used by BRI reviewers to locate animals within a frame. Example images of marked animals can be found in Appendix 4C.

d) On completion, 20% of the frames in each survey are re-reviewed (blind) by a second HiDef observer to determine the rate of agreement between observers. Agreement must be at least 90% for the audit to pass. All objects found by both the observer and the auditor are included in the final file sent to BRI regardless of whether the audit passes. If the audit is not passed, that observer's recent data are examined for consistent errors and issues are addressed.

e) Spreadsheets with marked data are sent to BRI for object identifications (Appendix 4B).

*II. Biodiversity Research Institute*

a) The BRI review team examines video frames which contain marked targets. Each target is identified to species or group, at the lowest possible taxonomic level, or as abiota of various types (Appendix 4C; also see "ID Category" section below). An assessment of the reviewer's certainty level is associated with each identification. If possible, ancillary data, such as the animal's behavior, direction of movement, and age and sex, are also noted (Appendix 4B, Table 4B-2). Direction of movement is noted in relation to the viewing screen (e.g., up, down, left or right).

b) The identification data spreadsheets are returned to HiDef in the UK.

c) Twenty percent of the objects originally categorized as animals by the HiDef review team are re-reviewed by additional BRI reviewers to determine the rate of agreement between observers. Agreement is defined according to relationships described in the QA/QC Review Protocol below. If <90% agreement is obtained for a given survey, supplementary audit and review processes are conducted as outlined in this protocol.

*III. HiDef Aerial Surveying Ltd.*

a) HiDef calculates flight altitude for flying targets listed in the identification data spreadsheets, using their proprietary parallax technology (Hatch et al., 2013), and georeferences each video frame containing target objects using GPS data from the survey flight. Direction of movement is translated into a cardinal direction, based on the direction in which each camera was pointed during the time of recording (Appendix 4B, Table 4B-2).

b) The spreadsheets are returned to BRI with added parallax, location, and direction of movement information.

*IV. Biodiversity Research Institute*

a) These data are joined to the audited data held by BRI and complete datasets are sent to the US Geological Survey, which is currently managing the federal Compendium of Avian Information database, as well as to project partners for statistical analysis.

Tasks II.a and II.c. are described in “Procedure for target identification” and “Quality assurance and quality control of data collection,” below. Detailed information on the survey and data management tasks completed by HiDef Aerial Surveying, Ltd. and their contractors are outside the scope of this protocol.

## Procedure for target identification

### *Video data storage*

Video footage is stored on external hard drives and shipped between the aerial operator, BRI, and HiDef. These hard drives are uploaded to the server at BRI upon arrival, and data are accessed by the review team through this server. When data analysis and management is completed for a survey, the video data are transferred to external hard drives for long-term storage in a fire safe at an external location.

### *Filename conventions for spreadsheets and sequence files*

Video file names are in the following format: ZoneID#_Month#_SurveyDay_CameraNumber_Year (example: Zone19_M03_S01_D01_C2_12). Sequences, which contain camera reels, are named in the format 10-15-43.796.

In the above examples, Zone 19 refers to the mid-Atlantic study region; M03 refers to the calendar month (e.g., 03=March); S01 refers to the first survey of that month; D01 refers to when the section was flown over the course of the survey (e.g., Day 1), and changes geographic location from survey to survey; C2 refers to one of the four cameras that the reel came from; and 12 refers to 2012, the year the survey was flown. Sequence 10-15-43.796 refers to the specific video reel and is named for the time that the survey plane started on that transect line.

### *Excel spreadsheets*

Each camera for each day has an associated spreadsheet for analysis. HiDef completes several fields for each identified object: Location (Zone), Date of flight, Camera Number, Resolution, Reel Name, Observer, Time, Frame number, Category of object, and Marker Number[2] (Appendix 4B, Table 4B-1). The spreadsheet is protected to prevent changes being made to these columns that would affect later processing. Columns in (Appendix 4B) are filled out by the review team. Spreadsheets are stored on the BRI server in their respective year, survey, month, day and camera folders and can be accessed by the entire review team.

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[2] For March through October of 2012, the marker number for each object identified within the frame was added manually by BRI reviewers. From December 2012 onwards, HiDef included the marker numbers for all objects in the data spreadsheets, eliminating the need for manual entry.

### *Selecting a reel*

The object marker files are named to match reel names in video sequences. If a video sequence is unable to open, it is possible to repair the corrupted reel on-site using a proprietary module that re-builds headers for each file. Review of video sequences is recorded in spreadsheets associated with each day and camera; reviewers track who is reviewing which reels on the video review room white board.

### *Viewing a frame*

Each object has a marker number and a frame number associated with it. The frame number refers to the frame that the HiDef reviewer has marked as containing an object for review (e.g., the frame in which the object is closest to the red center line that bisects the camera field of view; Appendix 4C). The start or end of a reel or a section of footage containing atmospheric interference (e.g., clouds) may also be assigned a frame number. Some frames may have zero markers or a single marker identified, while others will have multiple markers, if there are numerous animals in the frame (Appendix 4C). The frame number from the spreadsheet is equivalent to the number in the file header information located at the top of the proprietary video processing software.

### *Identifying a marked target*

For each frame that features a target marked by the HiDef review team, a BRI reviewer enters the frame number into the proprietary video processing software to view the object. The reviewer closely examines each target for features (size, shape, color, behavior, flight pattern) that will allow for identification to species according to defined criteria (Appendix 4E). If it cannot be identified to species, the object is categorized to a higher taxon level or a broad category (e.g., “UNKN; Unknown”). The aim is to identify targets to the lowest taxonomic level possible, with accuracy. Reviewers move through all of the frames in which that object is recorded to get a sense of the target’s movement, and to find the clearest images for review. Using proprietary video processing software, reviewers can adjust the image brightness and other qualities to create a clearer image of the object being identified, or to pick up lighter or darker colors that may be obscured (Appendix 4C).

### *Data fields completed by BRI*

Fields L-AA in the data spreadsheet (Appendix 4B, Table 4B-2) are filled out for each marked object. When a reviewer finishes reviewing the data for a spreadsheet, he or she checks their spreadsheet with a QA/QC checklist and makes required edits (see Appendix 4G).

#### *Marker number*

If an object is missed at the review stage and crosses the red line, a marker number is added to the spreadsheet and highlighted in bright yellow with a frame number filled in to the “Added Frame Number” column. A new marker number is added to the screen by clicking on the object and selecting “Ok” (Appendix 4C). Marker numbers are generated automatically by the proprietary video processing software and go up sequentially. The marker number is added to the marker number column in the spreadsheet.

### *ID category*

Objects are identified to species, if possible, based on the animal's size, coloring, movement, general shape, and movement/flight pattern. Options include bird, mammal, shark, ray, fish, and turtle species, as well as algae and abiotic objects (a complete listing of codes employed through May 2014 is included in Appendix 4D). Species group codes may also be used; these correspond to groups of species that may be difficult to differentiate. For example, the "SMTU" code ("small turtle") includes green, Kemp's ridley, hawksbill, and loggerhead sea turtle species, and is used in cases where more definitive species identifications are deemed to be impossible. Other group codes include:

> CESS; Cetacean/Seal/Shark - Animal is too obscured to tell if it is a cetacean, a seal, a shark, or a large fish
>
> UNBI; Unidentified Bird – Object is a bird but no further taxonomic distinctions can be made

In addition, non-object codes can be used for marked objects that cannot be placed in a biotic category:

> Nothing; Nothing – Something has been marked as an object, but there is nothing there. This is also used when a wave or feces is marked as an object.
>
> ERRO; Error – This is used to identify objects in an inoperable or damaged reel that is unable to be repaired. This is also used to identify objects on land.

### *Species confidence*

Each target identified has a confidence level associated with it (Table 4-1). All objects must have an associated confidence. For non-species based identifications (e.g., "DUPL; Duplicate," "NA; Not Applicable"), "Definite" is used as the confidence.

### *Behavior*

When a target is identified as an animal, the general behavior of the target is described using the options in the drop-down menu (Appendix 4B, Table 4B-2). Some categories of behavior refer specifically to avian or bat targets (sitting, flying, taking off), while others refer to aquatic animals (stationary, moving). Direction of the animal's movement is indicated when applicable.

### *Flying at sea level*

This designation is used for targets identified as birds or bats that are flying. Reviewers consider whether or not there is evidence that the animal in flight is flying close to the ocean surface. Splashing may indicate the bird has just taken off, or a shadow close to the target object may indicate it is low over the water (Appendix 4C).

### *Submerged*

Reviewers note whether the animal is submerged or surfaces within the recorded frames. This designator is only used for aquatic animals.

### *Approximate age*

If possible, reviewers note the approximate age of the animal based on measurements of size (mammals, turtles, rays) or plumage (birds).

### *Plumage*

Any details about plumage are noted in this text field. Options include gannet and fulmar plumages (see Appendix 4B, Table 4B-2).

### *Molt*

Molt stages are noted for birds if possible (see Appendix 4B, Table 4B-2).

### *Probable sex*

Probable sex is noted where possible. There are many species that cannot be identified to sex, so this is only marked when the reviewer is able to determine sex easily (e.g., scoters).

### *Measurements*

If an on-screen measurement of an object is taken during the ID process using the proprietary point-to-point caliper module, the measurement is recorded in the Measurements[3] column in centimeters (Appendix 4C). The types of measurements included in this field (Appendix 4B, Table 4B-2) are listed in Table 4-2; all other measurement types (i.e., sitting birds, caudal fin measurements, partial measurements) are placed in the comments field.

### *Outside zone*

Animals are occasionally marked that do not cross the red line. When that happens, reviewers indicate this by choosing “Yes” here.

### *Flag*

Flags are used to mark an animal that reviewers want to revisit for any reason. The reason for flagging must be noted in the comments.

### *Added frame*

When an unmarked object is found, this is where the position of the marked object (frame number) is noted.

### *Comments*

Reviewers fill out comments on the object when necessary. Reel names of any missing objects are included here. In addition, if changes are made to the data after they are sent back to HiDef for parallax and georeferencing (for example, as the result of an audit arbitration; see QA/QC Review Protocol below), reviewers use the following wording in the comments: Post-parallax edits-MM/DD/YYYY and any other comments associated with the post-parallax change along with their initials. In addition, reviewers change the ID Category fill color to dark green.

---

[3] For March 2012, the measurement tool was not available and objects were measured using a ruler. Prior to December 2012, measurements did not follow the definitions found in Table 4-2.

*Identification date*
Reviewers enter the date of identification for every line of data.

*Identifier*
Reviewers enter their initials here for every line of data at the time it is completed.

### Completion of data analysis

Following the data analysis outlined above, as well as the data collection and data completion QA/QC procedures outlined in the QA/QC Review Protocol below, BRI sends data spreadsheets to the HiDef head office in the United Kingdom. The UK office georeferences all frames with target objects, and estimates the approximate flight height of flying objects using a proprietary parallax technique (Hatch et al., 2013). The columns produced through this process are Latitude, Longitude, Flight Height, Flight Height Confidence, and a modified field for Behavior that includes cardinal direction of movement where applicable (Appendix 4B, Table 4B-3). While HiDef is completing these data analyses for the survey, BRI concurrently begins regular and Threatened and Endangered Species audit procedures as outlined in the QA/QC Review protocol below.

## Quality assurance and quality control of data collection

### Goals

1. Data are consistent, accurate, valid, and repeatable
2. Problem areas and successes are identified, addressed, documented, and reported
3. ID criteria and SOPs are up to date and applied consistently by each reviewer
4. Exceptional data quality is maintained for:
    a. Basic analysis/summary reports
    b. Statistical modeling
    c. Synchronization with current or similar datasets
    d. Collaborator analysis needs

### Filename conventions

13_M09_Audit_JGO

13_M09_Arbitration_JGO

In the examples above, "13" represents the year in which the survey was flown, "M09" represents the month in which the survey was flown. "Audit" or "Arbitration" is the task performed. "JGO" represents the initials of the auditor or arbitrator.

### Data collection QA/QC

To ensure consistency during data collection and the accuracy of data entry, spreadsheet formatting is locked and drop-down menus are used in fields with analyzable data. In addition, drop-down menus are extracted from a master code database, which is updated with definitions and codes on a regular basis (Appendix 4D). Ancillary data within the reviewer spreadsheet, such as measurements or comments, are recorded as text. To ensure repeatability and consistency, all reviewers refer to the same reference documents, such as measurement charts, seasonal distribution maps, and a "Confidence and Identification Criteria" document (Appendix 4E), which is based on a hierarchal matrix (Appendix 4F) that was developed from biota previously encountered on aerial and boat-based surveys conducted in the study area, as well as taxonomic pairings or groupings developed during HiDef's previous projects in Europe. In order for an object to be called a 'definite Dovekie', then all of the criteria for 'definite' and 'Dovekie' need to be met. Otherwise, it is either downgraded to a lower confidence level or a higher taxonomic grouping, such as 'Unidentified Alcid'.

### Data completion QA/QC

Reviewers check for common data errors using a checklist (Appendix 4G). After target identification is complete, the QA/QC manager compiles the data by month and double-checks for errors, such as those listed in Appendix 4G. Errors are corrected by the original identifier, if available, and any corrections to the data by the team leader or QA/QC manager are noted in the comments field in the original spreadsheet.

### Blind audit re-identifications

Following completion of the above steps for each survey's data, the compiled data are filtered for objects originally characterized by Hi-Def reviewers as biota. Buoys, boats, and reel locations do not qualify for audit. Twenty percent of the remaining objects are eligible for audit and this number is noted. In order to maximize the audit effort, and to reduce audit technical error, other objects are exempt from the audit, such as duplicate objects, outside zone objects, and objects that could not be identified due to reel or marker number errors. Next, a formula is used to assign a random number to all eligible objects. Once those numbers are generated, the spreadsheet is sorted in numerical order by the random-generated number. The top 20% are chosen and pasted into a new tab. The original compiled spreadsheet is sorted for threatened and endangered (T&E) species and any T&E objects that did not get chosen for the random-generated audit are also added to the random audit. The objects are assigned a second random number and sorted in ascending order by the random-generated number. The spreadsheet is filtered by each original reviewer and those objects are evenly distributed to other BRI reviewers. All original answers are removed, new fields for audit identification are added, and a new "blind" spreadsheet is generated for each auditor. Auditors follow the same identification protocol as for the target ID process above, and their identifications are compared to the original identifications to determine how often the first and second reviewers agree. A "pass" grade occurs when auditors agree with ≥90% of original reviewers' data in the random audit, and 100% for the T&E audit.

### Audit analysis: randomly chosen objects

Assessment of audit agreement rates is conducted via MS Access using pre-determined answer agreements (Appendix 4H; these are based on the 'Confidence and Identification Criteria' document and rules that apply to all biotic objects). The rules are (see Table 4-3 for examples):

1. Specific species identifications are considered to equal the next available higher taxonomic grouping as long as the next available grouping is not "UNBI; Unidentified Bird", "ID Impossible; ID Impossible", or "CESS; Cetacean/Seal/Shark". In those instances, the species can only equal itself. For example, some species such as "NOGA; Northern Gannet" and "REBA; Red Bat" are

singly defined in the audit answer status (see Appendix 4H) because they have no known similar species in the study area at this time. Fully defined species such as “BODO; Bottlenose Dolphin” can pass as an “SBCE; Small beaked Cetacean to 3m” in an audit.

2. Higher taxonomic grouping identifications are considered equal to the next lowest taxonomic level as well as next highest taxonomic grouping. For example, SBCE; Small beaked Cetacean to 3m can equal CODO; Common Dolphin or BODO; Bottlenose Dolphin, and it can also be considered to equal UNDO; Unidentified Dolphin. However, it would not be a match to an even broader taxonomic category such as UNCE; Unidentified Cetacean.
3. Biota will not equal abiota.
4. For higher flying birds, such as gulls and terns, measurements may have overlapping measurement error values, which can further be exacerbated by unknown flight height of the bird at the time of identification. Therefore, groupings with size designations can cross sizes in the audit agreement rules and also equal the next available higher taxonomic grouping. For example, UNMT; Unidentified Medium Tern: 32-45 cm can equal UNLT; Unidentified large Tern as well as UNTE; Unidentified Tern.
5. Except for sea turtles, which all have a T&E status, T&E species can only equal themselves.

Some ID Categories are not yet fully defined and, therefore, are more likely to change in audit answer composition or be more flexible with the audit answers. This mostly occurs with non-avian biota, such as sharks and cetaceans, where reviewers may be less certain of what species to expect, or whether there are enough ID criteria available in video footage to discern between higher and lower taxonomic groupings (see Appendix 4H).

If there is at least 90% agreement, then the audit is passed for that survey and no further analysis is needed. If the overall audit is in <90% agreement (meaning that for 100 objects, there was disagreement between the first and second reviewer on >10 objects), then biotic taxonomic groups that represent ≥20% overall object composition within the survey but have <90% agreement are discussed by the team to determine better methods for identification (see Table 4-4 for an example). After clarifying ID criteria and revising the ‘Confidence and Identification Criteria’ document, all objects from those taxonomic groups are re-reviewed in the original data. After repeating a review of those target taxa, 20% of the target taxa that were not in the original audit are audited. If 90% agreement is achieved in this second audit, no further analysis is needed. If audit disagreement continues, the taxonomic grouping goes into arbitration, whereby the object(s) in question are independently reviewed again by the entire team and a final answer is determined based on those results. Mismatches from taxa that do not represent ≥20% overall object composition within the survey are team-reviewed in order to improve identification methods and criteria.

### *Audit analysis: T&E species*

All objects that are identified as state- and federally-listed species are included in the audit alongside randomly chosen objects[4]. Audit agreement must be 100% for T&E species and in most cases, the T&E

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[4] The T&E audit was conducted separately from the random audit for the first five surveys, but became integrated with rest of the audit beginning with December 2012.

species must match exactly. Since all sea turtles are federally listed, audit agreement allows for a specific species of turtle to match SMTU; small turtle and vice versa. However, a specific species cannot match another species in the turtle grouping. All mismatches of T&E species automatically go to arbitration.

### *Arbitration*

Arbitration occurs when there is less than 20% agreement on biotic objects during the randomly chosen object audit and <100% agreement on T&E objects. Essentially, the objects in question are independently reviewed again by the entire team and final answers are determined based on those results.

Each arbitrator receives a new spreadsheet with all audit mismatches and each object is reviewed on the video footage again. Each object mismatch is reviewed by the original reviewer, original auditor, a new informed reviewer, and a new uninformed reviewer. For those objects where the arbitrator role is original reviewer, original auditor or informed reviewer, the arbitrator can view the original reviewer's identification category and comments, as well as the original auditor's identification category and comments. The arbitrators review the video footage, assess the original reviewer and auditor answers, and either choose one of those answers or an entirely new answer. For the uninformed arbitrator, the original reviewer's identification category and comments as well as the original auditor's identification category and comments are omitted.

Once the arbitration spreadsheets are complete, the final answers are determined by the level of Identification Category agreement. If the majority of the arbitrators chose the same Identification Category, then this is the final answer. If there is no majority agreement, then the Identification Category with the highest taxonomic value is chosen. If needed, corrections are then made to the original reviewer spreadsheet.

## Literature cited

Hatch, S.K., Connelly, E.E., Divoll, T.J., Stenhouse, I.J., Williams, K.A., 2013. Offshore observations of eastern red bats (Lasiurus borealis) in the mid-Atlantic United States using multiple survey methods. PLoS One 8, 1–8. doi:10.1371/journal.pone.0083803

## Figures and tables

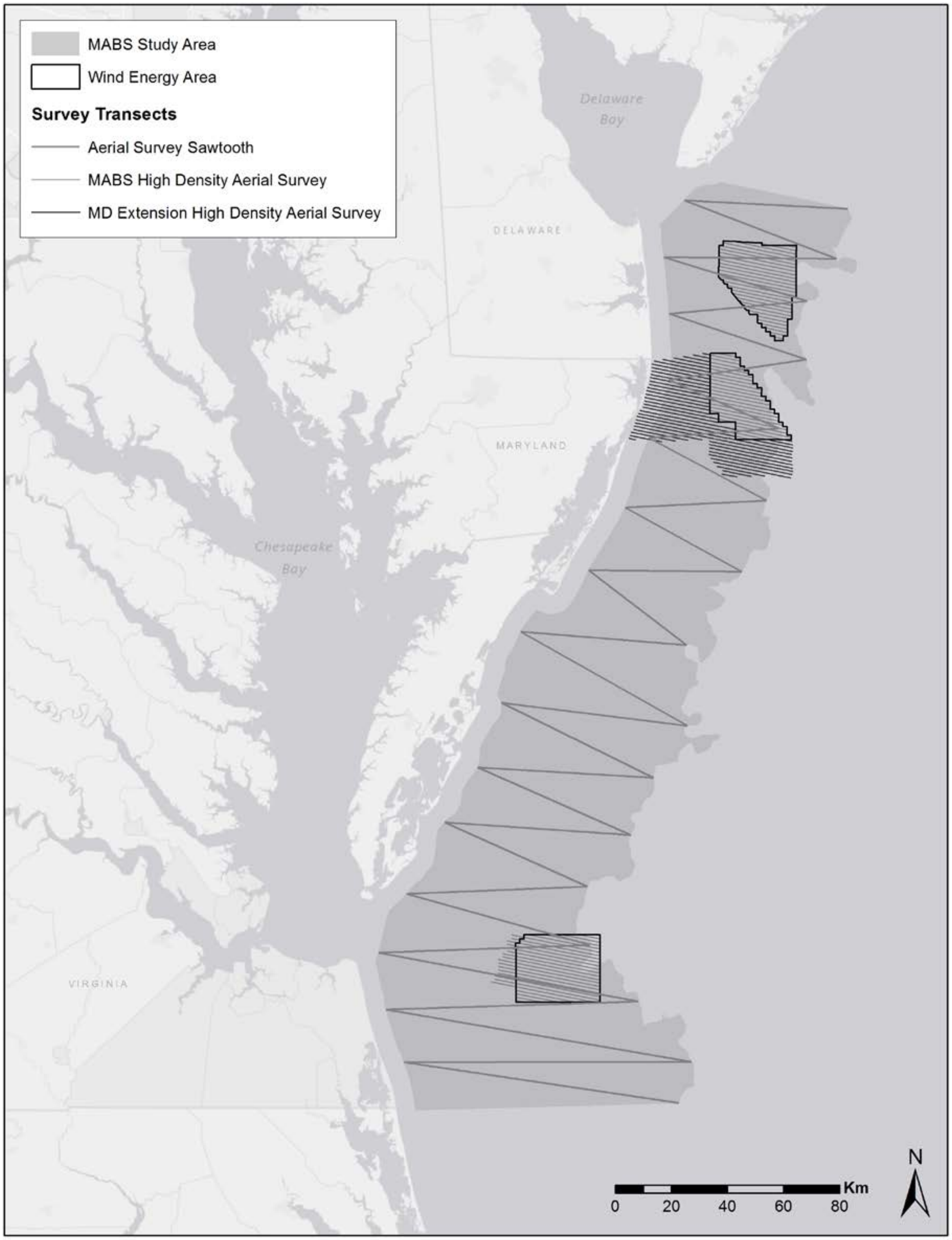

**Figure 4-1. Map of digital video aerial survey transects for the Mid-Atlantic Baseline Studies and Maryland Projects.** Mid-Atlantic Baseline Studies transects are shown in light gray. High-density Maryland extension transects are shown in dark gray.

**Table 4-1. Species identification confidence levels.**

| Option | Definition |
|---|---|
| Possible | less than 50% certain |
| Probable | greater than 50%, but less than 95% certain |
| Definite | greater than 95% certain |

**Table 4-2. Measurement definitions for various taxonomic groups.**

| Taxon Group | Measurement Format | Example |
|---|---|---|
| Birds in flight | Length (bill tip to tail tip) x Wingspan (wing tip to wing tip) | 60 x 114 |
| Bats in flight | Length (tip of head to tail tip) x Wingspan (wing tip to wing tip) | 15 x 40 |
| Sharks, Fish | Length (snout tip to caudal fin tip) | 157 |
| Cetaceans | Length (upper jaw tip to fluke notch) | 225 |
| Seals | Length (nose tip to tail tip) | 190 |
| Rays | Disc Width (pectoral fin tip to pectoral fin tip at the widest part) | 90 |
| Turtles | Straight Carapace Length (carapace top to carapace bottom at the midline) | 84 |

**Table 4-3. Excerpt from an audit showing examples of audit agreement and disagreement.** The complete rules are contained in Audit analysis: randomly chosen objects.

| Reviewer ID Category | Auditor ID Category | Audit Match? |
|---|---|---|
| CATE;Caspian Tern | UNTE;Unidentified Tern | Yes, rule 1 |
| GRSH;Greater Shearwater | UNBI;Unidentified Bird | No, rule 1 |
| TSMG;Tern/Small or Medium Gull | UNTE;Unidentified Tern | Yes, rule 2 |
| UNBI;Unidentified Bird | UNKN;unknown | No, rule 3 |
| UNMT;Medium Tern: 32-45 cm | UNLT; Unidentified large Tern | Yes, rule 4 |
| UNMT;Medium Tern: 32-45 cm | UNBI;Unidentified Bird | No, rule 2 |
| BODO;Bottlenose Dolphin | SBCE;Small beaked Cetacean to 3 m | Yes, rule 1 |
| BODO;Bottlenose Dolphin | CESS;Cetacean/Seal/Shark | No, rule 1 |
| COWR;Cownose Ray | UNRA;Unidentified ray | Yes, rule 1 |
| COWR;Cownose Ray | CESS;Cetacean/Seal/Shark | No, rule 1 |
| KRST;Kemp's Ridley Sea Turtle | SMTU;Small turtle | Yes, rule 5 |
| SMTU;Small turtle | LOTU;Loggerhead Turtle | Yes, rule 5 |
| SMTU;Small turtle | UNKN;unknown | No, rule 3&5 |
| SCHA;Scalloped Hammerhead | HASH;Hammerhead shark | Yes, rule 1 |

**Table 4-4. Example of disagreement in audit results.** Overall agreement from the example audit below is 80%. Loons represent ≥20% overall object composition and received <90% agreement, resulting in re-review of all loon objects (n=548) and subsequent 20% re-audit of loons not in the original audit.

| Taxonomic Grouping | n | Overall Object Composition | # Mismatches | # Matches | Total | % Agreement | Consequences |
|---|---|---|---|---|---|---|---|
| Egrets and Herons | 2 | 0% | 0 | 1 | 1 | 100% | None |
| Fish and Sharks | 209 | 11% | 5 | 42 | 47 | 89% | Team review of mismatches |
| Gannets (Sulidae) | 71 | 4% | 3 | 8 | 11 | 73% | Team review of mismatches |
| Gulls and Terns (Laridae) | 341 | 18% | 17 | 54 | 71 | 76% | Team review of mismatches |
| Jaegers and Skuas (Stercorariidae) | 4 | 0% | 0 | 0 | 0 | NA | Not applicable |
| Jellyfish (Cnidaria) | 1 | 0% | 0 | 0 | 0 | NA | Not applicable |
| **Loons (Gaviidae)** | **548** | **29%** | **31** | **76** | **107** | **71%** | **Re-review and re-audit** |
| Other Biota | 26 | 1% | 0 | 0 | 0 | NA | Not applicable |
| Pelicans (Pelicanidae) | 3 | 0% | 1 | 0 | 1 | 0% | Team review of mismatches |
| Rays (Batoidea) | 1 | 0% | 0 | 0 | 0 | NA | Not applicable |
| Scoters, Ducks, Geese (Anatidae) | 1 | 0% | 0 | 0 | 0 | NA | Not applicable |
| Toothed Whales (Odontoceti) | 200 | 11% | 5 | 34 | 39 | 87% | Team review of mismatches |
| **Turtles (Testudines)** | **293** | **16%** | **3** | **56** | **59** | **95%** | **Arbitration** |
| Unidentified Birds (Aves spp.) | 152 | 8% | 10 | 20 | 30 | 67% | Team review of mismatches |
| Unidentified Marine Mammal or | 20 | 1% | 1 | 2 | 3 | 67% | Team review of mismatches |
| Unidentified Whale (Cetacea) | 2 | 0% | 1 | 0 | 1 | 0% | Team review of mismatches |
| **Grand Total** | **1874** | 100% | **77** | **299** | **376** | **80%** | |

## Supplementary material

### Appendix 4A. Glossary

Audit – Inspection of data conducted by reviewers after each major step of the data analysis process. A minimum of 20% of the data from each survey month is audited by a second observer, and objects on which the reviewers disagree may be re-reviewed in an arbitration process (the exact process varies between the marking audit and identification audit; for details on the identification audit process, see the Target Identification Protocol). The selection of data for regular audits is random. Threatened and Endangered Species audits (in which all species initially identified as a listed species of concern at the state or federal level are reviewed by a second observer) are comprehensive, and include 100% of these identified species for each audit.

BRI – Biodiversity Research Institute, the nonprofit research organization based in Maine that is overseeing the Mid-Atlantic Baseline Studies Project (www.briloon.org).

Frame – individual image within a video reel. There are roughly 20,000 frames per reel. Frames are recorded at a rate of approximately one every 0.06 seconds of survey under normal circumstances.

GSD – ground sample distance, affects image resolution.

HiDef – HiDef Aerial Surveying Ltd., the organization based in the United Kingdom that developed the high resolution video camera system and captures and processes high resolution digital aerial video.

Marker number – number assigned by HiDef reviewers as a unique identifier for individual objects. This is recorded in the spreadsheet automatically during HiDef processing.

Maryland Project – Extension to the Mid-Atlantic Baseline Studies Project funded by the Maryland Department of Natural Resources. Expanded high density survey coverage south and west of the Maryland WEA, including into Maryland state waters. These surveys were flown March 2013-May 2014 in conjunction with the MABS surveys. One additional survey of the Maryland Project study area and the Maryland WEA occurred in August 2013 as a part of the extension project.

Mid-Atlantic Baseline Studies Project (MABS) – three-year (2012-2015) project funded by the Department of Energy. The project includes boat and digital video aerial surveys of animals in the mid-Atlantic outer continental shelf, among other studies (www.briloon.org/mabs).

Parallax – the apparent motion of an elevated object against a distant background due to the movement of the observer (used to estimate flight height).

QA/QC – quality assurance and quality control.

Red line – midline of the video footage, and over which an object must cross to be included within the survey area - this red line represents 50 meters wide for 2 cm GSD, 75 meters wide for 3 cm GSD.

Reel – continuous stream of video footage. ID# for a reel is the exact (GPS) start time. One camera records one reel along one transect.

SOP – standard operating procedure.

T&E – threatened and endangered species.

Transect – line flown by aircraft during surveys. There are 152 individually numbered transects under the current survey design for the Mid-Atlantic Baseline Studies Project and Maryland Project (as of March 2013).

Video sequence – sequence of video collected by HiDef, split into individual reels.

WEA – federally designated Wind Energy Area, or geographic region that the Bureau of Ocean Energy Management has identified as an area for potentially expedited permitting of offshore wind facilities.

## Appendix 4B. Quick Guide to Video Identification Spreadsheet Fields

**Table 4B-1. The fields completed by HiDef Review Team for every object identified.** *Required information for all records.

| Field | Description | *Example* |
|---|---|---|
| [redacted] | Zone surveyed. | *Zone 19* |
| [redacted] | Date of survey – mm/dd/yyyy. | *03/26/2012* |
| [redacted] | Number assigned to each camera in an array. | *1* |
| [redacted] | Ground sample distance in cm | *2cm* |
| [redacted] | Local time at start of reel. Noted as hours-minutes-decimal seconds in 24 hour time. | *11-36-07.796* |
| [redacted] | Initials of the HiDef reviewer. | *DC* |
| [redacted] | [redacted] | [redacted] |
| [redacted] | Frame number in which an object is marked. Frames are numbered sequentially at the beginning of each reel. | *159* |
| [redacted] | General category describing observation. Start and end of reels are also noted in this field. | *bird* |
| [redacted] | Number of the marker on the object to be identified. | *23* |

**Table 4B-2. The fields completed by BRI Review Team for relevant objects identified.** Unused fields (for non-required information) are left blank. *Required information for all records.

| Field | Description | Field Type | Drop-down Options or Text examples |
|---|---|---|---|
| **ID Category** | Code for ID of object. | Drop-down | See Appendix D |
| **Confidence** | Degree of certainty. | Drop-down | Definite, Probable, Possible |
| **Behaviour** | General behavior of identified animals. Include direction of animal's movement in relation to camera applicable. | Drop-down | Sitting, Sitting on object, Loafing, Taking Off, Feeding, Following Vessel, Flying (Direction Unknown), Flying up (etc.), Stationary, Moving left (etc.), Haul-out (pinnipeds) |
| **Flying at Sea Level** | Splashing or shadow at ocean surface. | Drop-down | Yes, No |
| **Submerged** | Under or at water's surface. | Drop-down | Submerged, Surfacing |

| Field | Description | Field Type | Drop-down Options or Text examples |
|---|---|---|---|
| **Approximate Age** | Adult= animals with adult plumage or mature body size; Immature= animals >1 year old that have not achieved adult plumage or full body size; Juvenile= young of the year, Hatch Year (HY) birds or any animal with known age <1 year. | Drop-down | Adult, Immature, Juvenile |
| **Plumage** | Gannet or Northern Fulmar plumages. | Drop-down | Light Phase, Dark Phase, Intermediate Phase, Gannet Plumage 1 – 6, Unknown |
| **Molt** | Bird molt stage. | Drop-down | Summer, Winter, Transitional, Primary Molt, No Primary Molt, Unknown |
| **Probable Sex** | Select appropriate option from list. | Drop-down | Male, Female |
| **Measurements** | Estimated length or wingspan, in cm. | Number | 105 |
| **Outside Zone** | Mark if object does not cross line. | Drop-down | Yes or blank |
| **Flag** | Entry marked for later examination. | Drop-down | Yes or blank |
| **Added Frame Number** | Frame number where missed object was marked. | Number | 485 |
| **Comments** | Other notable features, description of what you have seen, clarity of camera/frames. | Text | Too blurry to ID to species |
| **Identification Date** | Date of review – mm/dd/yyyy. | Number | 5/29/2013 |
| **Identifier** | Initials of the BRI reviewer. | Text | EC |

**Table 4B-3. Spreadsheet compiled by HiDef analysts in the parallax and georeferencing process.**

| Field | Description | *Example* |
|---|---|---|
| **Behaviour** | General behavior of identified animals. Direction of animal's movement is translated from the movement in relation to viewing screen (up, down, left, right) to cardinal direction when applicable. | Flying SE |
| **Flight Height** | Range of possible flight heights in meters for eligible objects. | 0 - 20 |
| **Flight Height Confidence** | Confidence of the flight height calculation. | 100% |
| **Latitude** | Latitude of the frame number or “play pos” in decimal degrees. | 36.93328 |
| **Longitude** | Longitude of the frame number or “play pos” in decimal degrees. | -75.56408 |

## Appendix 4C. Data Analysis Methods: example images from the proprietary video processing software.

**Figure 4C-1. Footage of a Northern Gannet (*Morus bassanus)* in flight.** The yellow circle with “65” inside (not visible) is the marker with a marker number. Animals are marked when they are close to the red midline, as in this picture.

**Figure 4C-2. Gain can be adjusted to help pick up different features on the object for identification.** Here, gain was increased from the base image in **Figure 4C-1**, causing the white on this bird to stand out.

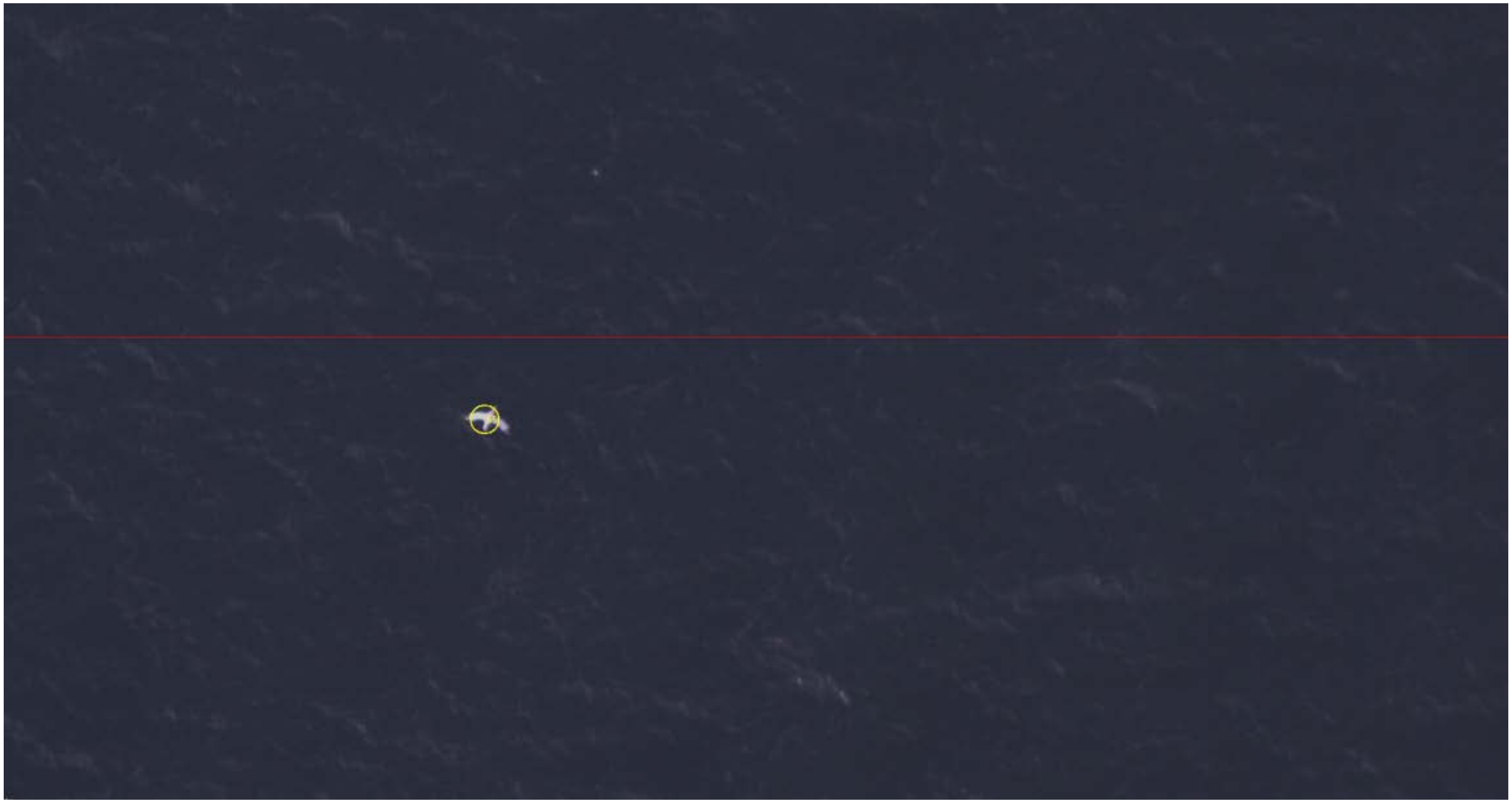

**Figure 4C-3. Adjusting the Gain and Gamma can give greater overall contrast especially when viewing flying objects vs. submerged objects.** The Gain adjusts the brightness of highlights or whites, while gamma can be adjusted to deal with the brightness of mid-tones. In this image the Gain has been lowered compared to **Figure 4C-2** and the Gamma has been decreased. The adjustments allow for the yellow coloration on the head and the black wing tips to stand out.

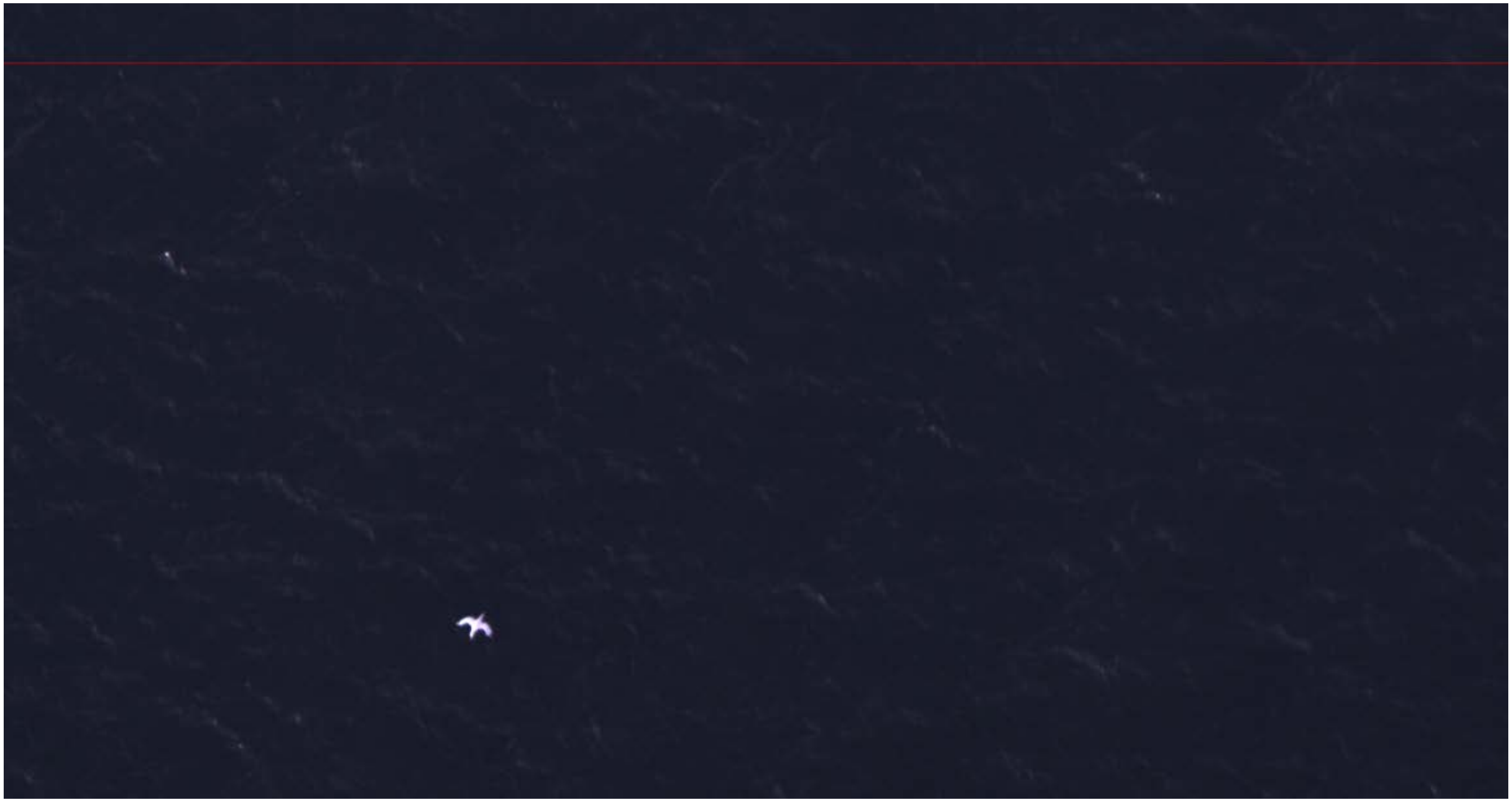

**Figure 4C-4. Moving the footage backward and forward from the frame in which the object is marked can allow reviewers to examine animal movements such as wing flapping, diving, or turning a head.** In this image the reviewer has reversed to an earlier frame from the one shown in **Figure 4C-1**.

**Figure 4C-5. It is important for reviewers to move through each frame when making identifications as some portions of the screen can be blurry.** These images of a Cownose Ray (*Rhinoptera bonasus*) can show how one image (left) can be clear, while the subsequent frame (right) can be blurry.

**Figure 4C-6. Black Scoters flying with shadows visible.** Each scoter in this image would be counted as flying at sea level.

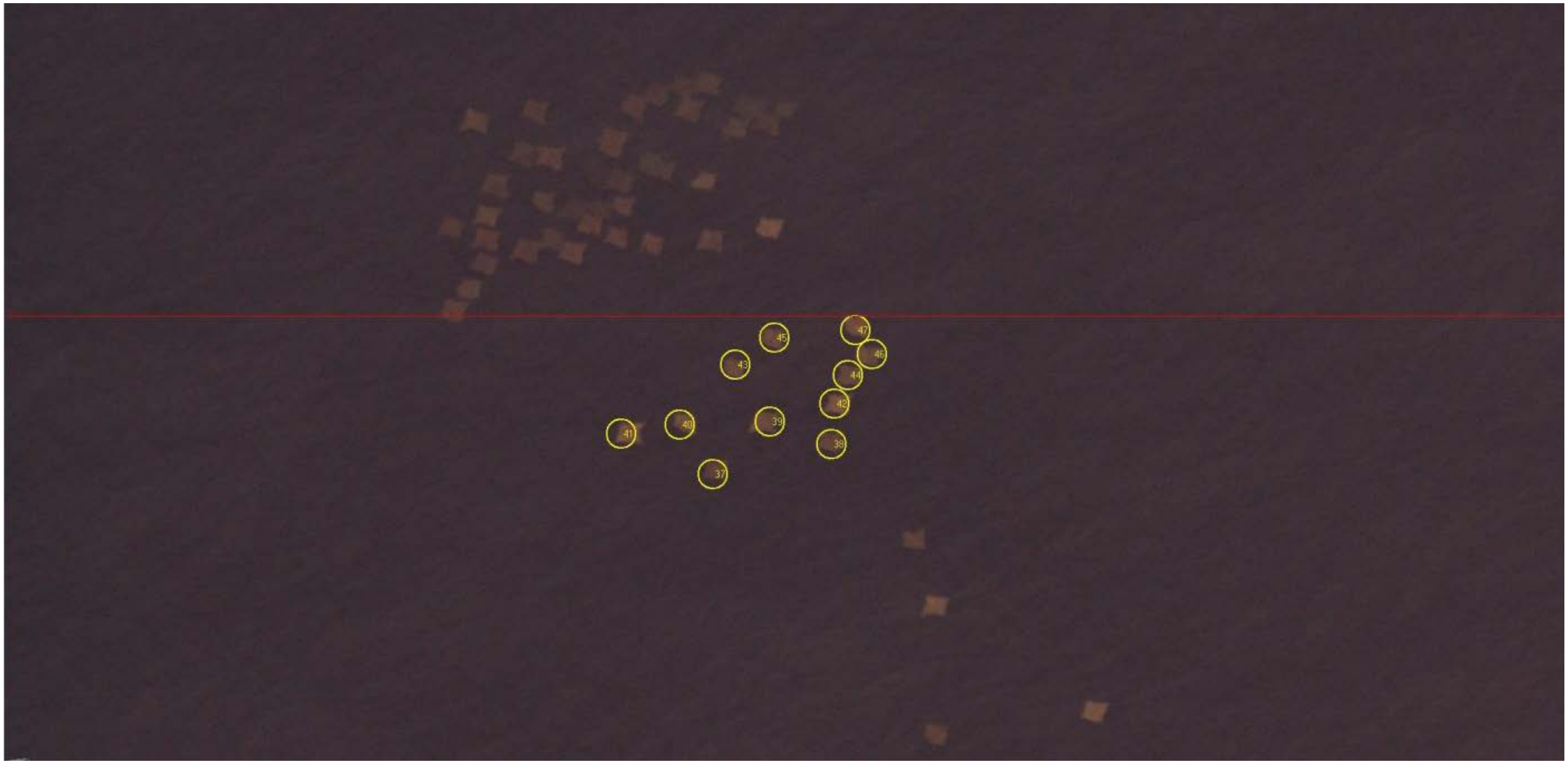

**Figure 4C-7. Cownose Rays in a large school. Individuals that are close to the red midline are marked while those that have passed or are approaching are marked in a different frame.** This helps with data processing and identifying individual animals.

**Figure 4C-8. Measurement of a Cownose Ray.** Cownose Rays are measured from their widest point fin-to-fin, so it is important to choose a frame where you can see both fin tips. Note that this is a screen capture of the same rays as **Figure 4C-7,** but a few frames before it, so a different group of rays is marked by the marker numbers.

## Appendix 4D. Identification Categories Used in Data Analysis

**Table 4D-1. Identification (ID) categories used for aerial analysis.** New categories or species are added as the need arises. All species codes used in the species included column can be found in the ID Category column. Bird species codes largely adhere to AOU (American Ornithologists' Union) four-letter alpha codes.

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| AKSH;Auk or Shearwater | Includes ATPU,AUSH,BLGU,COMU,DOVE,MASH,RAZO,TBMU | Yes |
| AMBI;American Bittern | | Yes |
| ARTE;Arctic Tern | | No |
| ASDO;Atlantic Spotted Dolphin | | No |
| ATPU;Atlantic Puffin | | Yes |
| AUSH;Audubon's Shearwater | | No |
| BAEA;Bald Eagle | | Yes |
| BAIT;bait ball | | Yes |
| BALN;balloon | | Yes |
| BAOR;Baltimore Oriole | | Yes |
| BARS;Barn Swallow | | Yes |
| BASH;Basking Shark | | Yes |
| BBWH;Blainville's Beaked Whale | | No |
| BCPE;Black-capped Petrel | | No |
| BEKI;Belted Kingfisher | | Yes |
| BLGU;Black Guillemot | | No |
| BLKI;Black-legged Kittiwake | | No |
| BLSC;Black Scoter | | Yes |
| BLSH;Blue Shark | | No |
| BLTE;Black Tern | | Yes |
| BLVU;Black Vulture | | Yes |
| BLWH;Blue Whale | | No |
| BOAT;Boat--unidentified | | Yes |
| BOBA;boat--barge/barge and tug | | No |
| BOCA;Boat--cargo | | Yes |
| BOCF;Boat--commercial fishing | | Yes |
| BOCG;Boat--Coast Guard | | Yes |
| BOCR;boat--cruise | | Yes |
| BOCS;boat--container ship | | Yes |
| BODO;Bottlenose Dolphin | | Yes |
| BOFE;boat--ferry | | No |
| BOFI;boat--fishing | | Yes |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| BOGU;Bonaparte's Gull | | Yes |
| BOLO;boat--lobster | | No |
| BOME;boat--merchant | | No |
| BOPL;boat--pleasure | | Yes |
| BOPS;boat--purseiner | | No |
| BORF;boat--recreational fishing | | Yes |
| BORV;boat--research vessel | | Yes |
| BOSA;boat--sail | | Yes |
| BOTA;boat--tanker | | Yes |
| BOTD;boat--trawler/dragger | | No |
| BOTU;boat--tug | | Yes |
| BOWW;boat--whale watch | | No |
| BOYA;boat--yacht | | Yes |
| BRAN;Brant | | Yes |
| BRBO;Brown Booby | | No |
| BRDO;Bridled Dolphin? | Members of genus Stenella. Includes ASDO,CLDO,LSSD,PSDO,STDO | No |
| BRPE;Brown Pelican | | Yes |
| BRTE;Bridled Tern | | No |
| BRWH;Bryde's Whale | | No |
| BUFF;Bufflehead | | No |
| BUOY;Buoy | | Yes |
| CANG;Canada Goose | | No |
| CASW;Cave Swallow | | No |
| CATE;Caspian Tern | | Yes |
| CBWH;Cuvier's Beaked Whale | | No |
| CEDW;Cedar Waxwing | | Yes |
| CESS;Cetacean/Seal/Shark | Includes all cetaceans/seals/sharks and fish. | Yes |
| CLDO;Clymene Dolphin | | No |
| CODO;Common Dolphin | Short-beaked or Long-beaked Common Dolphin. Short-beaked is the more likely common dolphin in the Mid-Atlantic target area. | Yes |
| COLO;Common Loon | | Yes |
| COME;Common Merganser | | No |
| COMU;Common Murre | | No |
| CONI;Common Nighthawk | | Yes |
| COSH;Cory's Shearwater | | Yes |
| COTE;Common Tern | | Yes |
| COWR;Cownose Ray | | Yes |
| DBSH;Dark-backed Shearwater | Includes AUSH,MASH,SOSH | No |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| DCCO;Double-crested Cormorant | | Yes |
| DOVE;Dovekie | | Yes |
| DOWI;Dowitcher spp. | Short-billed or Long-billed Dowitcher | Yes |
| DUPL;Duplicate | The same object marked in two different frames | Yes |
| ERRO;error | | Yes |
| FIGE;fishing gear | | Yes |
| FISH;Unidentified fish | All "Fish" | Yes |
| FISS;Unidentified fish school | All "Fish" schools | Yes |
| FIWH;Fin Whale | | Yes |
| FKWH;False Killer Whale | | No |
| FLJE;flotsam and jetsam | | Yes |
| FOTE;Forster's Tern | | No |
| FUMG;Fulmar or Medium Gull | Includes BLKI,LAGU,NOFU,RBGU,SAGU | Yes |
| GBBG;Great Black-backed Gull | | Yes |
| GBHE;Great Blue Heron | | Yes |
| GBWH;Gervais' Beaked Whale | | No |
| GLGU;Glaucous Gull | | No |
| GMRA;Giant Manta Ray | | Yes |
| GRBC;Great Shearwater or Black-capped Petrel (flying) | Includes BCPE,GRSH | Yes |
| GRCO;Great Cormorant | | No |
| GRCS;Great or Cory's Shearwater (on water) | Includes COSH,GRSH | No |
| GRSE;Gray Seal | | No |
| GRSH;Greater Shearwater | | Yes |
| GRSK;Great Skua | | No |
| GRTU;Green Turtle | | Yes |
| GSGO;Greater Snow Goose | | Yes |
| HAPO;Harbor Porpoise | | Yes |
| HASE;Harbor Seal | | No |
| HASH;Hammerhead shark | | Yes |
| HATU;Hawksbill Turtle | | Yes |
| HELI;Helicopter | | No |
| HERG;Herring Gull | | Yes |
| HOGR;Horned Grebe | | Yes |
| HOSE;Hooded Seal | | No |
| HUWH;Humpback Whale | | Yes |
| ICGU;Iceland Gull | | No |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| ID Impossible;ID Impossible | Biotic object lacking enough detail to place in a broad taxonomic grouping | Yes |
| JASK;Jaeger or Skua | Includes GRSK,LTJA,PAJA,POJA,SPSK | No |
| KIWH;Killer Whale | | No |
| KRST;Kemp's Ridley Sea Turtle | | Yes |
| LABA;balloon--Latex | | Yes |
| LAGU;Laughing Gull | | Yes |
| LASH;Large Shorebird sp. | | Yes |
| LBBG;Lesser Black-backed Gull | | Yes |
| LESP;Leach's Storm-petrel | | No |
| LETE;Least Tern | | No |
| LETU;Leatherback Turtle | | Yes |
| LFPW;Long-finned Pilot Whale | | No |
| LIGU;Little Gull | | No |
| LOTU;Loggerhead Turtle | | Yes |
| LSSD;Long-snouted Spinner Dolphin | | No |
| LTDU;Long-tailed Duck | | Yes |
| LTJA;Long-tailed Jaeger | | No |
| MACR;macroalgae | | Yes |
| MARA;Unidentified Manta Ray | | Yes |
| MASH;Manx Shearwater | | Yes |
| MBCE;Medium beaked Cetacean 3-10 m | Medium sized cetaceans with beaks. | No |
| MIWH;Minke Whale | | Yes |
| MNBC;Medium non beaked Cetacean 3-10 m | Medium sized Cetaceans with small or no beaks. | No |
| MOLA;Ocean Sunfish (Mola) | | Yes |
| MYBA;balloon--Mylar | | Yes |
| NA;Not Applicable | Used for the first and last frame of the reel, and any other descriptive tags (e.g., start and end of clouds). | Yes |
| NABW;North Atlantic Bottle-nosed whale | | No |
| NOFU;Northern Fulmar | | Yes |
| NOGA;Northern Gannet | | Yes |
| Nothing;Nothing | Use for objects that are waves or bird feces | Yes |
| OSPR;Osprey | | Yes |
| PAJA;Parasitic Jaeger | | Yes |
| PEFA;Peregrine Falcon | | No |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| PKWH;Pygmy Killer Whale | | No |
| POJA;Pomarine Jaeger | | Yes |
| PSDO;Pantropical Spotted Dolphin | | No |
| RAZO;Razorbill | | Yes |
| RBGU;Ring-billed Gull | | Yes |
| RBME;Red-breasted Merganser | | Yes |
| REBA;Red Bat | | Yes |
| REPH;Red Phalarope | | No |
| RIDO;Risso's dolphin | | Yes |
| RIWH;Right Whale | North Atlantic Right Whale | Yes |
| RNGR;Red-necked Grebe | | Yes |
| RNPH;Red-necked Phalarope | | No |
| ROST;Roseate Tern | | No |
| ROYT;Royal Tern | | Yes |
| RSST;Roughtail or Southern Stingray | | Yes |
| RTDO;Rough-toothed Dolphin | | No |
| RTLO;Red-throated Loon | | Yes |
| SAGU;Sabine's Gull | | Yes |
| SATE;Sandwich Tern | | Yes |
| SBCE;Small beaked Cetacean to 3 m | Smaller sized cetaceans with beaks. Includes ASDO,BODO,CLDO,CODO,LSSD,PSDO,RTDO,STDO | Yes |
| SBWH;Sowerby's Beaked Whale | | No |
| SCHA;Scalloped Hammerhead | | Yes |
| SEDO;Seal/Dolphin | True Seals and small cetaceans | Yes |
| SEWH;Sei Whale | | No |
| SFWH;Short-finned Pilot Whale | | No |
| SHAR;Unidentified shark | Members of Chondrichthyes | Yes |
| SMSH;Small Shorebird sp. | Includes REPH,RNPH | Yes |
| SMTU;Small turtle | Includes GRTU,HATU,KRST,LOTU | Yes |
| SNBC;Small non beaked Cetacean to 3 m | Smaller sized Cetaceans with small or no beaks | No |
| SNEG;Snowy Egret | | Yes |
| SOSH;Sooty Shearwater | | Yes |
| SOTE;Sooty Tern | | No |
| SPDO;Spinner Dolphins | Either Clymene, Short-snouted dolphin or long-snouted dolphin | No |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| SPSK;South Polar Skua | | No |
| SPWH;Sperm Whale | | No |
| STDO;Striped Dolphin | | No |
| SUSC;Surf Scoter | | Yes |
| SWAL;Unidentified Swallow | Includes BARS,CASW | Yes |
| TBMU;Thick-billed Murre | | No |
| TBWH;True's Beaked Whale | | No |
| THSH;Thresher Shark | Thresher Shark or Bigeye Thresher | Yes |
| TSMG;Tern/Small or Medium Gull | Includes ARTE,BLKI,BLTE,BOGU,BRTE,CATE,COTE,FOTE,LAGU,LETE,LIGU,RBGU,ROST,ROYT,SAGU,SATE,SOTE | Yes |
| UNAL;Unidentified Alcid | Includes ATPU,BLGU,COMU,DOVE,RAZO,TBMU | Yes |
| UNBI;Unidentified Bird | Includes all bird species | Yes |
| UNBW;Unidentified Baleen Whale | Members of Suborder Mysticeti | No |
| UNCE;Unidentified Cetacean | All whales and dolphins | Yes |
| UNCO;Unidentified Cormorant | Includes DCCO,GRCO | No |
| UNDO;Unidentified Dolphin | Members of Family Delphinidae | Yes |
| UNDT;Dark Tern | Includes BRTE,SOTE | No |
| UNDU;Unidentified Duck | Includes BLSC,BUFF,COME,LTDU,RBME,SUSC,UNME,UNSC,WWSC | Yes |
| UNFS;Unidentified Fin/Sei | Fin or Sei Whale | Yes |
| UNGR;Unidentified Grebe | Includes HOGR,RNGR | Yes |
| UNGU;Unidentified Gull | Includes BLKI,BOGU,GBBG,GLGU,HERG,ICGU,LBBG,LAGU,LIGU,RBGU,SAGU | Yes |
| UNJA;Unidentified Jaeger | Includes LTJA,PAJA,POJA | Yes |
| UNJE;Unidentified jellyfish | Members of Cnidaria | Yes |
| UNKN;unknown | Biotic or Abiotic objects | Yes |
| UNLA;Unidentified large alcid (Razorbill or Murre) | Includes COMU,RAZO,TBMU | Yes |
| UNLG;Unidentified Large Gull | Includes GBBG,GLGU,HERG,ICGU,LBBG,SAGU | Yes |
| UNLO;Unidentified Loon | Includes COLO,RTLO | Yes |
| UNLT;Unidentified large Tern | Includes CATE,ROYT | Yes |
| UNLW;Unidentified large whale | Large Cetacean > 10m | No |
| UNME;Unidentified Merganser | Includes COME,RBME | No |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| UNMG;Medium Gull: 38-53 cm | Includes BLKI,LAGU,RBGU,SAGU | Yes |
| UNMT;Medium Tern: 32-45 cm | Includes ARTE,BRTE,COTE,FOTE,ROST,SATE,SOTE | Yes |
| UNMW;Unidentified Medium Whale | Medium-sized Cetacea. Could include species BBWH,CBWH,FKWH,GBWH,KIWH,LFPW,MIWH,NABW,SFWH,SBWH,TBWH | Yes |
| UNPA;Unidentified Passerine | | Yes |
| UNPH;Unidentified Phalarope | Includes REPH,RNPH | Yes |
| UNRA;Unidentified ray | Includes members of superorder Batoidea | Yes |
| UNRO;Unidentified Rorqual | Members of Family Balaenopteridae | No |
| UNRS;Unidentified ray school | School of unidentified rays are marked (instead of individual animals within the schools) in situations where individuals are too small, deeply submerged, or otherwise poorly visible to be able to reliably distinguish individuals. | Yes |
| UNSA;Unidentified small alcid (Puffin/Dovekie) | Includes ATPU,BLGU,DOVE | Yes |
| UNSC;Unidentified Scoter | Includes BLSC,SUSC,WWSC | Yes |
| UNSG;Unidentified small gull | Includes BOGU,LIGU,SAGU | Yes |
| UNSH;Unidentified Shearwater | Includes AUSH,COSH,GRSH,MASH,SOSH | Yes |
| UNSK;Unidentified Skua | Includes GRSK,SPSK | No |
| UNSP;Unidentified Storm-petrel | Includes BRSP,LESP,WFSP,WISP | Yes |
| UNST;Unidentified small Tern | Includes BLTE,LETE | Yes |
| UNSW;Unidentified small whale | Small-sized Cetacea | No |
| UNTE;Unidentified Tern | Includes ARTE,BLTE,BRTE,CATE,COTE,FOTE,LETE,ROST,ROYT,SATE,SOTE | Yes |
| UNTW;Unidentified Toothed Whales | Odontoceti | Yes |
| UTSE;Unidentified True Seal | Members of Family Phocidae | No |
| WFSP;White-faced Storm-Petrel | | No |
| WHSH;Whale Shark | | No |
| WISP;Wilson's Storm-Petrel | | Yes |

| ID Category | Explanation or Species Included | Occurred in Project? |
|---|---|---|
| WSDO;Atlantic White-sided Dolphin | | No |
| WTTR;White-tailed Tropicbird | | No |
| WWSC;White-winged Scoter | | Yes |

## Appendix 4E. Excerpt from the “Confidence and Identification Criteria” document

Examples of criteria used for identifying avian and non-avian biotic targets at different confidence and taxonomic levels. If an object does not meet the “Definite” criteria for a particular ID category, then it goes to the next lower confidence level. If the object does not meet even the “Possible” level criteria, then it goes to a higher taxonomic grouping (e.g., from “Possible Black Scoter” to “Definite Unidentified Scoter”).

**Table 4E-1. Excerpt from the Avian Confidence and Identification Criteria.**

| AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite (Sitting)** | **Probable (Sitting)** | **Possible (Sitting)** | **Next Higher Taxonomic Group** |
| AKSH; Auk or Shearwater | To be determined as the need arises | Not used | Not used | UNBI; Unidentified Bird |
| DOVE; Dovekie | Dark bird < 21 cm (approximate sitting size[5] ) with some white in front. Reddish bill can be ruled out. | Dark bird < 21 cm (approximate sitting size) with some white in front. Bill is not seen. | To be determined | UNSA; Unidentified small alcid (Puffin/Dovekie) |
| NOGA; Northern Gannet | Adult: Large white, gannet-shaped bird with dark-tipped primaries and yellow to yellow-brown wash on head. | Adult: If the bird is obscured due to position, orientation, or blurriness and the definite features are hard to discern. | Adult: General size and shape are present and you can rule out buoy. | UNBI; Unidentified Bird or UNKN;unknown |
| SUSC; Surf Scoter | Can see white patches on head. Can see yellow orange to red bill pixel. Shape, size, and color is like a scoter. Female in close proximity to a definite male. | Shape, size, and color is like a scoter. Female or undetermined sex is in close proximity to a definite male or in all SUSC flock. Inconclusive bill color. | Shape, size, and color. Not enough frames to determine sex and species, but is in a SUSC flock. | UNSC; Unidentified Scoter |

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[5] Sitting size is the measurement of a resting bird (not stretched out).

| AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite (Sitting)** | **Probable (Sitting)** | **Possible (Sitting)** | **Next Higher Taxonomic Group** |
| UNAL; Unidentified Alcid | An auk of indeterminate size with general auk shape, dark plumage, white on the sides and head, visible bill and face characteristics, but unable to distinguish between species. | An auk of indeterminate size with general auk shape, dark plumage and white on the sides and head. | An auk of indeterminate size with general auk shape and color, but there are fewer frames or image obscurities that won't allow for higher identification confidence. | AKSH; Auk or Shearwater or UNBI;Unidentified Bird |
| UNBI; Unidentified Bird | Object has shape, color, head, bill, and bird-like movement. Image quality or other factors won't allow placement into a lower taxonomic grouping. | Shape, color, head, and bill with possible bird-like movement or posture. | Bird shape (body and head) and coloring. | ID Impossible;ID Impossible |
| UNSA; Unidentified small alcid (Puffin/Dovekie) | An auk between 15-30 cm (approximate sitting size) with general auk shape, dark plumage, white on the sides and head, but no bill or face details to distinguish between species. | An auk between 15-30cm (approximate sitting size) with general auk shape and color, but there are fewer frames or image obscurities that won't allow for higher identification confidence. | To be determined | UNAL; Unidentified Alcid |
| UNSC; Unidentified Scoter | Female or unknown sex in a mixed flock. Color and shape similar to scoters. No conclusive bill or head identifying features. | Single bird with color and shape similar to scoters. No conclusive bill identifying features. | To be determined | UNDU;Unidentified Duck or UNBI;Unidentified Bird |

**Table 4E-2. Non-Avian Confidence and Identification Criteria.**

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| BAIT; bait ball | Small fish-shaped objects in a group. There is definite movement. | A more submerged or blurry bait ball. | Hard to determine definite movement. There is speckling in the water but it is harder to determine if this is a bait ball. Reviewer are most likely assessing whether this is a bait ball, ocean spray or debris in the water. | ID Impossible; ID Impossible |
| BASH; Basking Shark | Large, odd shaped shark with a pointed snout. If feeding, can see the shape of the gills extended out. | Large, odd shaped shark that is more submerged or seen in less frames. | Not used. | SHAR; Unidentified shark |
| BODO; Bottlenose Dolphin | No distinct color patterns and dolphin >300 cm. | Unsure if size >300, but is associating with a definite BODO. | Large( >300, unless it is a juvenile) cetacean that is submerged or blurry cetacean and is non-descript. | SBCE; Small beaked Cetacean to 3 m |
| CODO; Common Dolphin | Can clearly see hourglass pattern on the side. | Can see hourglass pattern in a few frames. There is no identification by association with this species. | A submerged common dolphin with a probable CODO patterning on the side. Can rule out BODO and Striped Dolphin patterning. There is no identification by association with this species. Associated species without a hint of patterning should go to a SBCE identification. | SBCE; Small beaked Cetacean to 3 m |
| COWR; Cownose Ray | Cownose is visible and therefore direction of movement is known. Can be single or in a group. | An individual that cannot be identified to species, but is associating with a school member that is a definite cownose ray. | A deep or blurry individual that is associating with a cownose ray. | UNRA; Unidentified Ray |
| FIWH; Fin Whale | Slender whale with white under the lower right jaw. | Not used | Not used | UNFS; Unidentified Fin/Sei |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| GMRA; Giant Manta Ray | Dark ray with a disc width >122 cm. | Dark ray with a disc width >122 cm. Reviewer unfamiliarity with this species would cause a lower confidence. | Not used | MARA; Unidentified Manta Ray |
| GRTU; Green Turtle | Head width is consistently small through frames. SCL>90cm. Carapace shape is elliptical. | SCL >90 cm, head width smaller (up to 15 cm) and not broad | >90 cm, head width not consistent or unclear across some frames | SMTU; Small turtle |
| HAPO; Harbor Porpoise | Not used | Between 137-183 cm with no distinct markings and no beak. Also has a chunky appearance compared. | The "no beak" appearance is seen in fewer frames. Since it is harder to definitively determine that a cetacean is non-beaked, reviewers would most likely use the broader category, Unidentified Toothed Whale. | SNBC; Small non beaked Cetacean to 3 m |
| HASH; Hammerhead shark | Shark with a distinct hammer-shaped head consistent across frames | Submerged or blurry shark with a hammer-shaped head. | Deeply submerged shark with a hint of a hammer-shaped head. | SHAR; Unidentified shark |
| HATU; Hawksbill Turtle | Overlapping scutes, color is like a Hawksbill. SCL>65 and <90 cm, head width smaller and not broad, jagged edges of scutes. | SCL>65 and <90 cm, head width smaller and not broad, jagged edges of scutes. | SCL>65 and <90 cm, head width smaller and not broad. Large tail indicating a mature male is present. | SMTU; Small turtle |
| HUWH; Humpback Whale | Stocky body with relatively long white pectoral fins. | Not used | Not used | UNBW; Unidentified Baleen Whale |
| ID Impossible; ID Impossible | Biotic object that cannot be put into a species grouping. | ID Impossible Probable was used in the first month of data, but then was not used after that. This should be the same as ID Impossible Definite. | Not used | UNKN; unknown |
| KRST; Kemp's Ridley Sea Turtle | >56 cm, round shell (width is almost equal to length), broad head compared to SCL (up to 13cm head width), | Carapace looks round, but measurements indicate a more elliptical shape. | Turtle is more submerged or in fewer frames, but can still see shape and size. | SMTU; Small turtle |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| LETU; Leatherback Turtle | A turtle with a broad upper body with relatively long front flippers. | A more submerged individual or an individual in fewer frames. Dark coloring and overall shape is still present. | Large, dark-colored object that is mostly leatherback-shaped. Can rule out Molas and manta rays. Leatherback turtles that cannot be identified to Possible would most likely go to ID Impossible if there is movement and UNKN (Unknown) if there is no discernible movement. | ID Impossible; ID Impossible |
| LOTU; Loggerhead Turtle | SCL >90 cm, head width large (up to 28 cm) and broad, overall carapace is heart-shaped | SCL > 65 and <90, head is broad | Loggerhead features (broad head, carapace shape and etc.) are in fewer frames. Other species of sea turtles can be ruled out. | SMTU; Small turtle |
| MARA; Unidentified Manta Ray | Not used | A dark ray that is too submerged to get an accurate size. Overall shape and color of a Mylobatidae spp. Reviewer unfamiliarity with this grouping would cause a lower confidence. | Not used | CESS; Cetacean/Seal/Shark |
| MIWH; Minke Whale | Not used. | Slender, comparatively small whale with a pointed rostrum. Can see white band on flippers. | A more submerged cetacean with the shape and size of a Minke Whale. | UNRO; Unidentified Rorqual |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| MOLA; Ocean Sunfish (Mola) | Large, irregular shaped fish with fins near the posterior end. A definite MOLA would consistently and definitively be a MOLA in almost all frames. | MOLA that is angled a bit in the water column where it is hard to see the shape. A probable MOLA might be more submerged or blurry in some of the frames. | Either deeply submerged or seen in fewer frames. Must rule out small turtle and rays. Due to the irregular shape, MOLA not identified to the Possible confidence may to go to a broader category such as ID Impossible. If the fins are not seen and there is no discernible movement, it is possible for a MOLA to go to the UNKN (Unknown) identification category. | FISH; Unidentified fish |
| REBA; Red Bat | Tone or color is reddish or rusty brown. Body shape is oblong to oval giving it a chunky appearance. Wing coloration is grayish, white or blurry. Wing is angled proximally to the body giving it a triangular appearance. All aforementioned characteristics are consistent across frames or you can see the arm. | Tail shape is wedged or "V" shaped like a bat and there is a triangular appearance to the wings. Red color is present. | To distinguish from an UNBI or ID Impossible, tail shape is wedged or "V" shaped like a bat and there is a triangular appearance to the wings. | ID Impossible; ID Impossible |
| RIDO; Risso's dolphin | Not used | Not used | Used only once, the reviewer cited that the cetacean was at least 387cm with a comparatively large back fin, and light-colored belly and side. If cetacean was approximately 300 cm, then this would go to the broader group, SBCE (Small beaked Cetacean to 3m). If cetacean was >300cm, then this would go to the broader group, UNDO(Unidentified Dolphin). | SBCE; Small beaked Cetacean to 3 m |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| RIWH; Right Whale | A robust whale with callosities on the rostrum. | Not used | Not used | UNBW; Unidentified Baleen Whale |
| RSST; Roughtail or Southern Stingray | Rule out skate by size (disc width>107 cm) and shape (pointed wings). Disc width overlaps between species. There are not enough features such as tail length to identify the stingray to species. | Not used | Not used | UNRA; Unidentified Ray |
| SBCE; Small beaked Cetacean to 3 m | Beak present. Cetacean less than 3m. | Unknown if beak is present or unknown size, but is associating with a definite SBCE. | Beak may not be seen consistently across frames or single animal is deeply submerged. | UNDO; Unidentified Dolphin |
| SCHA; Scalloped Hammerhead | Not used | Hammerhead shark with a central notch on the head as well as smaller notches on either side of the central notch giving it a "scalloped" appearance. | "Scalloped" appearance may be harder to ascertain due to submergence, blurriness or number of frames. | HASH; Hammerhead shark |
| SHAR; Unidentified shark | Caudal fin is vertical and animal is greater than 198 cm. Or, if a smaller animal, there is consistent sinusoidal movement. | Shark is more submerged but size, shape or movement is shark-like. | Shark shape, size or movement is seen in fewer frames. | CESS; Cetacean/ Seal/Shark |
| SMTU; Small turtle | SCL>65 and <90 cm, head width smaller and not broad, lack of jagged edges. Or, definitely turtle-shaped (carapace shape with at least two alternate flippers or head is seen with the carapace shape), but other criteria are not met in order to classify to species. Or, SCL <50 cm, then immature SMTU. Mola and ray can be ruled out. | Turtle shape is not consistently seen in all frames, but in most frames. Mola and ray can be ruled out. | Mola and ray can be ruled out. There is often turtle-like movement. If it is a juvenile turtle, macroalgae can be ruled out. | ID Impossible; ID Impossible |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| SNBC; Small non beaked Cetacean to 3 m | Not used | Not used | Not used | UNTW; Unidentified Toothed Whales |
| THSH; Thresher Shark | Elongated upper caudal fin lobe that is almost longer than the entire body is consistently seen across many frames. | Shark is more submerged and caudal fin is not seen consistently across frames. | Elongated upper caudal fin lobe is seen in fewer frames, is more submerged or blurry. | SHAR; Unidentified shark |
| UNBW; Unidentified Baleen Whale | Not used | Not used | Not used | UNLW; Unidentified large whale or UNMW; Unidentified Medium Whale |
| UNCE; Unidentified Cetacean | Has cetacean tail. Unknown size or beak status. | A blurry or more submerged cetacean associating with a known cetacean. | Not used | SEDO; Seal/Dolphin or CESS; Cetacean/Seal/Shark |
| UNDO; Unidentified Dolphin | Unknown if a beak is present. Harbor Porpoise can be ruled out. | Associating with a definite UNDO. | Blurry or submerged cetacean that is associating with a group of unidentified dolphins. | UNTW; Unidentified Toothed Whales |
| UNFS; Unidentified Fin/Sei | Sizes overlap between Fin and Sei Whale. Rostrum and tail obscured. Overall slender whale. | Not used | Not used | UNRO; Unidentified Rorqual |
| UNKN; unknown | Shape is similar to flotsam/jetsam and animal. | NA | NA | NA |
| UNRA; Unidentified Ray | Single or group of ray-shaped objects where the nose shape or direction of movement is unknown. | A blurry or obscured individual with a ray shape or movement that is associating with definite rays. | A single ray or small group that consistently has a ray shape or ray movement. Turtle, Mola and trash can be ruled out. | CESS; Cetacean/Seal/Shark |
| UNRO; Unidentified Rorqual | Not used | Not used | Not used | MNBC; Medium non beaked Cetacean 3-10 m or UNBW; Unidentified Baleen Whale |

| NON-AVIAN | | | | |
|---|---|---|---|---|
| **ID Category** | **Definite** | **Probable** | **Possible** | **Next Higher Taxonomic Group** |
| UNTW; Unidentified Toothed Whales | Unknown if a beak is present. Unknown size. Harbor Porpoise cannot be ruled out. | A blurry or more submerged toothed whale with unknown size or beak, but is small enough to be a Harbor Porpoise. | Not used | UNSW; Unidentified small whale |

## Appendix 4F. Hierarchical Matrix of Target Taxonomic Groups

Prior to the start of video analysis, a list of anticipated biota was compiled from past boat surveys and bird and mammal surveys conducted in the project area. Groupings were either developed based on anticipated similarity in video or from boat survey codes and experience. Other codes were added as they were discovered (e.g AMBI;American Bittern and BEKI;Belted Kingfisher). This table guides the "Confidence and Identification Criteria" and development of audit rules. Starting with the ID Category, it shows how the animal or animal group moves from a lower taxonomic group (Group 1) to a higher taxonomic group (Group 6 or 8), depending on the quality of the image, certainty criteria, and other factors.

Group 1 consists of mixed species from the same genera. It has a lesser amount of species associated with the grouping than Group 2 and in general, the grouping contains <4 associated species. It is also the lowest taxonomic grouping. Group 2 consists of mixed species and mixed genus groupings. This group may have a size or color designation that further splits the family down into fewer species' associations. Group 3 consists of mixed genus groupings. It will have a lower number of associated species than Broad Group 4 and in general may contain a subset to all members of the family. Group 4 consists of mixed genus groupings. In addition, it may have a high number of species associated with this group. This may contain a subset to all members of the family. Group 5 consists of mixed order and mixed family groupings. Group 6 is the highest taxonomic bird grouping. It consists of all bird orders.

**Table 4F-1. Excerpt from Hierarchal Matrix of Avian Taxonomic Groups.** Birds that cannot be identified to Group 6 (Unidentified Bird) will either be identified to ID Impossible if there is movement or UNKN;Unknown if there is no discernible movement.

| AVIAN | | | | | | |
|---|---|---|---|---|---|---|
| **ID Category** | **Group 1** | **Group 2** | **Group 3** | **Group 4** | **Group 5** | **Group 6** |
| AKSH; Auk or Shearwater | | | | | | UNBI |
| DOVE;Dovekie | | UNSA | UNAL | | | UNBI |
| NOGA; Northern Gannet | | | | | | UNBI |
| SUSC; Surf Scoter | UNSC | | | UNDU or UNBI | | UNBI |
| UNAL; Unidentified Alcid | | | | | AKSH | UNBI |
| UNSA; Unidentified small alcid (Puffin/Dovekie) | | | UNAL | | AKSH | UNBI |
| UNSC; Unidentified Scoter | | | | UNDU or UNBI | | UNBI |

**Table 4F-2. Hierarchal Matrix of Non-Avian Taxonomic Groups.** Non-avian biota that cannot be identified to Group 8 (CESS;Cetacean/Seal/Shark) will either be identified to ID Impossible if there is movement or UNKN;Unknown if there is no discernible movement. Group definitions: Group 1 consists of mixed species and mixed genus groupings. Group 2 consists of mixed genus groupings. Group 3 consists of mixed genus and mixed family groupings. It will have a lower number of associated species than Broad Group 4 and in general may contain a subset to all members of the family. Group 4 consists of sub-order groupings. Group 5 consists of mixed sub-order groupings based on size class. Group 6 consists of order groupings. Group 7 consists of mixed order and some mixed class groupings. Group 8 consists of mixed classes and contains the most number of species.

| NON-AVIAN | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **ID Category** | **Group 1** | **Group 2** | **Group 3** | **Group 4** | **Group 5** | **Group 6** | **Group 7** | **Group 8** |
| BAIT;bait ball | | | | | | | | |
| BASH;Basking Shark | | | | | | | SHAR or FISH | CESS |
| BODO;Bottlenose Dolphin | | SBCE | UNDO | UNTW | UNSW | UNCE | SEDO | CESS |
| CESS;Cetacean/Seal/Shark | | | | | | | | |
| CODO;Common Dolphin | | SBCE | UNDO | UNTW | UNSW | UNCE | SEDO | CESS |
| COWR;Cownose Ray | UNRA | | | | | | | |
| FISH;Unidentified fish | | | | | | | | CESS |
| FISS;Unidentified fish school | | | | | | | | CESS |
| FIWH;Fin Whale | UNFS | UNRO | | UNBW | UNLW | UNCE | | CESS |
| GMRA;Giant Manta Ray | MARA | | | | | | | CESS |
| GRTU;Green Turtle | SMTU | | | | | | | |
| HAPO;Harbor Porpoise | | | SNBC | UNTW | UNSW | UNCE | SEDO | CESS |
| HASH;Hammerhead Shark | | | | | | | SHAR or FISH | CESS |
| HATU;Hawksbill Turtle | SMTU | | | | | | | |
| KRST;Kemp's Ridley Sea Turtle | SMTU | | | | | | | |
| LETU;Leatherback Turtle | | | | | | | | |
| LOTU;Loggerhead Turtle | SMTU | | | | | | | |
| MARA;Unidentified Manta Ray | | | | | | | | CESS |
| MIWH;Minke Whale | | UNRO | MNBC | UNBW | UNMW or UNLW | UNCE | | CESS |
| MNBC;Medium non beaked Cetacean 3-10 m | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| MOLA;Ocean Sunfish (Mola) | | | | | | | FISH | CESS |
| REBA;Red Bat | | | | | | | | |
| RIDO;Risso's Dolphin | | SBCE | UNDO | UNTW | UNSW | UNCE | SEDO | CESS |
| RIWH;Right Whale | | | | UNBW | UNLW | UNCE | | CESS |
| RSST;Roughtail or Southern Stingray | UNRA | | | | | | | CESS |

| NON-AVIAN | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **ID Category** | **Group 1** | **Group 2** | **Group 3** | **Group 4** | **Group 5** | **Group 6** | **Group 7** | **Group 8** |
| SBCE;Small beaked Cetacean to 3 m | | | UNDO | UNTW | UNSW | UNCE | SEDO or CESS | CESS |
| SCHA;Scalloped Hammerhead | HASH | | | | | | SHAR or FISH | CESS |
| SEDO;Seal/Dolphin | | | | | | | | CESS |
| SHAR;Unidentified shark | | | | | | | | CESS |
| SMTU;Small turtle | | | | | | | | |
| SNBC;Small non beaked Cetacean to 3 m | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| THSH;Thresher Shark | | | | | | | SHAR or FISH | CESS |
| UNBW;Unidentified Baleen Whale | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| UNCE;Unidentified Cetacean | | | | | | | SEDO or CESS | CESS |
| UNDO;Unidentified Dolphin | | | | | | | | CESS |
| UNFS;Unidentified Fin/Sei | | UNRO | | UNBW | UNLW | UNCE | | CESS |
| UNJE;Unidentified jellyfish | | | | | | | | |
| UNLW;Unidentified large whale | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| UNMW;Unidentified Medium Whale | | | | | | UNCE | | CESS |
| UNRA;Unidentified ray | | | | | | | | CESS |
| UNRO;Unidentified Rorqual | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| UNRS;Unidentified ray school | | | | | | | | CESS |
| UNSW;Unidentified small whale | NOT USED IN THE AERIAL SURVEY | | | | | | | |
| UNTW;Unidentified Toothed Whales | | | | | | | | CESS |

## Appendix 4G. QA/QC checklist for reviewed data

**Table 4G-1. QA/QC checklist for reviewed data – updated as of September 16, 2013.**

| |
|---|
| Missed objects are highlighted in yellow. |
| Missed objects have a frame number in the "Added Frame Number" column |
| All confidences should be filled out. |
| Birds and bats should be associated with appropriate behavior (flying, sitting, taking off, loafing, following vessel) |
| All objects in flight or taking off should have a yes or no filled in the "Flying at sea level" field. |
| Birds should not have the submerged field filled out. |
| Non-avian biota are associated with appropriate behavior (stationary, moving, haul out) |
| All seals, sharks, turtles, cetaceans, and fish should have the "Submerged" field filled out. |
| Make sure there are behaviors filled out for all animals. |
| Objects identified as Not Applicable, ID Impossible, UNKN, boats, balloons, FIGE, FLJE, MACR and buoys should not have behavior, flying at sea level, submerged, age, plumage, molt or sex filled out. Behaviors associated with UNKN, boats and ID Impossible can be put in comments. |
| ID Impossible should have a comment. |
| Make sure dates and initials are filled out for all lines in the spreadsheet. |
| In the "Category" column, all reel characterizations or bad condition comments such as Start of reel, end of reel, reached here, resumed here and end of cloud are classified as "NA; Not Applicable" in the "ID Category" column. |
| Make sure age and plumage match each other if both have been filled out. |
| Do a quick check in comments for misspellings. |
| Check for formatting in ID Category (lowercase vs uppercase). |

## Appendix 4H. Excerpt of allowed audit answers for a particular ID Category.

**Table 4 H-1. Excerpt of allowed audit answers for a particular ID Category.** Some ID categories such as "UNRA**;** Unidentified Ray" were not "Fully Defined" in the event that other species of rays were discovered during the project.

| ID Category | Allowed Audit Answers | Audit Answer Status |
|---|---|---|
| AKSH; Auk or Shearwater | AKSH,UNAL,UNBI,UNSH,UNLA,DBSH | Fully defined |
| BODO;Bottlenose Dolphin | BODO,SBCE | Fully defined |
| COWR; Cownose Ray | COWR,UNRA | Fully defined |
| DOVE;Dovekie | DOVE,UNSA,UNAL | Fully defined |
| LOTU; Loggerhead Turtle | LOTU,SMTU | Fully defined |
| NOGA; Northern Gannet | NOGA | Single Defined-No similar spp |
| REBA;Red Bat | REBA | Single Defined-No similar spp |
| SBCE;Small beaked Cetacean to 3 m | BODO,CODO,SBCE,UNDO | Partial Defined-More Information is needed |
| SMTU; Small turtle | GRTU,HATU,KRST,LOTU,SMTU | Fully defined |
| SUSC; Surf Scoter | SUSC,UNSC | Fully defined |
| UNAL; Unidentified Alcid | AKSH,ATPU,BLGU,COMU,DOVE,RAZO,TBMU,UNAL,UNBI,UNLA,UNSA | Fully defined |
| Unidentified Bird | AKSH,DBSH,FUMG,GRBC,GRCS,JASK,LASH,SMSH,TSMG,UNAL,UNCO,UNDU,UNDT,UNGR,UNGU,UNJA,UNLA,UNLG,UNLO,UNLT,UNME,UNMG,UNMT,UNPH,UNSA,UNSC,UNSG,UNSH,UNSK,UNSP,UNST,UNTE,UNBI | Fully defined |
| UNDO; Unidentified Dolphin | SBCE,UNDO,UNTW | Partial Defined-More Information is needed |
| UNMT;Medium Tern: 32-45 cm | ARTE,BLTE,BRTE,CATE,COTE,FOTE,LETE,ROST,ROYT,SATE,SOTE,TSMG,UNDT,UNLT,UNMT,UNST,UNTE | |
| UNRA; Unidentified Ray | COWR,MARA,RSST,UNRA,UNRS | Partial Defined-More Information is needed |
| UNSA; Unidentified small alcid (Puffin/Dovekie) | ATPU,BLGU,DOVE,UNAL,UNSA | Fully defined |
| UNSC; Unidentified Scoter | WWSC,BLSC,SUSC,UNDU,UNBI,UNSC | Fully defined |
| UNTW; Unidentified Toothed Whales | UNDO,UNTW | Partial Defined-More Information is needed |

# Chapter 5: Summary of high resolution digital video aerial survey data

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 5 Highlights

*Results from high resolution digital video aerial survey data collected in the mid-Atlantic study area.*

### Context[1]

High resolution digital video aerial surveys are a recently developed method to collect animal distribution and abundance data, and our study was the first to use this method on a broad scale in the U.S. Chapters 3-4 focus on the methods used to collect the digital video aerial survey data, and Chapter 6 includes recommendations for future work using this technology. Chapter 5 reviews the results of these surveys for the mid-Atlantic study area, including data on observed counts and species identification rates for birds, marine mammals, sea turtles, and other wildlife. Flight heights for flying animals were also estimated from the video footage, allowing for analysis of animal altitude in relation to potential wind turbine heights.

Building off of this summary chapter, Chapter 13 discusses an experimental survey conducted to directly compare boat and aerial survey methodologies, while Chapter 14 examines the differences between the overall boat and aerial datasets. Subsequent chapters in Part IV of this report (Chapters 15-19) focus on integrating the two survey methods using a variety of techniques to better understand the distribution and abundance of wildlife within the mid-Atlantic study area.

### Study goal/objectives

Summarize animal distribution and abundance data that were collected using a novel surveying method in the mid-Atlantic study area.

### Highlights

- Over 100,000 animals were observed within the study area over two years of surveys, including over 46,000 birds and 60,000 aquatic animals.
- The most abundant animals observed in aerial video were rays (Batoidea), making up 44% of the study data. The most commonly observed birds were scoters (*Melanitta* spp., 19% of the data), primarily Black Scoters.
- Northern Gannets and Bonaparte's Gulls were also observed in large numbers.
- Notable animal sightings included many sea turtles, several species of baleen whales, and migrating Eastern Red Bats.
- Most of the animals with calculated flight heights flew below the predicted lower end of the rotor swept zone (20 m).

### Implications

Digital aerial surveys appear to have certain advantages for obtaining information on the distributions of animals within the marine environment, particularly for aquatic species such as sea turtles and rays. However, there are certain taxa that can more readily be identified than others using this technology.

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[1] For more detailed context for this chapter, please see the introduction to Part II of this report.

## Abstract

High resolution digital video aerial surveys are a relatively novel method for collecting information on marine wildlife distributions and abundances, and this study is the first to use these methods on a broad scale in the United States. Our study focused on collecting marine bird, mammal, and turtle data within the mid-Atlantic region, though we also documented the movements of rays and sharks, noted large schools of forage fish, and captured the migration of terrestrial species in the marine environment. We observed over 100,000 animals within the study area, including over 46,000 birds and 60,000 aquatic animals. The most abundant birds observed were scoters (*Melanitta* spp.), making up 19% of the data collected, and most of those observed were Black Scoters in the winter (*M. americana*). The most abundant animals overall were rays, making up 44% of the data; the Cownose Ray (*Rhinoptera bonasus*) was the most abundant species, which was primarily observed in the study area in the spring through early fall. Other abundant species included Northern Gannets (*Morus bassanus*) and Bonaparte's Gulls (*Chroicocephalus philadelphia*); less abundant but notable animals include several species of sea turtles and baleen whales, as well as diurnal migrations of Eastern Red Bats (*Lasiurus borealis*, n = 15) through the offshore study area in the fall. Identification rates for the video aerial surveys varied widely based on the quality of the footage, as well as the taxonomic group in question. Identification rates of small alcids were low, while scoters were more easily identifiable. Flight heights were estimable for 75% of flying animals, and showed that 59% of these animals were flying below the likely rotor swept zone for current offshore wind turbines (<20 meters). More detailed analyses of these data can be found in Part IV of this report.

## Introduction

The mid-Atlantic region is an important area for a broad range of marine wildlife species throughout the year. Some breed in the area, such as coastal birds and sea turtles, while others visit from the southern hemisphere in their non-breeding season, such as shearwaters and whales. In the fall, many summer residents migrate south to breed or winter in warmer climes, and they are replaced by species that have traveled from their northern breeding grounds to winter in the mid-Atlantic. Additionally, many pelagic, coastal, and terrestrial species make annual migrations up and down the eastern seaboard, and travel directly through the mid-Atlantic region in spring and fall. Thus, many species use or funnel through the mid-Atlantic region each year, resulting in a complex ecosystem where the community composition is constantly shifting, and the temporal and geographic patterns are highly variable.

In our study, we aimed to produce data to inform siting and permitting processes for offshore wind energy development in the mid-Atlantic. We collected information on bird, sea turtle, and marine mammal abundances and movements over a two-year time period (2012-2014) using a variety of technologies and methods to examine spatial patterns and trends, while simultaneously testing a new technology for the first time in the United States, high resolution digital video aerial surveys (hereafter digital video aerial surveys). Digital video aerial surveys are a relatively new method for collecting distribution and abundance data on animals in the marine ecosystem (Thaxter and Burton, 2009). Although digital video aerial surveys have become common practice for offshore wind energy planning and monitoring in Europe, this Department of Energy (DOE)-funded Mid-Atlantic Baseline Studies

Project (MABS) and state-funded Maryland Project are the first projects to use these methods on a large scale in the United States. We also conducted boat surveys for wildlife within the study area on the continental shelf, to accompany and compare with the data from the digital video aerial surveys. For details on boat survey approaches, and for comparisons between boat and aerial data, see Parts III and IV of this report, respectively. Here, we examine the digital video aerial survey results in detail, including discussion of observation rates, species identification rates, and flight height estimates for volant taxa.

We discuss results for the Cownose Ray (*Rhinoptera bonasus*) in particular detail, to highlight the utility of offshore digital video aerial surveys for aquatic taxa. This species is found along the coast of the western Atlantic Ocean from the northeastern U.S. to Brazil, and migrates seasonally, likely prompted by changes in water temperatures (Goodman et al., 2011). There are limited studies on Cownose Ray migration, but the mid-Atlantic may be an important area for migrating rays (Blaylock, 1993; Goodman et al., 2011). Their movements are of interest to fisheries regulators as they are commonly thought to predate bivalve aquaculture beds (Myers et al., 2007), though little evidence of this has been documented (Fisher, 2010). An unregulated cownose ray fishery exists in Virginia, and there are also high bycatch and discard rates of rays within other fisheries; population declines are predicted as a result (Barker, 2006; Goodwin, 2012). They are listed by the IUCN as “Near Threatened” globally largely due to heavy and unregulated fishing pressure in Central and South America (Barker, 2006). Aerial surveys have been used to study the species in Chesapeake Bay (Blaylock, 1993; Goodman et al., 2011), but rarely cover migration in the open ocean, and this is the first instance of using digital video aerial surveys to monitor their distributions and relative abundance.

## Methods

Between March 2012 and May 2014, HiDef Aerial Surveying, Ltd. conducted fifteen large-scale surveys using super high-definition digital video on an aerial platform (Table 5-1). For fourteen surveys, transects were flown at high densities within the federally-designated Wind Energy Areas (WEAs) off of Delaware, Maryland, and Virginia, while the remainder of the study area was surveyed on an efficient ‘sawtooth’ transect path to provide broad-scale context (Chapter 3). In the second year of surveys (March 2013-May 2014), additional high density transects were added to the west and south of the Maryland WEA (Figure 5-1), and the fifteenth survey was conducted in just the Maryland WEA and adjacent high-density Maryland survey areas (Table 5-1). Both MABS and Maryland Project survey data are presented in this report. Early surveys included video footage at 2 cm Ground Spatial Resolution (GSR) for transects within the WEAs, and 3 cm GSR for the broader sawtooth survey; beginning in September 2012, all transects were surveyed at 2 cm GSR.

Final geoprocessing of the data was completed in January 2015. The project team identified wildlife locations, taxonomic identities, behaviors, and flight heights from the video footage. Detailed data collection, analysis, and data management protocols can be found in Chapters 3-4 of this report.

This chapter presents summaries of raw count data from the digital video aerial surveys on a monthly, seasonal, and annual basis. We also discuss identification rates for the most common species groups. For these summaries, all identifications in the aerial data were taken at face value (e.g., an identified "possible Black Scoter [*Melanitta americana*]" was considered to be a Black Scoter, rather than an "Unidentified Scoter"; see Chapter 4 for additional information on certainty levels and identification criteria). Ray (Batoidea) densities were examined across the study area using counts of rays per Bureau of Ocean Energy Management (BOEM) 4.8 x 4.8 km lease block, corrected for survey effort within the lease blocks (km$^2$). All rays were included in the analysis, and the four survey periods with highest ray abundances were mapped using ArcGIS 10.1 (ESRI, Redlands, CA).

Flight heights were examined for different avian species groups to compare to the rotor-swept zone of offshore wind turbines. Flight heights were estimated using a proprietary flight height estimation method developed by HiDef Aerial Surveying, Ltd., which uses measurements of "parallax," or the apparent motion of an elevated object against a distant background due to the movement of the observer (Hatch et al., 2013). Flight heights of flying animals could not be estimated when the animal was flying directly parallel to the plane, rendering calculations of displacement impossible, or the animal was present in an unusually small number of frames (Hatch et al. 2013). Flight heights were estimated in altitude bands (0-20, 20-50, 50-100, 100-200, and 200+ m).

Part IV of this report presents additional information comparing digital aerial and boat survey results, and integrating data from both survey types into in-depth analyses of wildlife distributions and relative abundance.

## Results

A total of 107,003 animals were observed in the fifteen surveys of the MABS and Maryland Project study areas, including over 46,000 birds and 60,000 aquatic animals (including cetaceans, sea turtles, rays, sharks, and fish; see Appendix 5A). At least 48 species of birds and 19 species of non-avian animals were represented. Overall, 45% of the animals observed in the study were identified to species level. The greatest numbers of animals were observed in March, July, and September (Table 5-2). There were variations in data quality throughout the project, with low light in winter causing difficulty for identifications. It should be noted that data collected in each survey (as shown in Table 5-2 and Table 5A-1) are not entirely comparable across the duration of the study, as the study area was significantly expanded beginning in March of 2013. Additionally, the exact timing of surveys can have a huge effect on species counts, particularly during migration periods when large numbers of wintering birds could be moving in or out of the study area; a week's difference in survey dates could have a significant effect on observed overall abundance.

Quality assurance and quality control (QA/QC) protocols for analysis of the video data are presented in Chapter 4. An audit was not conducted for the first (March 2012) survey, as object identifications for those data were performed collectively among BRI biologists to develop a common identification process and pool their existing expertise. For all other surveys, object identifications were independently conducted by BRI biologists, and random audits (e.g., blind re-reviews of 20% of all objects, and 100% of objects identified as state- or federally-listed threatened and endangered species) were conducted for all identifications (Table 5-3). Species identifications were problematic for 3 cm footage in early surveys, due to poorer image clarity and color rendition, and this issue was addressed by project collaborators by discontinuing all use of 3 cm GSR for surveys beginning in September 2012 (Chapter 6, Table 6-2). Agreement rates for the random audit varied from 80-98% between surveys (Table 5-3); when agreement was less than 90% (for random audit objects) or less than 100% (for threatened and endangered species) in a survey, then partial re-review of survey data and/or arbitration of disagreements among reviewers occurred (as described in detail in Chapter 4).

### *Relative abundance of counts*

#### *Birds*

Scoters (*Melanitta* spp.) were the most abundant avian species observed in the digital video aerial surveys, making up over 19% of all observations (Figure 5-2 and Table 5-2). Black Scoters represented 9% of the total dataset and were the most commonly represented avian species. Another 9% of the data were classified as *Melanitta* sp. (either Black Scoter; Surf Scoter, *M. perspicillata;* or White-winged Scoter, *M. fusca*), but could not be identified to the species level. Scoters were generally present in the region in the winter and early spring months (Figure 5-4), and were often observed in large flocks spanning across multiple video frames and between cameras. Numbers of scoters observed varied from year to year, with the highest numbers of scoters observed in the first survey in late March of 2012 (Table 5-2, Figure 5-4).

Gannets (all identified to a single species, the Northern Gannet, or *Morus bassanus*) were the next most commonly observed avian family (6.7% of the total dataset), followed by loons (Gaviidae; 5%), with both groups predominantly observed in winter and spring (Table 5-2, Figure 5-4). Gannets were most abundant during the aerial survey in February of 2013, but the numbers of loon observations did not vary greatly from year to year (Table 5-2).

A variety of gull and tern species were observed throughout the year (Appendix 5A). Bonaparte's Gulls (*Chroicocephalus philadelphia*) were the most common (1%) and were predominantly seen in the winter and fall, while Herring Gulls (*Larus smithsonianus*, 0.09%) and Great Black-backed Gulls (*L. marinus*, 0.22%) were seen in lower numbers throughout the year (Appendix 5A) Few auks (Alcidae) were observed overall (Figure 5-2, Table 5-2).

#### *Non-avian animals*

Large numbers of animals were observed in digital video aerial surveys at or below the surface of the water (Figure 5-3). There were major seasonal differences for aquatic animal abundance, most notably with very large numbers of rays observed in summer and fall surveys (Figure 5-5). Rays were the most common species group observed in the digital video aerial surveys. Fish were the next most commonly observed non-avian animals; individually recognizable larger fish (>1 m in length) were counted as individual fish even if they were located within a school, and these are the only data presented in figures in this chapter. However, most fish observed in footage were in groups of small forage fish, or "bait balls," of varying sizes, which were observed mostly between May and September, primarily inshore. The majority of bait balls were seen on the September 2013 survey (4,142 schools of fish), and 7,514 schools were observed in all (61% were observed in the Maryland Project transects). Some schools were smaller than a square meter while others extended across all four cameras and spanned many frames of footage (school size was not quantified during video analysis). Additional discussion of bait ball geographic patterns may be found in Chapter 17.

Dolphins were the most commonly observed marine mammals in the aerial video. Dolphins were seen throughout the study period, but Bottlenose Dolphins (*Tursiops truncatus*, the most commonly identified species; Appendix 5A) were most abundant in the spring and summer. Large cetaceans were observed as well, with eight endangered North Atlantic Right Whales (*Eubalaena glacialis*) observed in February and March of 2012-2014. One instance provided still images of high enough quality to identify a known female, named "Blackheart", with her calf on their migration north to the Gulf of Maine (Figure 5-6). Three Minke Whales (*Balaenoptera acutorostrata*), two Humpback Whales (*Megaptera novaeangliae*), and one Fin Whale (*B. physalus*) were also observed. Whales were observed predominantly in the autumn, winter, and early spring (Table 5-2).

A notable number of sea turtles were observed (1.63% of total survey observations), primarily in the spring, summer, and autumn. Loggerhead Turtles (*Caretta caretta*) and Leatherback Turtles (*Dermochelys coriacea*) were the two most commonly identified to species, with some observations of rarer species (Kemp's Ridley Turtle, *Lepidochelys kempii*; Green Turtle, *Chelonia mydas;* and Hawksbill Turtle, *Eretmochelys imbricata*; Appendix 5A).

### Identification rates

Identification rates varied by survey and season. March surveys had the highest rate of birds identified to species due to the number of highly identifiable Black Scoters observed. However, image quality, observer bias, and other factors could also have varied through time and influenced identification rates.

Identification rates for Anatidae (geese, swans, and ducks) were strong relative to the rates for other avian groups (Figure 5-7), with 53% of anatids identified to species. Only 15% of loons were identified as either Red-throated Loon (*Gavia stellata*) or Common Loon (*G. immer,* Appendix 5A), as the video footage was not always clear enough to distinguish plumage coloration, and there is significant size overlap between Red-throated Loons and Common Loons in the mid-Atlantic study area (Gray et al., 2014). Gulls and terns were identified to species 35% of the time, with Bonaparte's Gulls the most common of identified gull species (1% of total observed animals). Small birds like auks and terns were seldom identified to species (6%, Figure 5-7), often due to difficulty in picking out fine details in plumage variation.

Few individual fish were identified to species, as this taxon was not a focus of the current study, but video data will remain archived in case additional analysis of species identities or baitfish school sizes is warranted. Most non-Leatherback Turtles remained unidentifiable at the species level because of inconclusive carapace length measurements and/or insufficient detail visible on the carapace (often due to the animal being too deeply submerged in the water column to allow for detailed observation). While most turtles were not identified to species (79%, Figure 5-8), all species observed in the area are federally endangered. Of all toothed whales (Odontoceti), including dolphins, 51% were not identified to species level, again in part due to animals being submerged to varying depths in the water column.

### *Flight heights*

Flight heights were estimated for 75% of flying animals (or 5,299 animals). Of all birds with estimable flight heights, 59% were estimated to be flying within 0-20 meters of the water's surface. Thirty eight percent of observations occurred between 20 and 200 m in altitude (1,990 observations), a range that was used in one recent study to cover a variety of possible turbine types, foundations, and variations in tidal heights (Willmott et al., 2013). We observed nearly every avian taxonomic group flying within this zone, though the proportion of individuals in this latitude band varied by taxon. Within this range, 19% of birds were flying from 20-50 m, 12% were flying from 50-100 m, and 6% were flying from 100-200 m. An additional 3% of birds were flying above 200 m.

Of the birds with estimated flight heights, the seven most commonly observed avian families were all marine birds that forage in the study area and spend some time on the surface of the water, and were by far most commonly observed in the lowest 0-20 m altitude band (Figure 5-9, Figure 5-10). Gulls and terns were the most commonly observed species aloft, followed by Northern Gannets. Gulls and terns were observed flying at the 20-50m flight band 19% of the time, 50-100 m 12% of the time, and 100-200 m 7% of the time. Gannets had a similar distribution, and were observed flying at 20-50 m 22% of the time, 50-100 m 15% of the time, and 100-200 m 6% of the time. Scoters, ducks, and geese were generally observed flying lower, at 20-50 m 12% of the time, 50-100 m 3% of the time, and 100-200 m 3% of the time. Loons were also flying lower, in the 20-50 m altitude band 22% of the time, 50-100 m 4% of the time, and 100-200 m 1% of the time (for more details see Figure 5-10). Species groups that were less commonly observed in digital video aerial surveys had a more varied altitudinal distribution (Figure 5-11); cormorants and shorebirds both showed a split distribution between the lowest 0-20 meter band and the >100 meter bands. Passerines were observed at all height bands, but most were observed at 200+ meters, as were most shorebirds, egrets and herons (Figure 5-11).

Fifteen Eastern Red Bats (*Lasiurus borealis*) were detected by observers in the September 2012 and 2013 digital video aerial surveys (Appendix 5A; Hatch et al. 2013). Fourteen of the bats were observed in one survey day in September of 2012, while an additional bat was seen on the September 2013 survey. Flight heights were estimated for seven of the 15 bats observed, and all fell into the >200 m flight height category, meaning that all bats with calculated flight heights were flying higher than the likely rotor-swept zone for offshore wind turbines. Bats were observed between 16.9 and 44 km offshore of Delaware, Maryland, and Virginia. These observations were notable as they provided new evidence of offshore migrations of red bats, how high they fly while on migration, and the time of day the migrations may occur. Additional information may be found in Hatch et al. (2013).

### *Rays*

Rays (Batoidea) represented over 44% of all observations from the digital video aerial surveys (Table 5-2). Cownose Rays were the most common ray species observed (54% of all rays, and almost 100% of all rays identified to species; Figure 5-8). Rays were not identified to species unless they were individually identifiable and their characteristic noses were clearly visible, so many of the rays present in Cownose Ray schools (though they were not identifiable as Cownose Rays themselves) were likely also of the same species; the overwhelming majority of rays in video footage are thought to have been

Cownose Rays. Some schools of rays were so densely packed and submerged that individuals could not be discerned, and these were identified as schools rather than as individuals (78 schools). These schools were primarily found in September (53 schools) when rays migrate through the study area (Goodman et al., 2011).

Rays were primarily observed during the summer and fall surveys (Figure 5-5), though there was a high level of variation between the two survey years: many more rays were observed in 2013 compared to 2012, with nine times as many rays observed in September 2013 than September of 2012 (Table 5-2). The differences in observations between the two years may reflect variation in water temperatures, timing of migration movements relative to our survey dates, or differences in migration behaviors. Rays additionally showed distinct monthly variation in abundance and distribution. Rays were distributed more broadly in the early summer surveys, June 2012 and July 2013 (Figure 5-12). More rays were seen in the July 2013 survey, and they were predominantly found further north up the coast of Virginia and Maryland compared to June 2012, when they were mostly found off the coast of Virginia and Chesapeake Bay. Rays in the September surveys were much more densely packed in high density pockets throughout the study area, but the 2013 survey had densities up to fifteen times those of the 2012 survey (Figure 5-12).

## Discussion

### *Digital video aerial surveys and aquatic taxa*

Aerial surveys, and particularly digital video aerial surveys, have been noted to reduce glare and increase visibility for aquatic animals such as sea turtles when compared to boat-based surveys (Normandeau Associates Inc. 2013), and we saw similar results in our study (Chapter 14). The high altitude of digital aerial survey aircraft also reduces disturbance compared to low-flying observation planes or survey vessels (Chapter 13), which may play a role in increased detections of aquatic animals (Normandeau Associates Inc. 2013). We discuss these differences in more detail in Chapters 13-14, where we directly compare the results of the two survey approaches.

We examine ray distributions and abundances in some detail in this chapter, as they were the most abundant animal in aerial surveys, and provided a good example of the use of digital video aerial surveys to monitor aquatic animals. Our study was the first to use digital video aerial surveys to monitor ray distributions and densities. Our findings not only illustrate the utility of the digital video aerial surveys for documenting the distributions of Cownose Rays, and aquatic animals in general, but add to the limited knowledge of Cownose Ray migratory movements in the mid-Atlantic (Blaylock, 1993; Goodman et al., 2011). There is a continued risk of overfishing Cownose Rays, and a need to establish a baseline population assessment and develop an effective conservation and management plan (Goodwin, 2012). Additionally, rays could be affected by the formation of artificial reefs, as turbine foundations provide new habitats for benthic organisms (Andersson, 2011; Zucco et al., 2006). However, it is not clear whether Cownose Rays forage offshore during migration (e.g., in locations where turbines would be placed), so the potential for indirect effects to this taxon from such ecological changes is likewise unclear.

Many elasmobranchs are both magnetosensitive and electrosensitive, senses which are thought to be used to locate prey, predators, or conspecifics, as well as for navigation (Normandeau Associates Inc. et al., 2011). As a result, elasmobranchs can detect electromagnetic fields (EMF) produced by power transmission cables in the marine environment, including cables associated with offshore wind development (Gill et al., 2009; Normandeau Associates Inc. et al., 2011). The strength of the electric and magnetic fields emitted by a cable, and thus the distance from the cable at which the fields are detectable, depends on a variety of factors, including the type of cable (e.g., AC vs. DC) and whether it is buried or sheathed (Normandeau Associates Inc. et al., 2011). It has been hypothesized that EMF could affect the navigation or foraging behaviors of these species, possibly causing disruption of migratory routes or influencing foraging patterns, although evidence of such effects is limited, and the results of the limited experimental studies on rays have been mixed (Boehlert and Gill, 2010; Gill, 2005; Gill et al., 2009). Experiments using EMF of similar types and intensities to those emitted by sub-sea cables showed some response by the EM-sensitive benthic Thornback Ray (*Raja clavata*), with some individuals showing increased searching effort for prey in the presence of EMF (presumably because the EMF were similar to those emitted by prey), but the response was not predictable (Gill et al., 2009). Cownose Rays do use electroreception to detect their prey, but their ability to detect and tendency to react to EMFs from sub-sea cables have not yet been determined. In addition, the species could only be affected by EMF if they are at or near the ocean floor, within range of the fields (Boehlert and Gill, 2010). While the species is known to forage for mollusks on the seafloor in coastal bays during the summer breeding season (Smith and Merriner, 1985), it is unknown whether they behave similarly during migration, as we were only able to observe rays in the upper few meters of the water column.

### *Distribution and relative abundance patterns*

Scoters were the most abundant bird group observed in the aerial data, with gannets and loons also observed in large numbers. This pattern was similar to that found in the boat-based surveys (Chapter 8), though much higher numbers of birds were found in boat surveys (Chapter 14). The mid-Atlantic region is an important wintering ground for these species (Barr et al., 2000; Bordage and Savard, 2011; Mowbray, 2002; Savard et al., 1998), and in this study, all three of these species groups were most commonly found in the study area in the winter and spring. The timing of seasonal variations in abundance of scoters, gannets, and loons was similar to that of migratory movements indicated by individual tracking of Surf Scoters, Northern Gannets, and Red-throated Loons (Chapters 20-23). Gulls and terns were the most abundant bird group in the summer and fall, when several species were breeding onshore and foraging in the study area, though Bonaparte's Gulls were most abundant in the winter (Nisbet et al., 2013). Fewer aquatic animals were seen in the study area in the winter, but many fish were observed in spring, and rays were extremely abundant in the summer and fall. Toothed whales were observed in similar numbers throughout the year, though the species composition changed over the course of each year, with Bottlenose Dolphins present in the warmer months, and Common Dolphins (*Delphinus delphis*) more abundant in the cooler months (Chapter 15).

A few large whales were seen in the study area, mostly in cooler months, including several observations of endangered North Atlantic Right Whales. Though a large number of documented mortalities have occurred in the mid-Atlantic region (Firestone et al., 2008), and North Atlantic Right Whales are known to use the study area as a migration corridor, little is known of their movements in the region, and even the few observations made in this study provide useful new information about a period in their life cycle that is not well understood. Some passerines were observed migrating through the study area as well, more commonly from the boat platform than in aerial video. Most passerines migrate at night, however, when surveys did not occur (Chapters 26-27).

### *Bats*

Other notable observations included Eastern Red Bats, which were seen migrating in the daytime through the study area during the fall, providing evidence for offshore migration of this species. While the number of bats observed was small overall (n = 15), most of those sightings occurred on one survey day, likely because it was flown during a peak migration time for Eastern Red Bats. Little is known about the migration routes of migratory tree bats in the U.S., but Eastern Red Bats are the most frequently encountered bat species off the east coast during fall migration; autumnal sightings at sea have been recorded dating back to 1890, usually as individuals but sometimes in large flocks (Hatch et al., 2013). Bats likely prefer to migrate at night, and most of the previous offshore observations took place at night, but there have been previous daytime fall migration sightings as well.

Marine surveys are poorly designed to detect bats and other small migrants aloft, due to their size (and thus low detectability, particularly if they are flying more than a few tens of meters above sea level), as well their primarily nocturnal migration behaviors. The flight heights at which bats were observed in our surveys was unexpected, given the limited data available on bat movements in the offshore environment (Ahlen et al., 2009), and suggests that some bats migrating offshore may fly at altitudes that prevent visual detection and identification under most circumstances. The altitude at which our surveys were flown, in combination with our ability to zoom in on small high-flying organisms for identifications, may have allowed for higher detection rates than other survey methods (Hatch et al., 2013).

Bat fatalities have been documented at offshore wind facilities in Europe (European Environmental Agency, 2009). Eastern Red Bats migrate over land in large numbers, where they make up the greatest proportion of bats killed at terrestrial commercial-scale wind facilities (Arnett et al., 2008), and they are also the species most often observed at sea in the eastern U.S. While the proportion of the population that migrates offshore remains unknown, Eastern Red Bats are probably the bat species most likely to interact with future offshore wind developments in the mid-Atlantic (Hatch et al., 2013).

## ***Flight heights and collision risk***

Flight height data is often used alongside information on avoidance behaviors, turbine specifications, and other data in models that attempt to estimate avian collision risk for offshore wind energy projects in Europe (e.g., Band 2012), although there is still debate in the European literature regarding the factors that best predict this risk (e.g., Cook et al. 2012, Douglas et al. 2012, Langston 2013, Furness et al. 2013). Flight heights are suspected to vary in relation to weather and time of day, for example, so collision risk is likely to be highest at night, and in particular on nights with poor visibility (Dirksen et al., 2000; Hill et al., 2014; Hüppop et al., 2006). Our surveys were limited to daytime hours and periods of clear weather, when cameras had adequate visibility for observing and identifying animals (Chapter 3), which may limit the applicability of these flight height data for estimating collision risk. Despite these limitations, the suspected importance of flight height data in predicting collision risk means that these data are used in assessments of relative vulnerability of various taxa to offshore wind energy development (e.g., Furness et al., 2013; Garthe and Hüppop, 2004; Willmott et al., 2013).

In our study, we compared the estimated flight heights of birds and bats in relation to the potential rotor-swept zones (RSZ) of offshore wind turbines. The RSZ depends on the turbine type and may be specific to each project; for example, the RSZ for Siemens 3.6 MW offshore turbines is about 28-132 m, while the RSZ for Siemens 6 MW turbines is about 27-177 m[2], though specific altitudes will vary based on site-specific attributes. Larger turbines are also possible, and prototypes have already been deployed in some locations in Europe[3]. While the majority of birds were observed flying below 20 meters, and thus below the expected RSZ, 38% of observations occurred between 20 and 200 m in altitude (1,990 observations), and nearly every avian taxonomic group was observed within this zone at some point in our study. Gannets, gulls and terns, and loons all had high proportions of birds within this altitude range (Figure 5-10). Most passerines that were detected were flying above 200 meters.

### *Species identifications*

Identification rates for some animal groups were low in this study. In future, it is likely that many of the issues related to identification rates and lower-confidence observations that occurred in this study will be addressed through technical improvements to the camera systems. The newer generation of cameras currently used in Europe have a greatly improved identification rate as compared to those used in this study (HiDef Aerial Surveying Ltd. unpubl. data).

Analytical approaches can also help address this issue. Chapter 16 uses boat data and environmental covariates to develop species-specific estimates of distribution and abundance from the digital video aerial dataset. Additionally, the development and use of a metric for image quality, which could be applied to all video data, would be helpful for assessing identification rates relative to changing atmospheric conditions (see recommendations in Chapter 6). Inter-observer and inter-survey bias in species identifications could also be examined using a double observer approach during video analysis. This approach would be relatively straightforward to incorporate into existing audit protocols for object location and species identification.

Early digital video aerial surveys were conducted at 2 cm GSR in some areas, and 3 cm GSR in others. Initial review of these video data indicated that, despite the high number of easily identifiable scoters in early surveys, the clarity of the 3 cm video was not sufficient to identify many taxa to species (Table 5-3; also see Chapter 6, Table 6-2). The study design was adjusted beginning in September 2012 to conduct all survey flights at 2 cm GSR. While this reduced the sampled area for the sawtooth transects from roughly 3.2% of the study area to 2.1% (since a higher GSR necessitates a narrower transect strip),

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[2] http://www.energy.siemens.com/hq/en/renewable-energy/wind-power/platforms/

[3] http://www.windpowermonthly.com/10-biggest-turbines

project collaborators felt it was necessary to improve video clarity and species identification rates. Newer generations of these camera systems, currently in operation in Europe, have a wider strip width and better clarity and color rendition, thus rendering this tradeoff largely unnecessary (A. Webb pers. comm.).

### *Other analyses of digital video aerial survey data*

Chapters in Part IV of this report further analyze digital video aerial data, either separately or alongside boat survey data. Several chapters focus on contrasting boat and digital video aerial survey approaches (Chapters 13-14 and 18). In other cases, digital video aerial survey data and boat survey data are used jointly (Chapters 15-17 and 19) to describe distributions and abundance of animals across the study area.

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## Figures and tables

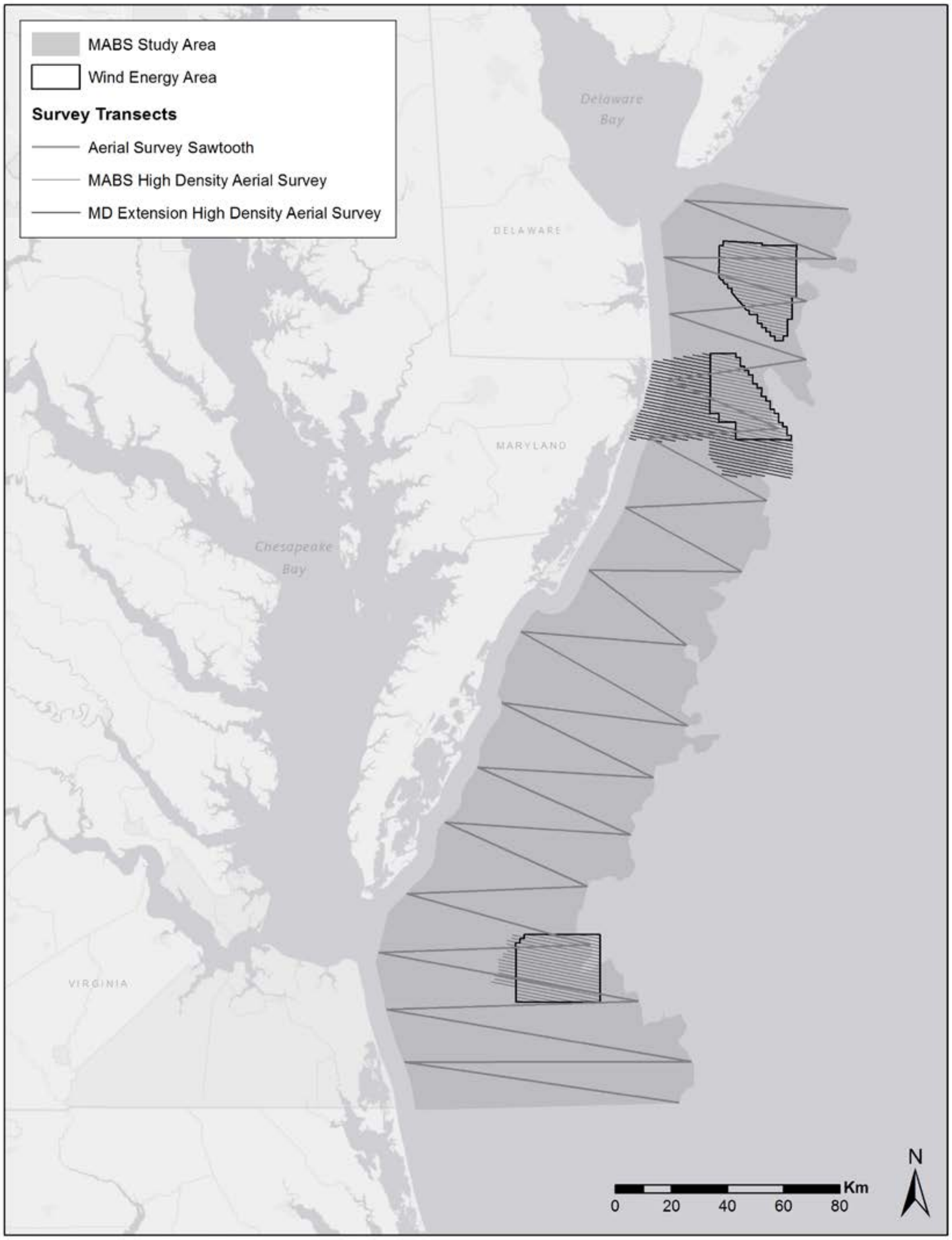

**Figure 5-1. Aerial survey transects for the Mid-Atlantic Baseline Studies and Maryland projects (2012-2014).** Light grey transects are part of the Mid-Atlantic Baseline Studies; darker grey transects off the coast of Maryland are part of the Maryland Project (surveys conducted in March 2013-May 2014).

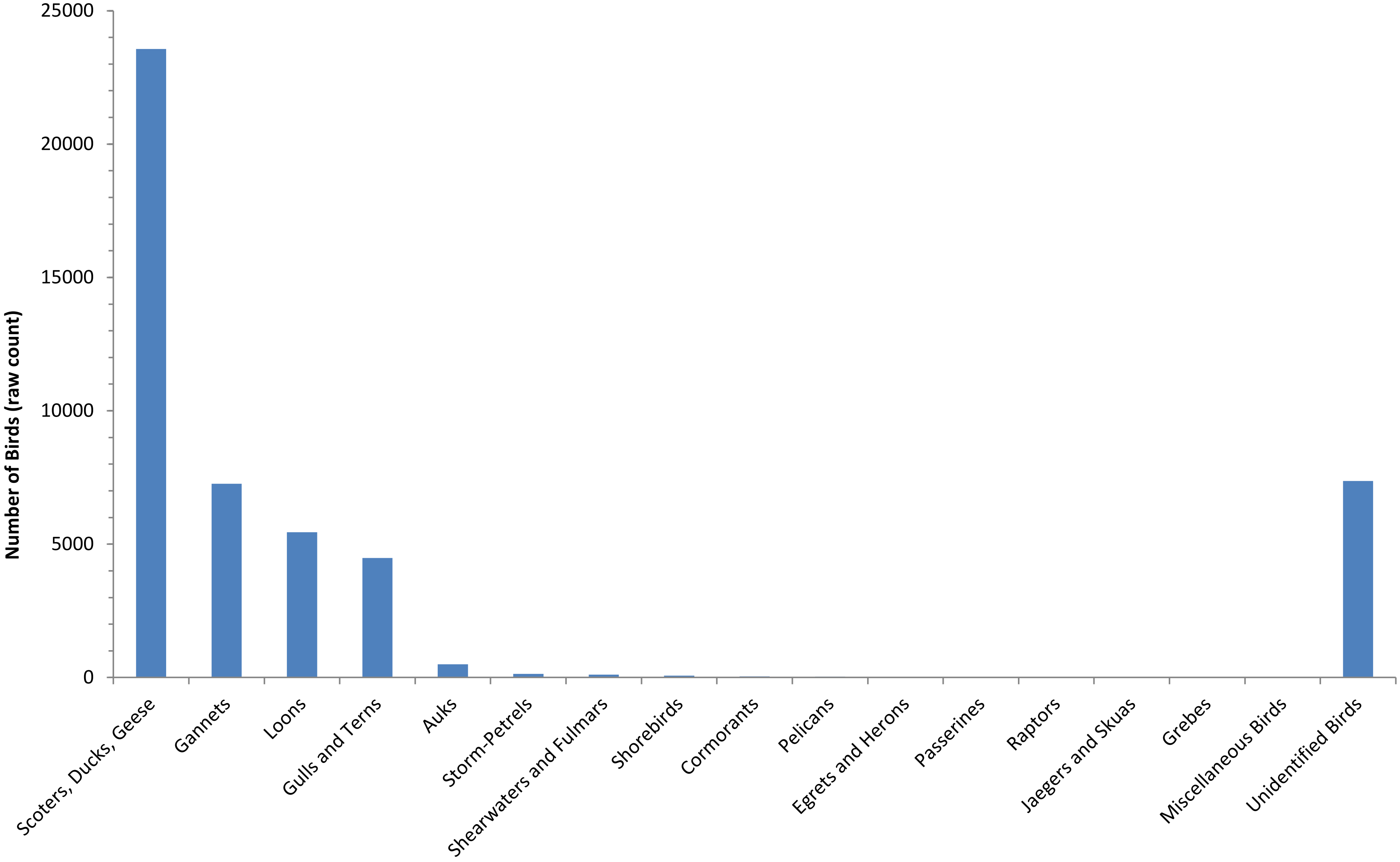

**Figure 5-2. Avian observations (raw counts) from the digital video aerial surveys, by family (March 2012 – May 2014).** Unidentified birds are all birds not identified to species or to any higher level taxonomic group. Birds from all levels of identification are taken at face value (e.g., a “possible” Northern Gannet is counted as a Northern Gannet; see Chapters 3-4).

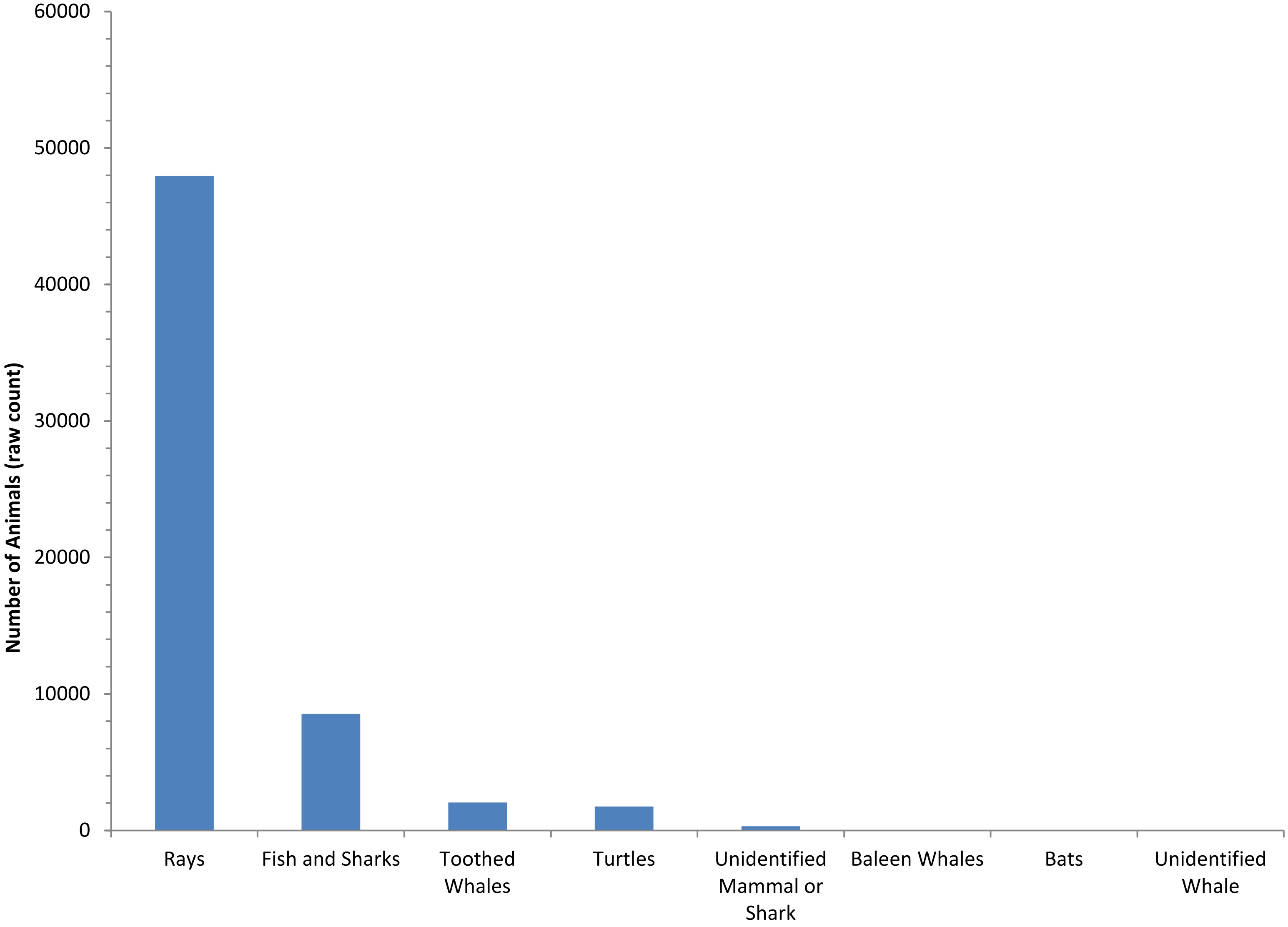

**Figure 5-3. Observations from the digital video aerial surveys of other non-avian animals by family group (March 2012 – May 2014).** Numbers do not include schools of rays or fishes, so these data are an underestimate of the total counts for these animals.

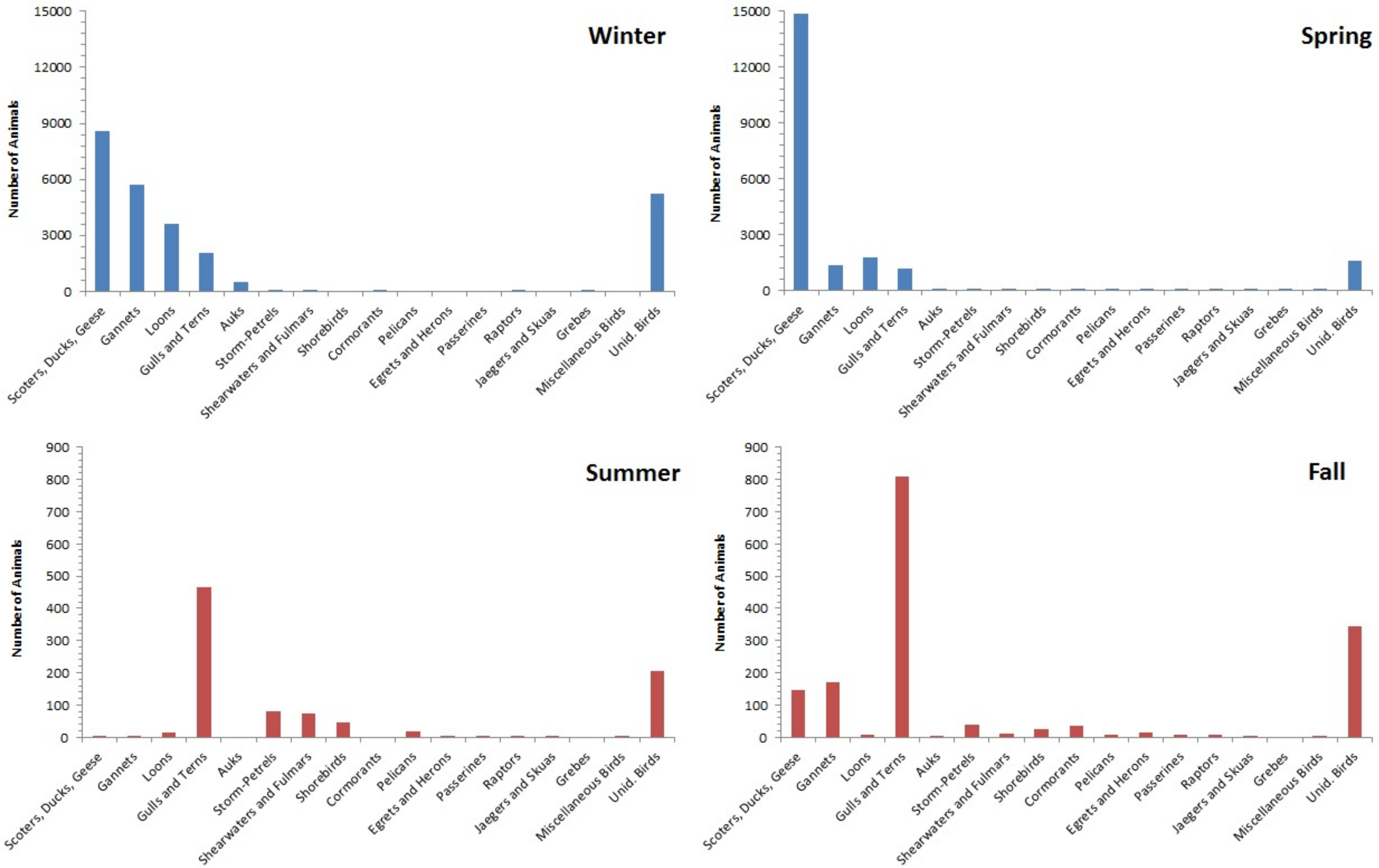

**Figure 5-4. Abundance of birds by family or group in winter (December to February), spring (March and May), summer (June, July, August), and fall (September and October).** Note different y-axes between the top and bottom graphs. X-axes are in order of overall abundance by family or group across all surveys.

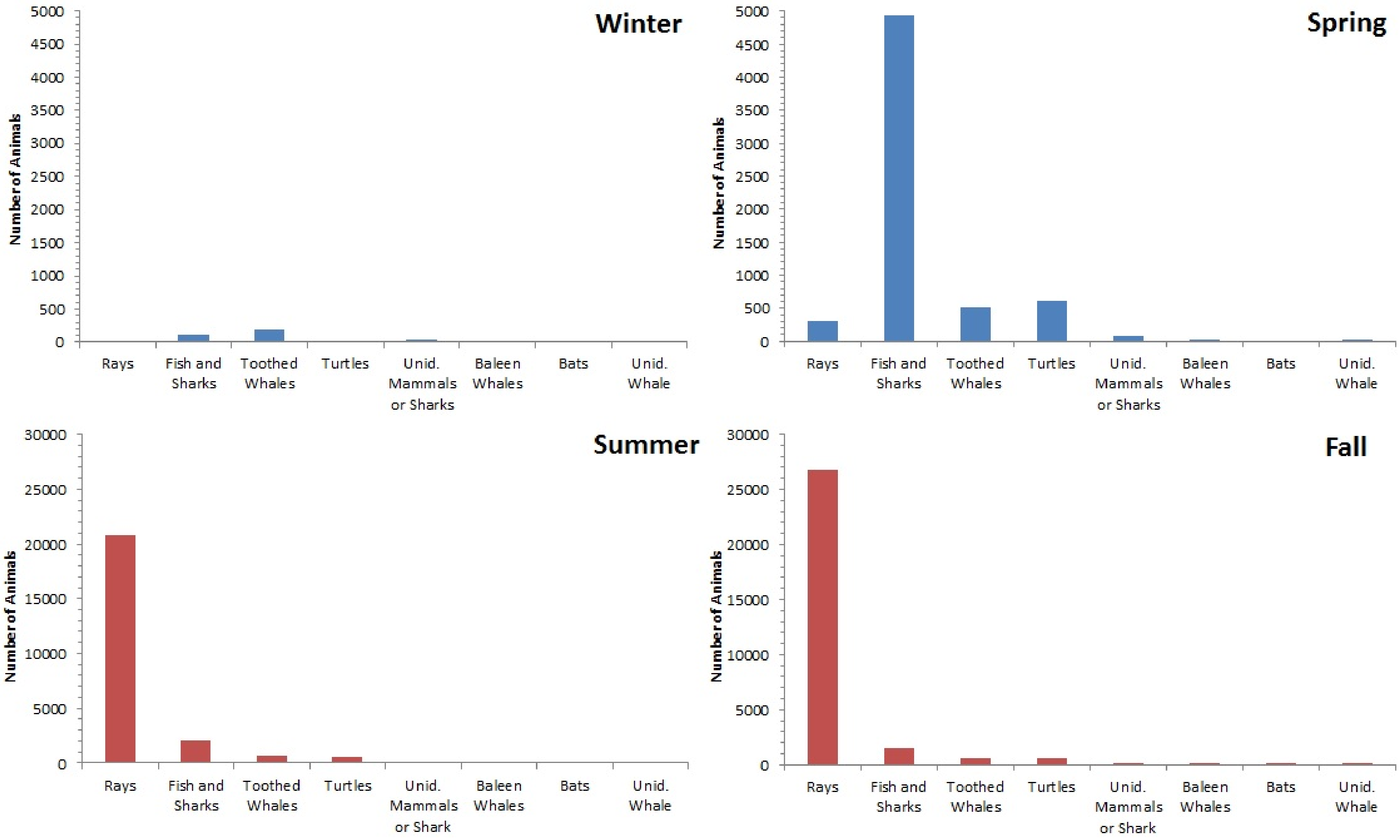

**Figure 5-5. Abundance of non-avian animals by group in winter (December to February), spring (March and May), summer (June to August), and fall (September and October**). Note different y-axes between top and bottom graphs. X-axes are in order of overall abundance by family or group across all surveys.

**Figure 5-6. Two of the eight North Atlantic Right Whales (*Eubalaena glacialis*) observed in the aerial footage.** Blackheart (#3390) and her calf were observed on February 16, 2013, 56 kilometers offshore of Virginia. All recorded footage of Right Whales was passed on to the North Atlantic Right Whale Consortium (NOAA) and the New England Aquarium.

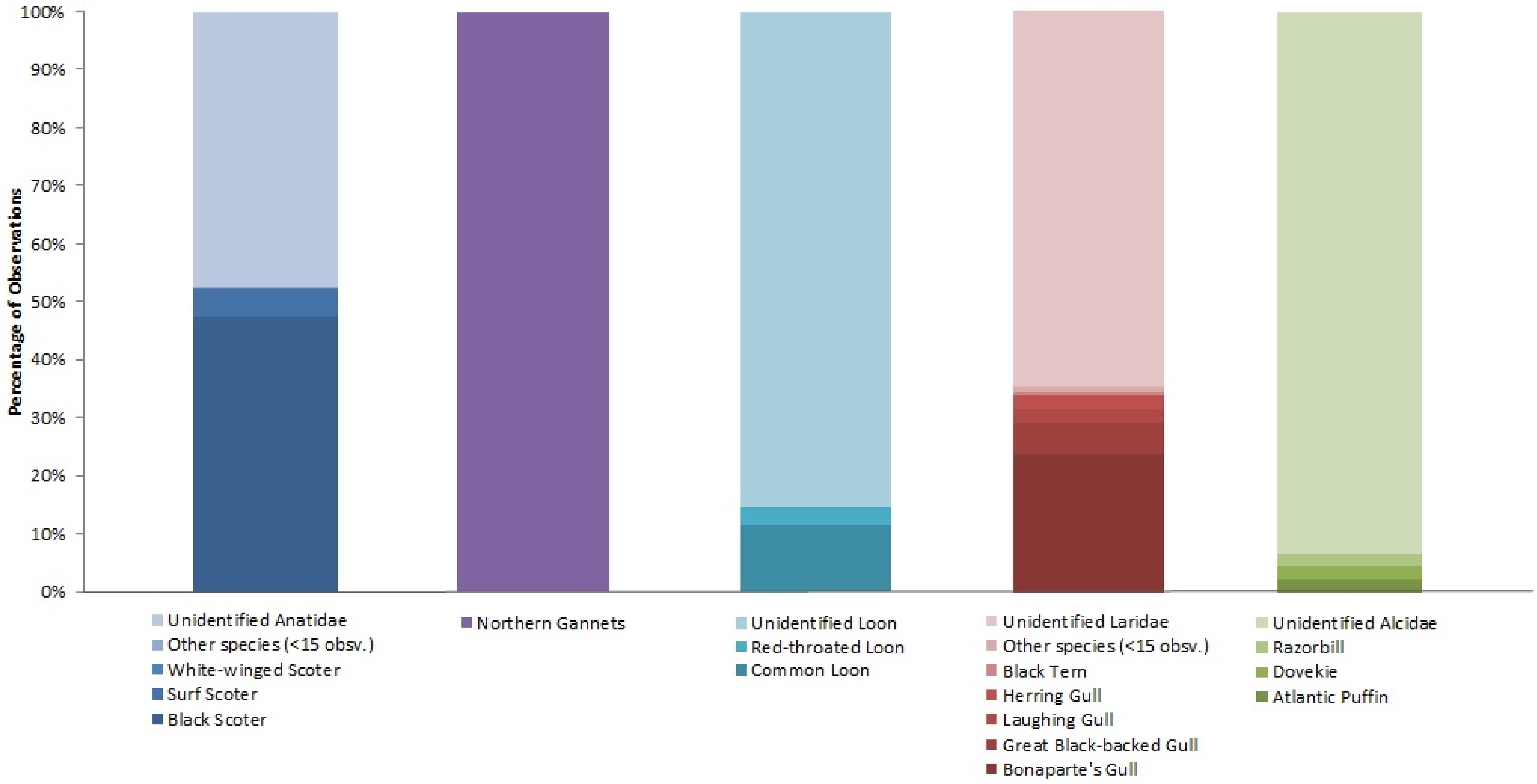

**Figure 5-7. Identification rates for the most abundant avian families in the digital video aerial surveys.** Identifications to species level are shown in darker colors. “Other Species” in the Laridae (red, n = 4475) and Anatidae (dark blue, n=21,146) columns can be found in Appendix 5A. Sample sizes for gannets, loons, and auks are 7,126, 5,407, and 495 respectively. Birds from all levels of identification are taken at face value (see Chapters 3-4).

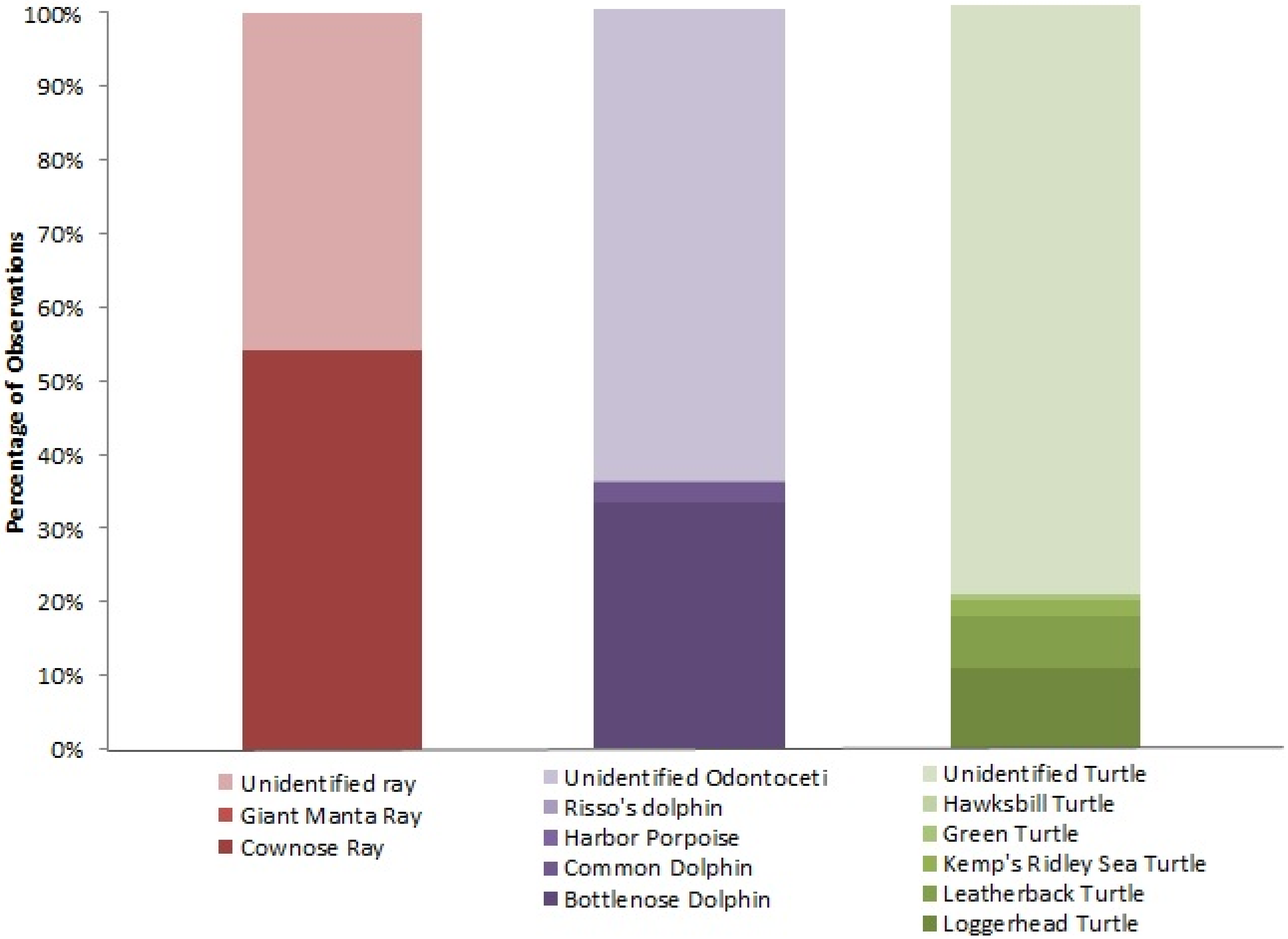

**Figure 5-8. Identification rates for common aquatic animal groups in the digital video aerial surveys.** Identifications to species level are shown in darker colors. Sample sizes are 47,945 for rays (red), 2,028 for Odontoceti (purple), and 1,748 for turtles (green).

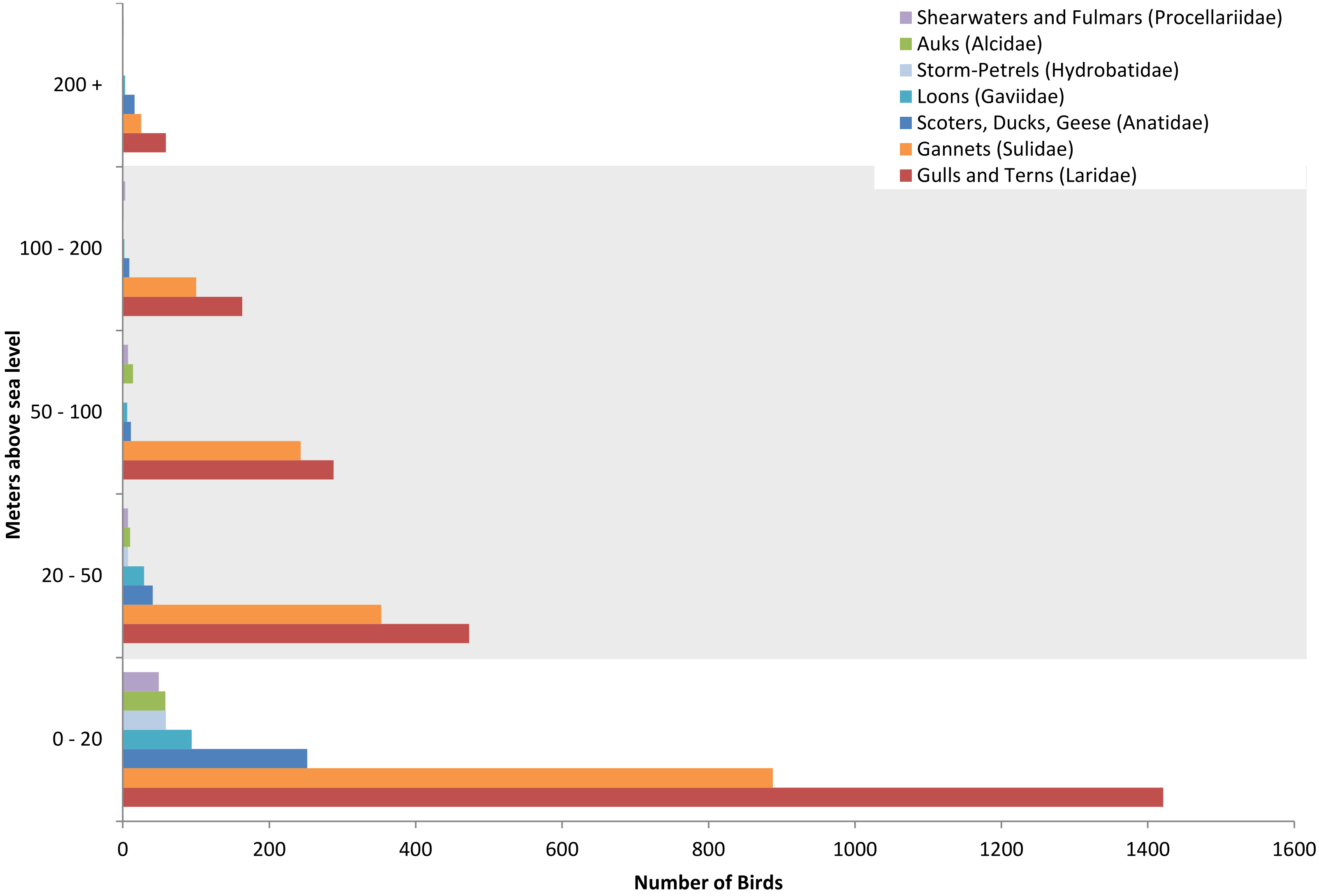

**Figure 5-9. Flight height above sea level (meters) of the most abundant bird families from the digital video aerial surveys, presented as raw counts.** Data are presented as number of animals observed at the given height range. All confidence levels of animal identifications and flight height estimates are included for this figure. Grey hatch marks indicate an approximate range of altitudes for the rotor-swept zone for offshore wind turbines (see text).

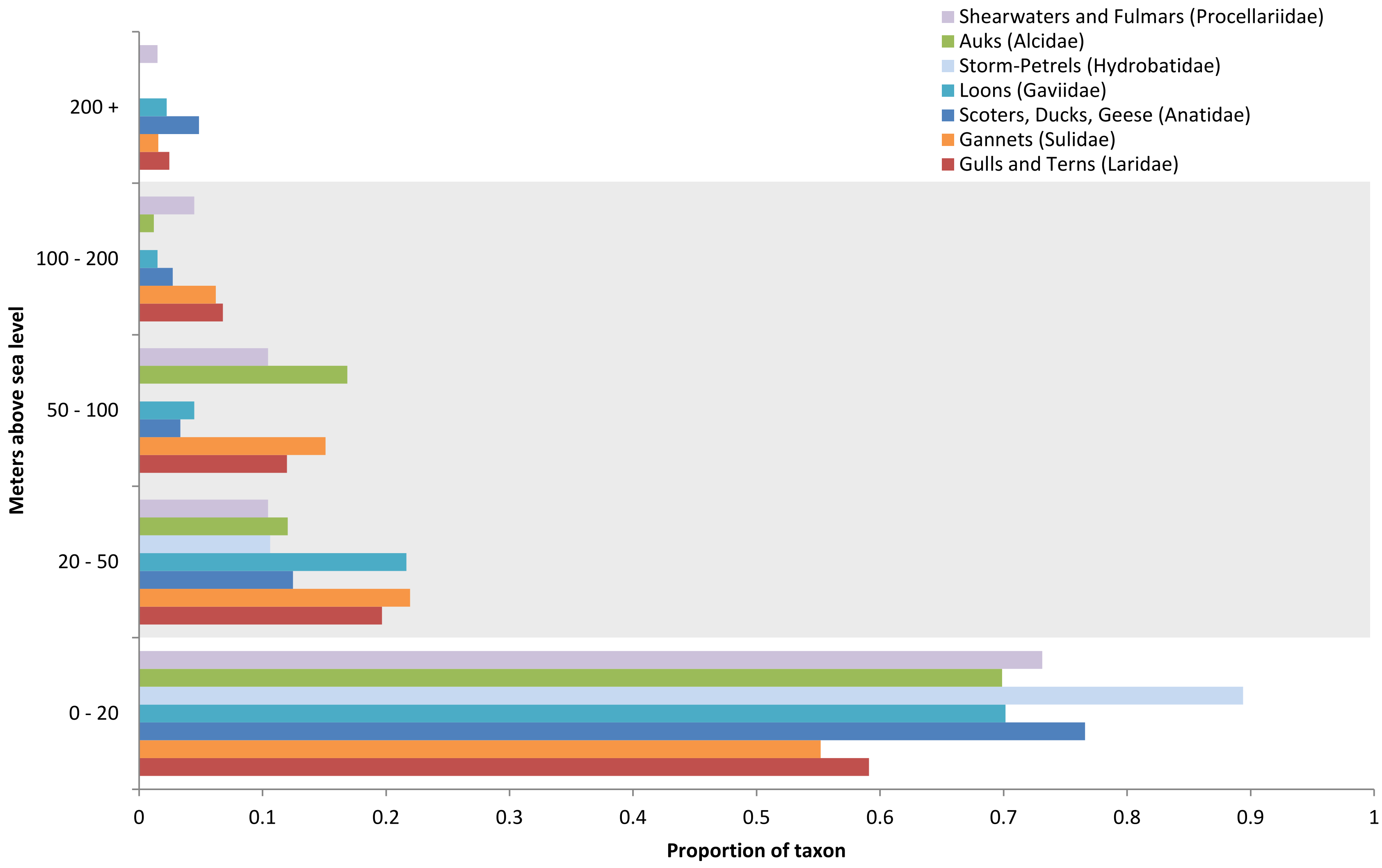

**Figure 5-10. Flight height above sea level (meters) of the most abundant bird families from the digital video aerial surveys, presented as proportions of each taxon.** Data are presented as the proportion of each species group observed at the given height range. All confidence levels of animal identifications and flight height estimates are included for this figure. Grey hatch marks indicate an approximate range of altitudes for the rotor-swept zone for offshore wind turbines (see text).

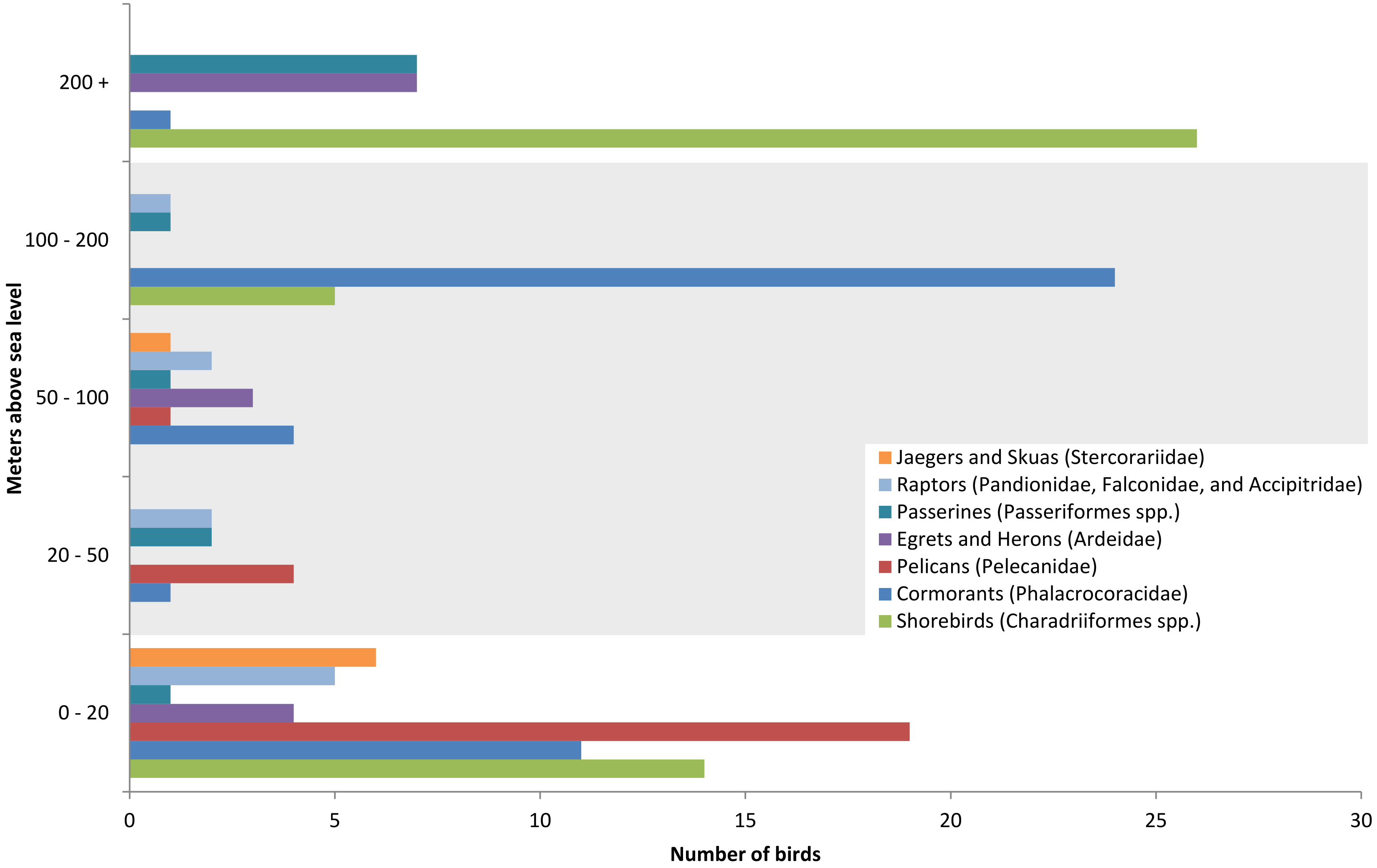

**Figure 5-11. Flight height above sea level (meters) for seven less abundant bird families or groups from the digital video aerial surveys, presented as raw counts.** In several cases, less common families have been combined into broader taxonomic categories (e.g., “passerines”). Data are presented as number of animals observed at the given height range. All confidence levels of animal identifications and flight height estimates are included for this figure. Grey hatch marks indicate an approximate range of altitudes for the rotor-swept zone for offshore wind turbines (see text).

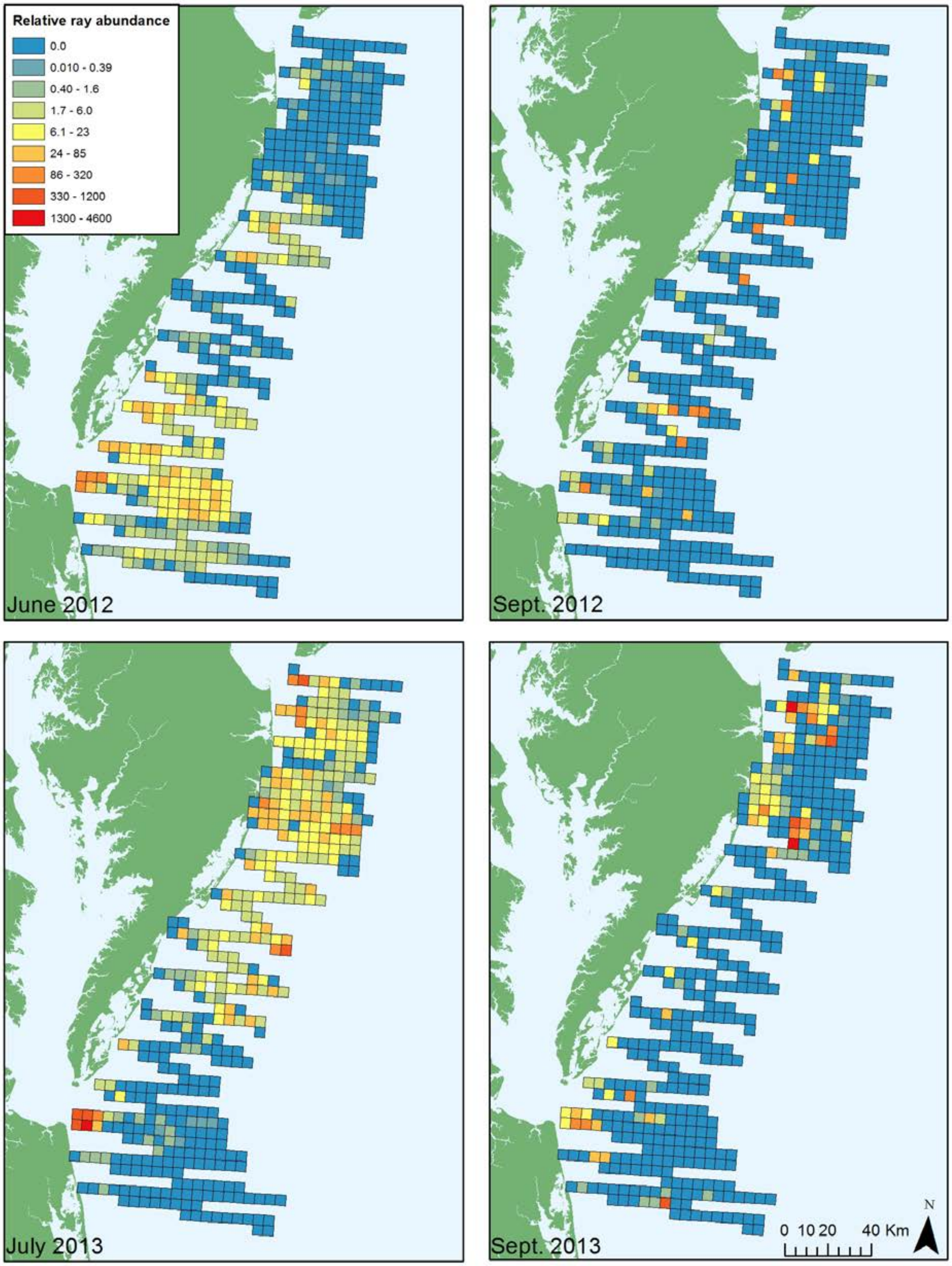

**Figure 5-12. Effort-corrected counts of rays (Batoidea) within Bureau of Ocean Energy Management (BOEM) lease blocks (4.8 x 4.8 km), for the four surveys when they were the most abundant.** Count data were corrected by area surveyed (km$^2$) within each lease block. Values have not been corrected for detection bias, and should be considered as relative estimates of density, not as estimates of actual ray densities.

**Table 5-1 Weeks in which digital video aerial surveys were completed during the Mid-Atlantic Baseline Studies Project.** Each survey took from one to eleven survey days to complete, depending upon weather, plane availability, and other factors. Surveys colored in gray only included Mid-Atlantic Baseline Studies transects; surveys in blue included Maryland Project transects as well. The survey noted in pale blue (August 2013) included only Maryland Project transects and coverage of the Maryland WEA.

| | Jan. | | | | Feb. | | | | Mar. | | | | Apr. | | | | May | | | | Jun. | | | | Jul. | | | | Aug. | | | | Sep. | | | | Oct. | | | | Nov. | | | | Dec. | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **2012** | | | | | | | | | | | | ■ | | | | | ■ | | | | | | ■ | | | | | | | | | | | ■ | | | | ■ | | | | | | | | ■ | ■ | |
| **2013** | | | | | | ■ | ■ | | | | 🟦 | | | | | | | | | | | | | | 🟦 | 🟦 | 🟦 | | | | | ▧ | 🟦 | | | | | | 🟦 | | | | | | | | 🟦 | 🟦 |
| **2014** | 🟦 | 🟦 | | | | | 🟦 | 🟦 | 🟦 | | | | | | | | | | 🟦 | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |

**Table 5-2. Summary data for the digital video aerial surveys (by species group).** Data from the aerial-boat comparison study (Chapter 13) are not presented. Data are presented in order of abundance based on the total count from all surveys. Counts include definite, probable, and possible identifications (Chapters 3-4). Surveys from March 2013 onwards included an additional ~21% ground coverage from Maryland Project transects (surveys noted in blue); the August 2013 survey included only the Maryland WEA and Maryland Project transects, but excluded the remainder of transects offshore of Delaware and Virginia (as well as the sawtooth transects throughout the study area) conducted in the remainder of surveys.

| Animal Group | Mar. 2012 | May 2012 | Jun. 2012 | Sept. 2012 | Oct. 2012 | Dec. 2012 | Feb. 2013 | Mar. 2013 | Jul. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Feb. 2014 | May 2014 | Grand Total | % of Total* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Scoters, Ducks, Geese (Anatidae) | 9408 | 1 | 1 | 0 | 0 | 487 | 1691 | 2876 | 0 | 0 | 0 | 147 | 4653 | 1737 | 145 | 21146 | 19.76% |
| Unidentified Birds (Aves spp.) | 545 | 218 | 81 | 120 | 48 | 736 | 1852 | 538 | 41 | 83 | 99 | 78 | 1667 | 972 | 209 | 7287 | 6.81% |
| Gannets (Sulidae) | 337 | 72 | 0 | 0 | 52 | 421 | 3730 | 821 | 1 | 0 | 1 | 119 | 728 | 839 | 5 | 7126 | 6.66% |
| Loons (Gaviidae) | 614 | 460 | 8 | 0 | 3 | 719 | 967 | 496 | 3 | 2 | 1 | 2 | 907 | 1047 | 178 | 5407 | 5.05% |
| Gulls and Terns (Laridae) | 552 | 332 | 73 | 120 | 178 | 737 | 118 | 69 | 172 | 222 | 302 | 210 | 1148 | 60 | 182 | 4475 | 4.18% |
| Auks (Alcidae) | 0 | 0 | 0 | 0 | 0 | 154 | 193 | 13 | 0 | 0 | 0 | 1 | 78 | 59 | 0 | 498 | 0.47% |
| Storm-Petrels (Hydrobatidae) | 1 | 0 | 53 | 38 | 0 | 12 | 0 | 0 | 21 | 5 | 0 | 1 | 0 | 0 | 5 | 136 | 0.13% |
| Shearwaters and Fulmars (Procellariidae) | 0 | 0 | 74 | 5 | 2 | 14 | 4 | 0 | 0 | 0 | 1 | 3 | 2 | 6 | 1 | 112 | 0.10% |
| Shorebirds (Charadriiformes spp.) | 2 | 0 | 0 | 20 | 0 | 0 | 0 | 0 | 39 | 7 | 4 | 2 | 0 | 0 | 0 | 74 | 0.07% |
| Cormorants (Phalacrocoracidae) | 0 | 0 | 0 | 1 | 25 | 1 | 0 | 0 | 0 | 0 | 0 | 8 | 1 | 0 | 6 | 42 | 0.04% |
| Pelicans (Pelecanidae) | 0 | 2 | 2 | 3 | 1 | 0 | 0 | 0 | 17 | 0 | 2 | 2 | 0 | 0 | 1 | 30 | 0.03% |
| Egrets and Herons (Ardeidae) | 0 | 2 | 0 | 7 | 7 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 17 | 0.02% |
| Passerines (Passeriformes spp.) | 0 | 0 | 1 | 2 | 4 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 7 | 17 | 0.02% |
| Raptors (Pandionidae, Falconidae, and Accipitridae) | 0 | 1 | 1 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 2 | 6 | 0 | 0 | 2 | 14 | 0.01% |

| **Animal Group** | **Mar. 2012** | **May 2012** | **Jun. 2012** | **Sept. 2012** | **Oct. 2012** | **Dec. 2012** | **Feb. 2013** | **Mar. 2013** | **Jul. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Feb. 2014** | **May 2014** | **Grand Total** | **% of Total*** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Jaegers and Skuas (Stercorariidae) | 0 | 4 | 3 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 0.01% |
| Grebes (Podicipedidae) | 4 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0.01% |
| Nighthawks | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 3 | 0.00% |
| Kingfishers | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.00% |
| Vultures | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.00% |
| **Avian Total** | **11463** | **1092** | **297** | **316** | **322** | **3284** | **8556** | **4814** | **295** | **321** | **414** | **580** | **9184** | **4720** | **741** | **46399** | **43.36%** |
| Rays (Batoidea) | 0 | 1 | 5562 | 2663 | 428 | 16 | 0 | 0 | 14903 | 374 | 23292 | 404 | 0 | 1 | 301 | 47945 | 44.81% |
| Fish and Sharks | 2527 | 205 | 389 | 346 | 168 | 10 | 4 | 3 | 1526 | 96 | 945 | 31 | 86 | 2 | 2206 | 8544 | 7.98% |
| Toothed Whales (Odontoceti) | 39 | 196 | 61 | 117 | 136 | 75 | 77 | 38 | 347 | 284 | 361 | 21 | 12 | 27 | 237 | 2028 | 1.90% |
| Turtles (Testudines) | 29 | 282 | 192 | 151 | 184 | 1 | 0 | 0 | 360 | 25 | 183 | 50 | 0 | 1 | 290 | 1748 | 1.63% |
| Unidentified Marine Mammal or Shark | 4 | 24 | 13 | 4 | 8 | 7 | 20 | 32 | 48 | 16 | 107 | 2 | 4 | 0 | 5 | 294 | 0.27% |
| Baleen Whales (Mysticeti) | 0 | 0 | 0 | 0 | 1 | 1 | 4 | 4 | 0 | 0 | 0 | 0 | 1 | 3 | 1 | 15 | 0.01% |
| Bats (Chiroptera) | 0 | 0 | 0 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 15 | 0.01% |
| Jellyfish (Cnidaria) | 0 | 1 | 3 | 1 | 1 | 0 | 1 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 9 | 0.01% |
| Unidentified Whale (Cetacea) | 0 | 2 | 0 | 2 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 6 | 0.01% |
| **Non-Avian Total** | **2599** | **711** | **6220** | **3298** | **926** | **110** | **106** | **77** | **17186** | **795** | **24890** | **508** | **103** | **35** | **3040** | **60604** | **56.64%** |

**Table 5-3. Audit results for digital video aerial surveys.** Audits consist of 20% of eligible biota, as well as objects with a Threatened and Endangered (T&E) status that are not part of the random audit (see text). All T&E observations—whether part of the random 20% or added afterwards—were held to a 100% agreement criterion, while the remainder of the audits were required to be in ≥ 90% agreement. If these percentages were not met, then there were associated consequences (1 = audit was passed, no additional processes required; 2 = refinement of ID criteria for taxon that represented the majority of mismatches, complete re-review of that taxon, and then 20% audit of re-reviewed objects that were not included in first audit; 3 = all objects in disagreement went to arbitration to develop final identifications; see Chapter 4 for more information). An audit was not conducted for the March 2012 survey, as object identifications were performed collectively to develop a common identification process among reviewers. The first three surveys were flown with the sawtooth transects at 3 cm ground spatial resolution (GSR). Difficulties with identifications resulted in changing all transects to 2 cm GSR from September 2012 onwards.

| Survey | Ground Spatial Resolution (GSR) | Audit | No. of objects | % agreement | Consequences |
|---|---|---|---|---|---|
| Mar. 2012 | 2 cm (WEAs); 3 cm (sawtooth) | 20% Audit | 0 | N/A | N/A |
| | | T&E Audit | 0 | N/A | N/A |
| May 2012 | 2 cm (WEAs); 3 cm (sawtooth) | 20% | 376 | 80% | 2 |
| | | T&E | 251 | 94% | 3 |
| Jun. 2012 | 2 cm (WEAs); 3 cm (sawtooth) | 20% | 1,506 | 93% | 1 |
| | | T&E | 209 | 88% | 3 |
| Sep. 2012 | 2 cm (all transects) | 20% | 868 | 96% | 1 |
| | | T&E | 177 | 86% | 3 |
| Oct. 2012 | 2 cm (all transects) | 20% | 335 | 88% | 3 |
| | | T&E | 210 | 93% | 3 |
| Dec. 2012 | 2 cm (all transects) | 20% | 861 | 86% | 3 |
| | | T&E | 4 | 25% | 1 |
| Feb. 2013 | 2 cm (all transects) | 20% | 2,228 | 85% | 3 |
| | | T&E | 4 | 100% | 1 |
| Mar. 2013 (comparison) | 2 cm (all transects) | 20% | 559 | 96% | 1 |
| | | T&E | 0 | N/A | N/A |
| Mar. 2013 | 2 cm (all transects) | 20% | 1,186 | 92% | 1 |
| | | T&E | 4 | 100% | 1 |
| Jul. 2013 | 2 cm (all transects) | 20% | 3,910 | 95% | 1 |
| | | T&E | 370 | 95% | 3 |
| Aug. 2013 | 2 cm (all transects) | 20% | 586 | 94% | 1 |
| | | T&E | 26 | 96% | 3 |
| Sep. 2013 | 2 cm (all transects) | 20% | 6,248 | 98% | 1 |
| | | T&E | 188 | 97% | 3 |
| Oct. 2013 | 2 cm (all transects) | 20% | 297 | 88% | 3 |
| | | T&E | 52 | 96% | 3 |
| Dec. 2013 | 2 cm (all transects) | 20% | 1,226 | 87% | 3 |
| | | T&E | 2 | 50% | 3 |
| Feb. 2014 | 2 cm (all transects) | 20% | 1,934 | 94% | 1 |
| | | T&E | 4 | 75% | 3 |
| May 2014 | 2 cm (all transects) | 20% | 878 | 91% | 1 |
| | | T&E | 280 | 98% | 3 |

# Supplementary material

## Appendix 5A.

**Table 5A-1. Summary of animals observed during 15 digital video aerial surveys in 2012-2014.** Data are presented in order of abundance by family, based on the total count from all surveys. Surveys in blue include Maryland Project surveys. Note the August 2013 survey included only the Maryland WEA and Maryland extension area (Figure 5-1).

| **Common Name** | **Mar. 2012** | **May 2012** | **Jun. 2012** | **Sep. 2012** | **Oct. 2012** | **Dec. 2012** | **Feb. 2013** | **Mar. 2013** | **Jul. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Feb. 2014** | **May 2014** | **Grand Total** | **% Total** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Black Scoter | 8272 | 1 | 0 | 0 | 0 | 284 | 537 | 374 | 0 | 0 | 0 | 67 | 277 | 176 | 1 | 9989 | 9.34% |
| Unidentified Scoter | 607 | 0 | 0 | 0 | 0 | 121 | 928 | 2362 | 0 | 0 | 0 | 30 | 4312 | 1458 | 144 | 9962 | 9.31% |
| Surf Scoter | 526 | 0 | 0 | 0 | 0 | 59 | 226 | 129 | 0 | 0 | 0 | 2 | 50 | 79 | 0 | 1071 | 1.00% |
| Unidentified Duck | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 2 | 0 | 0 | 0 | 48 | 3 | 21 | 0 | 80 | 0.07% |
| White-winged Scoter | 3 | 0 | 0 | 0 | 0 | 15 | 0 | 3 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 23 | 0.02% |
| Red-breasted Merganser | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 8 | 1 | 0 | 15 | 0.01% |
| Brant | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 0 | 4 | 0.00% |
| Greater Snow Goose | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Long-tailed Duck | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Scoters, Ducks, Geese (Anatidae) Total** | **9408** | **1** | **1** | **0** | **0** | **487** | **1691** | **2876** | **0** | **0** | **0** | **147** | **4653** | **1737** | **145** | **21146** | **19.76%** |
| Unidentified Bird | 545 | 218 | 81 | 120 | 48 | 735 | 1852 | 538 | 41 | 83 | 99 | 78 | 1667 | 971 | 209 | 7285 | 6.81% |
| Auk or Shearwater | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Fulmar or Medium Gull | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0.00% |
| **Unidentified Birds (Aves spp.) Total** | **545** | **218** | **81** | **120** | **48** | **736** | **1852** | **538** | **41** | **83** | **99** | **78** | **1667** | **972** | **209** | **7287** | **6.81%** |
| Northern Gannet | 337 | 72 | 0 | 0 | 52 | 421 | 3730 | 821 | 1 | 0 | 1 | 119 | 728 | 839 | 5 | 7126 | 6.66% |
| **Gannets (Sulidae) Total** | **337** | **72** | **0** | **0** | **52** | **421** | **3730** | **821** | **1** | **0** | **1** | **119** | **728** | **839** | **5** | **7126** | **6.66%** |
| Unidentified Loon | 551 | 170 | 3 | 0 | 2 | 568 | 824 | 481 | 2 | 2 | 1 | 2 | 836 | 1004 | 167 | 4613 | 4.31% |
| Common Loon | 53 | 245 | 4 | 0 | 1 | 87 | 89 | 10 | 1 | 0 | 0 | 0 | 71 | 43 | 10 | 614 | 0.57% |
| Red-throated Loon | 10 | 45 | 1 | 0 | 0 | 64 | 54 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 180 | 0.17% |
| **Loons (Gaviidae) Total** | **614** | **460** | **8** | **0** | **3** | **719** | **967** | **496** | **3** | **2** | **1** | **2** | **907** | **1047** | **178** | **5407** | **5.05%** |
| Bonaparte's Gull | 116 | 0 | 0 | 0 | 0 | 497 | 14 | 10 | 0 | 0 | 0 | 0 | 418 | 11 | 0 | 1066 | 1.00% |
| Tern/Small or Medium Gull | 218 | 64 | 15 | 10 | 20 | 124 | 40 | 1 | 3 | 14 | 10 | 4 | 448 | 10 | 49 | 1030 | 0.96% |
| Unidentified Gull | 107 | 82 | 5 | 16 | 42 | 32 | 40 | 48 | 31 | 50 | 93 | 74 | 201 | 26 | 41 | 888 | 0.83% |
| Unidentified Tern | 1 | 38 | 12 | 63 | 34 | 3 | 0 | 0 | 55 | 75 | 89 | 34 | 4 | 0 | 71 | 479 | 0.45% |
| Great Black-backed Gull | 2 | 16 | 4 | 2 | 31 | 45 | 11 | 5 | 4 | 2 | 13 | 54 | 37 | 9 | 5 | 240 | 0.22% |
| Unidentified Large Gull | 6 | 26 | 5 | 3 | 19 | 13 | 5 | 0 | 7 | 16 | 59 | 18 | 23 | 1 | 0 | 201 | 0.19% |
| Laughing Gull | 0 | 1 | 5 | 3 | 9 | 1 | 0 | 1 | 52 | 7 | 11 | 7 | 0 | 0 | 7 | 104 | 0.10% |
| Herring Gull | 18 | 5 | 4 | 2 | 12 | 9 | 1 | 3 | 0 | 3 | 7 | 15 | 14 | 3 | 5 | 101 | 0.09% |
| Medium Tern: 32-45 cm | 7 | 76 | 12 | 5 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 101 | 0.09% |

| Common Name | Mar. 2012 | May 2012 | Jun. 2012 | Sep. 2012 | Oct. 2012 | Dec. 2012 | Feb. 2013 | Mar. 2013 | Jul. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Feb. 2014 | May 2014 | Grand Total | % Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Unidentified large Tern | 36 | 9 | 3 | 1 | 1 | 0 | 1 | 0 | 3 | 1 | 7 | 0 | 0 | 0 | 4 | 66 | 0.06% |
| Unidentified small gull | 27 | 2 | 0 | 0 | 1 | 0 | 0 | 0 | 11 | 13 | 0 | 0 | 3 | 0 | 0 | 57 | 0.05% |
| Medium Gull: 38-53 cm | 8 | 6 | 2 | 4 | 3 | 11 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 39 | 0.04% |
| Unidentified small Tern | 0 | 7 | 1 | 4 | 0 | 0 | 0 | 0 | 4 | 14 | 4 | 0 | 0 | 0 | 0 | 34 | 0.03% |
| Black Tern | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 26 | 6 | 0 | 0 | 0 | 0 | 33 | 0.03% |
| Caspian Tern | 0 | 0 | 2 | 5 | 4 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 13 | 0.01% |
| Lesser Black-backed Gull | 2 | 0 | 2 | 0 | 1 | 1 | 1 | 0 | 0 | 0 | 2 | 2 | 0 | 0 | 0 | 11 | 0.01% |
| Royal Tern | 3 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.00% |
| Sabine's Gull | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 2 | 0 | 0 | 0 | 4 | 0.00% |
| Ring-billed Gull | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.00% |
| Common Tern | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Gulls and Terns (Laridae) Total** | **552** | **332** | **73** | **120** | **178** | **737** | **118** | **69** | **172** | **222** | **302** | **210** | **1148** | **60** | **182** | **4475** | **4.18%** |
| Unidentified Alcid | 0 | 0 | 0 | 0 | 0 | 102 | 127 | 11 | 0 | 0 | 0 | 0 | 33 | 14 | 0 | 287 | 0.27% |
| Unidentified small alcid (Puffin/Dovekie) | 0 | 0 | 0 | 0 | 0 | 7 | 47 | 2 | 0 | 0 | 0 | 0 | 44 | 44 | 0 | 144 | 0.13% |
| Unidentified large alcid (Razorbill or Murre) | 0 | 0 | 0 | 0 | 0 | 27 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 34 | 0.03% |
| Atlantic Puffin | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 12 | 0.01% |
| Dovekie | 0 | 0 | 0 | 0 | 0 | 8 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 0.01% |
| Razorbill | 0 | 0 | 0 | 0 | 0 | 1 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 0.01% |
| **Auks (Alcidae) Total** | **0** | **0** | **0** | **0** | **0** | **154** | **193** | **13** | **0** | **0** | **0** | **1** | **78** | **59** | **0** | **498** | **0.47%** |
| Wilson's Storm-Petrel | 0 | 0 | 51 | 35 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 5 | 93 | 0.09% |
| Unidentified Storm-petrel | 1 | 0 | 2 | 3 | 0 | 12 | 0 | 0 | 19 | 5 | 0 | 1 | 0 | 0 | 0 | 43 | 0.04% |
| **Storm-Petrels (Hydrobatidae) Total** | **1** | **0** | **53** | **38** | **0** | **12** | **0** | **0** | **21** | **5** | **0** | **1** | **0** | **0** | **5** | **136** | **0.13%** |
| Greater Shearwater | 0 | 0 | 57 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 60 | 0.06% |
| Unidentified Shearwater | 0 | 0 | 6 | 3 | 0 | 9 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 20 | 0.02% |
| Cory's Shearwater | 0 | 0 | 8 | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 1 | 3 | 0 | 0 | 0 | 16 | 0.01% |
| Northern Fulmar | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 6 | 0 | 10 | 0.01% |
| Great Shearwater or Black-capped Petrel (flying) | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Manx Shearwater | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Sooty Shearwater | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| **Shearwaters and Fulmars (Procellariidae) Total** | **0** | **0** | **74** | **5** | **2** | **14** | **4** | **0** | **0** | **0** | **1** | **3** | **2** | **6** | **1** | **112** | **0.10%** |
| Dowitcher spp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 39 | 0 | 0 | 0 | 0 | 0 | 0 | 39 | 0.04% |
| Unidentified Phalarope | 2 | 0 | 0 | 8 | 0 | 0 | 0 | 0 | 0 | 3 | 4 | 2 | 0 | 0 | 0 | 19 | 0.02% |
| Small Shorebird sp. | 0 | 0 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 15 | 0.01% |

| Common Name | Mar. 2012 | May 2012 | Jun. 2012 | Sep. 2012 | Oct. 2012 | Dec. 2012 | Feb. 2013 | Mar. 2013 | Jul. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Feb. 2014 | May 2014 | Grand Total | % Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Large Shorebird sp. | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Shorebirds (Charadriiformes spp.) Total** | **2** | **0** | **0** | **20** | **0** | **0** | **0** | **0** | **39** | **7** | **4** | **2** | **0** | **0** | **0** | **74** | **0.07%** |
| Double-crested Cormorant | 0 | 0 | 0 | 1 | 25 | 1 | 0 | 0 | 0 | 0 | 0 | 8 | 1 | 0 | 6 | 42 | 0.04% |
| **Cormorants (Phalacrocoracidae) Total** | **0** | **0** | **0** | **1** | **25** | **1** | **0** | **0** | **0** | **0** | **0** | **8** | **1** | **0** | **6** | **42** | **0.04%** |
| Brown Pelican | 0 | 2 | 2 | 3 | 1 | 0 | 0 | 0 | 17 | 0 | 2 | 2 | 0 | 0 | 1 | 30 | 0.03% |
| **Pelicans (Pelecanidae) Total** | **0** | **2** | **2** | **3** | **1** | **0** | **0** | **0** | **17** | **0** | **2** | **2** | **0** | **0** | **1** | **30** | **0.03%** |
| Great Blue Heron | 0 | 0 | 0 | 7 | 4 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 12 | 0.01% |
| American Bittern | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.00% |
| Snowy Egret | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| **Egrets and Herons (Ardeidae) Total** | **0** | **2** | **0** | **7** | **7** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **0** | **0** | **0** | **17** | **0.02%** |
| Cedar Waxwing | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 7 | 0.01% |
| Unidentified Swallow | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.00% |
| Unidentified Passerine | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 4 | 0.00% |
| Baltimore Oriole | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Barn Swallow | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Passerines (Passeriformes spp.) Total** | **0** | **0** | **1** | **2** | **4** | **0** | **0** | **0** | **0** | **0** | **2** | **1** | **0** | **0** | **7** | **17** | **0.02%** |
| Osprey | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 2 | 4 | 0 | 0 | 2 | 11 | 0.01% |
| Bald Eagle | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 3 | 0.00% |
| **Raptors (Pandionidae, Falconidae, and Accipitridae) Total** | **0** | **1** | **1** | **0** | **0** | **1** | **0** | **0** | **1** | **0** | **2** | **6** | **0** | **0** | **2** | **14** | **0.01%** |
| Unidentified Jaeger | 0 | 2 | 2 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0.00% |
| Parasitic Jaeger | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Pomarine Jaeger | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Jaegers and Skuas (Stercorariidae) Total** | **0** | **4** | **3** | **0** | **1** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **8** | **0.01%** |
| Unidentified Grebe | 3 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.00% |
| Horned Grebe | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Red-necked Grebe | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Grebes (Podicipedidae) Total** | **4** | **0** | **0** | **0** | **0** | **2** | **1** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **7** | **0.01%** |
| Belted Kingfisher | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Black Vulture | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Common Nighthawk | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Miscellaneous Birds Total** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **1** | **0** | **1** | **0** | **0** | **0** | **0** | **0** | **3** | **0.00%** |
| **Avian Total** | **11463** | **1092** | **297** | **316** | **322** | **3284** | **8556** | **4814** | **295** | **321** | **414** | **580** | **9184** | **4720** | **741** | **46399** | **43.36%** |

| Common Name | Mar. 2012 | May 2012 | Jun. 2012 | Sep. 2012 | Oct. 2012 | Dec. 2012 | Feb. 2013 | Mar. 2013 | Jul. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Feb. 2014 | May 2014 | Grand Total | % Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cownose Ray | 0 | 0 | 3345 | 1699 | 78 | 0 | 0 | 0 | 9328 | 97 | 11005 | 268 | 0 | 0 | 143 | 25963 | 24.26% |
| Unidentified ray | 0 | 1 | 2216 | 963 | 350 | 16 | 0 | 0 | 5574 | 277 | 12280 | 136 | 0 | 1 | 158 | 21972 | 20.53% |
| Giant Manta Ray | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 8 | 0.01% |
| Unidentified Manta Ray | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Roughtail or Southern Stingray | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Rays (Batoidea) Total** | **0** | **1** | **5562** | **2663** | **428** | **16** | **0** | **0** | **14903** | **374** | **23292** | **404** | **0** | **1** | **301** | **47945** | **44.81%** |
| Unidentified fish | 2525 | 147 | 284 | 296 | 114 | 4 | 4 | 1 | 1435 | 58 | 706 | 7 | 85 | 2 | 2176 | 7844 | 7.33% |
| Unidentified shark | 1 | 11 | 97 | 32 | 3 | 1 | 0 | 0 | 57 | 33 | 190 | 2 | 1 | 0 | 9 | 437 | 0.41% |
| Ocean Sunfish (Mola) | 1 | 45 | 3 | 7 | 51 | 5 | 0 | 0 | 17 | 1 | 10 | 21 | 0 | 0 | 15 | 176 | 0.16% |
| Hammerhead shark | 0 | 1 | 4 | 7 | 0 | 0 | 0 | 0 | 10 | 3 | 18 | 1 | 0 | 0 | 0 | 44 | 0.04% |
| Thresher Shark | 0 | 1 | 0 | 3 | 0 | 0 | 0 | 0 | 7 | 1 | 20 | 0 | 0 | 0 | 6 | 38 | 0.04% |
| Scalloped Hammerhead | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 3 | 0.00% |
| Basking Shark | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| **Fish and Sharks Total** | **2527** | **205** | **389** | **346** | **168** | **10** | **4** | **3** | **1526** | **96** | **945** | **31** | **86** | **2** | **2206** | **8544** | **7.98%** |
| Small beaked Cetacean to 3m | 7 | 91 | 13 | 65 | 90 | 25 | 16 | 25 | 153 | 213 | 204 | 2 | 0 | 14 | 126 | 1044 | 0.98% |
| Bottlenose Dolphin | 12 | 104 | 48 | 51 | 36 | 10 | 0 | 2 | 178 | 39 | 84 | 5 | 0 | 0 | 108 | 677 | 0.63% |
| Unidentified Dolphin | 20 | 0 | 0 | 1 | 1 | 1 | 52 | 7 | 9 | 32 | 59 | 2 | 5 | 0 | 1 | 188 | 0.18% |
| Unidentified Toothed Whales | 0 | 0 | 0 | 0 | 9 | 18 | 1 | 3 | 0 | 0 | 10 | 11 | 3 | 6 | 2 | 63 | 0.06% |
| Common Dolphin | 0 | 0 | 0 | 0 | 0 | 21 | 8 | 2 | 7 | 0 | 4 | 0 | 4 | 6 | 0 | 52 | 0.05% |
| Harbor Porpoise | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 3 | 0.00% |
| Risso's dolphin | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0.00% |
| **Toothed Whales (Odontoceti) Total** | **39** | **196** | **61** | **117** | **136** | **75** | **77** | **38** | **347** | **284** | **361** | **21** | **12** | **27** | **237** | **2028** | **1.90%** |
| Small turtle | 22 | 216 | 138 | 71 | 137 | 0 | 0 | 0 | 276 | 24 | 174 | 43 | 0 | 1 | 285 | 1387 | 1.30% |
| Loggerhead Turtle | 6 | 52 | 42 | 40 | 30 | 0 | 0 | 0 | 7 | 1 | 5 | 3 | 0 | 0 | 2 | 188 | 0.18% |
| Leatherback Turtle | 0 | 1 | 2 | 31 | 6 | 0 | 0 | 0 | 76 | 0 | 2 | 3 | 0 | 0 | 1 | 122 | 0.11% |
| Kemp's Ridley Sea Turtle | 0 | 11 | 9 | 7 | 5 | 1 | 0 | 0 | 1 | 0 | 2 | 0 | 0 | 0 | 2 | 38 | 0.04% |
| Green Turtle | 1 | 2 | 1 | 2 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 11 | 0.01% |
| Hawksbill Turtle | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| **Turtles (Testudines) Total** | **29** | **282** | **192** | **151** | **184** | **1** | **0** | **0** | **360** | **25** | **183** | **50** | **0** | **1** | **290** | **1748** | **1.63%** |
| Cetacean/Seal/Shark | 4 | 23 | 13 | 4 | 8 | 7 | 20 | 32 | 48 | 16 | 107 | 2 | 4 | 0 | 5 | 293 | 0.27% |
| Seal/Dolphin | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Unidentified Marine Mammal or Shark Total** | **4** | **24** | **13** | **4** | **8** | **7** | **20** | **32** | **48** | **16** | **107** | **2** | **4** | **0** | **5** | **294** | **0.27%** |
| Right Whale | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 3 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 8 | 0.01% |
| Minke Whale | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 3 | 0.00% |
| Humpback Whale | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 2 | 0.00% |

| **Common Name** | **Mar. 2012** | **May 2012** | **Jun. 2012** | **Sep. 2012** | **Oct. 2012** | **Dec. 2012** | **Feb. 2013** | **Mar. 2013** | **Jul. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Feb. 2014** | **May 2014** | **Grand Total** | **% Total** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fin Whale | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Unidentified Fin/Sei | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Baleen Whales (Mysticeti) Total** | **0** | **0** | **0** | **0** | **1** | **1** | **4** | **4** | **0** | **0** | **0** | **0** | **1** | **3** | **1** | **15** | **0.01%** |
| Red Bat | 0 | 0 | 0 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 15 | 0.01% |
| **Bats (Chiroptera) Total** | **0** | **0** | **0** | **14** | **0** | **0** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **0** | **0** | **15** | **0.01%** |
| Unidentified jellyfish | 0 | 1 | 3 | 1 | 1 | 0 | 1 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 9 | 0.01% |
| **Jellyfish (Cnidaria) Total** | **0** | **1** | **3** | **1** | **1** | **0** | **1** | **0** | **1** | **0** | **1** | **0** | **0** | **0** | **0** | **9** | **0.01%** |
| Unidentified Cetacean | 0 | 2 | 0 | 2 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0.00% |
| Unidentified Medium Whale | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0.00% |
| **Unidentified Whale or Dolphin (Cetacea) Total** | **0** | **2** | **0** | **2** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **0** | **0** | **1** | **0** | **6** | **0.01%** |
| **Non-Avian Total** | **2599** | **711** | **6220** | **3298** | **926** | **110** | **106** | **77** | **17186** | **795** | **24890** | **508** | **103** | **35** | **3040** | **60604** | **56.64%** |
| **Grand Total** | **14062** | **1803** | **6517** | **3614** | **1248** | **3394** | **8662** | **4891** | **17481** | **1116** | **25304** | **1088** | **9287** | **4755** | **3781** | **107003** | **100.00%** |

# Chapter 6: Recommendations for high resolution digital video aerial surveys in the U.S.

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 6 Highlights

*Recommendations for conducting high resolution digital video aerial surveys in the U.S.*

## *Context*[1]

High resolution digital video aerial surveys are one of several different survey methodologies for quantifying animal densities. Digital aerial approaches were developed in Europe, but this study represented the first broad-scale application of these technologies in North America. This novel approach to wildlife surveying presents several challenges, which can be addressed through management, technological advances, or analytical approaches.

## *Study goal/objectives*

Present advantages and challenges of high resolution digital video aerial surveying, and provide management, technology, and analysis recommendations for addressing these challenges in order to advance this method as an option in broad-scale surveying.

## *Highlights*

- We compare the advantages and challenges of boat, visual aerial, and digital aerial survey methods.
- We recommend that future application of digital video aerial surveys in the U.S. include the continued development of standardized data formats, as well as transparent data management and QA/QC protocols.
- This technology continues to be improved. Advances in camera resolution, GPS integration, and performance in poor weather are important for the application of this technology.
- There are tradeoffs between ground spatial resolution (GSR) of video and the ground coverage (e.g., strip width) of transects. For the camera technology used in this study, we recommend a minimum ground spatial resolution (GSR) of 2 cm.
- Digital aerial survey data are not distance-biased, unlike visual survey approaches from both boats and aircraft. However, there may be other sources of detection bias for these data, and this question has remained largely unexamined in Europe to date. It will be important to examine this issue, and if necessary, develop detectability and availability metrics for digital aerial survey data.

## *Implications*

High resolution digital aerial surveying has largely replaced other survey approaches for offshore wind energy development in Europe. With recent and continuing technological advances, digital aerial surveying could be a cost effective, time-efficient and repeatable option for performing broad-scale surveys to inform siting and permitting of offshore wind energy development in North America.

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[1] For more detailed context for this chapter, please see the introduction to Part II of this report.

## Abstract

High resolution digital video aerial surveys were conducted in the mid-Atlantic U.S. in 2012-2014, using two twin-engined Cessnas outfitted with four super high-resolution cameras. As a result of these survey efforts, we have significant experience with the advantages and challenges of digital video aerial survey approaches. Digital aerial surveys are a useful approach to many situations because they are safe, can be conducted both pre- and post-construction, and are efficient for covering large survey areas. In this chapter, we present management, technology and analysis recommendations for consideration in future high resolution video aerial surveys if implemented as a survey method in North American offshore wind development.

## Overview

The optimal approach for quantifying animal densities, both temporally and spatially, is dependent on project goals and the extent of the geographical area in question, among other factors. In recent years, digital aerial survey approaches have become commonly used methods in planning and assessing offshore wind energy development in Europe (Buckland et al. 2012). Comparing different offshore wildlife survey approaches, there are clear advantages and disadvantages to each method (Table 6-1). Boat-based surveys, for example, are known to cause disturbance to wildlife, are not generally repeatable pre- and post-construction, and have lower detections of submerged marine animals (i.e., sea turtles, rays and sharks) than either visual aerial or, in particular, digital aerial approaches (Normandeau Associates Inc. 2013; Chapter 14). Although not investigated in our study, traditional low-flying aerial surveys with visual observers are less safe than digital aerial surveys, are also known to cause disturbance, and are not repeatable post-construction, all due to the low altitude required to visually detect marine wildlife. Unlike observer-based approaches, digital aerial surveys are repeatable post-construction, and the data produced is auditable. However, digital aerial surveys are dependent on video quality and atmospheric conditions, have limited strip width, unknown detection bias (though no distance bias in detections, unlike boat and visual aerial surveys), and require greater technological infrastructure for video data management and storage.

High resolution digital video aerial surveys in this study were conducted in 2012-2014 by HiDef Aerial Surveying, Ltd., using two twin-engined Cessnas outfitted with four super high-resolution cameras. Fifteen surveys were flown at 610 m (2,000 ft) over a 13,245 km$^2$ study area (Chapter 3). Wildlife observed in the digital video footage was identified to the lowest taxonomic level (Chapter 4) and georeferenced. Additional data were also collected (e.g., direction of movement) or calculated (e.g., flight height), according to HiDef's standard protocols (Hatch et al. 2013).

As a result of these survey efforts, we have substantial experience with the advantages and challenges of digital video aerial survey approaches. Barring significant technical and legal advancements in the civilian use of drones, digital aerial surveys may be the most useful survey approach to many situations in the marine environment, because they are safe, can be conducted both pre- and post-construction, and are efficient for covering large survey areas. They have largely replaced other survey approaches for offshore wind energy development in Europe for these reasons. However, there are challenges to this approach as well. Some of these are inherent to the methodology, such as the inability to collect detailed behavioral data. Many challenges of digital video aerial surveys, however, are due primarily to the relative novelty of the method, and can be addressed through management, technological advances, or analytical approaches. We outline some of these possibilities here, with the intent of advancing the understanding and broad-scale use of this approach in North America.

## Recommendations

### *Management recommendations*

*Flights*
When planning digital aerial surveys over large geographic areas, prior consideration should be given to the frequency of naval and air traffic in the area that may impede or cancel planned survey flights. In addition, aerial surveying companies and pilots should maintain flexibility to conduct surveys at the first window of opportunity in case long periods of poor weather conditions develop that would prevent flying. If necessary, multiple planes should be used to complete surveys in brief time windows.

*Data standards*
We recommend working with federal, state, and private partners involved with at-sea survey data to develop a set of standard data fields and outputs to insure wide acceptance and use. This is a critical step in helping with future data aggregation and analysis. Given that this method is in its infancy in the U.S., it is important to develop these standards early, before many surveys have been conducted. We recommend consultation with managers of the two largest federal databases, the USFWS Northwest Atlantic Seabird Catalog (formerly the Avian Seabird Compendium) and the USGS North Pacific Seabird Pelagic Bird Database, to develop these standards with a view towards final deposition into these databases.

*Object tracking*
A unique identifier should be applied to each object observed in the video, and this identifier should be maintained throughout all data management and processing activities during the course of a study, including object location, object identification, audits, georeferencing, and flight height estimation. This will help with quality assurance and quality control (QA/QC) processes, and allow the tracking of individual records throughout data management and analysis efforts.

*Detailed QA/QC process*
We also suggest that a publicly available QA/QC process is developed for the video data. This guiding document should include methods for object detection, as well as identification and final data processing. We developed a data management and QA/QC protocol for BRI’s responsibilities during this study (Chapter 4) that could provide the basis for a larger protocol and improve data standardization between studies.

### *Technology recommendations*

*Ground spatial resolution*
We recommend a minimum ground spatial resolution (GSR) of 2 cm for all aerial video surveys. Two cm GSR digital aerial surveying had higher identification rates for scoters and higher detection of aquatic

animals when compared with 3 cm GSR (Chapter 5). While the 3 cm GSR allowed for a wider recorded strip width, video quality (clarity and color accuracy) was substantially lower, leading to poor identification rates (Table 6-2) and longer times required to make identifications. In the UK, HiDef has achieved 90% identification rates with 0.5 GSR (HiDef 2011), and a recent trial of a different digital aerial survey technology in the United States achieved higher identification rates for 1.5 GSR, compared to 2cm GSR (Normandeau Associates, Inc. 2013). While there is a tradeoff between GSR and strip width (and, thus, ground coverage of surveys), we believe that in many cases, it is worth considering the prioritization of GSR when designing studies to meet project goals. It should be noted, however, that recent technological advances in high resolution digital video camera systems have increased the strip width, as well as substantially improving identification rates beyond what is reported here, while keeping other survey characteristics constant (Webb and Hawkins 2013, HiDef unpubl. data).

#### *Improve camera response in poor weather*

Poor weather, such as low cloud and fog, may have been a factor in detection rates and identification rates of animals, particularly during winter surveys. Some animals behave very differently during adverse weather conditions, however, and may even change flight heights (Shamoun-Baranes et al 2006), emphasizing the importance of better data collection under these conditions. Predictive models of distribution and abundance are also hampered by variable (and unquantified) variation in detection and identification rates between surveys, possibly due to weather.

#### *Night-time imagery*

Currently, there are no surveys capturing data at night, which leaves a gap in our understanding of behaviors of marine fauna. There is evidence that animals make directed movements at night, and increasing knowledge of these movements would improve our understanding of animals in the offshore environment. Improvements in thermal and/or low light imaging should allow data collection at night or at least extend imagery later at night and earlier in the morning.

#### *Improved color fidelity, clarity, and contrast in video*

We recommend continued advances in video quality for surveys. Video technology used during these surveys at times suffered from poor color fidelity, clarity, and contrast, making it difficult to identify some animals to species. Recent advances in digital camera and optical technology have clearly improved video quality (Webb and Hawkins 2013), but need to be deployed in the United States to take advantage of these benefits. The improvement in quality should help improve identification rates, as well as sex and age determinations. This may be particularly useful for particular taxa, such as loons and small turtles, which were difficult to identify consistently in the mid-Atlantic.

#### *Standardization of video file types*

This survey used a proprietary video file type that was not viewable by most video review programs. While there may be reasons to develop such proprietary file types, we do not believe it is in the best interest of broadening our understanding of digital video systems or transparency in data, because data cannot be distributed and reviewed without specialized software. We recommend that a standardized file format be adopted, and, if necessary, an open-source video review software package be developed, in order to allow anyone to review such imagery and support further development and enhancements of such a product.

### *Integrated GPS and camera sensor data*

The importance of GPS positioning linked to aerial video data is paramount and must be tightly integrated into the video frame data. Much time and effort was spent post-processing video and GPS data to generate proper effort data for aerial observation data. We recommend that camera systems have an array of sensors on board to capture plane height, camera orientation (compass direction), camera down angle, zoom, sensor resolution, and other metrics necessary to calculating exact frame position on the ground. On-board processing should calculate exact field of view and spatial position during capture, and this data should be encoded in the video frame so that location information can be easily extracted and used for analysis. We also highly recommend an integrated backup GPS system that takes over in case of failure, and provides clear warning to pilots when systems are down, so that flights are not conducted without recording spatial data.

### *Improved parallax algorithm*

HiDef uses the principle of parallax to determine flight heights for flying birds and bats (Hatch et al. 2013). Flight heights were reported in categories – 0-20 m, 20-50 m, 50-100 m, 100-200 m, and 200+ m. For the purposes of informing siting and permitting of marine wind turbines (with yet to be determined turbine design) and to properly assess collision risks, we recommend the use of narrower, more precise flight height ranges for future surveys, where possible.

## ***Analysis recommendations***

### *Develop detection and availability metrics*

We were unable to derive true estimates of abundance from the aerial video data due to the way it is currently collected and processed. Boat surveys use distance sampling methods (Buckland 2001; Buckland et al. 2005) to estimate detection bias related to distance to observer. Using distance sampling allows for calculation of corrected abundance values which are then comparable with other surveys, times, and conditions. Video aerial surveys assume 100% detection across the video screen, which seems reasonable given the camera angle and relatively narrow strip width. Currently, however, they do not incorporate other potential sources of detection bias, such as variations in weather and image quality, or observer bias. We recommend that additional methods be developed to allow estimation of detection relative to changing atmospheric conditions (for example, the development and use of a metric for image quality that can be applied to all video data). Further research is needed to develop a suitable method for this. Inter-observer and inter-survey bias in detections and species identifications could be examined using a double observer approach during video analysis. This approach would be relatively straightforward to incorporate into existing audit protocols for object location and species identification.

To date, there is no way to calculate depth and visibility for submerged animals in digital aerial surveys. Flying over fixed points in the study area and using known objects of different sizes placed at different depths, measured under varying water turbidity and sea state conditions (Pollock et al. 2006), could provide a maximum depth of visibility under varying conditions and potentially maximum depths at which identifying marks and coloration disappear.

Given the greatly increased flight speeds of the aerial video, there is less time for diving animals to appear at the surface in the frame, potentially resulting in a lower availability for detection. This availability bias is also a common issue in other types of surveys (Thomson 2013). To offset this availability bias, which varies with season, region, depth, and temperature for some marine animals (Thomson 2013), it may be possible to use diving rates for some species based on behavioral and telemetry studies to obtain a better estimate of availability, since we know exactly the time and space covered by video (Thomson 2013; Southall et al 2005).

### *Additional comparison studies*

We were able to conduct a targeted comparison between boat and high resolution digital video aerial surveys (Chapter 13), but were limited to survey overlap of one day. Conditions change throughout the year, and video technologies are continuing to improve, so it would be useful to perform a number of overlapping survey runs to develop a more comprehensive and rigorous comparison and continue to improve our understanding of how best to integrate data developed using these different survey methods. Recent comparison efforts in Europe (Burt et al. 2009; Burt et al. 2010; Buckland et al. 2012; Webb and Hawkins 2013) have added to our understanding of the capabilities different survey approaches, and additional exploration of this topic in North America will be essential for establishing digital approaches in North America and integrating new survey datasets with those generated using traditional methods (Chapter 13). Furthermore, an analysis comparing results of the various digital aerial survey methods and technologies currently in existence would be helpful for determining exactly how these methods compare. The sole study to conduct this type of comparison (Thaxter and Burton 2009) is now outdated, due to recent advances in digital technologies. It is particularly important to conduct comparison studies in North America for taxa that have remained largely unexamined in similar studies to date, either because they do not occur frequently in European waters (e.g., sea turtles and some North American cetaceans), or because the North American populations exhibit different morphological or behavioral characteristics than European populations (e.g., loons in the mid-Atlantic study area; Chapters 5 and 16).

## Conclusions

Cost effective, time-efficient, repeatable survey methods are a priority for siting (i.e., baseline studies) and permitting (i.e., pre- and post-construction wildlife studies) of offshore wind energy development. Normandeau Associates, Inc. (2013) found that their digital aerial surveys were cost effective for areas >149 km$^2$. With technological advances in camera resolution and improvements in aircraft fuel efficiency, digital aerial surveying will continue to improve in cost-effectiveness, accuracy, and efficiency. Digital aerial surveys are also flown at much higher altitudes than visual aerial surveys, which provides several advantages; they are repeatable for direct comparisons pre- and post- construction (because flights are conducted above turbine height), they cause no discernible disturbance for most wildlife, and they are much safer for pilots and biologists. However, there are also limitations to digital aerial surveys. To ensure the success of digital aerial survey methods in the United States, wildlife managers and holders of historical databases must reach a consensus on the methods for image analysis and data incorporation. Additional comparison surveys using the latest technology should also be conducted. Some challenges, like the lack of detailed behavioral data compared to what can be collected from boat surveys (Normandeau Associates, Inc. 2013), are probably inherent to the survey method. Others can and should be addressed through adjustments to technology, management processes, or analytical approaches as this survey method is refined.

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**Table 6-1. Comparison of common survey approaches for marine wildlife**

| | **Advantages** | **Challenges** |
|---|---|---|
| **Visual Aerial Surveys** | • Fast survey pace<br>• Can record both in and outside the strip width<br>• Higher identification rates | • Safety issues related to low flight height<br>• Observer biases<br>• Not repeatable pre- and post- construction<br>• Not auditable<br>• Disturbance to animals<br>• Detection rates affected by ambient conditions and distance from observer |
| **Boat Surveys** | • Can record both in and outside the strip width<br>• Higher identification rates<br>• Behavioral details can be seen<br>• Comparable with historic datasets | • Observer biases<br>• Not repeatable pre- and post- construction<br>• Not auditable<br>• Disturbance (attraction and displacement) to animals<br>• Detection rates affected by ambient conditions and distance from observer<br>• Slow survey pace |
| **Digital Aerial Surveys** | • Fast survey pace<br>• Archivable<br>• Auditable (Observer identification variability, false negatives and observer biases can be identified and alleviated through audits and QA/QC processes)<br>• Ability to obtain replicable flight height estimates<br>• Minimal disturbance to animals<br>• Repeatable pre- and post- construction<br>• Technology will continue to improve | • Image quality is affected by atmospheric conditions<br>• Low identification rates for some taxonomic groups<br>• Detection and identification rates are dependent on video quality<br>• Strip width is limited (controlled by camera resolution and plane altitude) |

**Table 6-2. Comparison of 2 cm vs. 3 cm GSR video data from the March, May and June 2012 surveys in the mid-Atlantic study area.** The percentage of birds that were not identified to the species level for 2cm GSR was 53%, while the percentage of birds not identified to species in 3cm GSR was 74%.

| Statistic | 2 cm GSR | 3 cm GSR |
|---|---|---|
| Percentage of all birds from each resolution that were not identified to species (excluding scoters)* | 53% | 74% |
| Percentage of all loons from each resolution that were not identified to species | 43% | 79% |
| Percentage of all gulls and terns from each resolution that were not identified to species | 67% | 87% |
| Percentage of the sea turtles from each resolution that were not classifiable to species (e.g., all SMTU) | 61% | 88% |
| Percentage of the marine mammals from each resolution that were not classifiable to species (e.g., all unknown cetacean, unknown dolphin) | 18% | 50% |
| Percentage of the sharks and rays from each resolution that were not classifiable to species | 10% | 78% |
| Disagreement rate among observers during audits (as percentage of all biota audited; includes only May and June data) | 7% | 12% |
| Percentage of all animals not identified to species from each resolution** | 53% | 81% |

* Excluded because scoters occurred disproportionately in 3 cm footage.

**Does not include scoters and gannets (species with high identification rates despite GSR)

# Introduction to Part III
# Examining wildlife distributions and abundance using boat-based surveys

## Report structure

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure I).

Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure I. In Parts II (Chapters 3-6) and III (Chapters 7-12), we describe methods and results for high resolution digital video aerial surveys and boat surveys, respectively. Part IV of this report (Chapters 13-19) combines data from both survey approaches to develop a comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area. Part V (Chapters 20-25) focuses on individual movements and habitat use of focal avian species, tracked via satellite telemetry; and Part VI (Chapters 26-27) focuses on population-level migratory movements over the oceans, using several approaches for studying nocturnal avian migration. An additional study effort, which further explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

## Part III: Examining wildlife distributions and abundance using boat surveys

Standardized boat-based surveys with distance estimation are a widely used and well-established method of obtaining density data for birds, sea turtles, and marine mammals. There are six chapters in Part III of this report, focused on the use of boat surveys to examine wildlife distributions and abundance:

Chapter 7. Protocol for conducting boat surveys for wildlife.

Chapter 8. Basic summary of boat survey observation data.

Chapter 9. Scientific echo sounding study to obtain aquatic biomass data (includes data management and analysis protocols as well as a basic data summary).

Chapter 10. Examination of spatial associations between feeding seabirds and aquatic biomass.

Chapter 11. Development of a multi-species model for estimating seabird abundance and distributions.

Chapter 12. Prediction of seabird densities across the study area by season, based on an incorporation of environmental data into the multi-species modeling approach.

The survey protocol (Chapter 7) explains our boat survey study design in detail, and is referenced throughout the following chapters (also see Figure II). Surveys were particularly optimized for avian species, and detected a wide variety of seabird species as well as raptors, passerines, shorebirds, and other avian taxa. Boat surveys also recorded marine mammals, sea turtles, rays, sharks, fish, and bats (Chapter 8). Data collected on boat surveys provided some substantial advantages in species identification over digital data collected from aircraft. Species-specific information can be important, as even closely related species often have differences in their conservation status, ecology, and habitat requirements.

While conducting surveys, we collected environmental covariate data in order to assess fine-scale patterns of environmental variables in relation to wildlife densities. In particular, fisheries sonar (a scientific echo sounder) was used to estimate relative biomass of aquatic prey in the same areas as boat survey observations (Chapter 9). These data were used to examine spatial associations between feeding seabirds and acoustically detected prey (Chapter 10). Identifying the spatial and temporal associations and lags between aquatic biomass and seabird behavior is helpful for understanding how these birds are making decisions in the marine environment, and the simultaneous collection of in situ data on seabirds and their prey can allow for a better understanding of the ecological drivers of seabird distributions (e.g., by allowing analysis of co-occurrence at very fine geographic and temporal scales, or linking predator distributions to specific prey species; Veit et al. 1993, Santora et al. 2010) Improving our understanding individual-scale decisions and movements in relation to prey may help managers to determine the behaviors or environmental conditions that present the highest risk of interactions between seabirds and offshore wind energy development, as well as determine the suitability of potential mitigation approaches. In this regard, the goals of Chapter 10 are in some ways more similar to the individual movement chapters in Part V of this report than to the other chapters in Part III (and most similar to the state-space model describing Northern Gannet foraging and movement patterns in Chapter 24).

A broader geographic and temporal scale of analysis is required to develop wildlife data appropriate for siting future development projects, or to fully assess exposure to wildlife from proposed projects. These goals also require correction of certain biases associated with boat survey data, such as distance bias, in which observers are less likely to see animals located farther from the survey vessel. Hierarchical Bayesian statistical approaches, as applied to survey data in Chapters 11-12, are useful for situations where distribution patterns or resource use vary with scale, and where species of interest are highly mobile and may be periodically unavailable for detection (Mordecai et al. 2011). These modeling methods allow distribution models to be chosen to fit the observed data (Gardner et al. 2008, Zipkin et al. 2010), and incorporate distance estimation and environmental covariates into the model structure, in order to predict animal distributions and abundance on a broad geographic scale.

Project collaborators first focused on the development of a community distance sampling (CDS) model for seabirds, using data from the first boat survey in April 2012 (Chapter 11). This is a novel multi-species approach for estimating seabird abundance and distributions that explicitly estimates detection as well as abundance parameters. By sharing information across species, this community model allows us to make inferences about abundance, distribution, and response to environmental variables of rare species for which there would not be enough data to run individual models.

Building on the CDS model developed in Chapter 11, Chapter 12 examines survey data from 15 boat surveys to develop geospatial models that predict seabird densities by season. By incorporating remotely collected environmental covariate data into the hierarchical modeling structure in this expanded analysis, Chapter 11 predicts seabird abundance throughout the study area, including areas that were not directly surveyed. The seasonal abundance maps presented in Chapter 12, for both seabird communities and individual species, predict animal distributions and abundance on a broad geographic scale and are useful for identifying important habitat use areas and seasonal patterns. Unlike several chapters in Part IV of this report, which utilize approaches for combining boat and digital aerial survey data, Chapter 12 focuses on using data from a single, well understood survey method to do the best possible job of describing patterns of abundance. In this regard, though it builds directly from the CDS model development chapter (Chapter 11), Chapter 12's products may be most similar in potential application to the Bottlenose Dolphin Generalized Additive Model (GAM) presented in Chapter 16.

### *Implications*

These survey results on the geographic distributions and relative abundance of wildlife in the mid-Atlantic are expected to be useful for minimizing impacts to wildlife populations from offshore wind energy development in that they:

- Inform the siting of future projects, by incorporating wildlife patterns into marine spatial planning and decision making, and by using exposure data as a first step towards defining relative risk by location;
- Inform the permitting process for projects, by contributing data towards National Environmental Protection Act (NEPA) and other regulatory requirements, and by helping to define target taxa or research priorities on which to focus on during site-specific pre- and post-construction monitoring studies; and

- Inform mitigation efforts and construction and operations plans, by presenting temporal data on community composition, distributions, and abundance that can be used to time certain activities to coincide with reduced potential for exposure of certain populations.

Boat survey data and analyses can also be used to assess changes to wildlife populations as a result of offshore wind energy development, climate change, and other factors. Results from this project represent a baseline that can be used for comparison with compatible future surveys, and to assess changes due to development or other causes. Future research to fill data gaps on hazards and vulnerability can be targeted towards species with high levels of exposure, as well as species most likely to be impacted due to their conservation status or life history.

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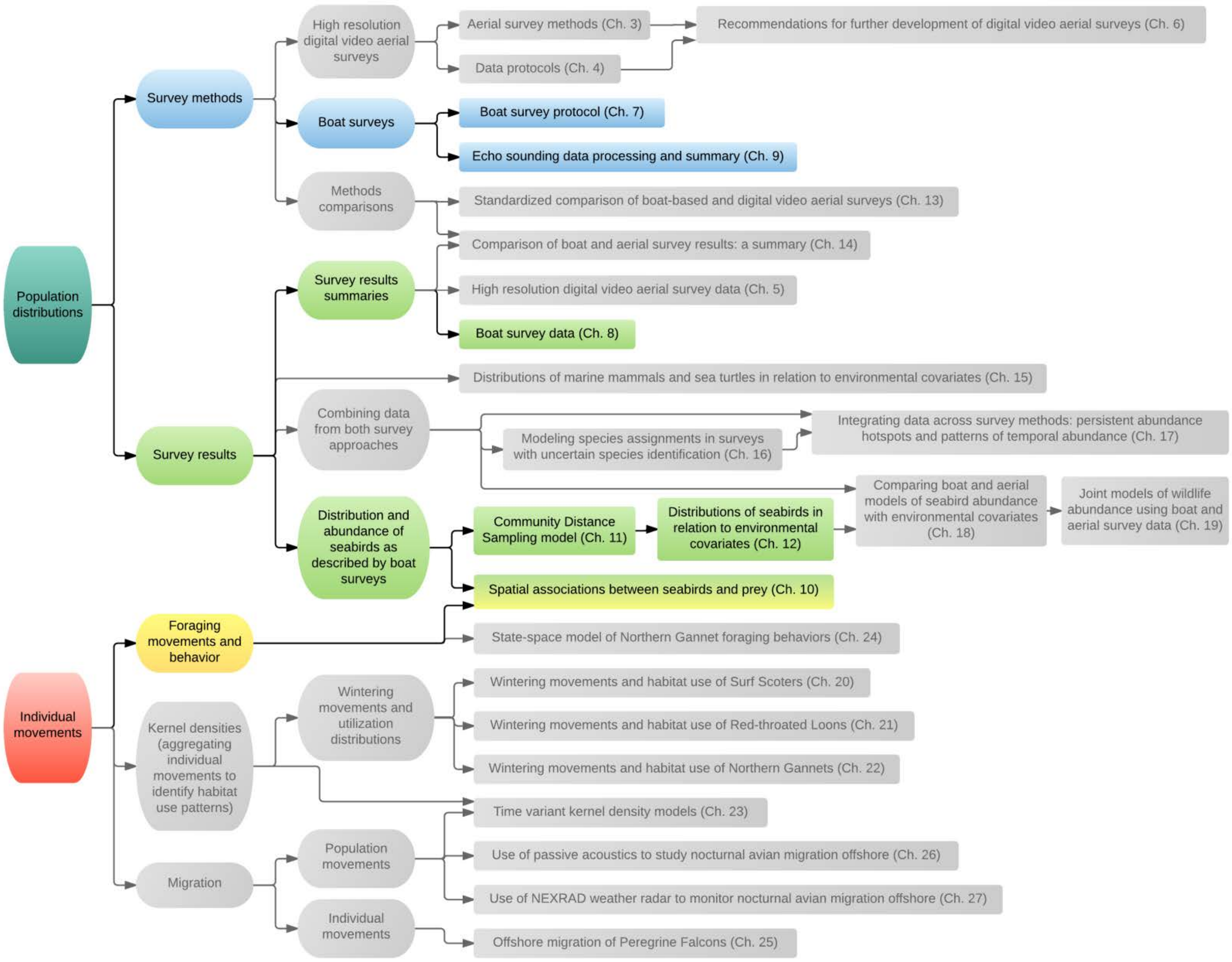

**Figure I. Organization of chapters within this final report.**

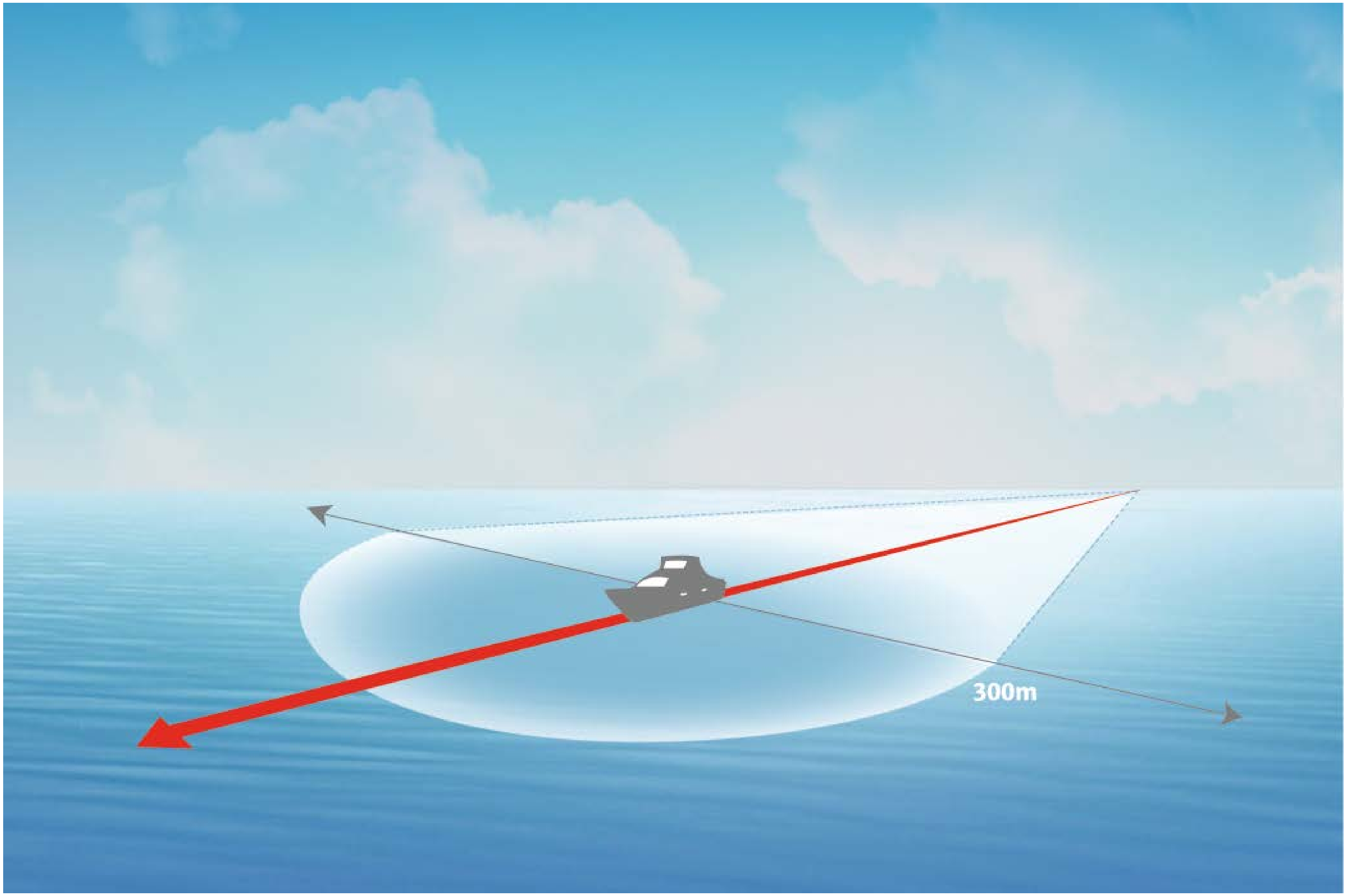

**Figure II. Diagram showing the field of view available during boat surveys.** The boat transect had an intended minimum strip width of 300 m on one side of the vessel, although observations of animals were generally recorded from both sides of the vessel and up to 1,000 m away.

# Chapter 7: Boat survey protocol for Mid-Atlantic Baseline Studies

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 7 Highlights

*The study design, data collection, and data processing protocols for boat-based surveys*

### *Context*[1]

Boat-based surveys with distance estimation are a well-established method of surveying marine wildlife such as seabirds and marine mammals. This chapter describes the study design and protocols used while collecting and post-processing boat survey data for the Mid-Atlantic Baseline Studies Project. These survey data are summarized in Chapter 8, and are used in subsequent analyses in Chapters 10-12. Chapter 10 examines the spatial relationship between feeding seabirds and acoustically detected prey (detected via use of a scientific echo sounder) along boat survey transects. Chapters 11 and 12 use data from boat-based surveys alongside environmental covariate data to predict seabird abundance throughout the study area (with Chapter 12 building off of model development presented in Chapter 11). This boat survey protocol is also referenced alongside digital video aerial survey data in Part IV of this report (Chapters 13-19), including comparisons of the two survey methods and efforts aimed at integrating data from both surveys.

### *Highlights*

- Sixteen surveys were conducted over two years between 2012 and 2014.
- Each survey included 12 linear transects in the vicinity of three Wind Energy Areas (WEAs) in the mid-Atlantic region, totaling approximately 559 km (572 km in the second year of surveys).
- Surveys were conducted on a 55-foot charter vessel during daylight hours, by teams of two observers using a combination of strip and line transect sampling.
- Observers recorded all observed species within a 360˚ arc during normal survey conditions. During inclement weather or when too many animals were present to accurately record observations in 360˚, observers recorded all observed species within a 90˚ arc and within 300 m from the boat.
- Data collected per observation included species, number of individuals seen, behavior, direction of movement, radial distance, degree of the animal's angle to the bow of the boat, and where possible, age and molt state.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

This chapter outlines the protocol followed while conducting boat-based surveys within the Mid-Atlantic Baseline Studies study area. Sixteen surveys were conducted over two years between 2012 and 2014. Each survey included 12 transects within the vicinity of three Wind Energy Areas offshore of Delaware, Maryland, and Virginia, USA, totaling 559 km in 2012-2013 and 572 km in 2013-2014 (with the additional of approximately 12.5 km of transect in Maryland state waters in 2013). Surveys were conducted on a 55-foot charter vessel during daylight hours, using distance sampling. Data collected per observation included species, number of individuals seen, behavior, direction of movement, radial distance, degree of the animal's angle to the bow of the boat, and, where possible, age and molt state. Additionally, environmental covariate data including water temperature, salinity, and hydroacoustic data were collected. Data were post-processed and quality assured/quality controlled prior to use in data analyses discussed elsewhere in this report.

## Boat survey protocol

Boat survey transects extended perpendicularly to the coastline from three nautical miles offshore to the 30 m isobath or the eastern extent of the Wind Energy Areas (WEAs), whichever was furthest (Figure 7-1). Each survey included 12 transects spaced 10 kilometers apart. The transect lines extended at least one transect north and south of each WEA. Total transect distance was approximately 559 km, with an additional 13 km included in the extensions of the three transects in Maryland state waters.

We conducted eight surveys per year on a scheduled basis as the weather allowed. Each survey was conducted during a 2-4 week window (Table 7-1); generally a survey was conducted at the earliest opportunity (based on weather) during each window. Surveys took 4 to 5 days to complete. Our surveys were conducted on a 55-foot charter vessel, the *Stormy Petrel II*, and were staffed with four observers who worked in teams of two. Surveys were run from Ocean City, Maryland, and Virginia Beach, Virginia. Surveys began at first daylight, once light enough for correct species identification. Survey speed was 10 knots unless weather conditions or boat traffic dictated otherwise. The boat returned in the evenings to overnight in port during surveys, with the exception of 7 nights which were spent offshore of Maryland and Virginia (Figure 7-1). Surveys were postponed due to foul weather, for safety reasons, at the discretion of the Captain.

Because of the narrow beam of the *Stormy Petrel II* and unobstructed view from the flying bridge, we recorded all birds seen on a 360° scan (Figure 7-2). Since most surveys were conducted in good weather, this method enabled us to scan both sides of the boat with ease and enabled extensive collection of "distance sampled" data. This was a fairly unusual situation for observations, however; on most survey vessels (and on the *Stormy Petrel II* in periods with high numbers of animals, where we could not accurately count all individuals in a full 360° scan) our protocol was to focus scanning efforts in a 90° arc within a 300 m strip to one side of the boat (Figure 7-2). The side of the vessel used for counts was noted in the data, so that it would be possible to separate observations within the area of primary focus from observations made in the remaining 270° degrees of visibility, if needed. The transect was surveyed continuously using the naked eye or binoculars. Observers regularly scanned ahead for marine mammals and diving birds, or for sitting birds that may have been flushed off the water. Animals within 300 m received priority for recording, though animals outside of the 300 m strip transect were also usually recorded.

One observer continuously scanned for birds, mammals, turtles, and fish, while the second recorded all observations into a Toughbook computer (Panasonic Corp. of North America, Newark, NJ) using the dLOG data entry program (Ford, 1999). Locations, date, and time were automatically recorded every 5 seconds and observations were individually georeferenced. At the beginning of each survey, the recorder entered sea state data using the Beaufort scale (Table 7-2), transect number, observer's initials, visibility, survey ID, station, and platform, changing each as needed throughout the survey (Table 7-3). The second observer/recorder also scanned ahead and to the horizon for marine mammals and sea turtles when possible. We used distance sampling under most conditions, recording distance and angle to every bird or flock of birds, but used the more restricted strip transect (width = 300 m) in areas of high bird density. We defined "high density" as being a greater number of birds in the area than the observers could keep track of, impeding the ability to count all birds present. For each animal, the team recorded: species, number seen, behavior, direction of movement, radial distance, degree of the animal's angle to the bow of the boat, and, where possible, age and molt state (see Table 7-3 for more details about data entry). Behavior codes were assigned based on the behavior initially displayed when the animal was first observed (Table 7-4). Four-letter codes were used to record species identification, as well as abiotic objects of interest such as boats (Table 7-5).

Animal movements were recorded in such a way as to allow for vector analysis (Spear et al., 1992). Radial distance was estimated from the observer to the animal or the center of a group of animals, based on the initial observation. Distance estimates were calibrated between observers using a handmade distance gauge (Gjerdrum et al., 2012) by estimating distances to stationary objects and were estimated as closely as possible (often to the nearest tens of meters for animals closer to the boat, and to the nearest 50 m for animals farther from the boat). The two teams of observers alternated every two hours. At the end of each day of surveying, all data were backed up on portable hard drives.

When weather deteriorated to the point that salt spray threatened the computer, observers moved inside the pilot house (Figure 7-2). Any changes in observer location were recorded in the data (Table 7-3). Although observing from the pilot house restricted the observer's field of vision, compared to observing from the bridge, observers were able to maintain complete view of the strip transect (300 m distance and 90˚ arc).

Environmental data were recorded during the surveys. Sea state and visibility were recorded hourly. Sea surface temperature and salinity were recorded every half hour, using a YSI Pro30 conductivity device (Yellow Springs, OH), with water drawn up from the ocean through the vessel's saltwater pump (located in the vessel's hull) into a bucket. These measurements were taken by off-duty observers to allow the active observers to maintain their positions. Biomass densities were recorded continuously using a Simrad EK60 scientific echo sounder (Kongsberg Maritime AS), employing a 120 kHz transducer. Echo sounding data were processed using Echoview (Myriax Software Pty, Ltd., Hobart, Australia) processing software (Chapter 9). When staying offshore overnight, a passive acoustic monitoring system was operated from the deck of the vessel to record flight calls of nocturnally migrating birds (Chapter 26).

We photographed cetaceans, whenever possible, and submitted these for individual identification using the established North Atlantic Fin Whale[2], and North Atlantic Right Whale[3] catalogues. Surveys were conducted in passing mode, where the boat stayed on transect and at survey speed except when complying with National Marine Fisheries Service (NMFS) rules regarding approaching marine mammals, including rules on vessel speed and encounters with endangered North Atlantic Right Whales (*Eubalaena glacialis*). Surveys in passing mode have been shown to have reduced bias in estimated encounter rates of marine mammals, though they also have lower rates of species identification and poorer estimation for pod size (Barlow, 1997; Douglas et al., 2014). Our research group believes passing mode surveys to be the best method available to ensure that we obtain accurate counts for all other taxa.

Data were collected using methods allowing for vector analysis, or analysis of relative movement (Spear et al., 1992). Constantly counting flying birds can, in some situations, provide a measure of bird “flux” or an overestimation, rather than a true density of birds (Spear et al., 1992). Using the vector method, however, one can estimate the distance of an animal from the observer; the angle from the observer, and the direction of movement. If desired, these data can be combined with known average flight speeds for avian species to calculate a correction factor during data analysis and find absolute densities of flying birds.

Data from boat surveys were consolidated and subject to a quality assurance/quality control (QA/QC) process. Each day of data collection produced an individual .csv file. These data were compiled in Excel for each survey and examined for errors. Common adjustments made during this QA/QC process included: concatenation of split fields; correction of longitude values to make values negative; addition of temperature and salinity measurement fields and incorporation of those data into the table from the comment fields; and ensuring that sighting data were entered in to the correct columns and were checked for consistency. After the initial QA/QC process, data were entered in to an Access database. Data managers then added a unique ID field for each survey and for individual records within each survey, concatenated date and time fields, converted GMT to EST for actual track time, checked for and corrected errors in species codes used, and added taxonomic information for each observation (for full list of species codes and corresponding common and Latin names, see Table 7-5). Finally, the data were geoprocessed to check for locational errors and missing data points. Fully QA/QC’d data were entered in to the Northwest Atlantic Seabird Catalog (formerly the USGS Compendium of Avian Occurrence Information; O’Connell et al., 2009).

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[2] http://www.coa.edu/nafwc.htm

[3] http://rwcatalog.neaq.org/Terms.aspx

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## Figures and tables

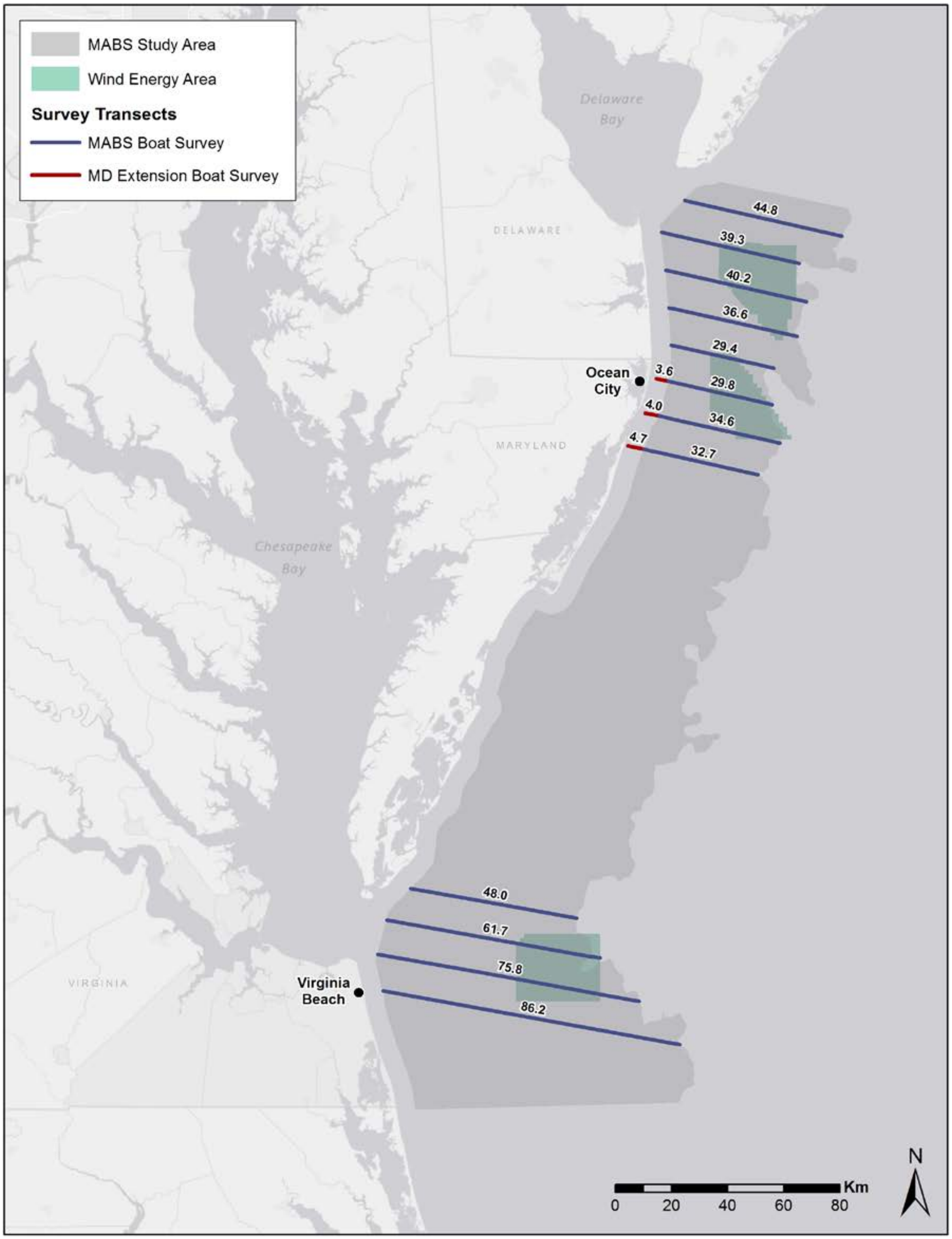

**Figure 7-1. Boat survey map with transect length (km) indicated on each line.** Mid-Atlantic Baseline Studies transects are shown in blue; Maryland extension transects are shown in red. Surveys were conducted out of Ocean City, MD and Virginia Beach, VA.

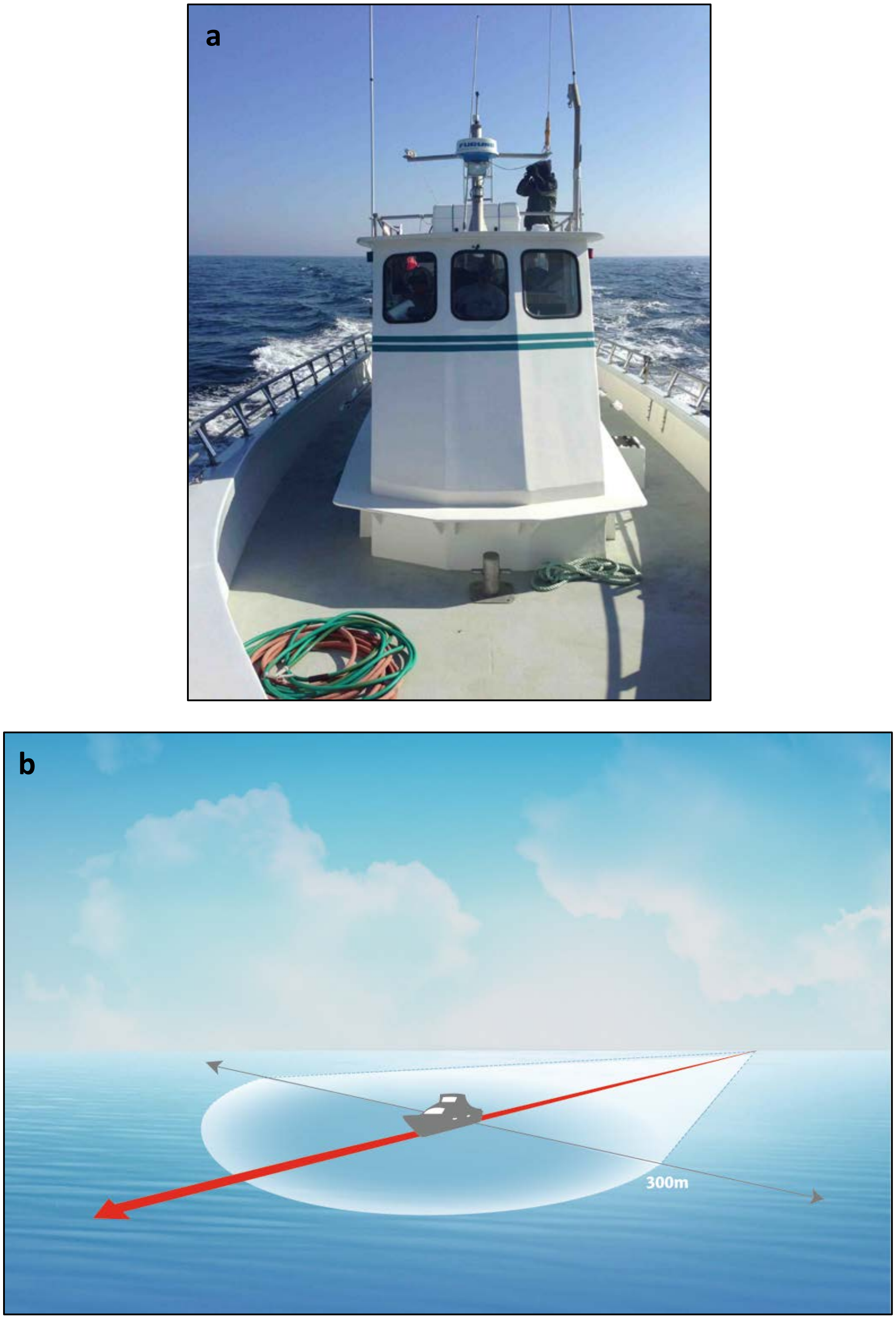

**Figure 7-2. (a) Photograph of the Stormy Petrel with an observer surveying from the flying bridge, above the pilot house. (b) Diagram illustrating the field of view available during boat surveys.** The boat transect has an intended minimum strip width of 300 m, although observations were made up to 1,000 m from the vessel. In most conditions, observers surveyed the full 360˚ from the flying bridge. In inclement weather (when surveying from within the pilot house), and when it was not possible to continuously survey the entire 360˚ arc, observers focused survey efforts within a 90˚arc on a single side of the boat.

**Table 7-1. Ideal schedule for boat surveys.**

| Survey Period | Timing for Boat Surveys |
|---|---|
| Early spring | Late March (into early April if needed) |
| Late spring | Early to mid-May |
| Early summer | June (pref. late June) |
| Late summer | Early August |
| Early fall | September (pref. early to mid-Sept.) |
| Late fall | October (pref. mid- to late Oct.) |
| Early winter | December (pref. mid-Dec.) |
| Late winter | Late Jan. or early Feb. |

**Table 7-2. Beaufort scale.**

| Beaufort Wind Force | Wind Speed (knots) | Sea state description | Beaufort wind force and description |
|---|---|---|---|
| 0 | 0 | Calm, mirror-like | Calm |
| 1 | 1 – 3 | Ripples with the appearance of scales are formed, but without foam crests. | Light air |
| 2 | 4 – 6 | Small wavelets, still short but more pronounced. Crests have glassy appearance and do not break. | Light breeze |
| 3 | 7 – 10 | Large wavelets. Crests begin to break. Foam of glassy appearance. Perhaps scattered white caps. | Gentle breeze |
| 4 | 11 – 16 | Small waves, becoming longer, fairly frequent white caps. | Moderate breeze |
| 5 | 17 – 21 | Moderate waves, taking a more pronounced long form; many white caps formed. Chance of some spray. | Fresh breeze |
| 6 | 22 - 27 | Large waves begin to form; the white foam crests are more extensive everywhere. Probably some spray. | Strong breeze |
| 7 | 28 – 33 | Sea heaps up and white foam from breaking waves begins to be blown in streaks along the direction of the wind. | Moderate gale |
| 8 | 34 – 40 | Moderate high waves of greater length; edges of crests begin to break into spindrift. The foam is blown in well-marked streaks along the direction of the wind. | Fresh gale |
| 9 | 41 – 47 | High waves. Dense streaks of foam along the direction of the wind. Crests of waves begin to topple, tumble and roll over. Spray may affect visibility. | Strong gale |
| 10 | 48 – 55 | Very high waves with long over-hanging crests. Foam blown in dense white streaks along the direction of the wind. Sea takes a white appearance. Visibility affected. | Whole gale |
| 11 | 56 – 63 | Exceptionally high waves (small and medium-sized ships might be for a time lost behind the waves). The sea is completely covered with long white patches of foam lying along the direction of the wind. Edges of the wave crests are blown in to froth. Visibility affected. | Storm |
| 12 | 64 + | The air is filled with foam and spray. Sea completely white with driving spray; visibility very seriously affected. | Hurricane |

**Table 7-3. Data fields, descriptions, and examples for each field.** All fields were required, unless noted otherwise in the description. Fields in italics were automatically entered by dLOG every 5 seconds. Fields in bold and italics were entered by the recorder at the start of the survey and changed when necessary (i.e., sea state changes, observers switch). Fields in bold were entered by the recorder when entering a record for a sighting, temperature and salinity readings, or other notes.

| Field | Description | Example |
|---|---|---|
| *Latitude* | GPS-entered Latitude | 36.944967 |
| *Longitude* | GPS-entered Longitude | -75.93854 |
| *Hour* | GPS-entered Hour | 11 |
| *Minute* | GPS-entered Minute | 8 |
| *Second* | GPS-entered Second | 0 |
| *Year* | GPS-entered Year | 2013 |
| *Month* | GPS-entered Date - Month | 3 |
| *Day* | GPS-entered Date - Day | 20 |
| *Type* | Notes whether data are entered by user or by computer. “GPS” data are entered by the computer every 5 seconds from the built in GPS. When a user enters a sighting or any other information this field is coded as USER. | USER |
| **Spp** | Four letter species code (Table 7-5). Also includes codes for boats and other items of interest. | HERG |
| **Count** | Number of animals seen. | 2 |
| **Behavior** | Add one word describing animal's behavior. Field is not locked. Common behaviors are described in Table 7-4 below. | FLYING |
| **Direction** | Optional. Direction of movement. Possible options are N, NE, E, SE, S, SW, W, or NW. Not applicable for birds that are milling, feeding, or sitting, or for other animals that are stationary. | N |
| **Distance** | Radial distance from the observer to the animal or the center of the group of animals. This distance is based on the observer’s estimate, calibrated to other observers. Estimate based on the first instance you see the animal. Distance estimate is rounded to the nearest 50 or 100 m, unless the animal is within 50 meters of the boat and a more accurate estimate is possible. | 100 |
| **Degree** | The animal’s location in degrees from the bow of the boat. The bow is 0°, one quarter around towards the starboard is 90°, directly off the stern is 180°, and three quarters around off of port is 270°. Estimate is based on the first instance that you see the animal, and is rounded to the nearest 10°. | 350 |
| **Plumage** | Optional. Describe the stage of the bird’s molt, whether it is a light or dark morph bird, or the gannet plumage type. | PRIMARY MOLT |
| **Age** | Optional. Adult, Immature, or Juvenile - or go in to more detail about the bird’s age (e.g., 1S for a first summer gull). | 2W |
| **Comment** | Optional. Any additional comments about the sighting. Also use this field for temperature and salinity measurements or any notes as needed. | WITH GBBG |
| ***Beaufort*** | Approximate description of the current sea state using the Beaufort Scale (Table 7-2). Update every hour. | 3 |
| ***Transect*** | Transect number. If recording between transects use T11 – T12 to indicate which transects you are moving between. Update at the start and end of each transect. | T11 |
| ***Obs*** | Observer’s three initials. Update every time the observer changes. | ETB |
| ***Visib*** | Visibility. Options are 5 = 5 miles plus, 4= 3-5 miles, 3= 1-3 miles, 2= 500 m-1 mile; 1 = 300-500 m visibility. Update as visibility changes. | 5 |

| Field | Description | Example |
|---|---|---|
| ***SurveyID*** | ID code for the survey taking place. Remains the same throughout the survey. Use the formula DOE(MONTH)(YEAR) as in DOEFEB2013. | DOEMARCH2013 |
| ***Station*** | Report which side of the boat the observer is standing on using PORT or STARBOARD. Update when the observer changes sides. | STARBOARD |
| ***Platform*** | Name of the vessel used for surveys. On the *Stormy Petrel II*, we conduct surveys on the flying bridge. If weather conditions deteriorate and observers must go inside the pilot house, indicate that by changing the platform to ONBRIDGE and make a note explaining the change in location. If you return to the flying bridge make a note and change back to STORMYPET | STORMYPET |

**Table 7-4. Common behavior codes.** Codes apply to an animal's initial behavior when first observed.

| Behavior | Description |
|---|---|
| FLYING | Bird or bat is flying – indicate flight direction |
| MILLING | Not flying in any specific direction but circling around above the water. This can indicate birds in a feeding flock. |
| SITTING | Bird sitting on the water. If sitting on an object use this code and make a note saying what it is on. |
| DIVING | Animal at the surface dives under water. |
| TAKING OFF | Bird taking off from the water. |
| FEEDING | Active feeding behaviors observed. |
| PLUNGE DIVING | Specific to birds that feed by plunge diving (e.g., gannets, terns, shearwaters, gulls) |
| PATTERING | Flying low and hitting the water's surface with feet, mainly storm-petrels |
| FOLLOWING | Bird following the boat. This code is used when a bird has already been counted, but continues to follow along with the boat. |
| STATIONARY | Non-avian animal stationary in the water. |
| SWIMMING | Non-avian animal swimming – indicate direction of movement. |
| PORPOISING | Marine mammal moving through the water like a porpoise, up and down through the water. |
| BREACHING | Whale rising up and breaking through the surface of the water and splashing back down into the water. |
| BLOWING | Whale blowing from its spout at the surface. |
| DEAD | Dead animal. |

**Table 7-5. Boat survey species code list for the Mid-Atlantic Baseline Studies and Maryland Projects.** Codes are listed in alphabetical order of common name.

| Species Code | Common Name | Latin Name |
|---|---|---|
| ALGA | Algal bloom | |
| ABDU | American Black Duck | *Anas rubripes* |
| AMCO | American Coot | *Fulica americana* |
| AMPI | American Pipit | *Anthus rubescens* |
| AMRE | American Redstart | *Setophaga ruticilla* |
| AMRO | American Robin | *Turdus migratorius* |
| HOCR | Atlantic Horseshoe Crab | *Limulus polyphemus* |
| ATPU | Atlantic Puffin | *Fratercula arctica* |
| ASDO | Atlantic Spotted Dolphin | *Stenella frontalis* |
| AUSH | Audubon's Shearwater | *Puffinus lherminieri* |
| BAIT | bait ball | |
| BAEA | Bald Eagle | *Haliaeetus leucocephalus* |
| BALN | balloon | |
| BARS | Barn Swallow | *Hirundo rustica* |
| BLSC | Black Scoter | *Melanitta nigra* |
| BLSK | Black Skimmer | *Rynchops niger* |
| BLTE | Black Tern | *Chlidonias niger* |
| BLVU | Black Vulture | *Coragyps atratus* |
| BLKI | Black-legged Kittiwake | *Rissa tridactyla* |
| BLPW | Blackpoll Warbler | *Dendroica striata* |
| BTBW | Black-throated Blue Warbler | *Dendroica caerulescens* |
| BOAC | boat--aircraft carrier | |
| BOBA | boat--barge/barge and tug | |
| BOCA | Boat--cargo | |
| BOCF | Boat--commercial fishing | |
| BOCS | boat--container ship | |
| BOCR | boat--cruise | |
| BOFI | boat--fishing | |
| BOPL | boat--pleasure | |
| BORF | boat--recreational fishing | |
| BORV | boat--research vessel | |
| BOSA | boat--sail | |
| BOSU | boat--submarine | |
| BOTA | boat--tanker | |
| BOTD | boat--trawler/dragger | |
| BOAT | Boat--unidentified | |
| BOGU | Bonaparte's Gull | *Larus philadelphia* |

| Species Code | Common Name | Latin Name |
|---|---|---|
| BODO | Bottlenose Dolphin | *Tursiops truncatus* |
| BRAN | Brant | *Branta bernicla* |
| BRPE | Brown Pelican | *Pelecanus occidentalis* |
| BHCO | Brown-headed Cowbird | *Molothrus ater* |
| BUFF | Bufflehead | *Bucephala albeola* |
| CANG | Canada Goose | *Branta canadensis* |
| CATE | Caspian Tern | *Sterna caspia* |
| CEDW | Cedar Waxwing | *Bombycilla cedrorum* |
| CHAN | Change in personnel, station, transect, etc. | |
| CODO | Common Dolphin | *Delphinus delphis* |
| COGO | Common Goldeneye | *Bucephala clangula* |
| COLO | Common Loon | *Gavia immer* |
| COMU | Common Murre | *Uria aalge* |
| COTE | Common Tern | *Sterna hirundo* |
| COSH | Cory's Shearwater | *Calonectris diomedea* |
| DASC | Dark scoter - either black scoter or surf scoter | |
| DEJU | Dark-eyed Junco | *Junco hyemalis* |
| DCCO | Double-crested Cormorant | *Phalacrocorax auritus* |
| DOVE | Dovekie | *Alle alle* |
| DUNL | Dunlin | *Calidris alpina* |
| FIWH | Fin Whale | *Balaenoptera physalus* |
| FLJE | flotsam and jetsam | |
| FOTE | Forster's Tern | *Sterna forsteri* |
| GLGU | Glaucous Gull | *Larus hyperboreus* |
| GCKI | Golden-crowned Kinglet | *Regulus satrapa* |
| GBBG | Great Black-backed Gull | *Larus marinus* |
| GBHE | Great Blue Heron | *Ardea herodias* |
| GREG | Great Egret | *Ardea alba* |
| GRSH | Greater Shearwater | *Puffinus gravis* |
| GRHE | Green Heron | *Butorides Virescens* |
| GWTE | Green-winged Teal | *Anas crecca* |
| HERG | Herring gull | *Larus argentatus* |
| HOGR | Horned Grebe | *Podiceps auritus* |
| HUWH | Humpback Whale | *Megaptera novaeangliae* |
| LAGU | Laughing Gull | *Larus atricilla* |
| LESA | Least Sandpiper | *Calidris minutilla* |
| LETE | Least Tern | *Sterna antillarum* |

| Species Code | Common Name | Latin Name |
|---|---|---|
| LETU | Leatherback Turtle | *Dermochelys coriacea* |
| LBBG | Lesser Black-backed Gull | *Larus fuscus* |
| LEYE | Lesser Yellowlegs | *Tringa flavipes* |
| LIGU | Little Gull | *Larus minutus* |
| LOTU | Loggerhead Turtle | *Caretta caretta* |
| LTDU | Long-tailed Duck | *Clangula hyemalis* |
| MALL | Mallard | *Anas platyrhynchos* |
| MASH | Manx Shearwater | *Puffinus puffinus* |
| MERL | Merlin | *Falco columbarius* |
| MIWH | Minke Whale | *Balaenoptera acutorostrata* |
| MOON | Moon Jellyfish | *Aurelia aurita* |
| MOWA | Mourning Warbler | *Oporornis philadelphia* |
| MYWA | Myrtle Warbler | *Dendroica c. coronata* |
| NONE | none | |
| NOFL | Northern Flicker | *Colaptes auratus* |
| NOFU | Northern Fulmar | *Fulmarus glacialis* |
| NOGA | Northern Gannet | *Morus bassanus* |
| NOHA | Northern Harrier | *Circus cyaneus* |
| NOWA | Northern Waterthrush | *Seiurus noveboracensis* |
| NOTE | note | |
| MOLA | Ocean Sunfish (Mola) | *Mola mola* |
| OSPR | Osprey | *Pandion haliaetus* |
| PAWA | Palm Warbler | *Dendroica palmarum* |
| PAJA | Parasitic Jaeger | *Stercorarius parasiticus* |
| POJA | Pomarine Jaeger | *Stercorarius pomarinus* |
| PONY | Pony | |
| PMOW | Portuguese Man o' War | *Physalia physalis* |
| PUMA | Purple Martin | *Progne subis* |
| RAZO | Razorbill | *Alca torda* |
| REBA | Red Bat | *Lasiurus borealis* |
| REPH | Red Phalarope | *Phalaropus fulicaria* |
| RBME | Red-breasted Merganser | *Mergus serrator* |
| RBNU | Red-breasted Nuthatch | *Sitta canadensis* |
| RNGR | Red-necked Grebe | *Podiceps grisegena* |
| RNPH | Red-necked Phalarope | *Phalaropus lobatus* |
| RTLO | Red-throated Loon | *Gavia stellata* |
| RWBL | Red-winged Blackbird | *Agelaius phoeniceus* |
| RIWH | Right Whale | *Eubalaena glacialis* |
| RBGU | Ring-billed Gull | *Larus delawarensis* |

| Species Code | Common Name | Latin Name |
|---|---|---|
| ROST | Roseate Tern | *Sterna dougallii* |
| ROYT | Royal Tern | *Sterna maxima* |
| RCKI | Ruby-crowned Kinglet | *Regulus calendula* |
| RUTU | Ruddy Turnstone | *Arenaria interpres* |
| SAGU | Sabine's Gull | *Xema sabini* |
| SAND | Sanderling | *Calidris alba* |
| SEWH | Sei Whale | *Balaenoptera borealis* |
| SEPL | Semipalmated Plover | *Charadrius semipalmatus* |
| SESA | Semipalmated Sandpiper | *Calidris pusilla* |
| SBDO | Short-billed Dowitcher | *Limnodromus griseus* |
| SMTU | Small turtle - Loggerhead, Green, Hawksbill, or Kemp's Ridley | |
| SOSP | Song Sparrow | *Melospiza melodia* |
| SOSH | Sooty Shearwater | *Puffinus griseus* |
| SUBU | Sulfur Butterfly spp. | *Coliadinae spp.* |
| SUSC | Surf Scoter | *Melanitta perspicillata* |
| TEWA | Tennessee Warbler | *Vermivora peregrina* |
| TBMU | Thick-billed Murre | *Uria lomvia* |
| TRAN | transect point (such as temperature/salinity values) | |
| TRES | Tree Swallow | *Tachycineta bicolor* |
| UNAL | Unidentified Alcid | *Alcidae spp.* |
| BUMB | Unidentified Bee | |
| UNBI | Unidentified Bird | *Aves* |
| UBUT | Unidentified Butterfly | |
| UNCO | Unidentified Cormorant | *Phalacrocorax spp* |
| UNDO | Unidentified Dolphin | *Unidentified Delphinidae* |
| UNDU | Unidentified Duck | |
| UNEI | Unidentified Eider | *Somateria spp.* |
| FISH | Unidentified Fish | *Osteichthyes* |
| UFFI | Unidentified Flying Fish | *Exocoetidae spp.* |
| UFOB | Unidentified flying object (animal-origin) | |
| UNGU | Unidentified Gull | |
| UNHU | Unidentified Hummingbird | |
| UNJA | Unidentified Jaeger | *Stercorarius spp.* |
| UNJE | Unidentified Jellyfish | *Scyphozoa spp.* |
| UNLA | Unidentified Large Alcid (Razorbill or Murre) | |

| Species Code | Common Name | Latin Name |
|---|---|---|
| UNLG | Unidentified Large Gull | |
| UNLW | Unidentified Large Whale | *Cetacea spp.* |
| UNLO | Unidentified Loon | *Gavia spp.* |
| UNMM | Unidentified Marine Mammal | *Mammalia* |
| UNPA | Unidentified Passerine | |
| PEEP | Unidentified Peep | |
| UNPH | Unidentified Phalarope | *Phalaropus spp.* |
| UNRA | Unidentified Ray | *Rajiformes spp.* |
| SCAU | Unidentified Scaup | *Aythya marila or A. affinis* |
| UNSC | Unidentified Scoter | *Melanitta spp.* |
| TURT | Unidentified Sea Turtle | |
| UNSH | Unidentified Shearwater | *Procellariidae spp.* |
| SHOR | Unidentified Shorebird | |
| UNSK | Unidentified Skua | *Stercorarius spp.* |
| UNSA | Unidentified Small Alcid (Puffin/Dovekie) | *Alle alle/Fratercula arctica* |
| UNSS | Unidentified Small Shearwater (Audubon's, Manx, or Little) | *Puffinus lherminieri , P. puffinus, or P. assimilis* |
| SPAR | Unidentified Sparrow | *Emberizidae spp.* |
| UNSP | Unidentified Storm-petrel | |
| SWAL | Unidentified Swallow | *Hirundinidae spp.* |
| TEAL | Unidentified Teal | |
| UNTE | Unidentified Tern | *Sterna spp.* |
| THSH | Unidentified Thresher Shark | |
| UNWA | Unidentified Warbler | |
| UNWH | Unidentified Whale | *Cetacea spp.* |
| UNKN | unknown | |
| WHIM | Whimbrel | *Numenius phaeopus* |
| WRSA | White-rumped Sandpiper | *Calidris fuscicollis* |
| WWSC | White-winged Scoter | *Melanitta fusca* |
| WILL | Willet | *Catoptrophorus semipalmatus* |
| WIPL | Wilson's Plover | *Charadrius wilsonia* |
| WISP | Wilson's Storm-petrel | *Oceanites oceanicus* |
| WODU | Wood Duck | *Aix sponsa* |

# Chapter 8: Summary of boat survey data

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 8 Highlights

*Results from boat-based survey data collected in the mid-Atlantic study area.*

### *Context*[1]

Standardized boat-based surveys are widely used to obtain density data for birds, sea turtles, and marine mammals. Chapter 7 describes the standardized protocol used to collect data. Chapter 9 describes the methods used to collect data on relative biomass of aquatic prey below the study vessel, using a scientific echo sounder that was deployed on all surveys.

This chapter describes the basic results of the boat surveys, including counts of various species and groups, and a discussion of identification rates. Subsequent chapters in Part III of the report present a more detailed analysis of the relationship between foraging seabirds and acoustically detected prey (Chapter 10), and use hierarchical Bayesian statistical approaches to estimate abundances and distribution patterns of seabirds in relation to habitat variables, while correcting for certain biases associated with boat methodologies (e.g., distance bias; Chapters 11-12). Part IV of this report (Chapters 13-19) combines data from boat-based surveys with data from digital video aerial survey approaches to develop a more comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area.

### *Study goal/objectives*

Summarize animal distribution and abundance data that were collected in the mid-Atlantic study area using a well-known and widely used survey method.

### *Highlights*

- There were 64,642 animals observed over two years of surveys; most of the animals were birds (over 62,000) though there were aquatic animals observed as well.
- The highest counts of animals occurred in December and January.
- The most abundant animals observed were scoters (*Melanitta* spp., 34% of the data), primarily Black Scoters (*M. americana*), most commonly in winter surveys.
- Other abundant or commonly observed animals included several species of gulls (*Laridae*), Northern Gannets (*Morus bassanus*), loons (*Gavia* spp.), and dolphins (*Odontoceti*).
- Rates of identification of animals to species were high for most animals, with the exception of scoters.

### *Implications*

Boat-based surveys are a well established means to collect distribution and abundance data for marine animals, and the study design used for these surveys may have been particularly useful for monitoring many species of birds. Many taxa were readily identified using this method, though there were few aquatic animals observed relative to birds.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

Information on bird, sea turtle, and marine mammal movements and abundance was collected from a boat platform using a standardized protocol. Between 2012 and 2014, 16 surveys conducted along twelve transect lines that focused on three offshore Wind Energy Areas (WEAs) in the mid-Atlantic U.S. A total of 64,642 animals were observed on the survey, including over 62,000 birds and 1,500 aquatic animals in a cumulative 10,698 km of transects. The most animals were observed in December and January, when large flocks of wintering birds were present in the study area. Scoters were the most abundant animal, making up 34% of observations; Gulls and Terns were the next most abundant animal group, at 23% of observations, and were primarily Bonaparte's Gulls and Laughing Gulls. Northern Gannets, loons, and auks were also commonly observed. Smaller numbers of aquatic animals such as dolphins, sea turtles, and baleen whales were observed. Rare observations included two Eastern Red Bats, three Roseate Terns, a Bald Eagle, and 35 whales of at least five different species. Most animals were successfully identified to species, save for scoters, which were often observed in large flocks at some distance from the vessel. More in-depth analyses of the boat survey data can be found in subsequent chapters in this report.

## Introduction

The mid-Atlantic region is important for a broad range of marine wildlife species throughout the year. Some breed in the area, such as coastal birds and sea turtles, while others visit from the southern hemisphere in their non-breeding season, such as shearwaters. In the fall, many summer residents migrate south to breed or winter in warmer climes, and they are replaced by species that have travelled south from their breeding grounds to winter in the mid-Atlantic. Additionally, many marine and landbird species make annual migrations up and down the eastern seaboard and travel directly through the mid-Atlantic region in spring and fall. Thus, many species use or funnel through the mid-Atlantic region each year, resulting in a complex ecosystem where community composition is constantly shifting, and the temporal and geographic patterns are highly variable.

In this study, we aimed to produce the wildlife data required to inform siting and permitting processes for offshore wind energy development in the mid-Atlantic. We collected information on bird, sea turtle, and marine mammal abundance and movements over a two-year period (2012-2014) using a variety of technologies and methods to examine spatial patterns and trends. Standardized boat-based surveys are a widely used method of estimating densities for coastal and marine birds, sea turtles, and marine mammals (Gjerdrum et al., 2012; Tasker et al., 1984), and are a key part of the Department of Energy (DOE)-funded Mid-Atlantic Baseline Studies Project and state-funded Maryland Project. We conducted boat surveys for wildlife within the study area on the Outer Continental Shelf to accompany and compare with the data from simultaneously conducted digital aerial surveys (for more information on aerial surveys, and for analyses synthesizing boat and aerial datasets, see Parts II and IV of this report). Boat surveys were particularly focused in and around three federally designated Wind Energy Areas (WEAs) off the coasts of Delaware, Maryland, and Virginia. Here, we examine the boat survey results in detail, including discussion of observation rates and species identification rates.

## Methods

Between April 2012 and April 2014, project partners conducted sixteen large-scale boat based visual surveys (Table 8-1) across the mid-Atlantic study area, focusing on three offshore WEAs (total transect length = 559 km, Figure 8-1). Details on survey design for our boat surveys can be found in Chapter 7. In the second year of surveys (March 2013 – January 2014), the western ends of three survey lines off of Maryland were extended into state waters (total transect length = 571 km, Figure 8-1). Both Mid-Atlantic Baseline Studies and Maryland Project survey data are presented in this report.

This chapter presents summaries of raw count data from the boat surveys on a monthly, seasonal, and annual basis. We also discuss identification rates for the most common species groups. Chapters 10-12 present additional analyses of the boat survey data. Chapters 13-19 present additional information on comparing digital video aerial to boat survey results, and integrating data from both survey platforms to develop in-depth analyses of wildlife distributions and relative abundances.

## Results

A total of 64,462 animals were observed in the sixteen boat surveys, including over 62,000 birds and over 1,500 aquatic animals (including cetaceans, sea turtles, sharks, and fish; Table 8A-1) in a cumulative 10,698 km of survey transects. At least 97 species of birds and 12 species of aquatic animals are represented in this dataset. Seventy-two percent of animals observed in the study were identified to species level; most unidentified animals were scoters, with an approximately 97% identification rate excluding this taxon. The greatest numbers of animals were observed in December and January, when large flocks of birds wintered in the study area (Table 8-2). It should be noted that data collected between the two years are not entirely compatible, as the exact timing of surveys can have a huge effect on species counts, particularly in migration periods when large numbers of wintering birds could be moving in or out of the study area, and a week's difference in survey dates could affect overall abundance observed.

### *Relative abundance of counts*

#### *Birds*

Scoters, a genus of sea ducks that in the mid-Atlantic includes Black Scoter (*Melanitta americana*), White-winged Scoter (*M. fusca*), and Surf Scoter (*M. perspicillata*), were the most abundant avian taxon observed in boat surveys (Figure 8-2), making up over 34% of all observations. Scoters were mostly in the region from December through April (Figure 8-4). There were annual fluctuations in the numbers of scoters observed; twice as many scoters were observed in January 2013 as in January 2014. Twelve other species of anatids (ducks and geese) were also observed in the study area (Table 8A-1).

Gulls and terns (*Laridae*) were observed throughout the year (Figure 8-4) and were the next most abundant animal group (23% of all data, Table 8-2). Bonaparte's Gull (*Choicocephalus philadelphia*) was the most common gull species observed (12% overall), and were most abundant in the winter months; in a single survey, in December 2013, over 5,500 were observed, many more than were observed in the December 2012 survey (Table 8A-1). Laughing Gull (*Leucophaeus atricilla*) was the next most abundant gull (4% overall), and was present in the study area in spring, summer and fall. Herring Gulls (*Larus* *smithsonianus,* 1.5%) and Great Black-backed Gulls (*L. marinus*, 1.6%) were observed consistently throughout the year in almost every survey, but with peaks in abundance in the winter. Six other gull species were also observed (Table 8A-1). Common Terns (*Sterna hirundo*) were abundant in several of the surveys, and present through the spring and fall (2%). Three endangered Roseate Terns (*Sterna dougallii*) were observed feeding on June 20, 2012, and May 9, 2013. Five other tern species were also observed in the study area (Table 8A-1).

Northern Gannets (*Morus bassanus*) were the next most abundant bird observed (22% of all data, Figure 8-2), and were most common in the winter to early spring months, with the highest numbers observed in January (Table 8A-1). Similar numbers of gannets were observed between survey years, with some fluctuations depending in part on timing of surveys in each year.

Other avian taxa observed in boat surveys included loons (Gaviidae), auks (Alcidae), storm-petrels (Hydrobatidae), shearwaters and fulmars (Procellariidae), shorebirds, and passerines. Loons made up 9% of all observations and were observed mostly in the winter and spring (Figure 8-4), with the highest number of loons observed in December of 2013. Common Loons (*Gavia immer*) were observed about three times as frequently as Red-throated Loons (*G. stellata*). Auks were observed in the winter and early spring (Figure 8-4). Razorbills were the most abundant (*Alca torda,* 1.7%), followed by Dovekies (*Alle alle,* 0.69%). Wilson's Storm-Petrels (*Oceanites oceanicus*) were observed in the study during summer surveys (1.3%). Thirteen species of shorebirds were also observed, the most abundant being Red Phalaropes (*Phalaropus fulicarius*) in spring surveys (0.59%). Six species of procellarids were observed, with very similar numbers of Cory's Shearwater (*Calonectris diomedea*), Sooty Shearwater (*Puffinus griseus*), Great Shearwater (*P. gravis*), and Manx Shearwater (*P. puffinus*) observed (all at 0.10% of all observations). Twenty-one different species of passerines were observed in the study, and the most common were Purple Martins (*Progne subis,* 0.09%; see Table 8A-1 for the full list of species). Fourteen Osprey (*Pandion haliaetus*) and one Bald Eagle (*Haliaeetus leucocephalus*) were observed.

### *Aquatic animals*

Dolphins were the most common non-avian animal group observed (Figure 8-5), with Bottlenose Dolphins (*Tursiops truncatus*) the most abundant (1.4% of all observations). They were observed predominantly in warmer months (Figure 8-5). Fewer numbers of Common Dolphins were observed (*Delphinus delphis*, 0.32%), and four Atlantic Spotted Dolphins (*Stenella frontalis*) were observed on one June survey (Table 8A-1). Large whales were also observed in the study in winter (Figure 8-5), including 12 Humpback Whales (*Megaptera novaeangliae*), three Fin Whales (*Balaenoptera physalus*) and three Minke Whales (*B. acutorostrata*), and one sighting each of North Atlantic Right Whales (*Eubalaena glacialis*) and Sei Whales (*B. borealis*). Two species of turtles were observed in the summertime (0.18%, Table 8A-1). Of the two species observed, Loggerhead Turtles (*Caretta caretta*) were the most common (0.14%), followed by Leatherback Turtles (*Dermochelys coriacea*, 0.02%). Fish were not a focus of the current study, but some were identified to Flying Fish spp. (Exocoetidae), and three Ocean Sunfish were observed (*Mola mola*; Table 8A-1).

#### *Bats*

Two bats were observed flying within normal line of sight (within 20 m of sea level), one in each September that the study took place (Table 8-2). Both were Eastern Red Bats (*Lasiurus borealis*). In 2012 the bat was seen 44 km east of Delaware, and in 2013 the bat was seen about 65 km off the coast of Virginia (see Chapter 14 for observation map). These observations were notable as they contributed towards evidence of offshore migrations of red bats in the study area, which was reported in a publication along with observations from our aerial survey data and historical records (Hatch et al., 2013).

### *Identification rates*

The bulk of scoters (73%) were not identified to species, but those identified were predominantly Black Scoters (*Melanitta nigra*; 4.5% of all animals, Figure 8-6). Scoters were often observed in large flocks, some far from the boat, which led to lower levels of identification to species (for additional discussion of this topic, see Chapter 12). Identification rates for other avian species were fairly high; most gull and tern observations were made to the species level, as were 92.2% of alcids and 91.4% of loons (Figure 8-6). Toothed whales were primarily identified to species (89%), most commonly Bottlenose Dolphins (Figure 8-7), though there were 15 sightings of unidentified large whales (Figure 8-7). Larger whales were identified to species 57% of the time, and Humpback Whales were the most commonly identified, but the “passing mode” in which surveys were conducted prevented accurate species identification and probably accurate estimation of group sizes for cetaceans in some cases (see Chapters 6 and 15 for more information). Sea turtles were almost always identified to species (88%), and most were Loggerhead Sea Turtles.

## Discussion

The most abundant animals observed in the boat-based surveys were scoters, gulls, Northern Gannets, and loons, which is similar to the high resolution digital video aerial surveys (Chapter 5). One notable difference between the results of the two study methods was the number of aquatic animals observed relative to the number of animals observed overall; a much higher number of aquatic animals were seen from the digital video aerial study, likely in part as a result of the differences between the observers’ perspectives. However, the boat observers’ perspective looking forward from the survey vessel (Chapter 7) appeared to provide an excellent means to spot distant large birds (e.g., Northern Gannets, shearwaters), large flocks of birds (e.g., scoters), the spouts and surfaced body parts of large whales, and pods of dolphins. Further examination of the differences in results from the two survey methods may be found in Chapters 13 and 14.

Rates of identification to species level were quite high for boat surveys, especially for avian groups. The notable exception was scoters, likely because many large flocks of scoters were visible at great distances from the boat, and were called either unidentified scoters or “dark scoters” (Black Scoter or Surf Scoter). The ability to see large flocks of birds at a great distance may be an advantage of boat surveys, but depending on the taxon, identifications to species level may be difficult in these cases. Even closely related species often have differences in their conservation status, ecology, and habitat requirements, so obtaining species-specific information on distributions, abundance, and habitat use is often important for identifying potential conflicts with anthropogenic activities in the marine environment.

Many of the subsequent chapters in Parts III and IV of this report use modeling approaches to investigate the distribution and abundance data from the boat-based surveys (including Chapters 11-13, 15-16, 18-19). These methods estimate detection as well as abundance, which helps correct for various types of observation bias, including distance bias, where observers are less likely to see animals located farther from the survey transect (Gardner et al., 2008; Spear et al., 2004). These methods can also incorporate environmental covariates into the model structure, in order to predict animal distributions and abundance on a broader geographic scale than where surveys were actually conducted.

Estimating spatial patterns in relative abundance in the offshore environment can be difficult, as these systems are extremely dynamic, animals tend to show high degrees of spatial autocorrelation or aggregative behaviors, and surveys are logistically challenging and more expensive than terrestrial equivalents. In the past century, offshore surveys have mostly been carried out by direct visual observation of wildlife from boats (or aircraft). Standardized methods using strip or line transects are common for monitoring marine species on boat-based surveys (Camphuysen and Garthe, 2004; Camphuysen et al., 2004; Gjerdrum et al., 2012; Tasker et al., 1984), and have been refined over the last few decades to achieve more accurate estimates of population size (Buckland et al., 2001, 1993; Evans and Hammond, 2004; Kaschner et al., 2012). These survey results on the geographic distributions and relative abundance of wildlife in the mid-Atlantic are expected to be useful for minimizing impacts to wildlife populations from offshore wind energy development in several ways. These data can inform the siting of future projects, and can also be used to inform the permitting process for projects, by contributing data towards National Environmental Protection Act (NEPA) and other regulatory requirements, and by helping to define target taxa or research priorities on which to focus on during site-specific pre- and post-construction monitoring studies. Detailed baseline survey data can also inform mitigation efforts, by presenting temporal data on community composition, distributions, and abundance that can be used to time certain activities to coincide with reduced potential for exposure of key populations.

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## Figures and tables

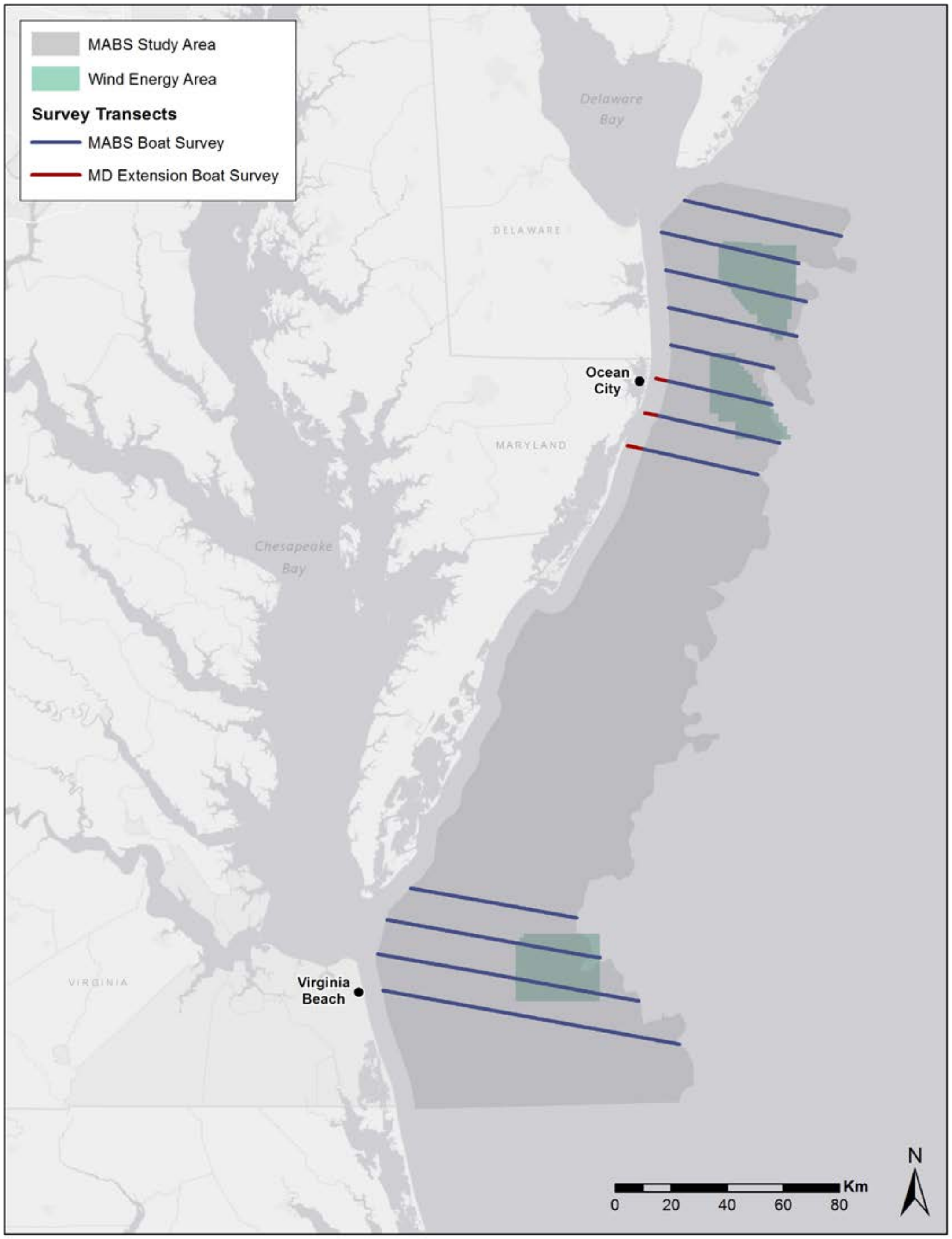

**Figure 8-1. Map of boat survey transects for the Mid-Atlantic Baseline Studies and Maryland projects.** Lines in blue are part of the Mid-Atlantic Baseline Studies, and lines red are part of the Maryland study (surveys conducted in March 2013-January 2014).

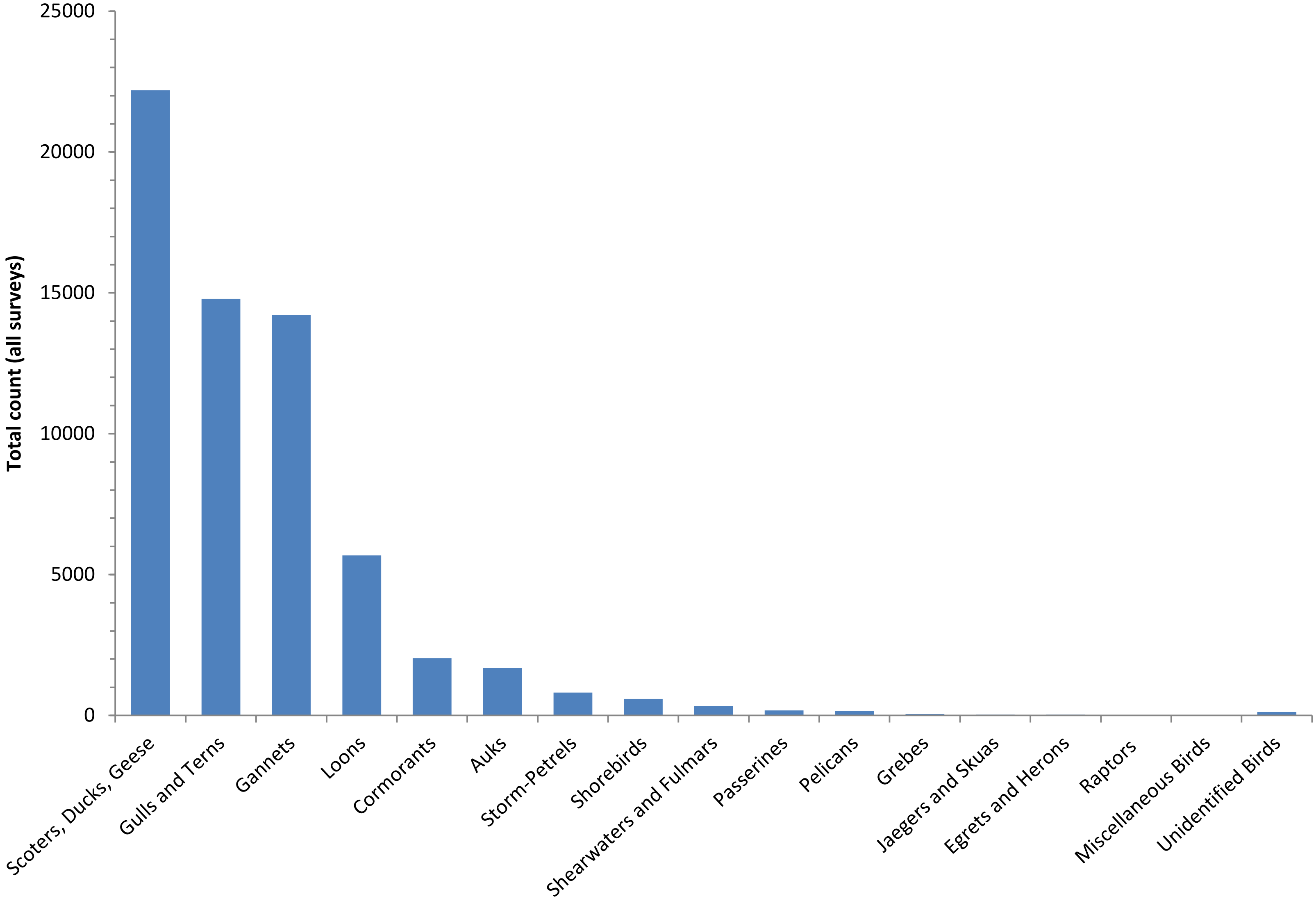

**Figure 8-2. Avian observations (raw counts) from April 2012-April 2014 boat surveys, by family or species group.** "Miscellaneous birds" include coots, hummingbirds, vultures, and woodpeckers. "Unidentified birds" were not identified to species or to a specific avian taxonomic group.

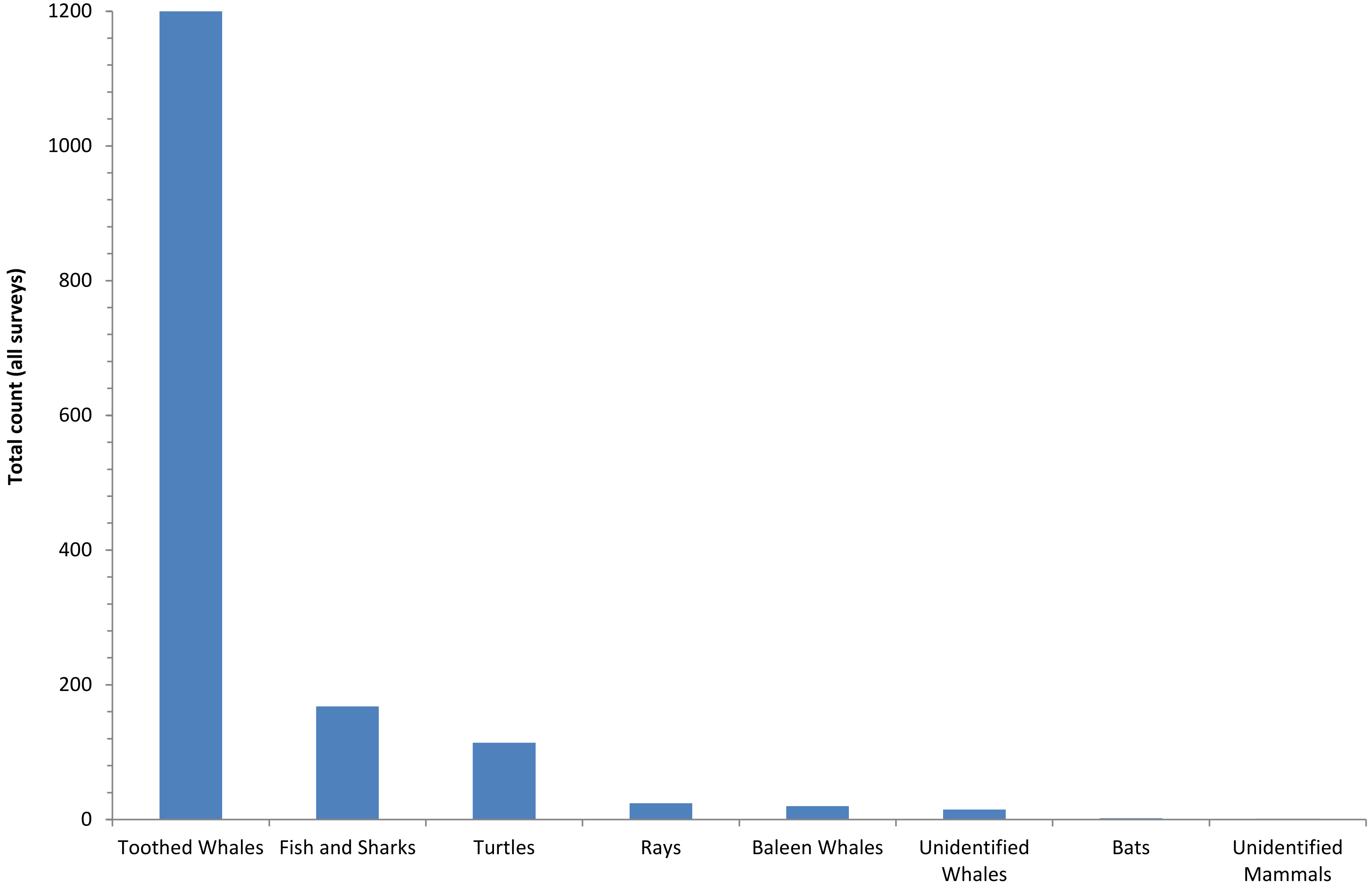

**Figure 8-3. Observations of aquatic animals and bats from April 2012-April 2014 boat surveys, by species group.** “Toothed whales” include dolphins and porpoises. Schools of fish were excluded from this figure.

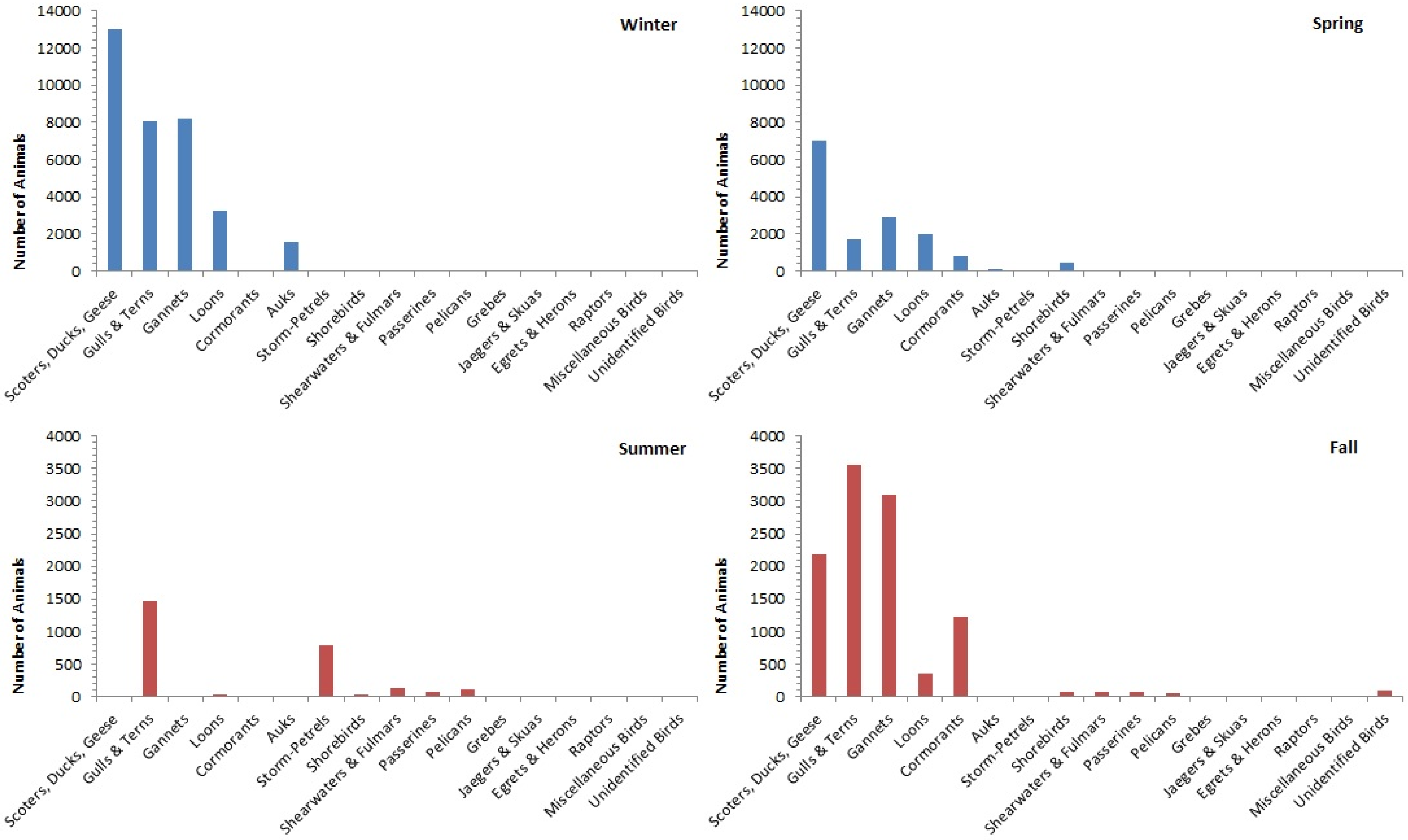

**Figure 8-4. Abundance of birds by family or group in a) winter (Dec.-Feb.), b) spring (Mar.-May), c) summer (Jun.-Aug.), and d) fall (Sep.-Nov.).** Note different y-axis between top (winter and spring) and bottom (summer and fall) graphs. X-axes are in order of overall abundance by family or group across all surveys.

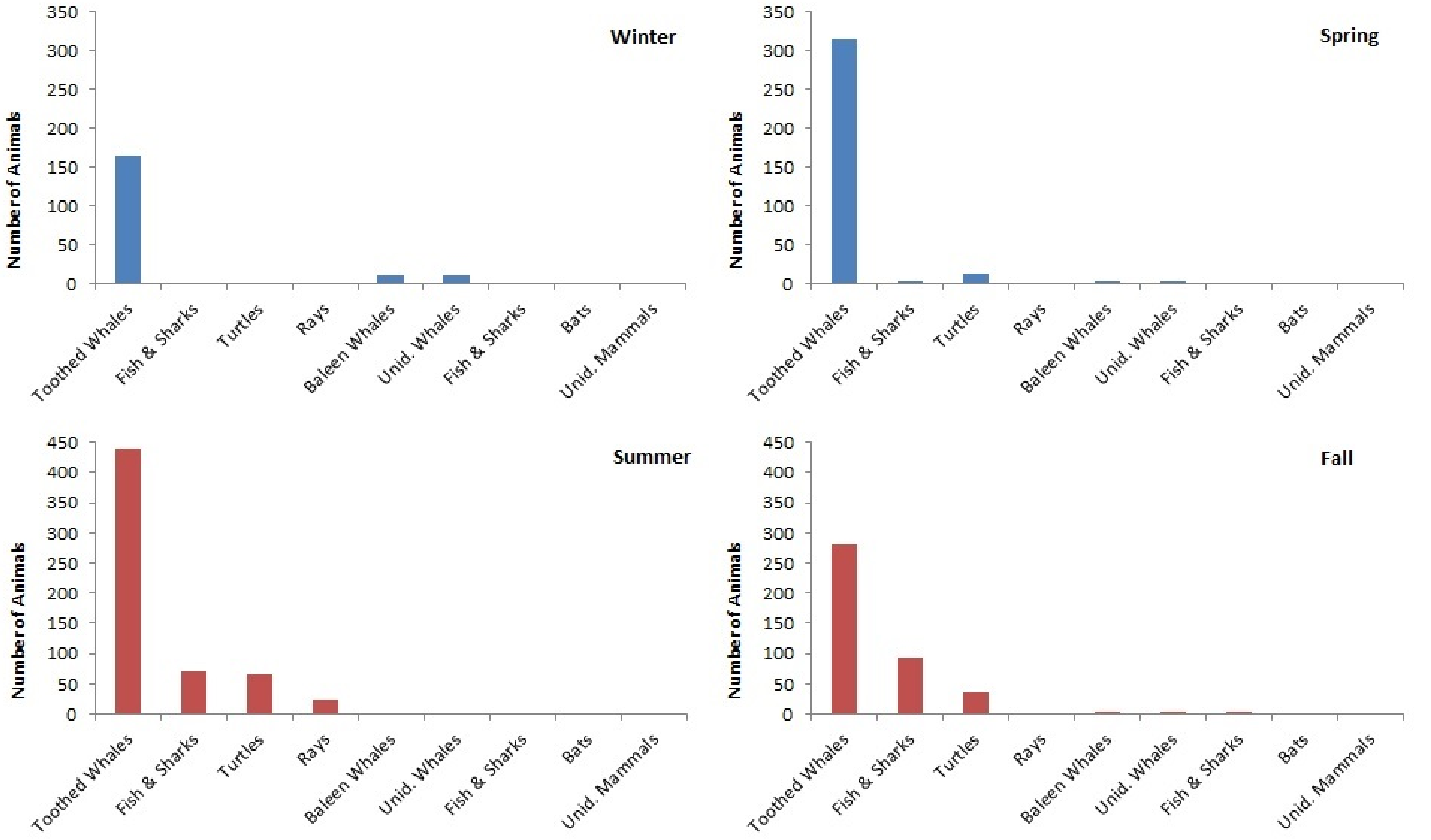

**Figure 8-5. Abundance of non-avian animals by family or group in a) winter (Dec.-Feb.), b) spring (Mar.-May), c) summer (Jun.-Aug.), and d) fall (Sep.-Nov.).** Note different y-axis between top (winter and spring) and bottom (summer and fall) graphs. X-axes are in order of overall abundance by family or group across all surveys.

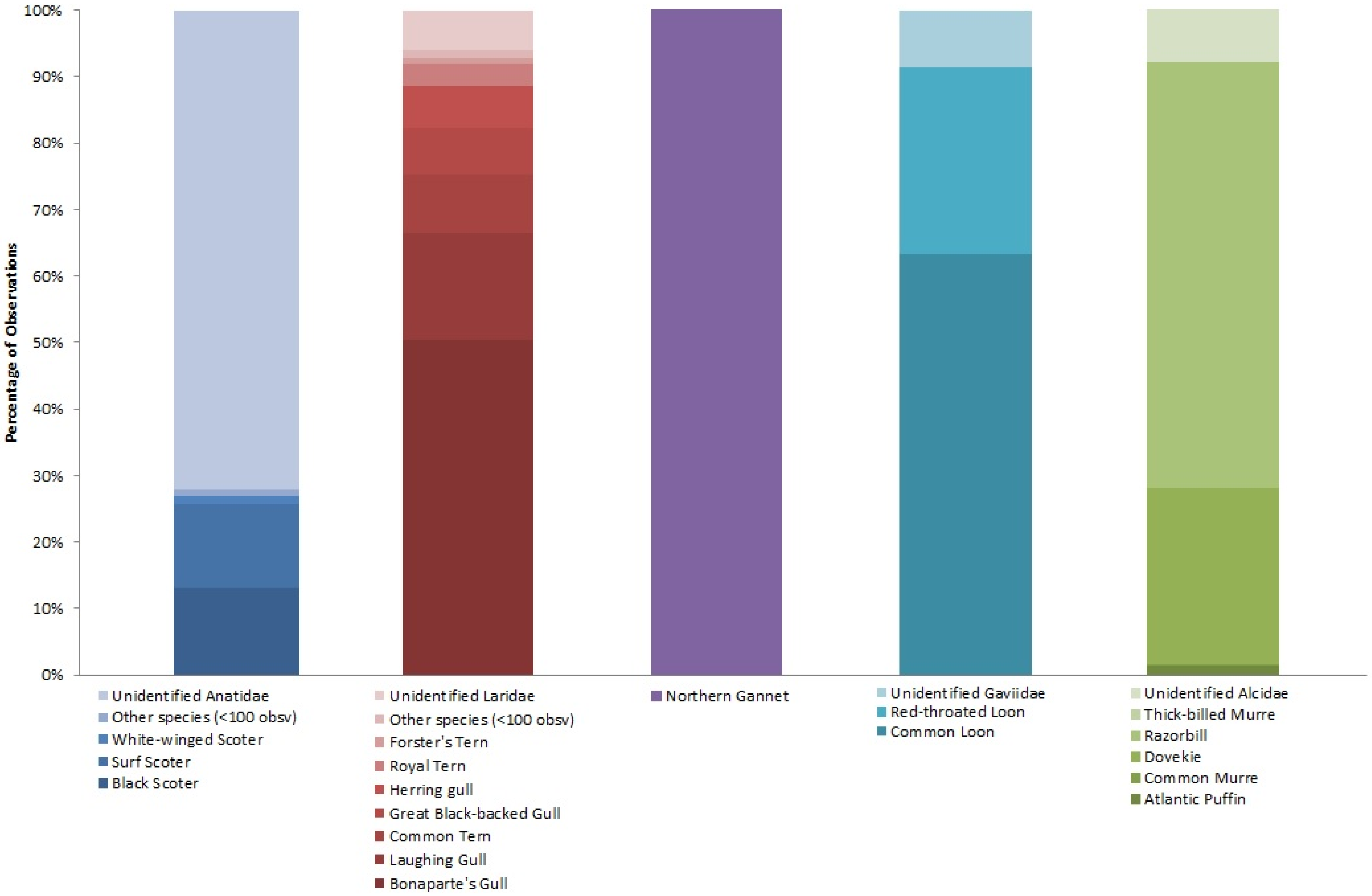

**Figure 8-6. Identification rates for the most common bird groups observed in boat surveys.** Identifications to species level are shown in darker colors. "Other species" in the Anatidae (blue; n=22,193) and Laridae (red; n=14,789) can be found in **Table 8A-1**. Sample sizes for gannets, loons, and auks are 14,091, 5,684 and 1,690, respectively.

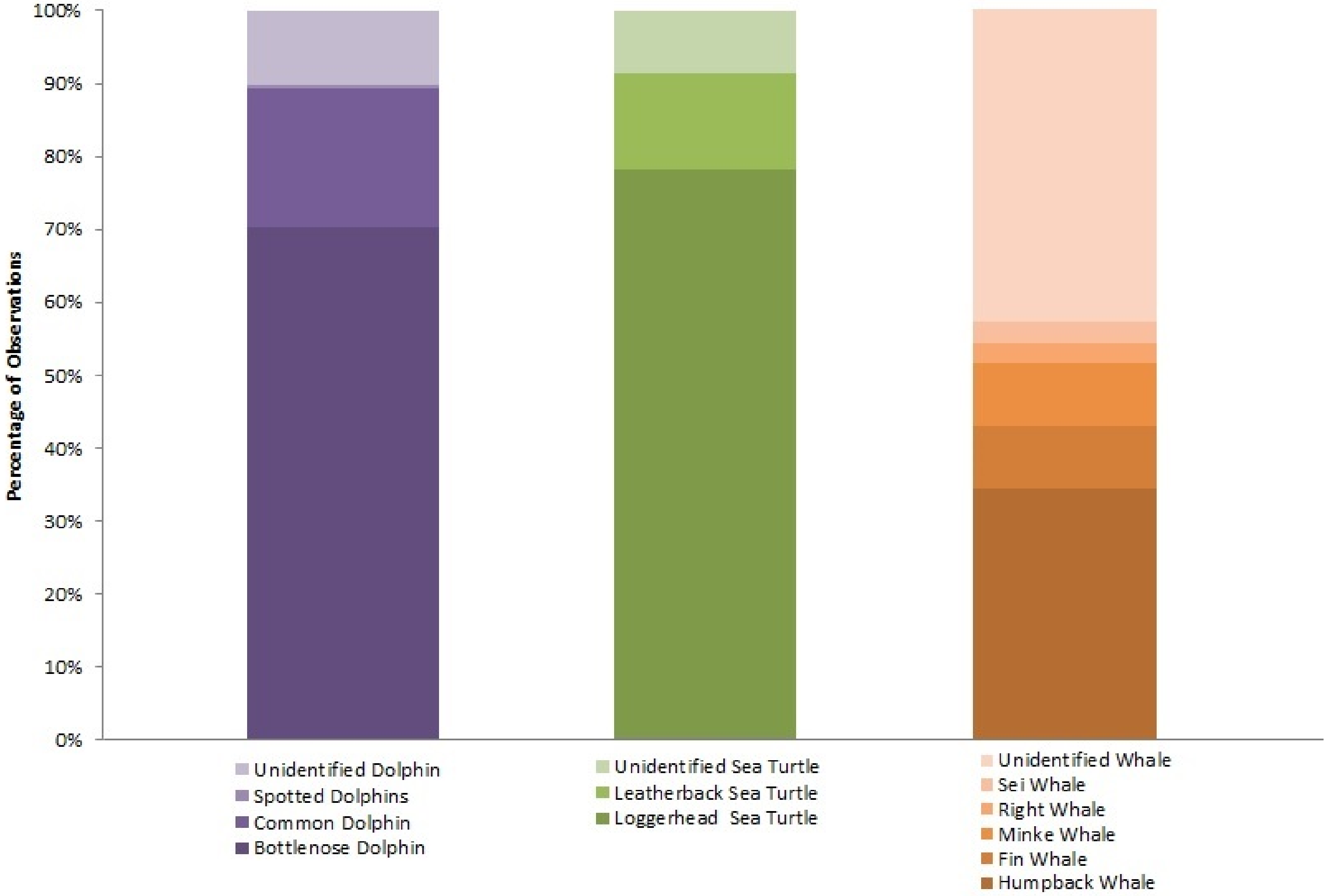

**Figure 8-7. Identification rates for common aquatic animal groups observed in the boat surveys.** Identifications to species level are shown in darker colors. Sample sizes for each taxon: dolphins (purple, n=1,200); sea turtles (green, n=114); whales (orange, n=35). A single “unidentified marine mammal” is not included in this figure (**Table 8A-1)**.

**Table 8-1. Weeks in which boat surveys were completed during the Mid-Atlantic Baseline Studies Project.** Each survey took from four to five days to complete, depending upon weather, ship availability, and other factors. Surveys colored in gray only included Mid-Atlantic Baseline Studies transects; surveys in blue also included Maryland study transects.

| | Jan. | | | | Feb. | | | | Mar. | | | | Apr. | | | | May | | | | Jun. | | | | Jul. | | | | Aug. | | | | Sep. | | | | Oct. | | | | Nov. | | | | Dec. | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **2012** | | | | | | | | | | | | | | | | ▨ | | | | | | | ▨ | | | | | | | ▨ | | | ▨ | ▨ | | | | | | | ▨ | ▨ | | | | | ▨ | |
| **2013** | ▨ | | | ▨ | ▨ | | | | | | 🟦 | 🟦 | | | | | 🟦 | 🟦 | | | | | 🟦 | | | | | 🟦 | 🟦 | | | | | 🟦 | | | | | | 🟦 | | | | | | 🟦 | 🟦 | |
| **2014** | | | | 🟦 | 🟦 | | | | | | | | ▨ | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |

**Table 8-2. Summary data for April 2012-April 2014 boat surveys (by species group).** Data are presented in order of abundance based on the total counts from all surveys. Gray survey headings and totals are surveys conducted solely within the Mid-Atlantic Baseline Studies Area, while surveys in blue also include the Maryland study transects.

| Animal Group | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Scoters, Ducks, Geese (Anatidae) | 1 | 6 | 0 | 30 | 447 | 707 | 7404 | 5438 | 94 | 0 | 0 | 0 | 1703 | 1682 | 3230 | 1451 | 22193 | 34.43% |
| Gulls and Terns (Laridae) | 294 | 215 | 484 | 608 | 1111 | 514 | 319 | 142 | 793 | 271 | 492 | 565 | 1272 | 6099 | 1139 | 471 | 14789 | 22.94% |
| Gannets (Sulidae) | 483 | 2 | 0 | 0 | 2344 | 1177 | 2209 | 1158 | 118 | 0 | 1 | 3 | 760 | 1345 | 3492 | 1127 | 14219 | 22.06% |
| Loons (Gaviidae) | 470 | 7 | 0 | 0 | 310 | 799 | 289 | 377 | 443 | 11 | 8 | 1 | 36 | 1476 | 715 | 742 | 5684 | 8.82% |
| Cormorants (Phalacrocoracidae) | 10 | 5 | 0 | 3 | 128 | 4 | 0 | 55 | 530 | 1 | 0 | 2 | 1095 | 0 | 0 | 202 | 2035 | 3.16% |
| Auks (Alcidae) | 0 | 0 | 0 | 0 | 0 | 344 | 608 | 76 | 2 | 0 | 0 | 0 | 0 | 24 | 623 | 13 | 1690 | 2.62% |
| Storm-Petrels (Hydrobatidae) | 3 | 238 | 130 | 7 | 0 | 0 | 0 | 0 | 9 | 118 | 308 | 1 | 0 | 0 | 0 | 0 | 814 | 1.26% |
| Shorebirds (Charadriiformes spp.) | 2 | 5 | 4 | 57 | 0 | 3 | 1 | 328 | 40 | 0 | 25 | 22 | 3 | 0 | 1 | 96 | 587 | 0.91% |
| Shearwaters and Fulmars (Procellariidae) | 0 | 44 | 1 | 5 | 2 | 21 | 0 | 5 | 66 | 74 | 10 | 17 | 43 | 18 | 17 | 2 | 325 | 0.50% |
| Passerines (Passeriformes spp.) | 13 | 2 | 48 | 49 | 14 | 0 | 0 | 0 | 2 | 0 | 33 | 0 | 4 | 0 | 0 | 15 | 180 | 0.28% |
| Pelicans (Pelecanidae) | 0 | 18 | 1 | 18 | 2 | 0 | 0 | 0 | 1 | 67 | 21 | 16 | 16 | 0 | 0 | 0 | 160 | 0.25% |
| Unidentified Birds (Aves spp.) | 1 | 0 | 0 | 10 | 82 | 11 | 8 | 1 | 1 | 1 | 0 | 1 | 1 | 1 | 0 | 0 | 118 | 0.18% |
| Grebes (Podicipedidae) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 32 | 10 | 43 | 0.07% |
| Jaegers and Skuas (Stercorariidae) | 9 | 2 | 0 | 1 | 2 | 0 | 0 | 0 | 8 | 2 | 0 | 0 | 4 | 3 | 0 | 0 | 31 | 0.05% |
| Egrets and Herons | 2 | 0 | 0 | 0 | 2 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 18 | 0 | 0 | 1 | 25 | 0.04% |

| Animal Group | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| (Ardeidae) | | | | | | | | | | | | | | | | | | |
| Raptors (Accipitridae, Falconidae, and Pandionidae) | 0 | 5 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 2 | 4 | 1 | 1 | 0 | 1 | 17 | 0.03% |
| Rails, Coots and Gallinules (Rallidae) | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 4 | 0.01% |
| Hummingbirds (Trochilidae) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Vultures (Cathartidae) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Woodpeckers and Sapsuckers (Picidae) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0.00% |
| **Avian Total** | **1289** | **549** | **668** | **789** | **4445** | **3581** | **10838** | **7582** | **2108** | **547** | **901** | **632** | **4957** | **10650** | **9249** | **4132** | **62917** | **97.60%** |
| Toothed Whales (Odontoceti) | 223 | 208 | 99 | 106 | 32 | 67 | 60 | 62 | 3 | 60 | 72 | 27 | 116 | 18 | 20 | 27 | 1200 | 1.86% |
| Fish and Sharks | 1 | 70 | 0 | 61 | 9 | 0 | 0 | 0 | 2 | 0 | 0 | 6 | 20 | 0 | 0 | 0 | 169 | 0.25% |
| Turtles (Testudines) | 13 | 13 | 22 | 8 | 2 | 0 | 0 | 0 | 0 | 8 | 22 | 13 | 13 | 0 | 0 | 0 | 114 | 0.18% |
| Rays (Batoidea) | 0 | 3 | 14 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 24 | 0.04% |
| Baleen Whales (Mysticeti) | 0 | 0 | 0 | 0 | 2 | 1 | 6 | 2 | 0 | 0 | 1 | 3 | 0 | 2 | 2 | 1 | 20 | 0.03% |
| Unidentified Whale (Cetacea) | 0 | 0 | 0 | 0 | 2 | 1 | 4 | 2 | 0 | 0 | 0 | 1 | 0 | 1 | 4 | 0 | 15 | 0.02% |
| Bats (Chiroptera) | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Unidentified Marine Mammals (Mammalia) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0.00% |
| **Total Non-Avian Animals** | **237** | **294** | **135** | **177** | **47** | **69** | **70** | **66** | **5** | **68** | **101** | **51** | **149** | **22** | **26** | **28** | **1545** | **2.40%** |
| **Grand Total** | **1526** | **843** | **803** | **966** | **4492** | **3650** | **10908** | **7648** | **2113** | **615** | **1002** | **683** | **5106** | **10672** | **9275** | **4160** | **64462** | **100.00%** |

# Supplementary material

## Appendix 8A.

**Table 8A-1. Animals observed during the boat survey.** Data are presented in order of abundance by family, based on the total count from all surveys, with avian species first. Gray survey headings and totals are surveys conducted solely within the Mid-Atlantic Baseline Studies Area, while surveys in blue also include the Maryland study transects (**Figure 8-1**).

| **Animals** | **Apr. 2012** | **Jun. 2012** | **Aug. 2012** | **Sep. 2012** | **Nov. 2012** | **Dec. 2012** | **Jan. 2013** | **Mar. 2013** | **May. 2013** | **Jun. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Jan. 2014** | **Apr. 2014** | **Total** | **Percent** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Unidentified Scoter | 0 | 0 | 0 | 0 | 222 | 269 | 6188 | 28 | 1 | 0 | 0 | 0 | 128 | 86 | 1918 | 2 | 8842 | **13.72%** |
| Dark scoter - either black scoter or surf scoter | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4099 | 75 | 0 | 0 | 0 | 685 | 1109 | 224 | 887 | 7079 | **10.98%** |
| Black Scoter | 0 | 0 | 0 | 0 | 112 | 340 | 706 | 981 | 2 | 0 | 0 | 0 | 158 | 322 | 116 | 179 | 2916 | **4.52%** |
| Surf Scoter | 1 | 0 | 0 | 0 | 31 | 5 | 461 | 269 | 0 | 0 | 0 | 0 | 685 | 111 | 886 | 346 | 2795 | **4.34%** |
| White-winged Scoter | 0 | 0 | 0 | 0 | 1 | 91 | 39 | 40 | 15 | 0 | 0 | 0 | 3 | 8 | 58 | 0 | 255 | **0.40%** |
| Unidentified Duck | 0 | 0 | 0 | 0 | 31 | 2 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 19 | 0 | 3 | 56 | **0.09%** |
| Red-breasted Merganser | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 30 | 49 | **0.08%** |
| Green-winged Teal | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 30 | 0 | 0 | 0 | 40 | **0.06%** |
| Brant | 0 | 0 | 0 | 0 | 25 | 0 | 10 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 39 | **0.06%** |
| Long-tailed Duck | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 28 | 2 | 37 | **0.06%** |
| Mallard | 0 | 0 | 0 | 20 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 20 | **0.03%** |
| Unidentified Eider | 0 | 0 | 0 | 0 | 18 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 18 | **0.03%** |
| Unidentified Teal | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 13 | 0 | 0 | 0 | 13 | **0.02%** |
| Canada Goose | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 11 | **0.02%** |
| American Black Duck | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 2 | 9 | **0.01%** |
| Bufflehead | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 6 | **0.01%** |
| Common Goldeneye | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 4 | **0.01%** |
| Unidentified Scaup | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 3 | **0.00%** |
| Wood Duck | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Scoters, Ducks, Geese (Anatidae) Total** | **1** | **6** | **0** | **30** | **447** | **707** | **7404** | **5438** | **94** | **0** | **0** | **0** | **1703** | **1682** | **3230** | **1451** | **22193** | **34.43%** |

| **Animals** | **Apr. 2012** | **Jun. 2012** | **Aug. 2012** | **Sep. 2012** | **Nov. 2012** | **Dec. 2012** | **Jan. 2013** | **Mar. 2013** | **May. 2013** | **Jun. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Jan. 2014** | **Apr. 2014** | **Total** | **Percent** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Bonaparte's Gull | 22 | 0 | 0 | 0 | 468 | 286 | 76 | 18 | 2 | 0 | 0 | 0 | 0 | 5534 | 799 | 255 | 7460 | **11.57%** |
| Laughing Gull | 113 | 81 | 140 | 110 | 348 | 0 | 1 | 4 | 30 | 188 | 292 | 318 | 699 | 24 | 0 | 41 | 2389 | **3.71%** |
| Common Tern | 38 | 38 | 265 | 233 | 12 | 0 | 0 | 0 | 482 | 33 | 87 | 86 | 0 | 0 | 0 | 0 | 1274 | **1.98%** |
| Great Black-backed Gull | 7 | 17 | 6 | 17 | 93 | 166 | 148 | 64 | 19 | 6 | 19 | 56 | 179 | 141 | 78 | 18 | 1034 | **1.60%** |
| Herring gull | 46 | 3 | 0 | 8 | 39 | 36 | 77 | 50 | 105 | 5 | 0 | 3 | 163 | 73 | 197 | 147 | 952 | **1.48%** |
| Unidentified Gull | 0 | 0 | 0 | 76 | 79 | 24 | 0 | 6 | 115 | 2 | 5 | 3 | 149 | 220 | 22 | 6 | 707 | **1.10%** |
| Royal Tern | 46 | 70 | 53 | 124 | 2 | 0 | 0 | 0 | 2 | 26 | 88 | 65 | 14 | 0 | 0 | 0 | 490 | **0.76%** |
| Unidentified Tern | 12 | 5 | 9 | 25 | 7 | 0 | 0 | 0 | 28 | 5 | 0 | 25 | 55 | 0 | 0 | 0 | 171 | **0.27%** |
| Forster's Tern | 4 | 0 | 0 | 1 | 47 | 0 | 0 | 0 | 0 | 0 | 1 | 4 | 0 | 56 | 0 | 2 | 115 | **0.18%** |
| Ring-billed Gull | 0 | 0 | 0 | 0 | 3 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 8 | 42 | 22 | 0 | 78 | **0.12%** |
| Black-legged Kittiwake | 0 | 0 | 0 | 0 | 5 | 1 | 14 | 0 | 0 | 0 | 0 | 0 | 4 | 8 | 9 | 0 | 41 | **0.06%** |
| Lesser Black-backed Gull | 6 | 0 | 3 | 0 | 8 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 2 | 29 | **0.04%** |
| Black Tern | 0 | 0 | 8 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 21 | **0.03%** |
| Least Tern | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 13 | **0.02%** |
| Caspian Tern | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 3 | 1 | 0 | 0 | 0 | 5 | **0.01%** |
| Roseate Tern | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | **0.00%** |
| Little Gull | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 2 | **0.00%** |
| Glaucous Gull | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 2 | **0.00%** |
| Sabine's Gull | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Unidentified Large Gull | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | **0.00%** |
| **Gulls and Terns (Laridae) Total** | **294** | **215** | **484** | **608** | **1111** | **514** | **319** | **142** | **793** | **271** | **492** | **565** | **1272** | **6099** | **1139** | **471** | **14789** | **22.94%** |
| Northern Gannet | 483 | 2 | 0 | 0 | 2344 | 1177 | 2209 | 1158 | 118 | 0 | 1 | 3 | 760 | 1345 | 3492 | 1127 | 14219 | **22.06%** |
| **Gannets (Sulidae) Total** | **483** | **2** | **0** | **0** | **2344** | **1177** | **2209** | **1158** | **118** | **0** | **1** | **3** | **760** | **1345** | **3492** | **1127** | **14219** | **22.06%** |
| Common Loon | 456 | 7 | 0 | 0 | 267 | 314 | 149 | 140 | 250 | 11 | 8 | 1 | 31 | 1281 | 373 | 318 | 3606 | **5.59%** |
| Red-throated Loon | 14 | 0 | 0 | 0 | 32 | 173 | 124 | 215 | 96 | 0 | 0 | 0 | 5 | 190 | 325 | 416 | 1590 | **2.47%** |
| Unidentified Loon | 0 | 0 | 0 | 0 | 11 | 312 | 16 | 22 | 97 | 0 | 0 | 0 | 0 | 5 | 17 | 8 | 488 | **0.76%** |

| Animals | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Loons (Gaviidae) Total** | **470** | **7** | **0** | **0** | **310** | **799** | **289** | **377** | **443** | **11** | **8** | **1** | **36** | **1476** | **715** | **742** | **5684** | **8.82%** |
| Double-crested Cormorant | 10 | 5 | 0 | 3 | 119 | 4 | 0 | 55 | 530 | 1 | 0 | 2 | 1095 | 0 | 0 | 202 | 2026 | **3.14%** |
| Unidentified Cormorant | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | **0.01%** |
| **Cormorants (Phalacrocoracidae) Total** | **10** | **5** | **0** | **3** | **128** | **4** | **0** | **55** | **530** | **1** | **0** | **2** | **1095** | **0** | **0** | **202** | **2035** | **3.16%** |
| Razorbill | 0 | 0 | 0 | 0 | 0 | 217 | 291 | 31 | 0 | 0 | 0 | 0 | 0 | 23 | 509 | 13 | 1084 | **1.68%** |
| Dovekie | 0 | 0 | 0 | 0 | 0 | 123 | 278 | 42 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 445 | **0.69%** |
| Unidentified Alcid | 0 | 0 | 0 | 0 | 0 | 3 | 14 | 3 | 1 | 0 | 0 | 0 | 0 | 1 | 51 | 0 | 73 | **0.11%** |
| Unidentified large alcid (Razorbill or Murre) | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 56 | 0 | 58 | **0.09%** |
| Atlantic Puffin | 0 | 0 | 0 | 0 | 0 | 0 | 21 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 22 | **0.03%** |
| Common Murre | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 5 | **0.01%** |
| Unidentified small alcid (Puffin/Dovekie) | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Thick-billed Murre | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Auks (Alcidae) Total** | **0** | **0** | **0** | **0** | **0** | **344** | **608** | **76** | **2** | **0** | **0** | **0** | **0** | **24** | **623** | **13** | **1690** | **2.62%** |
| Wilson's Storm-petrel | 0 | 236 | 129 | 6 | 0 | 0 | 0 | 0 | 9 | 117 | 308 | 1 | 0 | 0 | 0 | 0 | 806 | **1.25%** |
| Unidentified Storm-petrel | 3 | 2 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | **0.01%** |
| **Storm-Petrels (Hydrobatidae) Total** | **3** | **238** | **130** | **7** | **0** | **0** | **0** | **0** | **9** | **118** | **308** | **1** | **0** | **0** | **0** | **0** | **814** | **1.26%** |
| Red Phalarope | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 328 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 48 | 380 | **0.59%** |
| Unidentified Phalarope | 0 | 0 | 0 | 23 | 0 | 0 | 0 | 0 | 12 | 0 | 14 | 5 | 2 | 0 | 0 | 0 | 56 | **0.09%** |
| Dunlin | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 47 | 47 | **0.07%** |
| Red-necked Phalarope | 1 | 0 | 0 | 13 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 16 | 0 | 0 | 0 | 0 | 32 | **0.05%** |
| Unidentified shorebird | 0 | 0 | 1 | 14 | 0 | 0 | 0 | 0 | 9 | 0 | 4 | 0 | 1 | 0 | 0 | 0 | 29 | **0.04%** |
| Unidentified peep | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 8 | **0.01%** |

| Animals | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Short-billed Dowitcher | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | **0.01%** |
| Willet | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | **0.01%** |
| Wilson's Plover | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | **0.01%** |
| Sanderling | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 5 | **0.01%** |
| Semipalmated Plover | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 3 | **0.00%** |
| Semipalmated Sandpiper | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Whimbrel | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Least Sandpiper | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Black Skimmer | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Lesser Yellowlegs | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Ruddy Turnstone | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| White-rumped Sandpiper | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Shorebirds (Charadriiformes spp.) Total** | **2** | **5** | **4** | **57** | **0** | **3** | **1** | **328** | **40** | **0** | **25** | **22** | **3** | **0** | **1** | **96** | **587** | **0.91%** |
| Cory's Shearwater | 0 | 6 | 1 | 5 | 0 | 0 | 0 | 0 | 0 | 29 | 6 | 17 | 3 | 0 | 0 | 0 | 67 | **0.10%** |
| Sooty Shearwater | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 64 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 66 | **0.10%** |
| Greater Shearwater | 0 | 32 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 33 | 0 | 0 | 0 | 0 | 0 | 0 | 65 | **0.10%** |
| Manx Shearwater | 0 | 3 | 0 | 0 | 1 | 21 | 0 | 5 | 1 | 8 | 0 | 0 | 2 | 18 | 5 | 0 | 64 | **0.10%** |
| Unidentified Shearwater | 0 | 2 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 4 | 4 | 0 | 35 | 0 | 2 | 0 | 49 | **0.08%** |
| Northern Fulmar | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 8 | 2 | 12 | **0.02%** |
| Unidentified Small Shearwater (Audubon's, Manx, or Little) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | **0.00%** |
| Audubon's Shearwater | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Shearwaters and Fulmars (Procellariidae) Total** | **0** | **44** | **1** | **5** | **2** | **21** | **0** | **5** | **66** | **74** | **10** | **17** | **43** | **18** | **17** | **2** | **325** | **0.50%** |

| Animals | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Purple Martin | 0 | 1 | 45 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 58 | **0.09%** |
| Unidentified Swallow | 0 | 0 | 0 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 20 | 0 | 0 | 0 | 0 | 5 | 37 | **0.06%** |
| Unidentified Passerine | 0 | 0 | 0 | 8 | 5 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 20 | **0.03%** |
| Barn Swallow | 13 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 18 | **0.03%** |
| Unidentified Warbler | 0 | 0 | 1 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | **0.02%** |
| Tree Swallow | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | **0.01%** |
| Myrtle Warbler | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 3 | **0.00%** |
| Song Sparrow | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 2 | **0.00%** |
| Water Pipit | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Red-winged Blackbird | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Palm Warbler | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 2 | **0.00%** |
| Tennessee Warbler | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Golden-crowned Kinglet | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | **0.00%** |
| Dark-eyed Junco | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | **0.00%** |
| American Redstart | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| American Robin | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Blackpoll Warbler | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Cedar Waxwing | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | **0.00%** |
| Red-breasted Nuthatch | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Unidentified sparrow | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | **0.00%** |
| Black-throated Blue Warbler | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Mourning Warbler | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Ruby-crowned Kinglet | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Northern Waterthrush | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Brown-headed Cowbird | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |

| Animals | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Passerines (Passeriformes spp.) Total** | **13** | **2** | **48** | **49** | **14** | **0** | **0** | **0** | **2** | **0** | **33** | **0** | **4** | **0** | **0** | **15** | **180** | **0.28%** |
| Brown Pelican | 0 | 18 | 1 | 18 | 2 | 0 | 0 | 0 | 1 | 67 | 21 | 16 | 16 | 0 | 0 | 0 | 160 | **0.25%** |
| **Pelicans (Pelecanidae) Total** | **0** | **18** | **1** | **18** | **2** | **0** | **0** | **0** | **1** | **67** | **21** | **16** | **16** | **0** | **0** | **0** | **160** | **0.25%** |
| Unidentified Bird | 1 | 0 | 0 | 10 | 82 | 11 | 8 | 1 | 1 | 1 | 0 | 1 | 1 | 1 | 0 | 0 | 118 | **0.18%** |
| **Unidentified Birds (Aves spp.) Total** | **1** | **0** | **0** | **10** | **82** | **11** | **8** | **1** | **1** | **1** | **0** | **1** | **1** | **1** | **0** | **0** | **118** | **0.18%** |
| Horned Grebe | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 7 | 24 | **0.04%** |
| Red-necked Grebe | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 3 | 19 | **0.03%** |
| **Grebes (Podicipedidae) Total** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **0** | **0** | **0** | **0** | **32** | **10** | **43** | **0.07%** |
| Parasitic Jaeger | 9 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 2 | 2 | 0 | 0 | 19 | **0.03%** |
| Unidentified Jaeger | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 4 | 1 | 0 | 0 | 2 | 1 | 0 | 0 | 10 | **0.02%** |
| Unidentified Skua | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Pomarine Jaeger | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Jaegers and Skuas (Stercorariidae) Total** | **9** | **2** | **0** | **1** | **2** | **0** | **0** | **0** | **8** | **2** | **0** | **0** | **4** | **3** | **0** | **0** | **31** | **0.05%** |
| Great Blue Heron | 1 | 0 | 0 | 0 | 1 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 16 | 0 | 0 | 1 | 21 | **0.03%** |
| Green Heron | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | **0.00%** |
| Great Egret | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 2 | **0.00%** |
| **Egrets and Herons (Ardeidae) Total** | **2** | **0** | **0** | **0** | **2** | **1** | **0** | **1** | **0** | **0** | **0** | **0** | **18** | **0** | **0** | **1** | **25** | **0.04%** |
| Osprey | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 2 | 4 | 0 | 0 | 0 | 1 | 14 | **0.02%** |
| Northern Harrier | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | **0.00%** |
| Bald Eagle | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | **0.00%** |
| Merlin | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Raptors (Accipitridae, Falconidae, and Pandionidae) Total** | **0** | **5** | **0** | **1** | **0** | **0** | **0** | **0** | **0** | **2** | **2** | **4** | **1** | **1** | **0** | **1** | **17** | **0.03%** |
| American Coot | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 4 | **0.01%** |

| Animals | Apr. 2012 | Jun. 2012 | Aug. 2012 | Sep. 2012 | Nov. 2012 | Dec. 2012 | Jan. 2013 | Mar. 2013 | May. 2013 | Jun. 2013 | Aug. 2013 | Sep. 2013 | Oct. 2013 | Dec. 2013 | Jan. 2014 | Apr. 2014 | Total | Percent |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Rails, Coots and Gallinules (Rallidae) Total** | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 4 | 0.01% |
| Black Vulture | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Vultures (Cathartidae) Total** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| Northern Flicker | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0.00% |
| **Woodpeckers and Sapsuckers (Picidae) Total** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0.00% |
| Unidentified Hummingbird | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Hummingbirds (Trochilidae) Total** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.00% |
| **Aves Total** | 1289 | 549 | 668 | 789 | 4445 | 3581 | 10838 | 7582 | 2108 | 547 | 901 | 632 | 4957 | 10650 | 9249 | 4132 | 62917 | 97.60% |
| Red Bat | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| **Bats (Chiroptera) Total** | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 2 | 0.00% |
| Unidentified ray | 0 | 3 | 14 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 24 | 0.04% |
| Unidentified thresher shark | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 5 | 0.01% |
| Unidentified fish | 0 | 70 | 0 | 2 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 15 | 0 | 0 | 0 | 97 | 0.15% |
| Unidentified flying fish | 0 | 0 | 0 | 59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 64 | 0.10% |
| Ocean Sunfish (Mola) | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 3 | 0.00% |
| **Fish and Sharks total** | 1 | 73 | 14 | 62 | 9 | 0 | 0 | 0 | 2 | 0 | 6 | 6 | 20 | 0 | 0 | 0 | 193 | 0.30% |
| Bottlenose Dolphin | 221 | 180 | 94 | 87 | 28 | 0 | 0 | 12 | 2 | 54 | 72 | 26 | 86 | 6 | 2 | 4 | 874 | 1.36% |
| Common Dolphin | 0 | 0 | 0 | 0 | 0 | 64 | 60 | 47 | 0 | 0 | 0 | 0 | 0 | 10 | 10 | 18 | 209 | 0.32% |
| Unidentified Dolphin | 2 | 28 | 5 | 19 | 4 | 3 | 0 | 3 | 1 | 2 | 0 | 1 | 30 | 2 | 8 | 5 | 113 | 0.18% |
| Spotted Dolphins | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.01% |
| **Toothed Whales (Odontoceti) Total** | 223 | 208 | 99 | 106 | 32 | 67 | 60 | 62 | 3 | 60 | 72 | 27 | 116 | 18 | 20 | 27 | 1200 | 1.86% |
| Humpback Whale | 0 | 0 | 0 | 0 | 1 | 0 | 3 | 0 | 0 | 0 | 1 | 3 | 0 | 2 | 2 | 0 | 12 | 0.02% |
| Fin Whale | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 3 | 0.00% |

| **Animals** | **Apr. 2012** | **Jun. 2012** | **Aug. 2012** | **Sep. 2012** | **Nov. 2012** | **Dec. 2012** | **Jan. 2013** | **Mar. 2013** | **May. 2013** | **Jun. 2013** | **Aug. 2013** | **Sep. 2013** | **Oct. 2013** | **Dec. 2013** | **Jan. 2014** | **Apr. 2014** | **Total** | **Percent** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Minke Whale | 0 | 0 | 0 | 0 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | **0.00%** |
| Sei Whale | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| Right Whale | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | **0.00%** |
| **Baleen Whales (Mysticeti) Total** | **0** | **0** | **0** | **0** | **2** | **1** | **6** | **2** | **0** | **0** | **1** | **3** | **0** | **2** | **2** | **1** | **20** | **0.03%** |
| Unidentified Whale | 0 | 0 | 0 | 0 | 2 | 1 | 4 | 2 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 11 | **0.02%** |
| Unidentified large whale | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 4 | **0.01%** |
| **Unidentified Whale (Cetacea) Total** | **0** | **0** | **0** | **0** | **2** | **1** | **4** | **2** | **0** | **0** | **0** | **1** | **0** | **1** | **4** | **0** | **15** | **0.02%** |
| Unidentified Marine Mammal | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | **0.00%** |
| **Unidentified Marine Mammals (Mammalia) Total** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **0** | **1** | **0** | **0** | **1** | **0.00%** |
| Loggerhead Turtle | 11 | 11 | 19 | 3 | 2 | 0 | 0 | 0 | 0 | 6 | 16 | 11 | 10 | 0 | 0 | 0 | 89 | **0.14%** |
| Leatherback Turtle | 0 | 1 | 2 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 2 | 0 | 0 | 0 | 15 | **0.02%** |
| Small turtle - Loggerhead, Green, Hawksbill, or Kemp's Ridley | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 2 | 1 | 0 | 0 | 0 | 6 | **0.01%** |
| Unidentified Sea Turtle | 2 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | **0.01%** |
| **Turtles (Testudines) Total** | **13** | **13** | **22** | **8** | **2** | **0** | **0** | **0** | **0** | **8** | **22** | **13** | **13** | **0** | **0** | **0** | **114** | **0.18%** |
| **Non-Avian Animals Total** | **237** | **294** | **135** | **177** | **47** | **69** | **70** | **66** | **5** | **68** | **101** | **51** | **149** | **22** | **26** | **28** | **1545** | **2.40%** |
| **Grand Total** | **1526** | **838** | **803** | **965** | **4492** | **3650** | **10908** | **7648** | **2113** | **613** | **1000** | **679** | **5105** | **10671** | **9275** | **4159** | **64462** | **100.00%** |

# Chapter 9: Monitoring aquatic biomass via hydroacoustics: echo sounding data processing and summary of results

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 9 Highlights

*Outlines data collection and data processing protocols for echo sounding data collected during boat-based surveys, and provides a brief summary of results*

### *Context*[1]

Part III of this report focuses on boat-based surveys for wildlife in the offshore environment, including methodological reviews and data analyses. Most chapters within this section deal directly with the survey data itself (i.e., observations of marine birds, mammals, and sea turtles). While collecting survey data, however, various environmental covariate data were simultaneously collected, including sea state, sea surface water temperature and salinity, and hydroacoustic data.

This chapter focuses exclusively on the collection and data management of hydroacoustic data collected on boat survey transects, and provides a simple summary of results. These data provide us with the relative abundance of underwater biomass, and can be used to approximate prey (i.e., fish and plankton) biomass availability to seabirds and other marine predators. Chapters 10 and 11 of this report use these data to examine the spatial relationships of foraging seabirds and acoustically detected prey.

### *Study goal/objectives*

Estimate the relative abundance of hydroacoustically detected biomass along boat survey transects, using a scientific echo sounder.

### *Highlights*

- Data were collected along boat survey transects during 16 surveys conducted between 2012-2014, using a Simrad EK60 echo sounder unit (Kongsberg Maritime AS, Horten, Norway).
- Raw data were processed using Echoview 5.3 (Myriax Software Pty. Ltd., Hobart, Australia).
- Data were integrated by 1 x 500 m cells across the depth and length each survey, calculating a biomass index value per cell.
- Total biomass varied widely both within and between surveys, indicating a high level of spatial and temporal variation of prey biomass abundance across the study area and throughout the year.
- The mean depth of biomass did not vary significantly between seasons.
- Total biomass was higher in nearshore areas in the summer and fall, and in the southern end of the study area during winter surveys.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

This chapter outlines the methods used in the collection and processing of hydroacoustic echo sounding data collected as part of the Mid-Atlantic Baseline Studies Project, and provides a basic summary of results. Hydroacoustic data were collected during 16 boat-based surveys offshore of Delaware, Maryland, and Virginia, USA between 2012 and 2014, using a Simrad EK60 echo sounder unit. Raw data were processed by trained personnel using Echoview 5.3 software. Data were filtered to remove small particles, surface noise, bottom substrates, and anomalous data. Data were integrated into 500 m cells across the length of each survey and 1 m depth strata, calculating a biomass index value (Nautical Area Scattering Coefficient; NASC) per 1 x 500 m cell. Due to removal of surface noise and bottom substrates, data are limited to the water column between 2 m depth and the bottom substrate, and do not include surface and benthic biomass. Total biomass varied widely both within and between surveys, indicating a high level of spatial and temporal variation of prey biomass (i.e., fish and large plankton) abundance across the study area throughout the year. Total biomass was higher in western parts of the study area in the summer and fall, and in the southern end of the study area during winter surveys.

## Introduction

Non-invasive, quantitative estimates of fish abundance and aquatic biomass have been made possible in recent years with the development and subsequent improvement of acoustic echo sounding hardware, including split- and multi-beam transducers employing echo-counting and interpretation software. During the past decade, the development of stable, scientific echo sounders, multi-frequency applications, new transducer deployment techniques, standardized calibration procedures, and more realistic models of the sound-scattering properties of biological targets have improved accuracy of biomass estimations (Rudstam et al., 2013; Simrad, 2012).

While conducting boat-based surveys for higher trophic level wildlife (birds, marine mammals, sea turtles, and other taxa) in the mid-Atlantic region, we employed echo sounding technology in order to estimate the biomass and size classes of aquatic prey species (fish and zooplankton) present beneath the survey vessel. The echo sounder sends acoustic signals into the water column and detects resulting backscattered energy reflected from fish and other objects. Data from the Simrad EK60 scientific echo sounder were automatically processed using appropriate software, manually vetted, and integrated and summed by distance and depth intervals in order to estimate the contribution of backscattered energy from all targets within each sampling volume. These data were subsequently used to calculate estimates of fish size class and biomass by area and by volume along the survey transects.

## Data collection

Hydroacoustic data were collected during all 16 boat-based surveys, totaling 66 of 68 survey days. Data were not collected during the boat-based surveys conducted on February 3 and June 18, 2013 due to errors with equipment and surveyor oversight. Data were collected using a Simrad EK60 scientific echo sounder unit with a hull mounted 120 kHz split-beam transducer, transceiver, and a laptop computer with Simrad-ER60 echo sounder software, run off an external marine battery. A Garmin Map60CSX GPS (Garmin International, Inc., Olathe, KS) was attached to the data collection computer for georeferencing the echo sounder data. Transducer settings can be found in Table 9A-1. The unit was calibrated using a tungsten carbide calibration sphere in a monofilament harness, following calibration guidelines given in the Simrad EK60 reference manual (Simrad, 2012).

## Data processing

Raw data files were processed by trained personnel at BRI or Aquacoustics, Inc. (Sterling, AK). Data files were post-processed using Echoview 5.3, and the results summarized in Microsoft Excel. GPS data were reviewed to ensure spatial referencing was complete and accurate, and hydroacoustic data were calibrated for the speed of sound and absorption coefficients using mean sea surface temperature and salinity values collected every 30 minutes during boat-based surveys (Chapter 7).

Several steps were taken to filter and exclude data within the Sv fileset echogram that were generated from sources other than fish or zooplankton biomass. The Sv echogram is a visual representation of the volumetric backscattering of hydroacoustic signals sent and received by the echo sounder (Echoview, 2015). Data were initially filtered at -60 dB to exclude very small targets (< 2 cm) and low-intensity surface noise. A surface line was drawn at a depth of 2 m below the water surface (roughly 0.8 m below the surface of the transducer), and a bottom line was generated at roughly 20 cm above the ocean floor. Within the Sv echogram window, the bottom line was manually edited to exclude the bottom substrate and targets indistinguishable from the bottom substrate, as well as to ensure that the line was continuous from the beginning to the end of the survey. All backscattering signals occurring above the surface line or below the bottom line were excluded from analysis. Additionally, the Sv echogram was reviewed in order to exclude anomalous data from analysis, such as surface disturbances, non-fish objects, or other anomalies. After manual review, and per the recommendation of fishery acoustics specialists at Aquacoustics, Inc., data from surveys conducted in August, September, and October of 2013, and data for depths ranging from 25-40 m in April and June of 2012 were filtered at -54 dB rather than at -60 dB, to compensate for high densities of abnormal low-frequency signals possibly caused by small invertebrates or suspended particulate matter (D. Degan pers. comm.).

The Sv echogram was integrated by 1 m depth intervals (or “layers”) and 500 m distance intervals (or “intervals”), calculating the mean volume backscattering strength (Sv Mean), the area backscattering coefficient (ABC), and nautical area-scattering coefficient (NASC) value for each 1 x 500 m cell within the survey, among other variables and coefficients (Table 9B-1). Frequency distributions of ABC values were plotted and outliers were reviewed to ensure that the resulting ABC values were representative of biomass rather than an error in data filtering.

While the Sv echogram represents volumetric backscattering, the Single Target echogram represents individual targets (i.e., fish or large plankton) derived from single points. The Single Target echogram was also reviewed and integrated using the same exclusion criteria (surface line, bottom line, and anomalous data regions) established while vetting the Sv echogram. Single target detection variable properties defined prior to integration are listed in Table 9A-2.

The resulting integrated data gives the estimated number of individual targets per cell, as well as each target’s compensated target strength (TS Comp) value. The length of each target (cm) was calculated using a simplification for Love’s dorsal aspect equation for 120 kHz frequency (Love, 1971):

$$\text{Length} = (10^{(TS\_Comp + 26.1)/19.1})\bullet 100$$

Additionally, the backscattering cross-section ($\sigma_{bs}$) value for each target was calculated using the following equation (Echoview, 2015; Simmonds and MacLennan, 2005):

$$\sigma_{bs} = 10^{(TS\_Comp/10)}$$

The ABC value, Sv Mean, and mean backscattering cross-section value by layer ($\overline{\sigma}_{bs}$) were then used to calculate aerial density (number of targets/m$^2$) and volumetric density (number of targets/m$^3$) for each cell within the survey, using the following equations (Echoview, 2015; Simmonds and MacLennan, 2005):

$$\text{Aerial density} = ABC/(\overline{\sigma}_{bs})$$

$$\text{Volumetric density} = 10^{(Sv\_Mean/10)}/(\overline{\sigma}_{bs})$$

Data for each survey-day were processed separately and combined in a unified Microsoft Access database after undergoing QA/QC procedures outlined below.

## Quality assurance and quality control (QA/QC)

For each survey-day, the following post-processing steps were implemented to ensure that data within and between each survey were processed consistently and accurately:

1) GPS data were reviewed to ensure that correct spatial data were assigned to each dataset;
2) Calibration files were reviewed to ensure that correct temperature and salinity data were used in determining speed of sound and absorption coefficients;
3) Sv echogram cells with the highest ABC values were reviewed to ensure that values were representative of biomass; and
4) Integrated data were examined by interval and layer to look for instances where biomass was identified in Layer 3 (from 2-3 m in depth), and no biomass was identified in Layer 4, as this pattern may indicate the presence of surface noise that was not completely excluded from analysis. In these instances, the corresponding cell within the Sv echogram was reviewed to see if the values were representative of actual biomass.

If corrections were made during any of these four steps, cell integration of Sv and Single Target data as well as subsequent calculations were performed again, and corrected data were incorporated into the final dataset.

Additionally, surveys conducted on January 1-3, 2013 were independently analyzed by a BRI analyst and a fishery acoustics specialist from Aquacoustics, Inc., to determine the repeatability and comparability of analyses. This comparison was conducted by an expert at Aquacoustics, who concluded that analyses were highly comparable, and differences were within the expected margin of error.

## Data summary

Data below are summarized by total NASC (m$^2$/nmi$^2$), or the NASC values summed across all depths within an interval or survey. This metric represents an index of total prey biomass in the water column.

We chose to use this metric rather than fish density estimates as we are interested in representing total prey availability rather than estimated densities or numbers of individual fish. Total NASC was moderately variable between individual surveys (Table 9-1). Total NASC values per survey ranged from 11,667 in April 2014, to 1,501,620 in October 2013, with a mean (± SD) of 468,761 (± 377,811). Within each survey, Total NASC by 500 m interval was highly variable, indicating variable geographic distributions of prey biomass within the study area. For example, the mean total NASC per interval in October 2013 was 1,079, with a standard deviation nearly an order of magnitude higher (7,511). This spatial variability within surveys was typical across all surveys conducted (Table 9-1).

Total prey biomass within the water column also varied geographically by season, with higher near shore distributions in the summer and fall, and higher distributions in the southern end of the study area during winter surveys (Figure 9-1). The mean depth of biomass (± SD) did not vary significantly between seasons, ranging from 13.3 (± 6.8) m in fall surveys to 18.5 (± 8.3) m in spring surveys (Figure 9-2).

## Further analysis and caveats

These data paint a picture of the distribution and relative abundance of prey biomass within the study area throughout the year. They can also be used in combination with the boat-based survey observations to examine the relationship between acoustically detected biomass and observed predators such as gannets, gulls, and terns (Chapters 10 and 11). However, several limitations of these data should be noted prior to further explorations and interpretation of predator and prey correlations. First, it is important to keep in mind that the top several meters of the water column were excluded from integration due to surface noise backscatter. Surface noise typically extended to 2 m in depth during calm conditions, so a minimum of the top 2 m of the water column were excluded across all surveys. The depth to which the surface noise extended varied with sea state, however, and there were many instances where surface noise penetrated to greater depths, commonly requiring exclusion of the top 4-6 m of the water column for several kilometers within a survey, and on occasion requiring exclusion of the top 10-12 m. Similarly, this technique does not measure the biomass of benthic biota, such as shellfish, as they cannot be distinguished from the bottom substrate within the echogram. Thus, species that forage exclusively within the top few meters of water (such as storm-petrels, Hydrobatidae) and species that forage on benthos (such as scoters, *Melanitta* spp.) are unlikely to show direct correlations with distributions of biomass as detected by the echo sounder. Even for species which forage within our surveyed water depths, the relevance of aquatic biomass distributions will vary depending upon the species composition and size classes present in the water column. We did not directly measure the sizes or abundance of fish and plankton that would be consumed by our target species (e.g., seabirds, marine mammals, and sea turtles), as “ground truthing” the hydroacoustic data would have required substantial additional resources (and was not the focus of this study). However, measured aquatic biomass can be used as an index of prey availability (Santora et al., 2011, 2009; Simmonds and MacLennan, 2005). The relationship between acoustically detected biomass and observed seabird predators, along with these limitations, are further discussed in Chapters 10 and 11 of this report.

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## Figures and tables

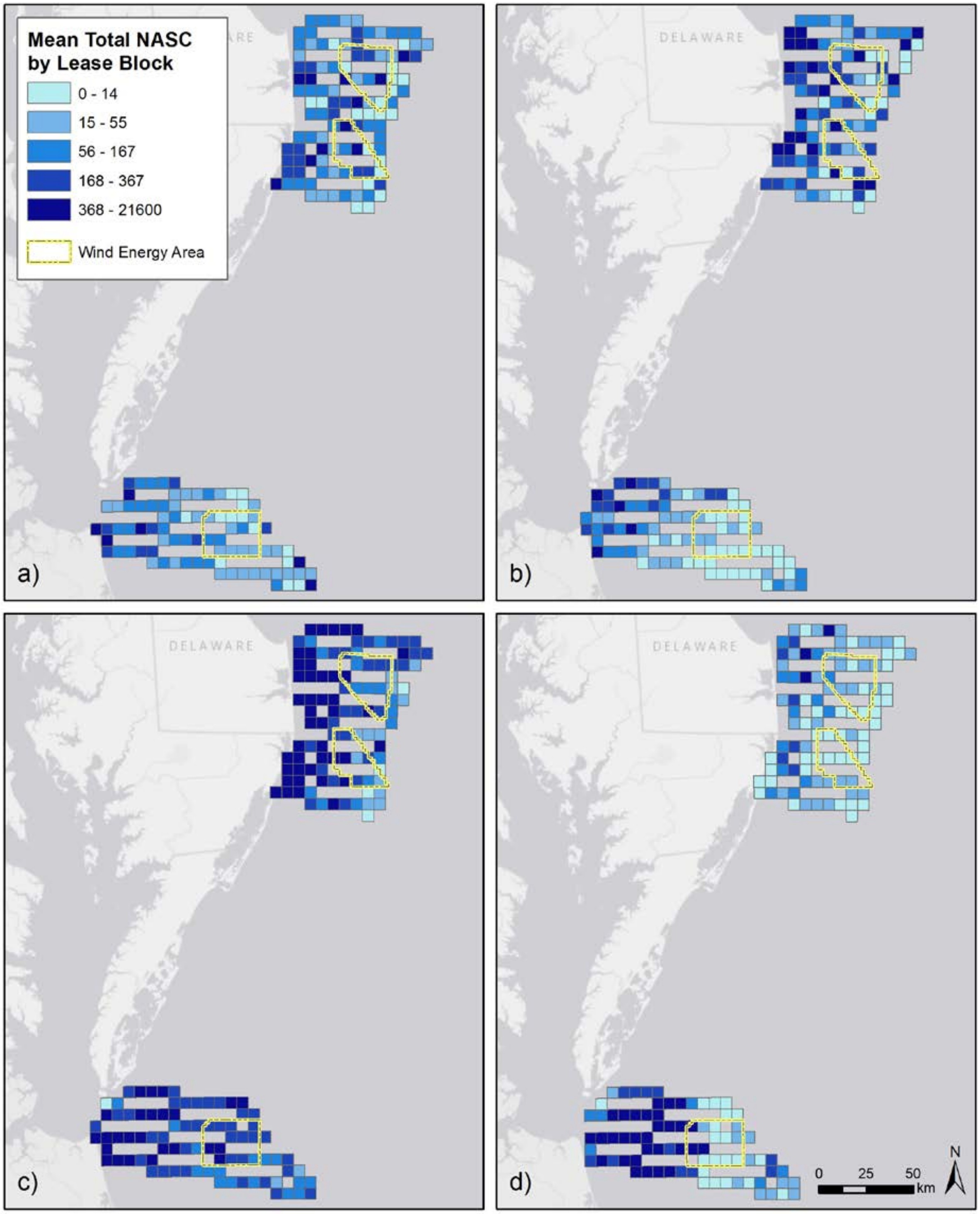

**Figure 9-1. Seasonal mean total NASC per 4.8 x 4.8 km BOEM lease block.** a) Spring, March 1 – May 31; b) Summer, June 1 – August 31; c) Fall, September 1 – November 30; d) Winter, December 1 – February 28. Total NASC was calculated by summing NASC across all depths for each 500 m interval within each survey. Total NASC values were binned and averaged by lease block. Mean total NASC is categorized by quintiles for mapping purposes.

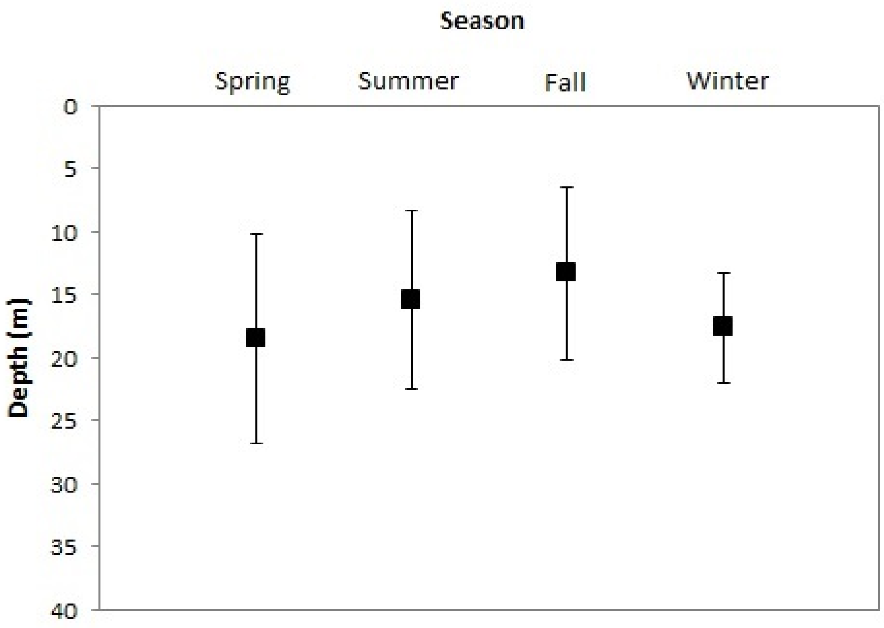

**Figure 9-2. Seasonal mean depth ± SD of biomass within the water column.** Total NASC by layer by season was calculated by summing NASC values within a layer across all survey intervals seasonally. Depth was weighted by the corresponding total NASC value in order to calculate the seasonal mean depth of biomass. Spring: March 1 – May 31; Summer: June 1 – August 31; Fall: September 1 – November 30; Winter: December 1 – February 28.

**Table 9-1. Total NASC by survey and interval, representing an index of total prey biomass within the water column.**

| Survey | Total NASC by Survey | Total NASC by Interval | | | |
|---|---|---|---|---|---|
| | | Mean | SD | Min. | Max. |
| Survey 1<br>April 2012 | 266,704 | 199 | 1,266 | 0 | 20,762 |
| Survey 2<br>June 2012 | 327,131 | 203 | 1,475 | 0 | 44,376 |
| Survey 3<br>August 2012 | 237,947 | 168 | 987 | 0 | 18,270 |
| Survey 4<br>September 2012 | 697,377 | 497 | 1,888 | 0 | 31,218 |
| Survey 5<br>November 2012 | 390,328 | 272 | 602 | 0.009 | 10,188 |
| Survey 6<br>Dec. 2012/Jan. 2013 | 678,388 | 501 | 2,210 | 0 | 37,824 |
| Survey 7<br>Jan./Feb. 2013 | 156,171 | 164 | 1,274 | 0 | 19,748 |
| Survey 8<br>March 2013 | 292,055 | 216 | 1,482 | 0 | 42,381 |
| Survey 9<br>May 2013 | 208,940 | 149 | 1,662 | 0 | 59,263 |
| Survey 10<br>June 2013 | 160,481 | 154 | 466 | 0 | 5,527 |
| Survey 11<br>July/Aug. 2013 | 245,312 | 176 | 821 | 0 | 15,602 |
| Survey 12<br>September 2013 | 721,390 | 529 | 1,497 | 0 | 27,431 |
| Survey 13<br>October 2013 | 1,501,620 | 1,079 | 7,511 | 0 | 123,833 |
| Survey 14<br>December 2013 | 699,640 | 510 | 3,887 | 0 | 96,674 |
| Survey 15<br>Jan./Feb 2014 | 905,020 | 653 | 8,673 | 0 | 209,625 |
| Survey 16<br>April 2014 | 11,667 | 9 | 59 | 0 | 1,345 |

# Supplementary material

## Appendix 9A. Transducer settings and integration variable properties

**Table 9A-1. Split-beam transducer settings used while collecting hydroacoustic data during boat surveys.**

| Field Name | Setting |
|---|---|
| Transducer draft (m) | 0.000 |
| Sample interval (s) | 0.000064 |
| Transmit power (W) | 250.0 |
| Pulse length (ms) | 0.256 |
| Transducer gain (dB) | 27.000 |
| Sa correction (dB) | 0.000 |
| Minor-axis beam width (degrees) | 7.000 |
| Major-axis beam width (degrees) | 7.000 |
| Frequency (kHz) | 120.000 |
| Two-way beam angle (dB re 1 Steradian) | -21.000 |

**Table 9A-2. Single target detection variable properties parameters set prior to single target cell integration.**

| Field Name | Setting |
|---|---|
| TS Threshold (dB) | -60.0 |
| Pulse length determination level (dB) | 6.0 |
| Minimum normalized pulse length | 0.7 |
| Maximum normalized pulse length | 1.75 |
| Beam compensation model | Simrad LOBE |
| Maximum beam compensation (dB) | 6.0 |
| Maximum standard deviation of minor-axis angles (degrees) | 0.6 |
| Maximum standard deviation of major-axis angles (degrees) | 0.6 |

## Appendix 9B. Exported data fields and definitions

**Table 9B-1. Sv data set field names and definitions.** Definitions are adapted from the Echoview glossary, through personal communications with specialists at Aquacoustics Inc. (Echoview, 2015; D. Degan, pers. comm.). Fields marked with an asterisk (*) were added to the dataset and calculated post cell-integration. All other fields were exported during the cell-integration process.

| Field | Example | Definition |
|---|---|---|
| Surv_Date* | 11/4/2012 | Date of survey. |
| ABC | 1.04E-07 | Area backscattering coefficient ($m^2/m^2$). Measure of area scattering rather than volume scattering. |
| NASC | 4.46 | Nautical area-scattering coefficient ($m^2/nmi^2$). Scaled version of ABC, equal to $4\pi(1852)^2(ABC)$. |
| Sigma* | 5.75E-06 | The back-scattering cross-section, or a measure of the backscatter strength from the target ($m^2$), calculated using data from the single target dataset. The mean sigma value per layer per day ($\overline{\sigma}_{bs}$) is presented here, and is used as a scalar when converting area and volume backscattering measurements to absolute numbers. |
| Aerial Density* | 0.0137 | Aerial fish density in the region (number of fish per square meter for a given thickness layer). Calculated as $ABC/(\overline{\sigma}_{bs})$. |
| Volumetric Density* | 0.0412 | Volumetric fish density in the region (number of fish per cubic meter). Calculated as $10^{(Sv\_Mean/10)}/(\overline{\sigma}_{bs})$. |
| Thickness_mean | 1.008047 | The mean thickness (m) of an analysis domain (i.e., the average thickness of each layer within the 500 m bin). |
| Interval | 1 | The sequentially numbered 500 m survey segment by which data is binned. |
| Layer | 3 | The layer or stratum number of the cell being analyzed (e.g., the number of the domain layer, counting from the water surface downwards). |
| Sv_mean | -55.74 | The linear mean Sv value for all samples in the 500 m bin, or domain, in ($m^2/m^3$). Another definition: the mean volume backscattering strength of the domain being integrated. |
| Height_mean | 1.008047 | The mean height (m) of the domain layer across the 500 m interval, or the projection of thickness mean onto the vertical axis taking transducer geometry into account. Height mean and thickness mean are equal for this project, due to the orientation of the transducer. |
| Depth_mean | 2.494063 | The mean depth (m) of the domain layer across the 500 m interval. |
| Layer_depth_min | 2 | The minimum depth (m) of the domain layer across the 500 m interval. |
| Layer_depth_max | 3 | The maximum depth (m) of the domain layer across the 500 m interval. |
| Ping_S, Ping_M, Ping_E | 15126 | A ping is the representation of the return signal (echo trace) measured after the transmission of a single acoustic pulse. Ping_S reports the sequential number of the first ping in the analysis domain (500 m interval) (S for start); Ping_M reports the number of the middle ping (M for middle); and Ping_E reports the number of the last ping (E for end). |
| Dist_S, Dist_E | 499.867146 | The distance (measured by GPS, in meters) from the first ping in the survey to the first ping (S for start) of the 500 m interval, or from the first ping in the survey to the last ping (E for end) in the 500 m interval. |
| Date_S, Date_M, Date_E | 20121104 | The date of the first ping (S for start), middle ping (M for middle), and last ping (E for end) in the 500 m interval. |

| Field | Example | Definition |
|---|---|---|
| Time_S, Time_M, Time_E | 10:49:40.70 | The time of day at which the first ping (S for start), middle ping (M for middle), and last ping (E for end) in the 500 m interval occurred. Time was recorded in GMT. |
| Lat_M | 36.93391333 | The latitude in decimal degrees of the middle ping in the analysis domain (i.e., the center latitude of the 500 m interval). |
| Lon_M | -76.04724667 | The longitude in decimal degrees of the middle ping in the analysis domain (i.e., the center longitude of the 500 m interval). |
| Exclude_below_line_depth_mean | 15.421739 | The mean depth of the bottom line, or exclude-below line, for the 500 m interval. |
| Minimum_Sv_threshold_applied | 1 | A value of 1 indicates that a minimum Sv threshold has been applied (see Minimum_integration_threshold), 0 indicates otherwise. |
| Minimum_integration_threshold | -60 | The value of the minimum threshold entered on the Data page of the Variable Properties dialog box for the variable which was analyzed (dB re $1 m^2/m^3$). For this project the threshold was set to -60 or -54 dB. |
| Maximum_Sv_threshold_applied | 0 | A value of 1 indicates that a maximum Sv threshold has been applied; 0 indicates otherwise. A maximum threshold was never applied for this project. |
| Exclude_above_line_applied | 1 | A value of 1 indicates that the exclude above line has been applied; 0 indicates otherwise. For this project the exclude above line was always applied. |
| Exclude_above_line_depth_mean | 2 | The mean depth (m) of exclude-above line across the 500 m interval. |
| Exclude_below_line_applied | 1 | A value of 1 indicates that the exclude-below line has been applied; 0 indicates otherwise. For this project the exclude below line was always applied. |
| Standard_deviation | 9.20E-09 | The standard deviation of all sample values in the analysis domain (1 x 500 m cell). This is calculated in the linear domain (not the dB domain). |
| Range_mean | 1.344063 | The distance (m) between the mean depth of the layer, and the depth of the center of the transducer face, within the 500 m interval. |
| Exclude_below_line_depth_min | 14.165446 | The minimum depth of the exclude-below line (or essentially the minimum bottom depth) within the 500 m interval. |
| Exclude_below_line_depth_max | 16.602294 | The maximum depth of the exclude below line (or essentially the maximum bottom depth) within the 500 m interval. |

# Chapter 10: Spatial association between seabirds and prey on the mid-Atlantic Outer Continental Shelf

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 10 Highlights

*Examines the spatial relationship between acoustically detected prey and foraging seabirds*

### Context[1]

In this chapter, we determine whether abundance of birds at sea can be predicted on the basis of abundance of their prey. In other chapters in Parts III and IV of this report, models incorporate a variety of physical (sea surface temperature) and biological (presence of other species of seabirds or marine mammals) variables as predictors of seabird abundance. Prey abundance is not generally included as a direct covariate in those models, which employ remotely sensed environmental covariates to predict seabird abundance across a larger spatial area. Rather, covariate data accessible via remote sensing, such as measures of primary productivity (chlorophyll *a*), are used as proxies for prey availability in these models.

The exception to this is Chapter 11, which examines the utility of hydroacoustic data on aquatic biomass (collected via a scientific echo sounder during boat surveys) to predict seabird distributions. It is possible that the low association between seabirds and biomass identified in Chapter 11's model is not because birds do not follow their prey, but rather because temporal and spatial lags between seabirds and prey obscure patterns of association that are in fact present. In this chapter, we use auto- and cross correlation analysis to identify patterns of seabird-prey association that may not have been evident in our other models.

### Study goal/objectives

Identify seabirds that show statistically significant association with acoustically detected prey.

### Highlights

- Four species of seabirds (Northern Gannets, Laughing Gulls, Common and Royal Terns) were statistically associated with patches of prey, as identified by the echo sounder.
- The strength of the statistical association depended on the spatial scale at which it was measured.
- The species of birds for which we detected significant association with their prey feed largely or entirely near the water's surface. Seabird-prey association was not detected for deep diving species such as loons and sea ducks.

### Implications

There are significant associations between seabirds and their prey, although these associations are taxonomically, temporally and spatially variable; spatial and temporal lags appear to be important in identification of seabird-seabird prey associations. Data on prey distributions may be important for delineating seabird hotspots and patterns of habitat use, particularly for foraging.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

We surveyed seabirds from a 55-foot charter vessel off the coasts of Delaware, Maryland, and Virginia over two years (April 2012 to April 2014), and quantified abundance of their prey using a hull-mounted echo sounder. Our objective was to identify areas of importance to foraging birds. As part of this objective, we sought to identify areas where seabirds concentrated due to the abundance or availability of prey. Most of the area we surveyed was within 75 km of the coast, and within the 30 m isobath, allowing us to collect hydroacoustic data in all but roughly the top 5 m of the water column. We found strong association between Northern Gannets (*Morus bassanus*) and Laughing Gulls (*Leucophaeus atricilla*) and acoustically detected prey; and significant but less consistent association for Common and Royal Terns (*Sterna hirundo* and *Thalasseus maximus*). Based on visual observations and existing knowledge of the pelagic fishes of the area, we suspect that much of the prey that we detected and gannets followed was menhaden and Atlantic Herring.

This analysis supplements other modeling efforts in Parts III-IV of this report by explicitly considering the role of seabird prey in determining the spatial distribution of birds. The time series methods presented here yield patterns not evident in other models, because we allow for spatial lags between seabirds and prey. Further, this analysis utilizes *in situ* data, collected in real time from the boat, allowing for the identification of fine-scale patterns that may not be discernible using remotely sensed covariate data. Distribution and relative abundance of prey fishes are clear drivers affecting seabird distributions. These populations should be more carefully considered when attempting to identify biologically important areas for seabirds, both generally during marine spatial planning efforts and specifically for siting offshore development on the Atlantic Outer Continental Shelf.

## Introduction

The mid-Atlantic region is used by a broad range of marine wildlife species across the entire annual cycle (Rowlett 1980). This is largely due to a relatively high level of productivity, as compared to the rest of the western North Atlantic (Yoder et al. 2001), as well as the region's central geographic location on the eastern edge of the continent. Areas near the mouths of the Delaware Bay and Chesapeake Bay typically have the highest offshore levels of primary productivity in the region, due to year-round mixing of saline and fresh waters through estuarine circulation, in combination with strong tidal currents. As water flows from the bays into the study area, nutrient and phytoplankton rich waters are swept southwards by the Labrador Current. In these shallow coastal waters, sunlight is able to penetrate a relatively high proportion of the water column (Xu et al. 2011; Schofield et al. 2008), fueling photosynthetic activity and growth of phytoplankton where nutrients are available.

The Mid-Atlantic Bight exhibits a large pulse in productivity each year due to a winter-spring bloom of phytoplankton, and occasionally an additional phytoplankton bloom in summer months (Yoder et al. 2001). This is followed by a pulse in secondary productivity – zooplankton species foraging on the phytoplankton – which in turn become food for larger predators, such as small fishes. The area is generally rich with these small, schooling epipelagic fishes (Pikitch et al. 2014). In the mid-Atlantic region, key fish species include the Atlantic menhaden (*Brevoortia tyrranus*), Atlantic mackerel (*Scomber scombrus*), Bay Anchovy (*Anchoa mitchilli*) and 'river herring', including the alewife (*Alosa* *pseudoharengus*) and blueback herring (*Alosa aestivalis*). Two large invertebrate species – the longfin inshore squid (*Loligo paeleii*) and the northern shortfin squid (*Illex illecebrosus*) – are also important prey items for a broad range of predators in the Mid-Atlantic Bight (Dawe et al. 2007; Hendrickson 2004). The presence of these pelagic fish populations indicate the elevated productivity of the area, and are likely responsible, in part, for the relatively high density of predators that use the area. Rowlett (1980) conducted shipboard surveys of the area during the 1970s and showed that this elevated productivity translates into high abundance of marine mammals and birds.

Non-invasive, quantitative estimates of fish abundance and aquatic biomass have been made possible in recent years with the development and subsequent improvement of acoustic echo sounding hardware, including split- and multi-beam transducers employing echo-counting and interpretation software. During the past decade, the development of stable scientific echo sounders, multi-frequency applications, new transducer deployment techniques, standardized calibration procedures, and more realistic models of the sound-scattering properties of biological targets have improved accuracy of biomass estimations (Rudstam et al. 2013; Simrad 2012). The echo sounder sends acoustic signals into the water column and detects resulting backscattered energy reflected from fish and other objects. These data can be integrated and summed by distance and depth intervals in order to estimate the contribution of backscattered energy from all targets within each sampling volume and to estimate prey abundance by area and by volume along survey transects.

Pelagic seabirds are often categorized into feeding guilds, based on their method of food capture and the depths to which they are able to dive (Ashmole 1971). While different seabirds very broadly differ in their physical ability to reach particular depths, the shallow (<= 40 m) waters of the US East coast largely obliterate these differences between species, as the fishes and plankton upon which the birds feed routinely travel from surface to bottom. There are probably some exceptions: the shellfish upon which scoters feed (Bordage and Savard 2011) certainly stay on the bottom and storm-petrels are rather strictly restricted to the immediate surface layers (Warham 1996). Loons may dive to depths of 70m to feed at or near the bottom of the water column (Evers et al. 2010). Gannets, shearwaters, and gulls are pelagic feeders and can likely access any fish or plankton within the 40 m water column.

Seabird prey have been successfully detected by echo sounders in the Antarctic (Veit et al. 1993; Veit 1999; Veit et al. 2008), in coastal waters of California (Santora et al. 2011), and in Newfoundland (Davoren et al. 2003a, b). While there are certainly correlations between seabirds and prey, these relationships can be complicated and difficult to discern because of the dynamic nature of predators searching for mobile prey. Relationships are strongly scale-dependent, with associations at differing scales showing different aspects of predator-prey interactions. (Veit et al. 1993; Rose and Leggett 1990).

Acoustic detection of prey is based upon the difference in density between that of seawater and that of prey, and detection of prey depends on their linear dimensions, as well as their density (Simmonds and MacLennan 2006). Thus, because we used a 120 kHz transducer, we have an approximate idea of the size of organisms detected (small fishes and large zooplankton). Nevertheless, we do not know for sure what the backscattering detected by our instrumentation consisted of. Due to substantial general knowledge of the marine biota of the area (e.g., Rowlett 1980; Overton et al. 2008; Scofield et al. 2008) we can be reasonably certain that a substantial fraction of the biomass we detected was from schooling pelagic fishes, especially menhaden. Indeed, many of the menhaden schools that we detected visually from the ship and identified through photography were accompanied by large acoustically-detected patches, similar to those found throughout the surveys. Thus, for purposes of this study, we refer to acoustically detected biomass as acoustically detected prey, assuming that the majority of detected biomass was comprised of prey species (while acknowledging that there were likely additional species detected that may not have been prey for the seabirds we examined).

In this study, we conducted two years of boat-based surveys for wildlife within a study area on the mid-Atlantic Outer Continental Shelf. Standardized boat-based surveys are a well-established and widely used method of obtaining abundance estimates for pelagic seabird species. While conducting these boat-based surveys, we also collected hydroacoustic data in order to index the relative abundance of aquatic prey species (fish and zooplankton) present beneath the survey vessel. We examine the relationship between acoustically detected prey and seabird predators on the mid-Atlantic Outer Continental Shelf at varying spatial and temporal scales, ranging from aggregated surveys (across the entire study area and two year study period) to individual transects within single surveys. We included eight seabird species in the  analysis, including one surface feeder (Wilson's Storm-Petrel), two benthic feeders (Common Loon, *Gavia immer;* and Red-throated Loon, *Gavia stellata*), and five pelagic feeders (Northern Gannet, *Morus bassanus*; Laughing Gull, *Leucophaeus atricilla*; Common Tern, *Sterna hirundo*; Royal Tern, *Thalasseus maximus*; and Dovekie, *Alle alle*).

## Methods

Sixteen boat-based surveys were conducted between April 2012 and April 2014, each covering 12 transects running perpendicular to shore and spaced roughly 10 km apart, offshore of Delaware, Maryland, and Virginia in the vicinity of three federally designated areas for potential offshore wind energy development (Wind Energy Areas, or WEAs; Figure 10-1). During each boat survey, observers recorded data on wildlife encountered, including species identification, number of individuals observed, locations of observations, and behavior. Detailed methods regarding boat-based survey protocols are found in Chapter 7 of this report.

Hydroacoustic data were simultaneously collected using a Simrad EK60 scientific echo sounder unit (Kongsberg Maritime AS, Horten, Norway) with a hull mounted 120 kHz split-beam transducer, documenting the relative abundance of acoustically detected prey along the survey transects throughout the water column. Hydroacoustic data were collected during all 16 boat-based surveys (a total of 66 survey days), and were calibrated for the speed of sound and absorption coefficients using mean water temperature and salinity values collected every 30 minutes during boat-based surveys. Data were filtered to exclude very small targets (< 2 cm), as well as surface and bottom noise potentially caused by non-biotic objects. A surface line was drawn at a depth of 2 m below the water's surface (roughly 0.8 m below the surface of the transducer), and a bottom line was manually edited to exclude the bottom substrate and targets indistinguishable from the bottom (roughly 20 cm above the ocean floor). All backscattering signals occurring above the surface line or below the bottom line were excluded from analysis. Remaining data were integrated by 1 m depth intervals (or "layers") and 500 m distance intervals (or "intervals") to calculate the nautical area-scattering coefficient (NASC) value for each 1 x 500 m cell within the survey (Chapter 9). The total NASC represented an index of total available biomass within the water column; additional details regarding hydroacoustic data collection and post-processing are found in Chapter 9 of this report.

Boat-based survey and hydroacoustic data were combined in ArcGIS version 10.2.2 (ESRI, Redlands, California) and binned into 2 km transect segments, summing the total number of individuals observed per species per segment, as well as total NASC per segment. Areas surveyed between transects, or "doglegs," were included in analyses if both survey and hydroacoustic data were consistently collected, and were assigned transect numbers for data management and analysis purposes (Transects 13-31; Figure 10-1).

We initially chose to examine associations between birds and acoustically detected prey for eight bird species, including Wilson's Storm-Petrel, Northern Gannet, Laughing Gull, Common Tern, Royal Tern, Dovekie, Common Loon, and Red-throated Loon. We chose these species as they were relatively abundant in our surveys. The five pelagic foragers (Northern Gannet, Laughing Gull, Common Tern, Royal Tern, and Dovekie) were also judged to be likely to feed upon pelagic prey. We partitioned the bird data into two groups for each species. The first group included all observed individuals. The second group consisted only of birds whose behavior suggested feeding activity, including diving, plunge diving, feeding, and milling (thus capturing observations where birds dove into the water, dove under the water, were seen with prey in the beaks, or were flying in tight circles as if in the process of searching for food; Veit 1999; Camphuysen and Garthe 2004). For each group of each species, we calculated Spearman rank correlation coefficients to assess spatial association (at a spatial lag of 0 km) between observed birds and acoustically detected prey across all surveys and locations. Species that showed no statistically significant association at this scale, such as Dovekie, were excluded from further analysis.

For species that showed significant association with prey across all surveys, we used cross-correlation analysis in Statistica (StatSoft© 2005) to assess spatial association between birds and their prey at spatial lags ranging from 0 to 15 lags (30 km) in either direction, and at geographic and temporal scales ranging from the entire study (16 surveys over two years, for a total of 10,698 linear transect km) to individual surveys or transects. We tested statistical significance of detected patterns using randomization of the bird relative to the hydroacoustic data (Rose and Leggett 1990; Veit et al. 1993; Veit et al. 2008).

Generalized Linear Models (GLMs; Zuur et al. 2009) were constructed for Northern Gannets and Laughing Gulls, the two species most highly associated with acoustically detected prey. Data were summarized by transect for the 12 long transects (Transects 13-31 were excluded, because they were shorter than what cross-correlation analyses suggested was required to detect many associated bird and prey aggregations). Data were included for survey months during which each species was most abundant (Northern Gannet: October through early May; Laughing Gull: mid-April through November). Resulting datasets contained 107 transects for Northern Gannets and 129 for Laughing Gulls. Three variables were calculated for each transect: mean bird abundance across 2 km bins, mean echo sounding backscatter (NASC), and maximum cross-correlation (ccf) for any scale (lag) between birds and prey. For maximum cross-correlation, values were restricted to those within spatial lags of < 10 km, or 20 km. We first calculated Spearman rank correlation coefficients among these quantities, and then built regression models for each species.

We then constructed Generalized Linear Models with negative binomial distribution of errors (Zuur et al. 2009, pp. 233-236) of the form:

Bird abundance = prey abundance + ccf (birds,prey) + error

using the logic that both prey abundance and the spatial correlation among birds and prey ought to combine to give useful information about where birds are abundant. We modeled data from the longer transects (20 + km) from the season during which modeled species was present. We counted the number of transects on which we found significant cross-correlation between gannets and prey, in an attempt to characterize persistence of such feeding aggregations.

## Results

At the largest spatial scale, that of the entire two-year study (16 surveys), Northern Gannets, Laughing Gulls, Common Terns and Royal Terns were statistically associated with acoustically detected prey; no statistically significant association was observed between acoustically detected prey and Common Loons, Red-throated Loons, Wilson's Storm-Petrels, or Dovekies (Table 10-1). This coarse scale analysis does not take into account spatial lags between birds and prey, as are bound to occur (Veit et al. 1993), due to feeding taking place sometime before the ship encounters the aggregation. To detect these relationships, we analyzed the entire dataset using cross-correlation analysis for species found to be statistically associated with prey. Of these, Northern Gannets and Laughing Gulls were most strongly associated in space with prey detected by the echo sounder; weaker associations were observed between acoustically detected prey and Common Terns and Royal Terns.

### *Northern Gannets*

Across all surveys, Northern Gannets were significantly correlated with acoustic prey at spatial lags of zero, 2, and 4 km, and also at lags of -14 to -18 km and positive 18 km to 22 km. The maximum correlation was at a lag of zero (Figure 10-2, Figure 10-3, and Figure 10-4).

Finally, much of the "noise" in the single-survey-scale pictured in Figure 10-4 is due to the spatial lags between birds and prey (Veit et al. 1993, Veit et al. 2008); if birds are encountered before prey patches on a transect at one place and after prey patches at another, these lags will obfuscate one another and lower the overall correlation. To circumvent this issue, we plotted cross-correlation function (CCF) and time series plot from single transects. Figure 10-5 and Figure 10-6 show examples of these analyses using data from two transects surveyed in March 2013.

### *Laughing Gulls*

Laughing Gulls were the next species most associated with prey as detected by acoustics. Laughing Gulls were most abundant May to October. As with Northern Gannets, we began by examining bird-prey association across all surveys (Figure 10-7, Figure 10-8). There was significant cross-correlation across a broad range of spatial scales, suggesting that this species feeding at a broader range of prey patches than Northern Gannets. As was true for Northern Gannets, Laughing Gulls track their prey even more closely when examined at the scales of single surveys (Figure 10-9) and individual transects (Figure 10-10). Also as for Northern Gannets, there were spatial lags between Laughing Gulls and the prey they were likely feeding upon. At the largest spatial scale, the most significant associations occurred at a lag of 10 km (Figure 10-8).

### *Common and Royal Terns*

The data on Common Terns show very little consistent association between birds and prey at large scales, due mainly to large numbers of patches of prey unattended by birds (Figure 10-11). There was no correlation between terns and prey at the scale of all surveys, or even within a single survey (Figure 10-12). However, at the scale of an individual transect, terns tracked fish schools well. Figure 10-13 shows associations along Transect 11, an enlargement of the far right end of the transects represented in Figure 10-12.

As with Common Terns, Royal Terns showed no statistical association with fish at the scale of all surveys or within a single survey. However, there was evidence of this association for the whole of the June 2012 survey (Figure 10-14), and a tight association along Transect 3 (Figure 10-15).

### *Models of Gannet and Laughing Gull Distributions*

We first calculated Spearman rank correlation coefficients among these quantities, and then built regression models for each species (Table 10-2 and Table 10-3).

The GLM modeling showed that inclusion of both prey abundance and the spatial correlation among birds and prey combine to give more useful information about where birds are abundant than prey abundance alone. Thus, by including spatial association between birds and prey one includes both prey abundance and prey availability to birds (Table 10-4).

Finally, we counted the number of transects on which we found significant cross-correlation between gannets and prey, in an attempt to characterize persistence of such feeding aggregations. There was not much variation among transects; all of them tended to have feeding gannets, so this approach was not useful in distinguishing among transects (Table 10-5).

## Discussion

The seabirds we surveyed on the mid-Atlantic Outer Continental Shelf fell into two groups: those which were clustered in association with acoustically detected prey, and those which were not. Northern Gannets were substantially more associated with acoustically detected prey than any other species. This is likely because Northern Gannets feed on relatively large (> 4”) fish located close to the surface. We suspect, but cannot prove directly, that the majority of these prey items were either Atlantic menhaden or possibly other similar schooling species such as Atlantic Herring (*Clupea harengus*; Reid et al. 1999). Laughing Gulls were also significantly associated with acoustically detected patches, but were less tightly clustered around big patches than were Northern Gannets. We do not know if Laughing Gulls eat the same fish as Northern Gannets, but in general they capture smaller individuals (Burger 1996). Common Terns and Royal Terns were associated with acoustically detected patches during some transects and surveys, but these correlations were less consistent than for Northern Gannets or Laughing Gulls.

Common Loons and Red-throated Loons, Dovekies, and Wilson's Storm-Petrels showed no statistically significant association with acoustic patches. Loons generally feed on fishes located at or close to the bottom (Evers et al. 2010), which are less likely to be detected by acoustics, as backscattering signals indistinguishable from the ocean floor were excluded from analysis during post-processing of hydroacoustic data (Chapter 9). Wilson's Storm-Petrels and Dovekies both feed on zooplankton (including small fishes) but of a size range that may fail to appear prominently in our acoustic data. Wilson's Storm-Petrels, moreover, feed at the immediate surface (top 6"), an area not detectable by our hull mounted echo sounder (Veit et al. 2008; Chapter 9).

Since a major objective of these surveys was to identify areas of the mid-Atlantic Outer Continental Shelf that are particularly important (or unimportant) to foraging birds, it is pertinent to determine how these acoustic data can be used to identify important foraging areas. It is clear that our acoustic data yields insight into important foraging areas for Northern Gannets and Laughing Gulls, and perhaps terns as well. From repeated sampling of our transects, we can estimate what proportion of those transects had 1) aggregations of feeding birds associated with prey and 2) patches of prey with similar characteristics to those that at other times were attractive to birds. The acoustic data on seabird prey is the only real time information we have that is immediately relevant to the foraging needs of the birds (although real time data on sea temperature and salinity, also collected from the survey boat, provide an indirect tool). Therefore more extended analysis of the bird-prey association is warranted as part of a habitat assessment for seabirds.

Using echo sounders to assess abundance of prey has been a controversial practice, although most controversy surrounds the conversion of backscattering strength to an absolute estimate of biomass, which we are not attempting (Simmonds and Maclennan 2006). We are concerned with the location of patches of prey, and their relative size, rather than the actual biomass of the organisms in the patches. Thus, we are using acoustic detection of prey as an index of abundance. This goal substantially minimizes the types of criticisms to which the use of acoustics has been exposed.

Nevertheless, we do have issues of species identification and bias in the portions of the water column sampled. To a certain extent, the size of organisms (Madureira et al. 1993; Logerwell and Wilson 2004; Lawson et al. 2008) can be determined from backscattering strength, and species of fishes and zooplankton can be identified especially if the acoustic data can be compared to net samples or local knowledge of the dominant species. Acoustic data were filtered to exclude objects less than about 2 cm in diameter (Chapter 9), and it is highly likely that that the overwhelming majority of aggregations we detected acoustically were fish rather than zooplankton, potentially menhaden and herring (Overholtz et al. 2000; Overton et al. 2008). However, we had no supplementary data on prey species or biomass composition for this region. Additionally, since our transducer was mounted on the keel (approximately 2 m below the waterline) and surface noise commonly reached depths of approximately 5 m, objects in the upper 0-5 m of the water column were invisible to the acoustics. This means that prey for the real surface feeders (certainly storm-petrels, perhaps terns as well) were not quantified (cf. Veit et al. 2008). For this reason, it is not surprising that Wilson's Storm-Petrel abundance was not correlated with acoustic data on our surveys.

By analyzing and modeling relative seabird and prey abundance at the transect level, we have provided statistical evidence that co-occurrence of feeding birds and prey swarms are an important indicator of quality habitat for Northern Gannets and Laughing Gulls. What remains to be done is to use the acoustic data on prey abundance to make predictions about what portions of the area we surveyed are most important to seabirds. One way to do this would be to incorporate either acoustic prey relative abundance or bird-prey co-occurrence (since some prey patches are inaccessible) as independent variables in a habitat model. The problem with that approach is that prey patches are highly mobile so there is a strong stochastic component to the exact location of the patches. An alternative might be to construct a probability-of-occurrence map for prey swarms, and include those values within a habitat model. In any event it seems critical to do this, at least for the species that this analysis shows to be statistically associated with prey. Thus, further analysis should focus on identification of hotspots incorporating real-time data on prey (as we have done here), and ascertain whether such prey-based hotspots are spatially coincident with hotspots identified on the basis of other environmental data, or from data on bird abundance by itself.

It is important to collect data on prey abundance during seabird surveys for two reasons, as supported by our results. First, predator-prey interactions are an essential component of ecological systems (Barbosa and Castellanos 2005), and the high levels of variability in marine systems may make prey distribution patterns particularly important for predicting the distributions of higher trophic level predators such as seabirds in the marine environment (Fauchald et al. 2000). Second, our acoustic data on seabird prey was collected simultaneously with the data on seabirds, and the two datasets are thus spatially and temporally linked to a closer degree than for most dynamic environmental variables we examined in this report (which were often at a ≥1 km spatial scale, and represented daily, weekly, or even monthly averages; Chapters 11-12, 15-16, 18-19). The high mobility of both birds and their prey make spatial associations complex and difficult to measure, as predator-prey interactions, ocean currents, and other dynamic factors ensure that there are often temporal and spatial lags between aggregations of seabirds and their prey. Nevertheless, such associations potentially provide the strongest indication of what parts of the ocean the birds find most important for foraging. It is critical to consider prey populations, including their distributions and the variability in those distributions, when attempting to identify important habitat use areas for upper trophic level predators.

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## Figures and tables

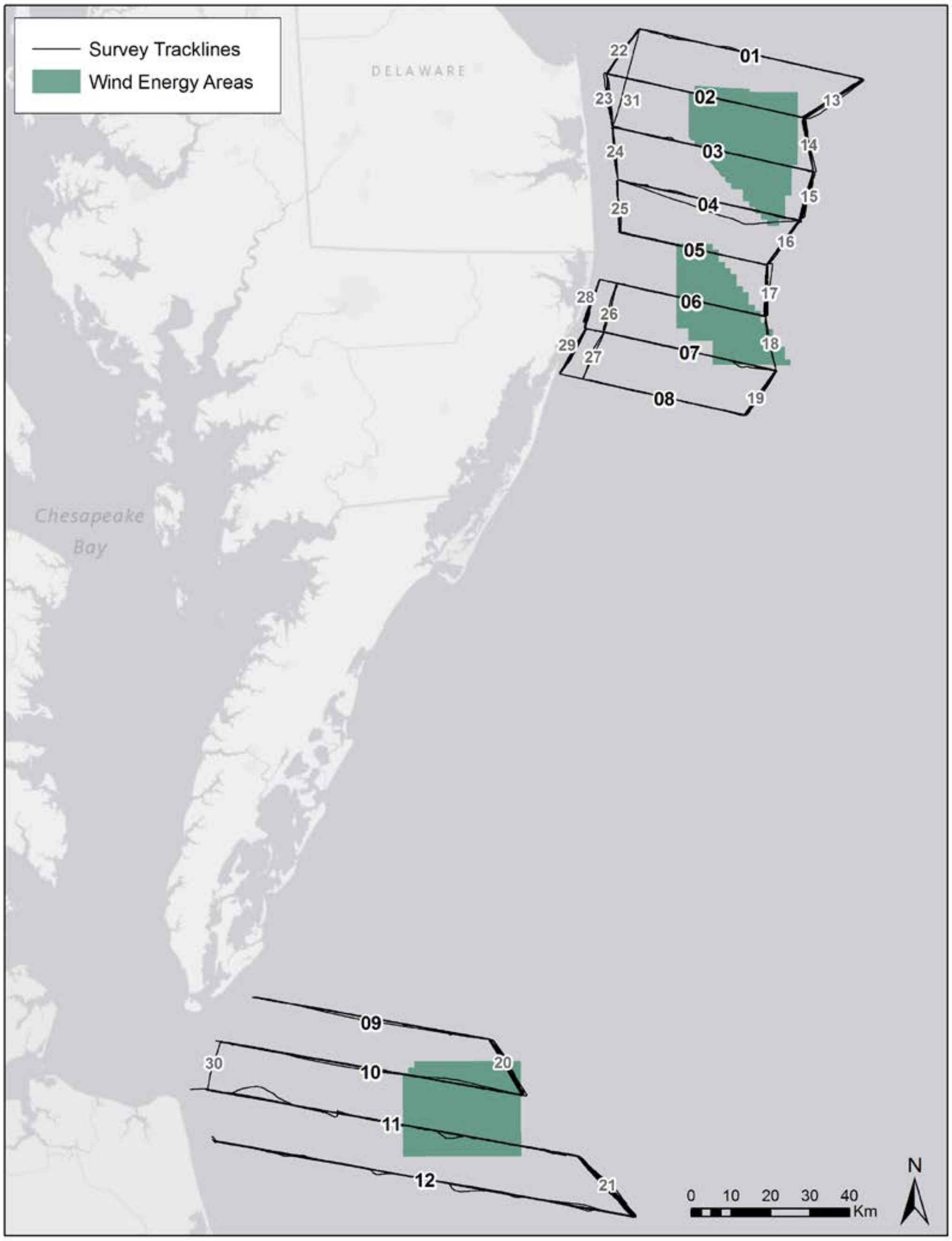


**Figure 10-1. Boat-based survey tracklines.** Survey transects are labeled by transect number. Transects 1-12 were repeated in each of the 16 surveys conducted between April 2012 and April 2014. Transects 13-31 are 'doglegs' connecting official transects, and were numbered for data management and analysis purposes where both survey data and hydroacoustic data were consistently collected. Exact transect locations varied slightly between surveys due to avoidance of military activities and other vessels, among other factors. Green areas are federally designated locations for potential offshore wind energy development.

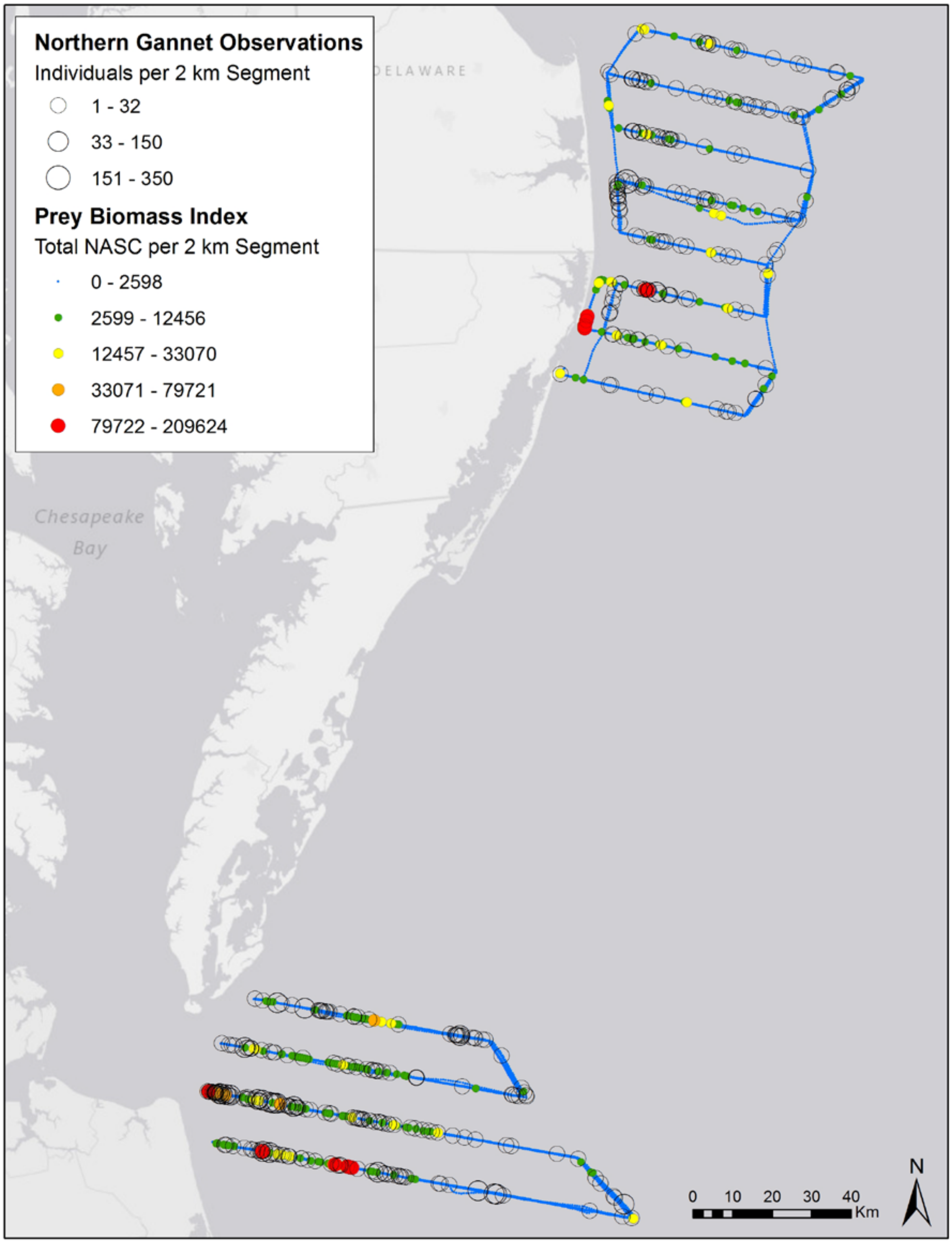


**Figure 10-2. Feeding Northern Gannet observations and prey biomass index per 2 km transect segment for data collected October-April, 2012-2014.** Observation counts and biomass index values are classified by natural breaks.

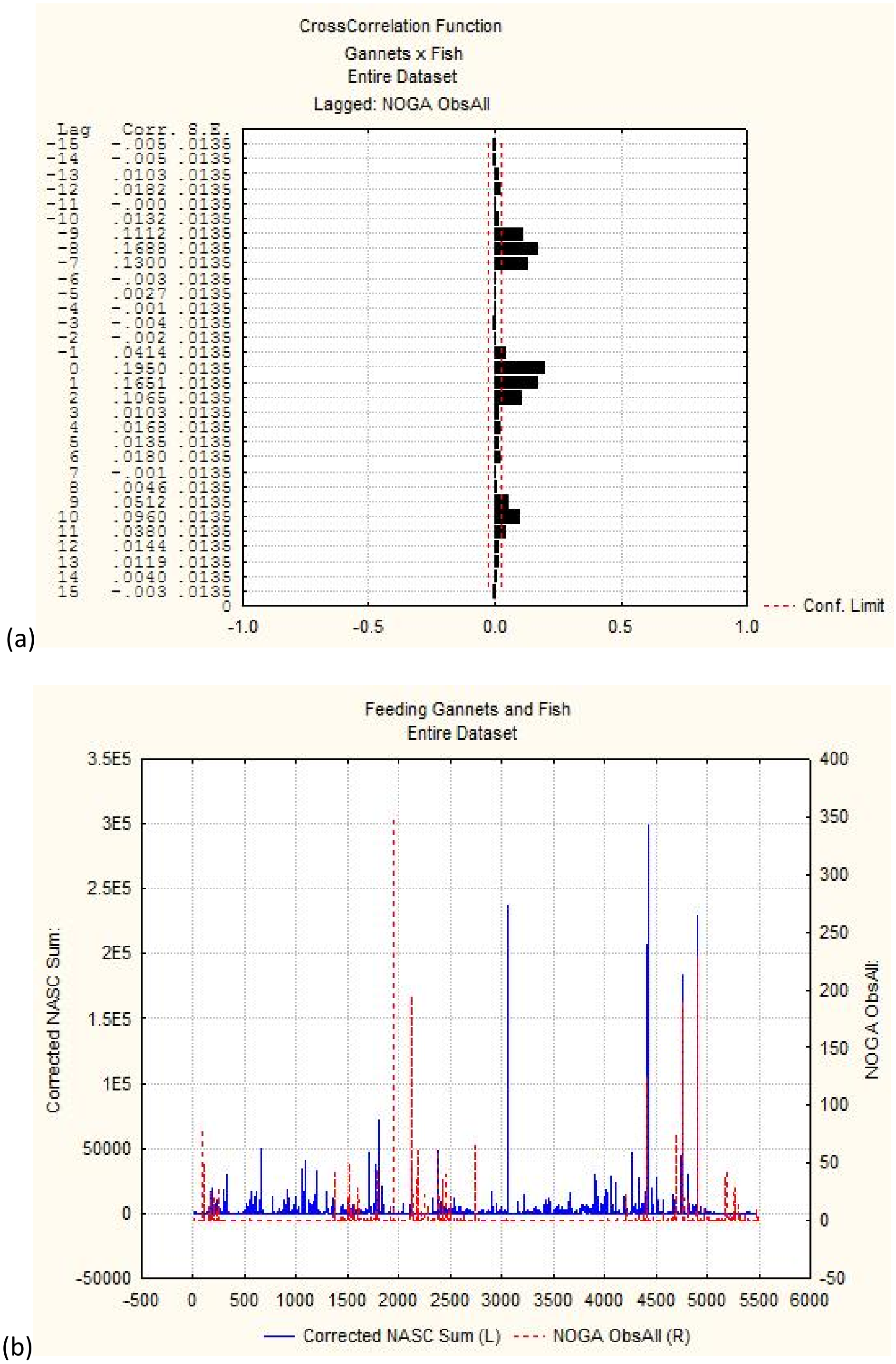
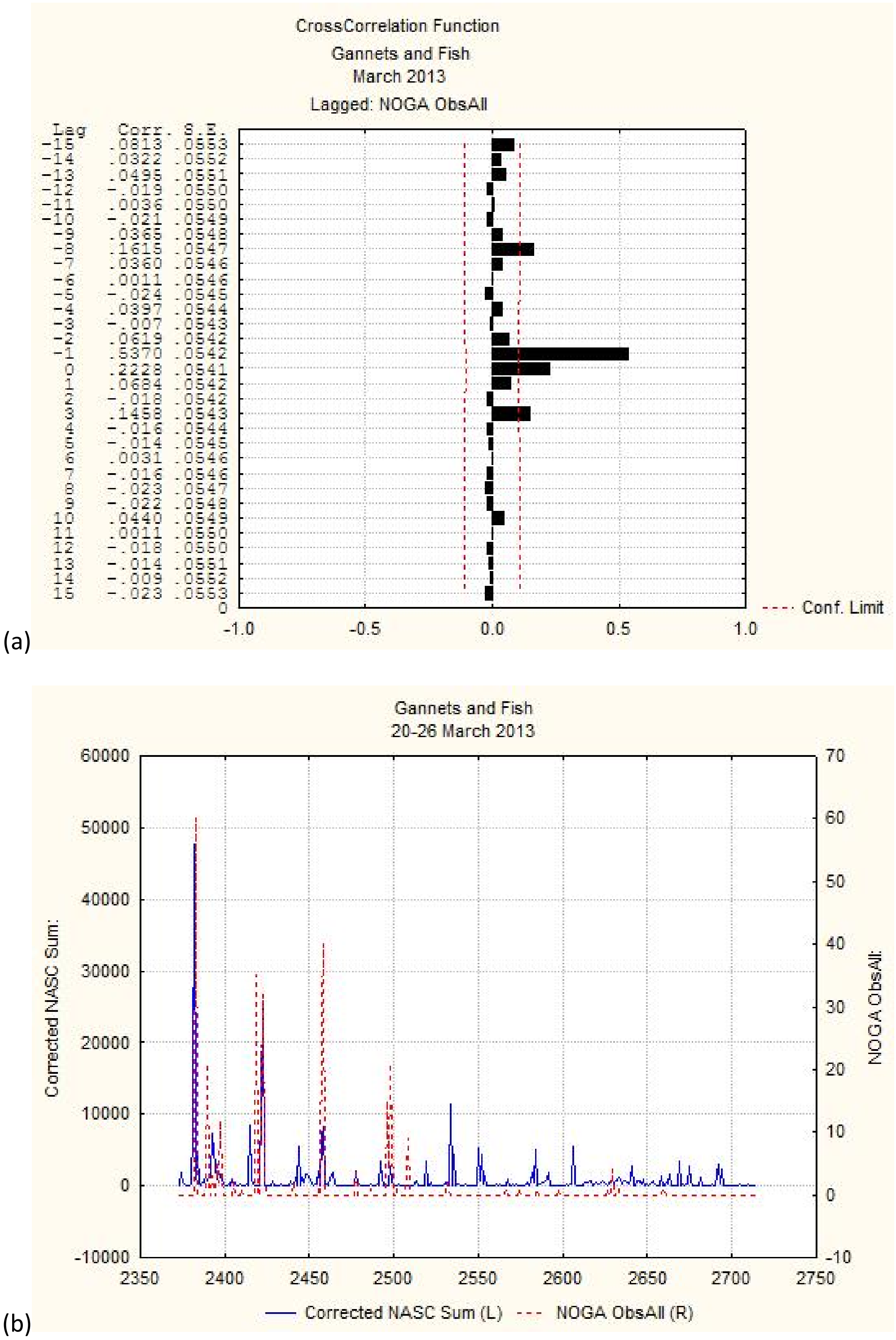
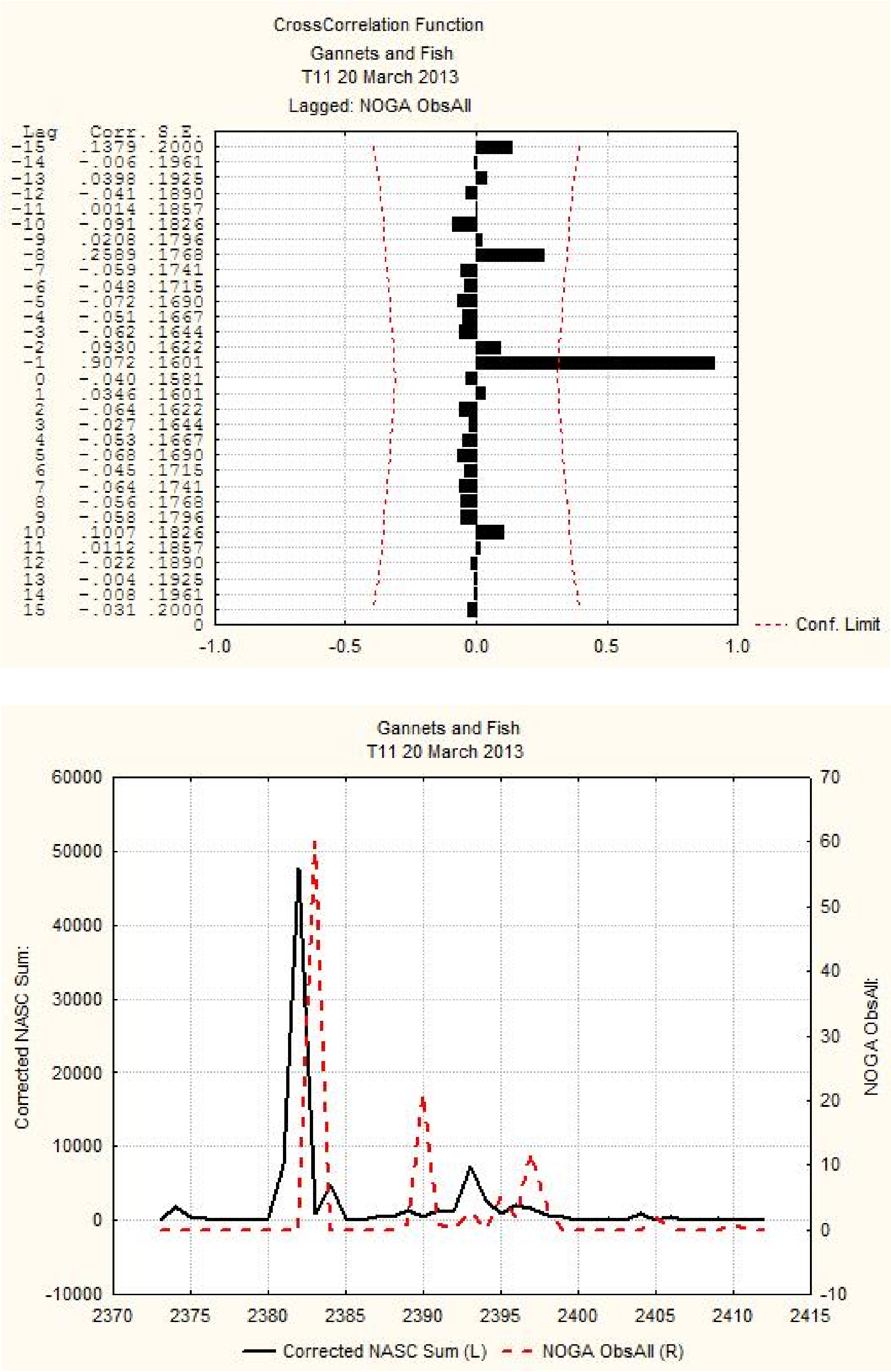


**Figure 10-3. (a) Cross-correlation and (b) time series plot between feeding Northern Gannets and acoustically detected prey across all surveys**. In (a), the “lag” value represents 2 km segments between gannet counts and NASC measurements of prey biomass; x-axis is correlation between these two values at different lag values. Red dotted lines are confidence limits. In (b), the x-axis is distance, and numbers represent consecutive 2 km bins along the transect lines. The two right hand bars reaching above 200 Northern Gannets represent feeding flocks closely associated with schools of fish detected by echo sounder.



| Lag | Corr. | S.E. |
|---|---|---|
| -15 | .0813 | .0553 |
| -14 | .0322 | .0552 |
| -13 | .0495 | .0551 |
| -12 | -.019 | .0550 |
| -11 | .0036 | .0550 |
| -10 | -.021 | .0549 |
| -9 | .0365 | .0548 |
| -8 | .1615 | .0547 |
| -7 | .0360 | .0546 |
| -6 | .0011 | .0546 |
| -5 | -.024 | .0545 |
| -4 | .0397 | .0544 |
| -3 | -.007 | .0543 |
| -2 | .0619 | .0542 |
| -1 | .5370 | .0542 |
| 0 | .2228 | .0541 |
| 1 | .0684 | .0542 |
| 2 | -.018 | .0542 |
| 3 | .1458 | .0543 |
| 4 | -.016 | .0544 |
| 5 | -.014 | .0545 |
| 6 | .0031 | .0546 |
| 7 | -.016 | .0546 |
| 8 | -.023 | .0547 |
| 9 | -.022 | .0548 |
| 10 | .0440 | .0549 |
| 11 | .0011 | .0550 |
| 12 | -.018 | .0550 |
| 13 | -.014 | .0551 |
| 14 | -.009 | .0552 |
| 15 | -.023 | .0553 |

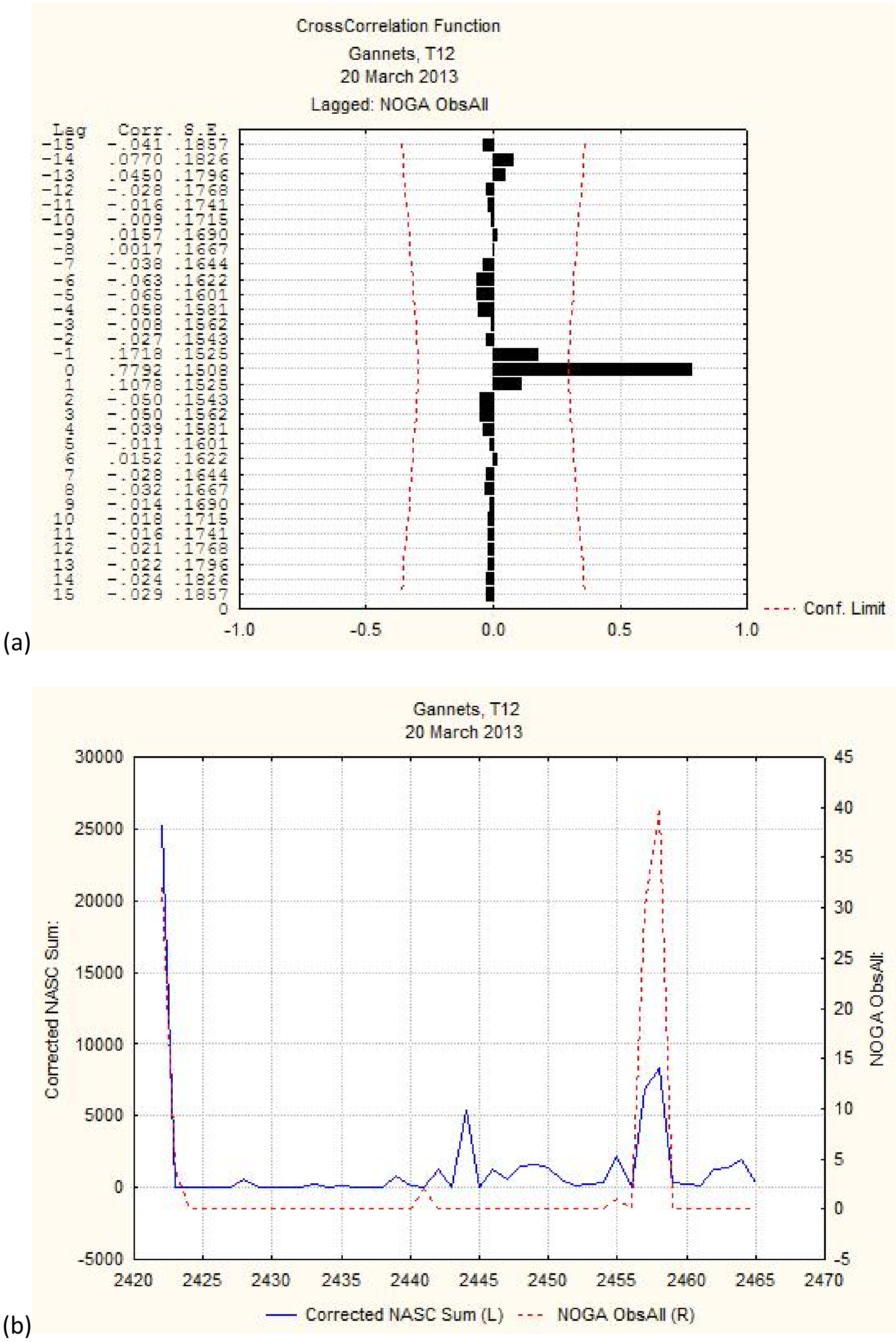
**Figure 10-4. (a) Cross-correlation and (b) time series plot between feeding Northern Gannets and acoustically detected prey within a single survey, 20-26 March 2013.** Note that the strongest correlation is at 1 spatial lag (2 km). Also note the tighter association between predators and prey than is shown in Figure 10-3(b).



(a)

(b)

**Figure 10-5. (a) Cross-correlation and (b) time series plot between Northern Gannets and acoustically detected prey along Transect 11, 20 March 2013.** Note that the largest prey patch is accompanied by the largest aggregation of Northern Gannets along the 76 km transect. The other two aggregations of Northern Gannets seem to flank the second largest prey patch. The tightness of the association is reflected in the correlation coefficient (0.9) in (a) above.



| Lag | Corr. | S.E. |
|---|---|---|
| -15 | -.041 | .1857 |
| -14 | .0770 | .1826 |
| -13 | .0450 | .1796 |
| -12 | -.028 | .1768 |
| -11 | -.016 | .1741 |
| -10 | -.009 | .1715 |
| -9 | .0157 | .1690 |
| -8 | .0017 | .1667 |
| -7 | -.038 | .1644 |
| -6 | -.063 | .1622 |
| -5 | -.065 | .1601 |
| -4 | -.058 | .1581 |
| -3 | -.008 | .1562 |
| -2 | -.027 | .1543 |
| -1 | .1718 | .1525 |
| 0 | .7792 | .1508 |
| 1 | .1078 | .1525 |
| 2 | -.050 | .1543 |
| 3 | -.050 | .1562 |
| 4 | -.039 | .1581 |
| 5 | -.011 | .1601 |
| 6 | .0152 | .1622 |
| 7 | -.028 | .1644 |
| 8 | -.032 | .1667 |
| 9 | -.014 | .1690 |
| 10 | -.018 | .1715 |
| 11 | -.016 | .1741 |
| 12 | -.021 | .1768 |
| 13 | -.022 | .1796 |
| 14 | -.024 | .1826 |
| 15 | -.029 | .1857 |

**Figure 10-6. (a) Cross-correlation and (b) time series plot between Northern Gannets and acoustically detected prey along Transect 12, 20 March 2013.**

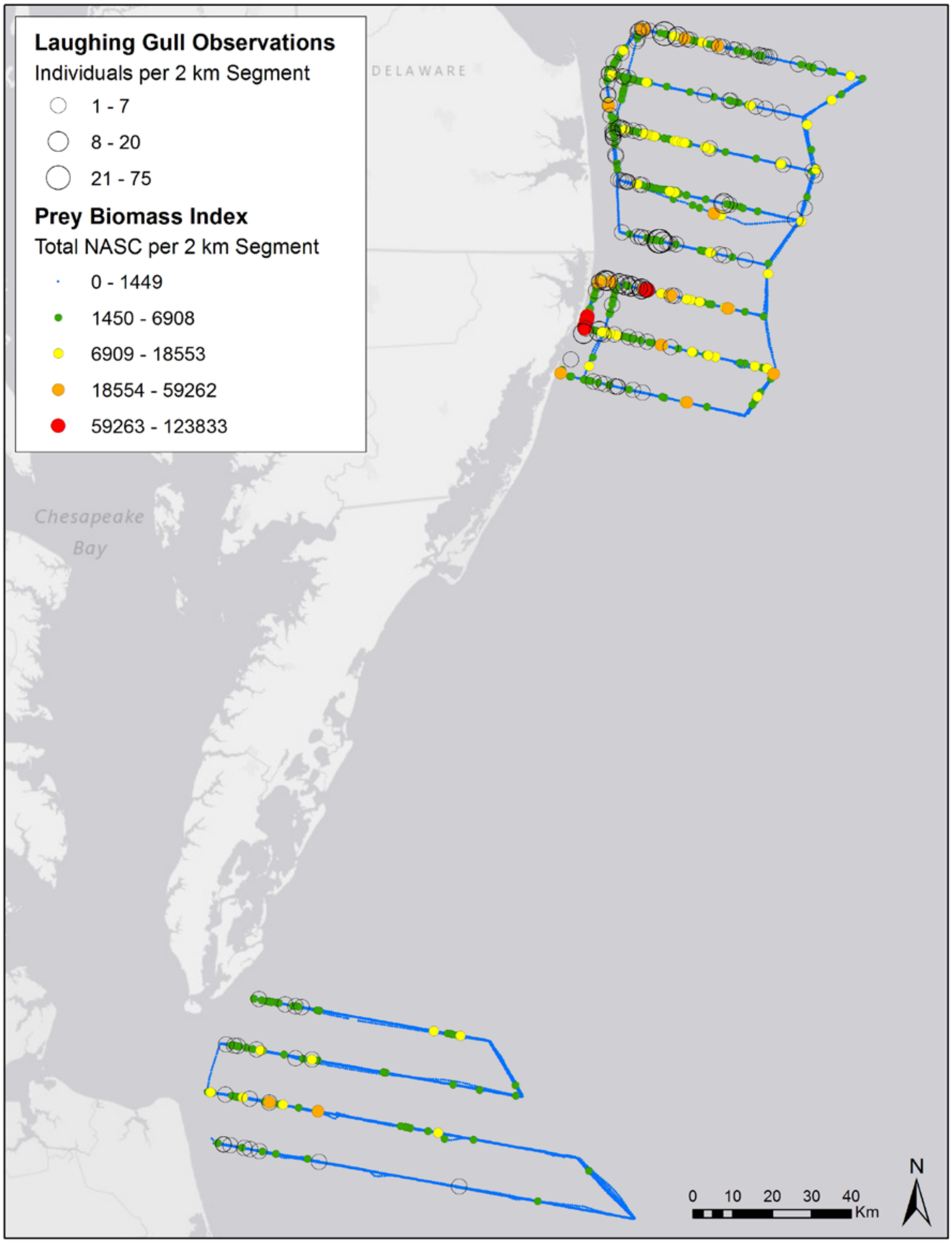


**Figure 10-7. Feeding Laughing Gull observations and prey biomass index per 2 km transect segment for data collected during 10 surveys between April and October (2012-2014).** Observation counts and biomass index values are classified by natural breaks.

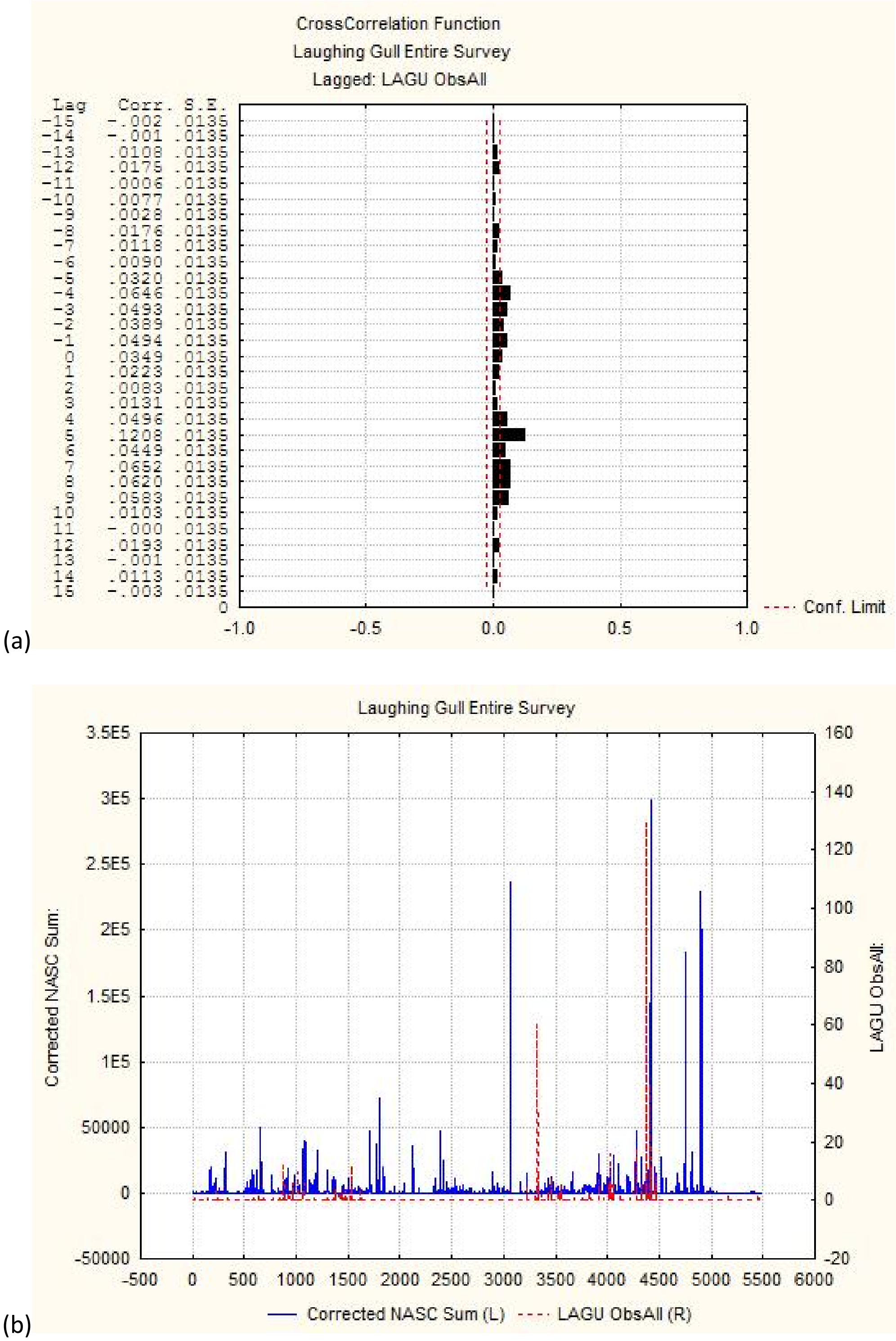
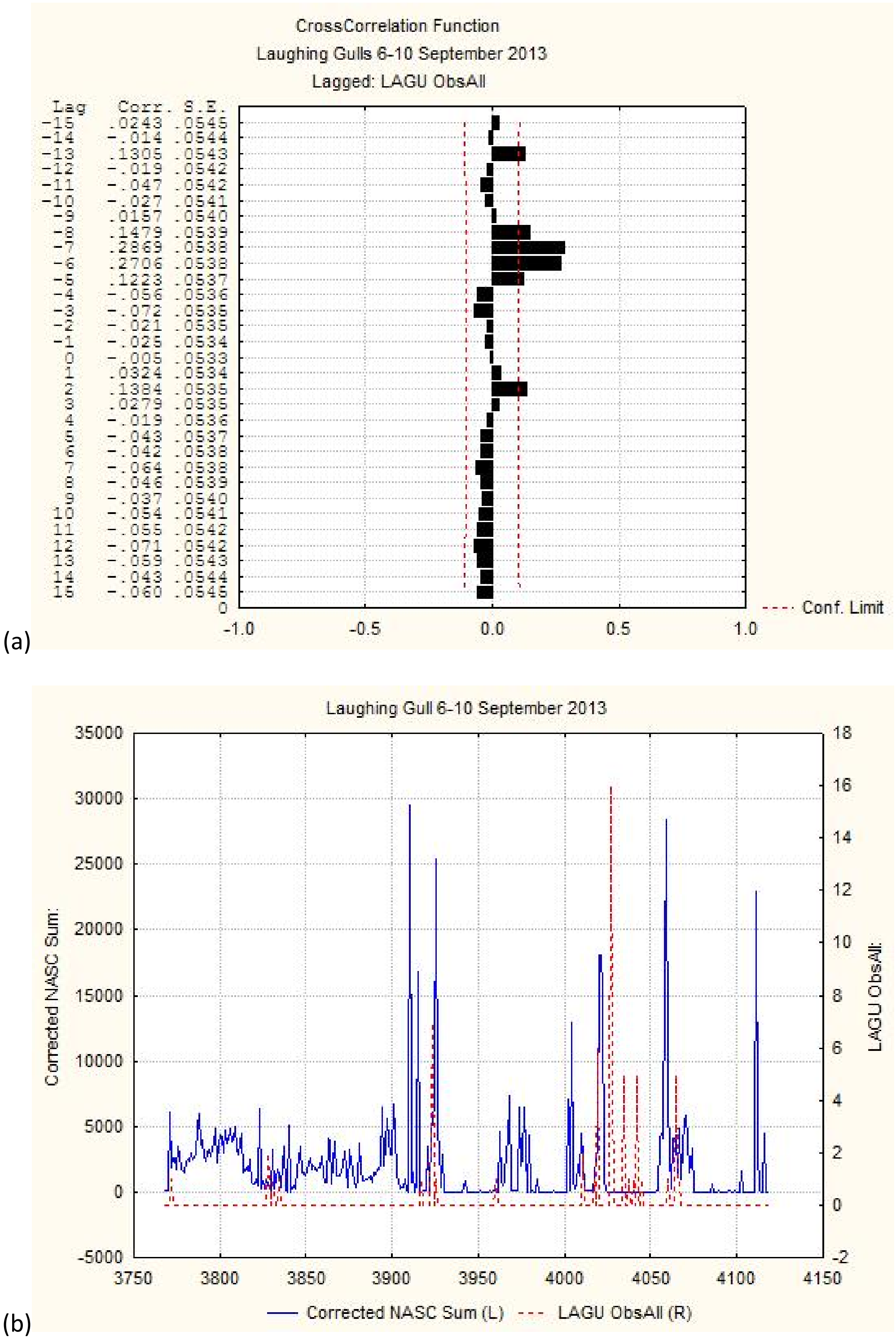


**Figure 10-8. (a) Cross-correlation and (b) time series plot between Laughing Gulls and acoustically detected prey across all surveys.** Note that Laughing Gulls have found 3 out of four of the major prey aggregations detected.



| Lag | Corr. | S.E. |
|---|---|---|
| -15 | .0243 | .0545 |
| -14 | -.014 | .0544 |
| -13 | .1305 | .0543 |
| -12 | -.019 | .0542 |
| -11 | -.047 | .0542 |
| -10 | -.027 | .0541 |
| -9 | .0157 | .0540 |
| -8 | .1479 | .0539 |
| -7 | .2869 | .0538 |
| -6 | .2706 | .0538 |
| -5 | .1223 | .0537 |
| -4 | -.056 | .0536 |
| -3 | -.072 | .0535 |
| -2 | -.021 | .0535 |
| -1 | -.025 | .0534 |
| 0 | -.005 | .0533 |
| 1 | .0324 | .0534 |
| 2 | .1384 | .0535 |
| 3 | .0279 | .0535 |
| 4 | -.019 | .0536 |
| 5 | -.043 | .0537 |
| 6 | -.042 | .0538 |
| 7 | -.064 | .0538 |
| 8 | -.046 | .0539 |
| 9 | -.037 | .0540 |
| 10 | -.054 | .0541 |
| 11 | -.055 | .0542 |
| 12 | -.071 | .0542 |
| 13 | -.059 | .0543 |
| 14 | -.043 | .0544 |
| 15 | -.060 | .0545 |

**Figure 10-9. (a) Cross-correlation and (b) time series plot between feeding Laughing Gulls and acoustically detected prey within a single survey, 6-10 September 2013.**

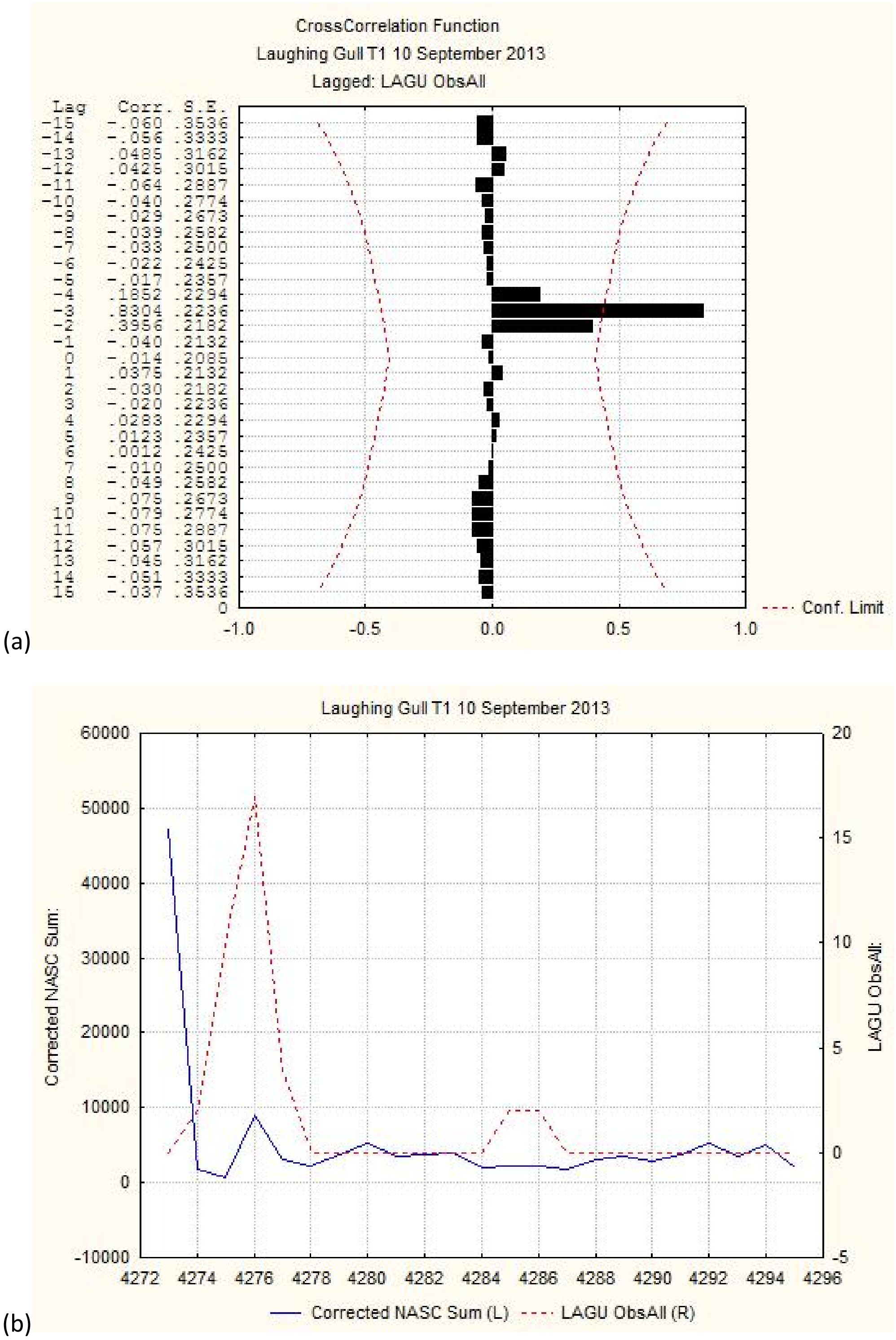


**Figure 10-10. (a) Cross-correlation and (b) time series plot between Laughing Gulls and acoustically detected prey along Transect 1, 10 September 2013.**

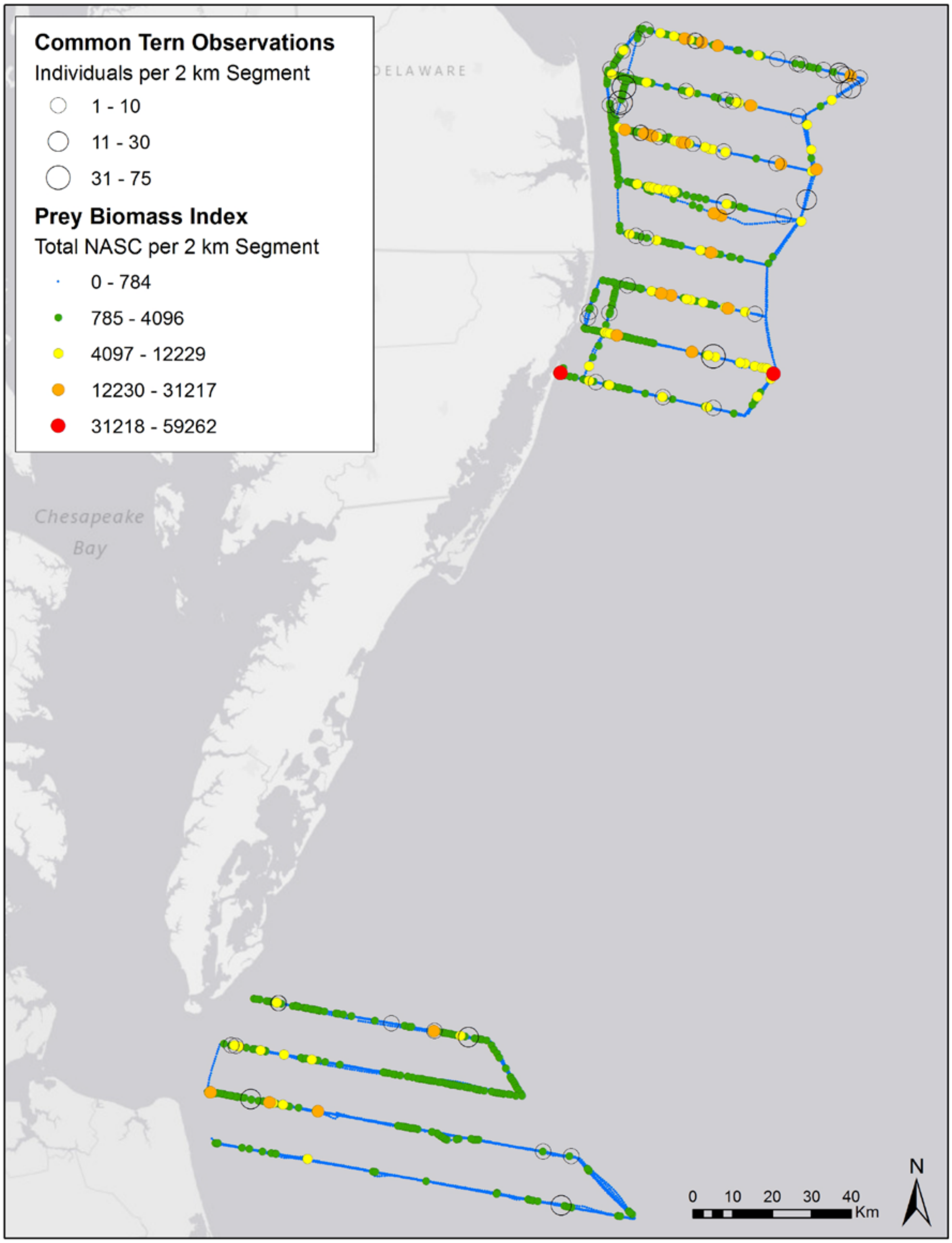


**Figure 10-11. Feeding Common Tern observations and prey biomass index per 2 km transect segment for data collected during 8 surveys between April 25 and September 30 (2012-2014).** Observation counts and biomass index values are classified by natural breaks.

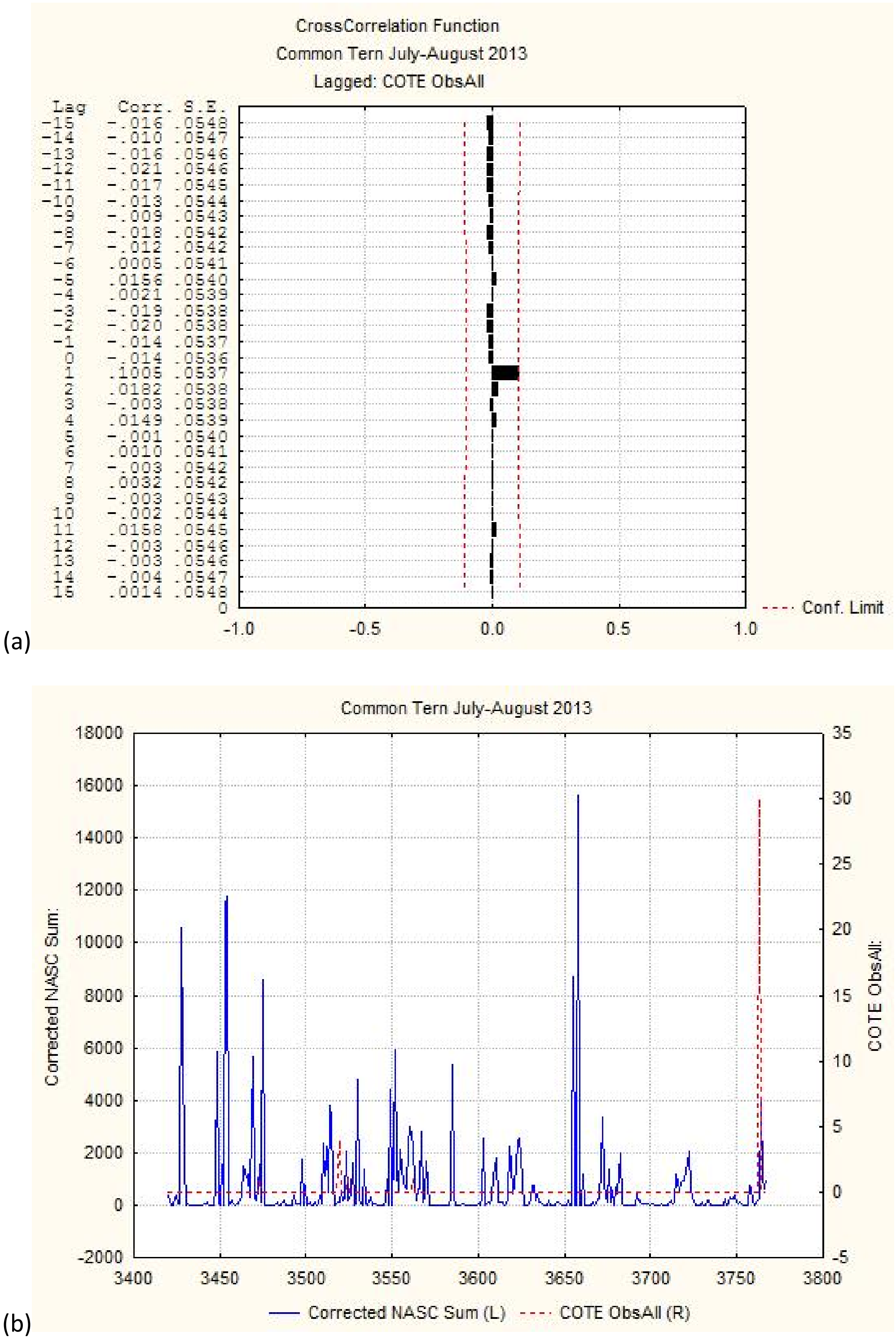


**Figure 10-12. (a) Cross-correlation and (b) time series plot between feeding Common Terns and acoustically detected prey within a single survey, 30 July – 2 August 2013.**

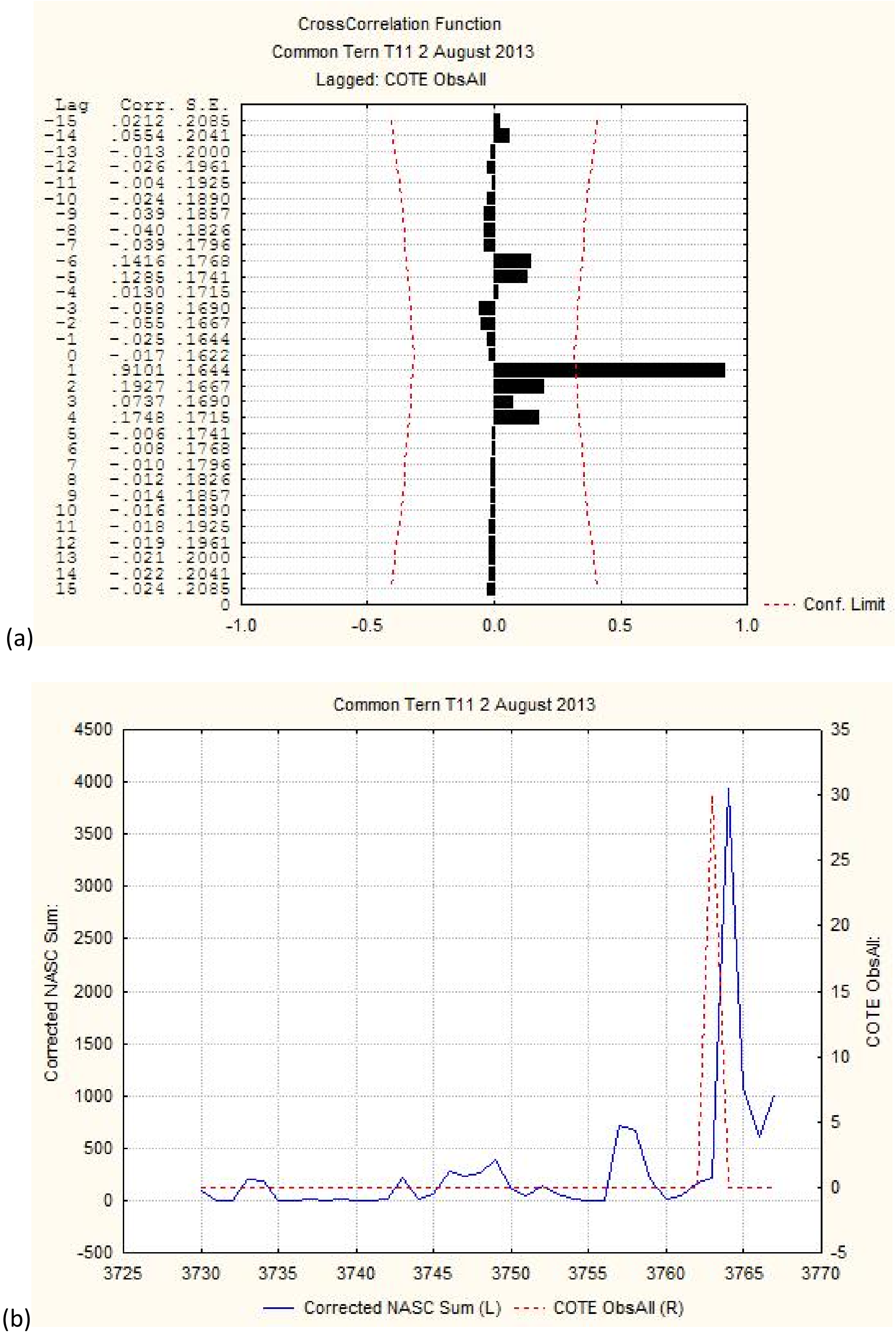


**Figure 10-13. (a) Cross-correlation and (b) time series plot between Common Terns and acoustically detected prey along Transect 11, 2 August 2013.**

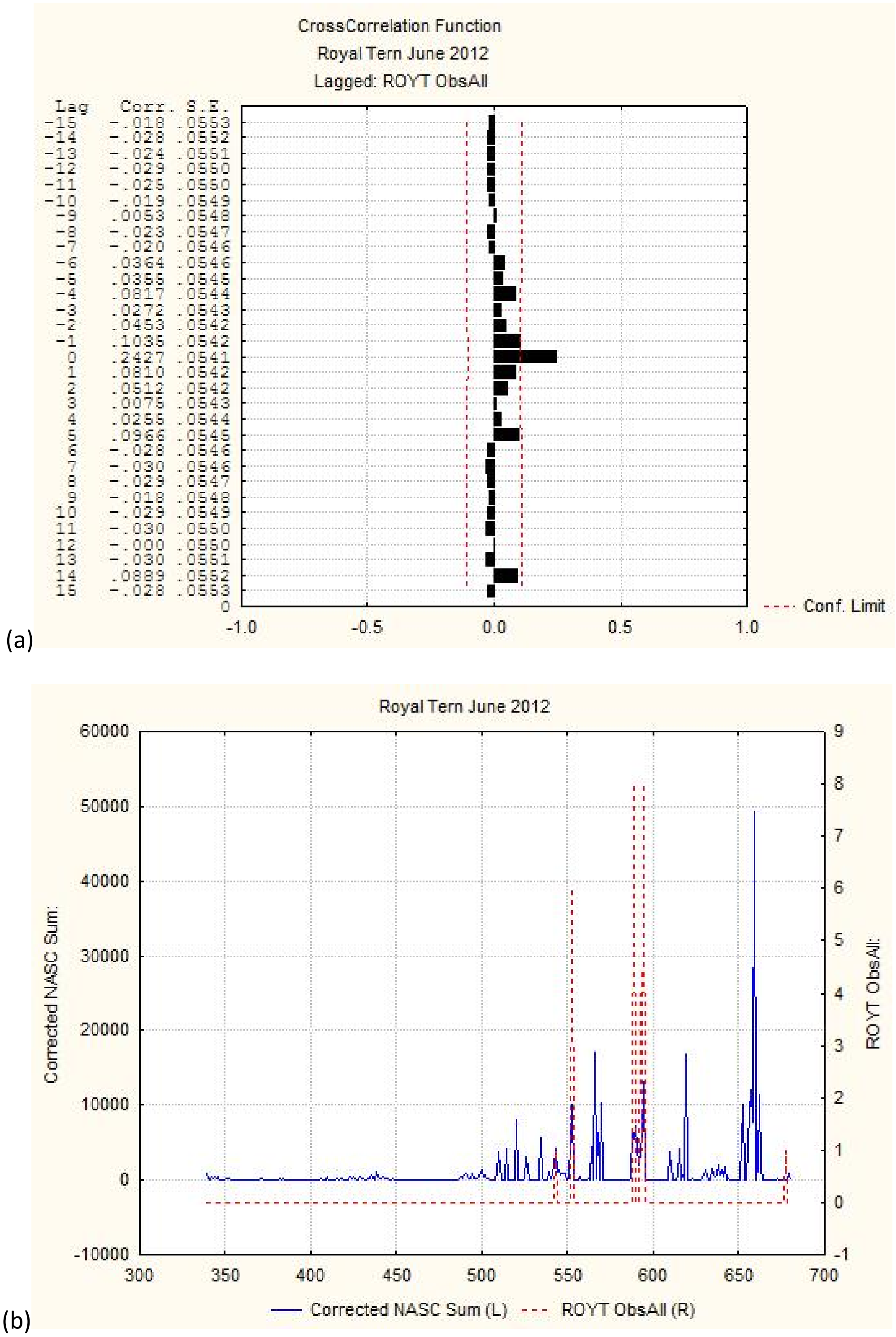
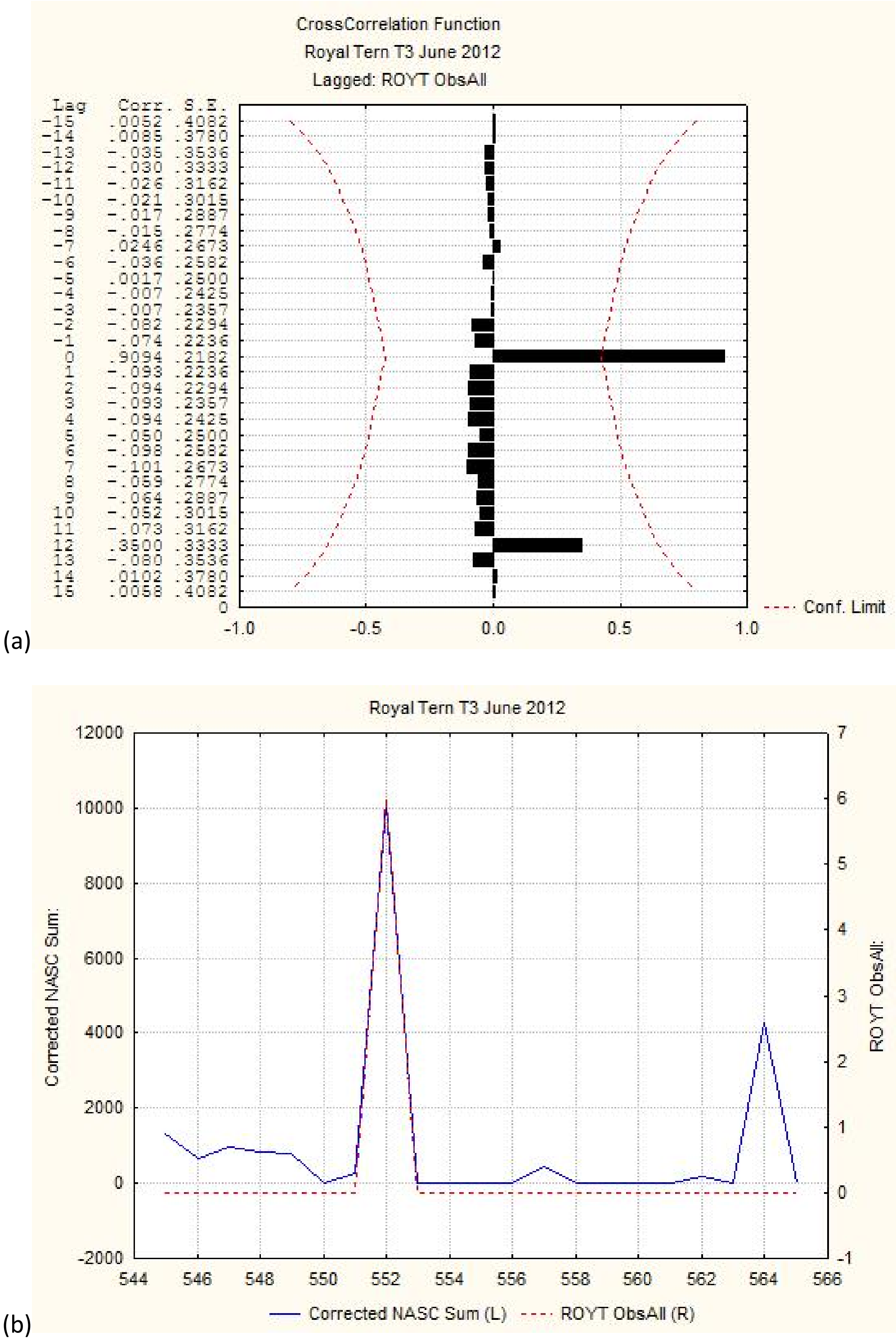


**Figure 10-14. (a) Cross-correlation and (b) time series plot between feeding Royal Terns and acoustically detected prey within a single survey, 18-21 June 2012.**



| Lag | Corr. | S.E. |
|---|---|---|
| -15 | .0052 | .4082 |
| -14 | .0085 | .3780 |
| -13 | -.035 | .3536 |
| -12 | -.030 | .3333 |
| -11 | -.026 | .3162 |
| -10 | -.021 | .3015 |
| -9 | -.017 | .2887 |
| -8 | -.015 | .2774 |
| -7 | .0246 | .2673 |
| -6 | -.036 | .2582 |
| -5 | .0017 | .2500 |
| -4 | -.007 | .2425 |
| -3 | -.007 | .2357 |
| -2 | -.082 | .2294 |
| -1 | -.074 | .2236 |
| 0 | .9094 | .2182 |
| 1 | -.093 | .2236 |
| 2 | -.094 | .2294 |
| 3 | -.093 | .2357 |
| 4 | -.094 | .2425 |
| 5 | -.050 | .2500 |
| 6 | -.098 | .2582 |
| 7 | -.101 | .2673 |
| 8 | -.059 | .2774 |
| 9 | -.064 | .2887 |
| 10 | -.052 | .3015 |
| 11 | -.073 | .3162 |
| 12 | .3500 | .3333 |
| 13 | -.080 | .3536 |
| 14 | .0102 | .3780 |
| 15 | .0058 | .4082 |

**Figure 10-15. (a) Cross-correlation and (b) time series plot between Royal Terns and acoustically detected prey along Transect 3, 20 June 2012.**

**Table 10-1. Spearman rank correlation coefficients ($r_s$) of association between birds and acoustically detected prey in 5,499 two km transect segments, 2012-2014.**

| Species | All Birds | | Feeding Birds | |
|---|---|---|---|---|
| Northern Gannet | $r_s=0.05$ | $p = 0.000021$ | $r^s=0.09$ | $P < 10^{-6}$ |
| Laughing Gull | $r_s=0.19$ | $p < 10^{-6}$ | $r^s = 0.09$ | $P < 10^{-6}$ |
| Common Loon | ns | ns | ns | |
| Red-Throated Loon | ns | ns | ns | |
| Dovekie | ns | ns | ns | |
| Wilson's Storm-Petrel | ns | ns | ns | |
| Common Tern | $r_s = 0.13$ | $p < 10^{-6}$ | $r^s = 0.04$ | $P = 0.0008$ |
| Royal Tern | $r_s= 0.15$ | $p < 10^{-6}$ | $r^s = 0.09$ | $P < 10^{-6}$ |

**Table 10-2. Spearman rank correlation among mean Northern Gannet abundance, mean prey abundance (NASC) and cross-correlation among birds and prey at the scale of individual transects.** Significant p values are highlighted in bold text.

| | N | Spearman R | p |
|---|---|---|---|
| Mean Abundance x Maximum cross-correlation | 107 | 0.47 | **$< 10^{-6}$** |
| Mean abundance x NASC | 107 | 0.24 | **0.013** |
| Maximum correlation x NASC | 107 | 0.168 | 0.083 |

**Table 10-3. Spearman rank correlation among mean Laughing Gull abundance, mean prey abundance (NASC) and cross-correlation among birds and prey at the scale of individual transects.** Significant p values are highlighted in bold text.

| | N | Spearman R | p |
|---|---|---|---|
| Mean abundance x Maximum cross-correlation | 129 | 0.79 | **$< 10^{-6}$** |
| Mean abundance x NASC | 129 | 0.33 | **0.00015** |
| Maximum cross-correlation x NASC | 129 | 0.24 | **0.0063** |

**Table 10-4. Including cross-correlation between bird and fish abundance yields a better model than using fish abundance alone**. GLMs calculated for data from individual transects. There were 107 2-km bins for Northern Gannets and 129 2-km bins for Laughing Gulls, as we restricted analysis to seasonally appropriate transects that were long enough to detect aggregations using time series methods. Probabilities are given under coefficients "NASC" and "Cross-correlation" and AIC values compare two models for each species.

| | NASC | Cross-Correlation | AIC |
|---|---|---|---|
| Northern Gannet | 0.0035 | 0.00017 | 276.3 |
| | 0.0027 | | 287.8 |
| Laughing Gull | 0.0047 | 2.54 x 10-6 | 114.2 |
| | 0.0087 | | 144.4 |

**Table 10-5. Number of surveys on which significant cross-correlation was found between Northern Gannets and acoustically detected prey, by transect.** This seems to not be a useful way to distinguish among transects, so was not attempted for other species.

| Transect | Number of surveys with significant cross-correlation |
|---|---|
| 1 | 3 |
| 2 | 3 |
| 3 | 5 |
| 4 | 6 |
| 5 | 6 |
| 6 | 5 |
| 7 | 5 |
| 8 | 5 |
| 9 | 7 |
| 10 | 5 |
| 11 | 5 |
| 12 | 4 |

# Chapter 11: A community distance sampling model to investigate the abundance and distribution of seabirds

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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*this chapter is under review at Methods in Ecology and Evolution

**Project webpage:** *www.briloon.org/mabs*

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# Chapter 11 Highlights

*Development of a multi-species modeling framework to predict seabird densities across the study area by season, incorporating environmental data and distance sampling*

### *Context*[1]

A broad geographic and temporal scale of analysis is required to assess exposure to wildlife from proposed development projects. Unlike several chapters in Part IV of this report, which utilize approaches for combining boat and digital aerial survey data, this chapter focuses on using data from a single, well understood survey method (distance sampling), to develop a multi-species model that includes infrequently observed species. Distance sampling generally requires a minimum sample size of 30-60 detections in order to produce valid results. With the data collected in the shipboard surveys, it was common to have fewer than 30 observations of a single species.

In order to estimate the abundance of those species with smaller sample sizes, project collaborators developed a community distance sampling (CDS) model. The technical components of the model development are provided in this chapter, and the model is illustrated on seabird data from a single survey (April 2012). This approach accounts for imperfect detection based on distance sampling, allows incorporation of rarely observed species through a hierarchical modeling structure, and estimates habitat relationships with abundance. Chapter 12 applies the model developed here to the full two years of data collection, and presents broader ecological findings.

### *Study goal/objectives*

Develop a model that makes efficient use of the boat survey data by allowing for estimation of abundance of all observed species, even those with a sparse number of observations.

### *Highlights*

- Successfully developed and implemented a hierarchical community distance sampling model
- Produced estimates of abundance and covariate relationships for all 14 species considered in the analysis of the April 2012 boat survey
- Created a flexible framework for analyzing all boat survey data
- Distance to shore was a significant predictor of abundance, consistent with other findings for the boat data (see Chapter 12).

### *Implications*

This model incorporates data from all species in to the analyses, which allows for making inferences about rare and infrequently observed species that would otherwise have to be discarded. By sharing information across species within a defined community, habitat relationships and abundance can be estimated for each species. In the context of conservation and management, rare and listed species are often of particular interest, and the ability to incorporate rare species into analyses provides important information about their abundance and distribution.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

Distance sampling is a common survey method in wildlife studies, because it allows accounting for imperfect detection, a known issue when trying to estimate abundance. The framework of distance sampling was employed during the shipboard surveys described in Chapters 7-8. Distance sampling generally requires a minimum sample size of 30-80 observations in order to produce valid results. With the data collected in the shipboard surveys, it was common to have fewer than 30 observations of a single species. In order to estimate the abundance of those species with smaller sample sizes, we developed a new multi-species model for distance sampling. In this chapter, we describe this model and provide an implementation example using data from the April 2012 survey. Non-technical readers may choose to skip to Chapter 12, in which all marine bird data from the boat survey were analyzed, for ecological insights provided using this model.

Specifically, in this chapter we develop a community distance sampling model that allows abundance to vary with environmental covariates. The model allows species-specific parameters, but these come from a common underlying distribution. This hierarchical model structure enables the incorporation of species with sparse data sets that would be otherwise discarded from analysis. We applied the model to marine bird data collected during the shipboard surveys in April 2012. The data set contained 14 species, out of which 10 yielded insufficient observations (< 30) for individual species models. The development of this new model allowed us to produce estimates of abundance and covariate relationships for all 14 species considered in the analysis. We found a strong negative association of community and species abundance with distance to shore. Sea surface temperature, and prey density in the top 3-5 m of the water column, both measured in situ, had weak effects on marine bird abundance. The model allowed us to make inference about ecology of the marine bird community, including rarely observed species, which is particularly important in a regulatory context.

## Introduction

The mid-Atlantic region is an extremely important area for a broad range of marine wildlife species throughout the year. This is due to relatively high levels of productivity, fed in part by nutrient inputs from Chesapeake Bay and Delaware Bay, as well as the region’s central location on the eastern edge of the continent and in the middle of an important migratory flyway (Chapter 1; Smith & Kemp 1995; Schofield et al. 2008). Seabirds species are globally more threatened (based on percent of bird species threatened) than comparable groups of birds, and the US ranks as a high priority area for conservation action based on species diversity, numbers of threatened species, and numbers endemic species present (Croxall et al. 2012; Sydeman et al. 2012). Current research indicates that globally monitored seabird populations have declined 70% over the last 60 years (Paleczny et al. 2015). Recently, the development of offshore wind energy facilities has raised additional concern about seabird conservation. Potential threats associated with the construction of offshore wind energy facilities include mortality through collisions, as well as displacement due to avoidance of the wind farms or the altered habitat they create (e.g., Garthe & Hüppop 2004; Petersen & Fox 2007). Minimizing the effects of future offshore wind energy development will require consideration of the distribution and abundance of seabirds across the area of interest, but studying these aspects of seabird ecology is challenging due to their large-scale movements, often clustered occurrence, and the vastness of the marine habitat in which they occur.

Evaluating potential impacts of environmental changes on this group is further complicated by the fact that seabirds comprise a diverse community of species, which are likely affected by environmental changes in different ways.

The shipboard surveys conducted in this study employed a method called distance sampling (Chapter 7). Distance sampling (Buckland 2001; Buckland et al. 2005) is a popular method to survey both terrestrial and marine wildlife species amenable to direct observation. In distance sampling, the probability of detecting an individual is assumed to decrease with increasing distance from the observer (see Figure 11A-1 for an example). This allows estimation of abundance and density while accounting for observation bias. The framework has been extended to accommodate the modeling of abundance at multiple survey sites as a function of site specific covariates (Hedley & Buckland 2004; Royle et al. 2004; Conn et al. 2012). This is also referred to as hierarchical distance sampling (HDS).

HDS provides a framework to investigate factors governing the distribution of individual species. Often, however, rare or elusive species will not yield sufficient observations to parameterize an individual model. The framework further does not allow for exploration of community-level effects of explanatory covariates. Community modeling provides a unified approach towards investigating community-level effects while maintaining the ability to model species-specific parameters (Dorazio & Royle 2005; Dorazio et al. 2006). In community models, species have individual parameters, but information is shared across species by assuming a common underlying distribution for these parameters, which in turn are governed by hyperparameters. The use of collective community data allows for inference about community and species-level patterns and processes, even for those species that are rare and elusive. This concept has been applied repeatedly in occupancy modeling (i.e., species-level detection/non-detection data, Dorazio & Royle 2005; Dorazio et al. 2006; Zipkin et al. 2009, 2010), but to our knowledge, no attempt has been made to combine community modeling with the framework of distance sampling.

Here, we develop a community distance sampling model that estimates both community-level and species-level parameters related to detection and abundance. We use the model to analyze seabird data collected from shipboard distance sampling surveys during April 2012 (analysis of all other surveys can be found in Chapter 12). Our analysis includes a community of 14 species, of which ten did not yield sufficient observations to be analyzed individually. By sharing information across species, the community distance sampling approach is able to estimate covariate effects for all 14 species of seabirds, providing important information on seabird abundance and distribution in areas actively explored for their wind energy potential. The method holds promise for many distance sampling applications to improve estimation of detection and abundance of species and communities.

## Methods

This section is based on developing methods in distance sampling and thus requires a fundamental understanding of distance sampling. For a review of distance sampling and hierarchical distance sampling please see Buckland 2001, Hedley & Buckland 2004, Buckland et al. 2005, Royle et al. 2004, and Conn et al. 2012.

### *Development of the community distance sampling model*

In distance sampling, the perpendicular distance of the object of interest to the observer is recorded along a transect or from a point. Detection on the transect line (or at the survey point) is assumed to be perfect and the detection probability of observation $m$ is defined by a declining function $f$ of its distance to the observer, $d_m$, for example, using a half-normal detection function

$$p_m = \exp\left(-\frac{d_m^2}{2\sigma^2}\right)$$

Here, $\sigma$ is the scale parameter of the half-normal function. In reality, observations are frequently grouped into $k = 1, 2, \ldots K$ distance categories. Let **b** be the $K$+1 break points of the $K$ distance categories, and $w$ be the width of the distance categories. Then, detection probability in $k$, $p_k$, is the integral of $f$(x) over $k$:

$$p_k = \frac{\int_{b_k}^{b_{k+1}} \exp\left(-\frac{x^2}{2\sigma^2}\right) dx}{w}$$

Individuals are assumed to be uniformly distributed in space, so that the probability of an individual occurring in distance band $k$, $\psi_k$, is the proportion of the sampled area covered by $k$ (note that in transect surveys with constant w, the area k is also constant across distance categories, but in point surveys, this area increases with increasing distance from the survey point). The vector of observations across all $K$ distance categories, **y**, is a multinomial random variable with size $n = \sum y$ and cell probabilities $\boldsymbol{\pi} = \mathbf{p}/\Sigma\mathbf{p}$.

We can link $n$ to the true abundance $N$ using the total detection probability $p.t = \sum_k p_k$:

$$n \sim Binomial(p.t, N)$$

When distance sampling surveys are carried out at $j = 1, 2, \ldots, J$ survey locations, observations are indexed by location:

$$n_j \sim Binomial\left(p.t_j, N_j\right).$$

Following Royle et al. (2004), we can assume $N_j$ to follow some probability mass function $f$ (e.g., Poisson or negative binomial) for $N$, and its expected value can be modeled as a function of covariates, **X**, e.g.,

$$N_j \sim f(\lambda_j)$$

$$\log\left(\lambda_j\right) = \alpha_0 + \boldsymbol{\alpha}'\mathbf{X}_{\boldsymbol{j}},$$

where $\alpha_0$ is the intercept and $\boldsymbol{\alpha}$ is a vector of coefficients associated with the covariates **X**. Analogously, detection parameters can be modeled as functions of site specific covariates (Marques & Buckland 2003; Oedekoven et al. 2013); for example, for the half-normal detection function:

$$\log(\sigma_j) = \beta_0 + \boldsymbol{\beta}'\mathbf{Y}_{\boldsymbol{j}},$$

Where $\beta_0$ is the intercept, and the vector **β** holds the coefficients associated with the detection covariates in **Y**.

To expand this approach to a community model for $i$ = 1, 2, … $S$ species, the parameters are further indexed by species, and we ascribe hyperdistributions to the resulting sets of parameters. For example, each species $i$ has an abundance intercept $\alpha_{0,i}$ such that:

$$\alpha_{0,i} \sim Normal(\mu_{\alpha 0}, \sigma_{\alpha 0}).$$

The hyperparameters of these distributions, here $\mu_{\alpha 0}$ and $\sigma_{\alpha 0}$, are estimated as part of the model and constitute the community parameters shared by all species.

### *Application: seabirds off the U.S. east coast*

Seabird distance sampling data were collected along 656.1 km of boat transects located off the coast of Delaware and Maryland (353.2 km), and Virginia (302.9 km) (Figure 11-1), sampled over the course of four days in April 2012. At any given survey, observations were restricted to one side of the boat (but side varied among surveys) and to the quadrant defined by the line of travel and a 90-degree angle to this line of travel. If seabirds occurred in flocks or clusters, angle and distance to the center of the cluster were estimated and the cluster size was noted. Each cluster was counted as a single record, and all records were individually georeferenced using dLOG (R.G. Ford Consulting, Inc.), a seabird and marine mammal observation program that also records the ship track. Details of the boat survey protocol are described in Chapter 7.

Sea surface temperature and salinity were recorded along these transects at 30-minute intervals using a YSI Pro30 handheld conductivity meter (YSI Inc.) with water drawn through the ship's salt water pump. Sea state and visibility were also recorded at these times. In addition, hydroacoustic data were recorded using a 120 kHz split beam echo sounder (Simrad EK60) with the transducers affixed to the hull of the boat. The acoustic backscatter measured by the echo sounder can be used as an index of prey abundance (e.g., Gurshin et al. 2009). Echo sounding data were binned into 500-m intervals for analysis; due to the position of the transducers on the boat, the first two meters of the water column were not recorded. Echo sounding data were processed using Echoview 5.3 (Myriax Software Pty Ltd., Hobart, Australia). Areas of interference were removed from the data prior to integration of the signal to estimate biomass.

### *Data preparation*

According to the observed distances, we set the maximum observation distance at 1000 m and binned observations into $K$ = 10 100-m distance categories. This binning smooths inaccuracies in distance estimation and reduces effects of movement of birds in response to observers.

Estimating abundance as a function of covariates from distance sampling requires spatial replication of surveys (Royle et al. 2004; Conn et al. 2012). To define these spatial replicates (or survey sites), we divided the 4 ship transects into 73 segments, using points at which environmental covariates were measured in situ as cut points (Figure 11-1). The resulting segments varied in length from 1.1 to 20.5 km (mean: 8.99 km, SD: 2.51 km). This relatively coarse spatial resolution was necessary to ensure that there were in-situ measurements of the covariates of interest for all segments. We accounted for these differences by using segment length as an offset in the abundance component of the model.

The survey yielded 681 records (i.e., clusters of 1 or more individuals) of 14 seabird species (defined as species that forage at sea). We excluded records with ambiguous species identification (6 observations) and observations without angle and/or distance recordings (43 observations). This may cause some negative bias in abundance estimates, but in the present case, this bias is likely negligible, due to the low number of excluded records. Records with these missing pieces of information appeared to come from across the range of observed species and group sizes, so we expect that the removal of these observations leads to equivalent levels of bias in abundance for all species, and does not bias estimates of group size or of covariate relationships. We discuss means of addressing missing information by model extension in the discussion. The final data set contained 632 records of 14 species (Table 11A-1 in Appendix 11A). As a rule of thumb, estimating abundance from distance sampling data requires at least 60 to 80 observations (Buckland et al. 1993). We applied a more liberal criterion of 30 observations to determine whether or not a HDS model could be applied to data of an individual species; only 4 of the species in this data set had >30 observation. We plotted the number of detections per distance category for all species, both separately and combined, to investigate whether data conformed to the distance sampling assumption of decreasing detection with distance from the observer. We found no indication for violation of this assumption (Appendix 11A).

### *Covariates*

We considered in situ collected temperature (*TEMP*, °C) and prey biomass density (*PD*) derived from echo sounding data, as well as distance to shore (*DTS*, km) as covariates on abundance. To define segment level values of *TEMP* and *DTS*, we took the mean of measurements from the start and end point of each segment. If a segment only had one associated measure (i.e., stretches leading up to the first or following the last measuring point), we used those single measurements as segment values. For *PD*, we averaged all measurements taken within a segment. Foraging depths for species in our dataset are likely to vary greatly with species, water clarity, water depth, and other factors. However, the majority of species in our dataset are visual hunters, and are likely responding to foraging cues from the top several meters of the water column. Therefore, we used prey density in the first 3 to 5 m of the water column (the first 2 m are missed by echo sounding devices). *TEMP* is considered an inverse proxy for prey availability (Hunt et al. 1981; Pinaud & Weimerskirch 2002), whereas echo sounding data gives a direct index of prey availability (e.g., Wiebe et al. 1990; Demer & Hewitt 1995).

As potential covariates on the detection parameter $\sigma$ we considered sea state (Beaufort values recorded in the survey from 1 = light air/water ripples, to 4 = moderate breeze/small waves and fairly frequent white caps; *BEAU*), visibility (categories from 1 = 300-500 m to 5 = 8000 m plus; *VIS*), and bird behavior (*BEHAV*; see below). For sea state and visibility, we applied the same procedure to obtain segment level values of the observation covariates as described above. For visibility, four segments had a value of 4, and all remaining segments had values of either 1 or 5. Therefore, we transformed visibility into a binary covariate of 'poor' = 1 or 'good' = 4 or 5.

It is conceivable that detectability of birds is reduced for observers on the bridge, as compared to the upper deck. The observation platform was recorded with each observation and changed within segments. Exploratory data analysis showed, however, that close to all observations at sea state 4 were made from the bridge, and almost all observations at lower sea states were made from the deck, suggesting that by accounting for sea state we implicitly account for observation platform. To account for the confounding of sea state and observation platform, we included sea state as a categorical covariate. Finally, because observers changed within segments, it is not straightforward to include an observer effect on detection. All observers were experienced in conducting seabird surveys and used rangefinders for distance estimation. Therefore, we believe that assuming relatively homogeneous skills across observers is reasonable.

Finally, birds flying might be more easily detected than birds on the water. To account for this source of variation in detection, we categorized bird behavior for each observation into “water adjacent” (diving, feeding, loafing, sitting) and “in the air” (flying, milling, following, plunge diving).

### *Parameterization of the community distance sampling model for the seabird dataset*

To fit the community distance sampling model to the seabird data set, we used a negative binomial distribution (with mean $\lambda_{ij}$ and overdispersion parameter $r$) for abundance and included all abundance covariates in the predictor, because these describe the ecological process we are interested in. We included a random species-specific intercept and random species-specific coefficients for these covariates in the abundance component:

$$N_{ij} \sim Negative\ Binomial(\lambda_{ij}, r)$$

$$\log(\lambda_{ij}) = \alpha_{0,i} + \beta 1_i TEMP_j + \beta 2_i PD_j + \beta 3_i DTS_j$$

To limit the number of parameters in the model due to sparseness in the data, we only included one detection covariate at a time. We assumed that the detectability of the different species is influenced in a similar way by these observation covariates and therefore estimated fixed coefficients $\beta$ for all species. Differences in detectability among species were accounted for by a random species specific intercept:

$$\log(\sigma_{ij}) = \beta_{0,i} + \beta Y_j,$$

where $Y$ is either *VIS*, *BEAU* or *BEHAV* (note that all covariates are categorical, so that a separate $\beta$ is estimated for each category minus the reference category, which is absorbed by the intercept). To accommodate *BEHAV*, which is an observation-level (as opposed to environmental, segment-level) covariate, we estimated abundance for the two behavioral categories $b$ using behavior-specific intercepts, $\alpha_{0,i,b}$, in the abundance model. This analysis also allows insight into which species are found near the water surface and which are more likely to be encountered in flight. We looked at effect strengths (Royle & Dorazio 2008) to choose among detection covariates: if the posterior distribution of a given $\beta$ strongly overlapped 0, we concluded that it was not an important covariate. We based inference on seabird abundance and distribution on a final model that contains all abundance covariates and those covariates on detection deemed influential.

### *Accounting for clusters of birds*

Seabirds are often observed in clusters. In this case, individuals are not observed independently, and clusters should be used as the unit of observation, so that $N_{ij}$ is the estimated number of clusters of species *i* at site *j*. To get an estimate of total abundance, we augmented the above described community distance sampling model with a component describing cluster size of observation *m*, $C_m$, to be a zero-truncated negative binomial variable, with a mean and dispersion parameter shared by all species:

$$C_m \sim zt\ Negative\ Binomial(\mu_C, \rho)$$

Note that $C_m$ is partially observed, i.e., known for observed clusters and unknown for $N_{ij} - n_{ij}$ unobserved clusters. Although it might seem biologically more appropriate to have a species-specific mean cluster size, 74% of all observations were of single individuals; 95% of all observations were of 4 or less individuals. We therefore decided against the additional complexity of a species-specific cluster size model. Because cluster size was predominantly small, we also refrained from adding it into the observation model as a covariate, but see the discussion for suggestions of how to include cluster size as a detection covariate. We calculated total abundance for a species at a site as the sum of all clusters – observed and estimated - for that species at that site, and total abundance in the survey area by summing over all (observed and estimated) clusters across all sites. The survey area is equivalent to a 1000-m strip along the combined boat transects.

### *Model fit*

We tested model fit using Bayesian p-values (Gelman et al. 1996). These values are obtained by calculating some fit statistic (e.g., a residual) that depends on the model parameters and the observed data, determining the same fit statistic for a new set of data generated from the model under consideration, and then calculating the portion of time the residuals from the newly generated data are larger (or smaller) than those of the original data. If the model fits the data appropriately, the resulting Bayesian p-value will be close to 0.5. We used Freeman-Tukey residuals, *R*, of the general form

$$R(\mathbf{y}, \boldsymbol{\theta}) = \sum \left(\sqrt{y} - \sqrt{E(y)}\right)^2,$$

where $\mathbf{y}$ is a collection of data, $\boldsymbol{\theta}$ are the parameters of the model describing $\boldsymbol{y}$ and E($\boldsymbol{y}$) is the expected value of $\boldsymbol{y}$.

Hierarchical models consist of several components, each of which can be evaluated for model fit. We calculated *R* and associated Bayesian p-values for the species and site-specific abundances, $N_{ij}$, to assess fit of the abundance component; for the observations **y** to assess fit of the observation component; and cluster size **C** to assess fit of the cluster size component. Note that all $N_{ij}$ are latent and subject to the specific assumptions of the distribution they are simulated from. It may therefore be more appropriate to use number of individuals observed at each site (generated from the model as $N_{ij} * p.t_j$) to evaluate fit of the abundance model based on observed data. This, however, confounds the detection and abundance models. In the present case, using $N_{ij}$ directly allowed us to determine that a Negative Binomial distribution provided a better fit that than a Poisson distribution. Formulas for residuals and associated Bayesian p-values are listed in Appendix 11C.

### *Implementation*

We implemented the community distance sampling model in a Bayesian framework using the software JAGS (Plummer 2003) accessed through R version 2.15.2 (R Core Team 2014). We ran three parallel Markov chains started at different initial values with a burn-in of 1,000 iterations and 50,000 post burn-in iterations. Because of the large number of parameters to be monitored, we thinned chains by 20 to reduce the size of the model output. We tested for chain convergence using the Gelman-Rubin statistic (Gelman et al. 2004). This statistic is a measure of among-chain versus between chain variance, and values < 1.1 indicate convergence. We report results as posterior means and standard deviations, as well as 2.5 and 97.5 percentiles, which represent the Bayesian equivalent to 95% Confidence Intervals (95BCI). We consider covariate effects as strong/significant if their 95% BCI do not overlap 0. Posterior distributions of total abundance estimates across all sites for the less abundant species tended to be right-skewed. Therefore, we provide the mode in addition to the mean in our summary statistic for species level abundances.

## Results

The community distance sampling model provided estimates of abundance of 14 seabird species, and identified covariates influencing their detectability and distribution. Neither visibility ($\beta$ = 0.015 ± 0.075, 95BCI: -0.136, 0.161) nor behavior ($\beta$ = 0.089 ± 0.068, 95BCI: -0.043, 0.222) had a strong effect on detectability (Table 11B-1 and Table 11B-5 in Appendix 11B), but sea states 3 and 4 had a strong negative effect on detectability of seabirds, relative to sea state 1 (Figure 11-3, panel B, Table 11B-3 in Appendix 11B). Therefore, we considered the model with *TEMP*, *DTS* and *PD* as covariates on abundance, a species-specific intercept for the detection parameter $\sigma$, and sea state as detection covariate as our final model and present results from this model only. Detailed results of all models can be found in Appendix 11B. According to the Bayesian p-values, the community distance sampling model fit the seabird data appropriately (Appendix 11C).

Distance to shore had a strong negative effect on seabird abundance across the entire community, with a mean, $\mu_{\alpha 1}$, of -0.999 ± 0.252 (Table 11B-3 in Appendix 11B). The effect was significantly negative for all but five species; for Surf Scoters, Lesser Black-backed Gull, Laughing Gulls, Herring Gulls and Forster's Terns the effect was negative but credible intervals overlapped 0 (Figure 11-2, panel A). Although the mean effect of sea surface temperature on the seabird community was negligible at -0.001 ± 0.170 (Table 11B-3 in Appendix 11B), it had a significantly positive effect on two species (Royal Tern, Common Tern; Figure 11-2B in Appendix 11B). The community mean effect of prey density was similarly weak (-0.018 ± 0.159) and was non-significant for all species (Table 11B-3 in Appendix 11B, Figure 11-2, panel C).

The mean detection parameter across species, $\mu_{\beta 0}$, was 216.269 ± 21.104 m at sea state = 1 and declined to 159.415 ± 18.680 m at sea state = 4 (Figure 11-3, panel B). Among species, $\sigma$ at sea state = 1 varied between 183.943 and 271.566 m (Figure 11-3, panel A). Mean cluster size for all species was 1.927 ± 0.107 individuals; the overdispersion parameter for the negative binomial cluster size model was 0.145 ± 0.016. Total abundance across all survey sites was highest for Common Loons (mode = 1677) and lowest for Forster's Terns and Surf Scoters (mode = 2) (Table 11B-4 in Appendix 11B).

## Discussion

The framework of community modeling so far has been applied mostly within occupancy models (Zipkin et al. 2009, 2010). We developed a community distance sampling model and used it to estimate relationships between abundance and environmental covariates for 14 species of seabirds, out of which only four species had enough observations to be modeled independently. Sharing information across ecologically similar species allows us to draw inferences about rare species, which are often of particular conservation concern.

Although information is shared by modeling covariate relationships of all species as coming from a common underlying distribution, there is still flexibility for species to have distinct covariate relationships. The community mean effect of *TEMP* on abundance, for instance, was close to 0 (Table 11B-3), yet *TEMP* had a significantly positive effect on abundance for two species (Figure 11-2, panel B). Similarly, the detection parameter $\sigma$ showed some variation across species, and Northern Gannets had a significantly larger $\sigma$ than the community average (Figure 11-3, panel A). This can be explained by the species' large size and predominantly bright white plumage, making it easily visible across longer distances.

### *Seabird abundance and distribution*

Common Loon, Northern Gannet and Laughing Gull had the highest estimated abundances in the study area (here, meaning a 1000-m strip along the ship transects). Laughing Gulls are generally common in the region at this time of year (Burger 2015), as are Northern Gannets, which leave for breeding grounds in the north later in the spring (Mowbray 2002), and Common Loons, which winter in the study area and tend to leave between March and June (Evers et al. 2010). In contrast, Surf Scoters and Forster's Terns were extremely rare during the April survey. Surf Scoters winter off the east coast of the US, but by April have mostly begun migrating to breeding grounds further north (Savard et al. 1998). Forster's Terns, on the other hand, are present year round (McNicholl et al. 2001), but stay very close to the shore and are therefore rare in the offshore community surveyed in the present study.

Abundance of all species decreased with increasing distance to shore (Figure 11-2, panel A), which can be a limiting factor for foraging activities (Weimerskirch 2007; Fauchald 2009). In addition, we expect *DTS* to correlate positively with ocean depth. Bathymetry has repeatedly been shown to be an important predictor of seabird foraging activity and abundance (e.g., Freeman et al. 2010; Nur et al. 2011), and the negative effect of *DTS* may be the result of seabirds actually responding to bathymetry.

Seabirds are top predators, and we expected a positive relationship with *PD* and an inverse relationship with *TEMP* (Hunt et al. 1981; Pinaud & Weimerskirch 2002). Contrary to our expectation, we observed mostly very weak effects of *TEMP* and *PD* on the abundance of seabird species (Figure 11-2, panels B and C). Lower sea surface temperatures are generally associated with higher primary productivity. There are, however, several intermediate trophic levels between primary production and top marine predators like seabirds (Barnes & Hughes 1988), which can lead to spatio-temporal lags in the response of seabirds to changes in these covariates.

It is also conceivable that the in-situ measures of *PD* and *TEMP*, taken immediately under the boat, do not adequately represent the environmental conditions in the 1000-m strip sampled. At an average resolution of approximately 10 km (i.e., the average lengths of transect segments), our study corresponds to a coarse-scale evaluation of factors driving seabird abundance (Haury et al. 1978; Hunt & Schneider 1987). At increasing spatial scales, temporal lags in the response of seabirds to environmental characteristics become more pronounced (Wiens 1989), and in situ measures cannot accommodate such lags. This indicates a need to determine the adequate spatial and temporal scale at which explanatory covariates are measured. A comparison of model fit between remote sensed data of different spatio-temporal scales and of in situ data could provide interesting insight for optimized choice of covariates.

### *Modeling detection probability*

Neither visibility nor whether birds were observed on the water or in the air ("behavior model") had a significant impact on detection distances of seabirds. In order to maintain reasonable sample sizes, we felt we could not subdivide observations into more than two behavioral classes. This coarse grouping may blur detection effects associated with certain behaviors. In spite of these ambiguities, we believe that the conceptual set-up of the "behavior" model allows for some interesting ecological insight into the percentage of individuals in a population performing certain behaviors. The approach of estimating abundance separately for two behavioral categories circumvents the issue of unknown behavioral category of unobserved individuals/clusters, but likely performs poorly with an increasing number of categories (due to low sample size per category), and breaks down completely for continuous individual covariates. In these cases, a different approach is to treat individual covariates of unobserved clusters as missing data, and specify a parametric model to estimate missing covariate values (e.g., Conn et al. 2014), which is equivalent to how the present model deals with cluster size of unobserved seabirds (see Accounting for clusters of birds).

Diving behavior makes birds unavailable for detection while under water, and failure to take into account availability <1 will lead to negative bias in abundance estimates. Availability can be estimated separately, for example from intensive observation studies or telemetry studies that allow inference on animal behavior (e.g., Diefenbach et al. 2007; Conn et al. 2014), and can be incorporated into the estimator of abundance (Buckland 2001) so that Eq. 1 becomes

$$n_{ij} \sim Binomial\left(p.t_{ij}, N.a_{ij}\right).$$

Here, $N.a_{ij}$ is the number of individuals of species $i$ at site $j$ that are available for detection, $p.t_{ij}$ is the total detection probability, and

$$N_{ij} = \frac{N.a_{ij}}{p.a_{ij}},$$

where $p.a_{ij}$ is the probability of being available, which can be species and site-specific.

Uncertainty about these estimates could readily be incorporated by treating availability as a parameter, rather than fixing it, and formulating an informative prior based on available information.

We did not explore differences in observer skills because the observer changed within a transect but time of change was not noted. If information was collected about where/when during the transect conditions changed, the detection parameter $\sigma$ could be modeled as a weighted mean of observer-specific $\sigma$ for each transect, considering which proportion of the transect was covered by each observer.

Finally, we did not include cluster size of birds as an explanatory covariate in the detection model because the present data set contained predominantly small clusters. This situation is not necessarily representative: many studies report occasional observations of large aggregations of seabirds. In these situations, the effect of cluster size on detectability (e.g., Smith et al. 1995; Pearse et al. 2008) should be considered. Observed cluster size can easily be included as a covariate on the log-linear predictor of $\sigma$. Further, different parameterizations of the cluster size model itself, for example in the form of species-specific means or distributions allowing for more variability in counts, may be required to adequately describe the observed data (Zipkin et al. 2014).

### *Missing individual covariate values*

In the present case, we excluded observations with uncertain species identification and/or missing distance-to-transect information. Rather than excluding such incomplete observations, which can negatively bias abundance estimates, species identity and distance can be viewed as missing individual covariates, which can readily be accommodated in a Bayesian framework. The present model could be augmented with a species identification model as developed by Conn et al. (2013, 2014). Here, species identity is treated as a latent variable with a multinomial distribution. Knowledge about the species-specific identification probabilities (e.g., from double-observer surveys or experiments with known species identity) can be used to formulate informative priors on these multinomial cell probabilities (with vague priors, unidentified observations will be distributed among species according to their proportion in the identified observations). Missing distances are naturally sampled from the multinomial model (for the present case of discrete distance bins) specified for the observations $\mathbf{y}$ (see Development of the community distance sampling model) if it is reasonable to assume that the probability of not recording a distance occurs at random across distance bins. Otherwise, additional information about this process would be necessary.

Distance sampling is employed in the study of a variety of taxa, and often, data on multiple species are collected (Jathanna et al. 2003; Somershoe et al. 2006; Williams & Thomas 2007). The present approach allows such studies to investigate community ecology and distribution of many species from within a flexible and coherent modeling framework. In the context of conservation and management, rare and listed species are often of particular interest, and the ability to incorporate rare species into analyses provides important information about their abundance and distribution.

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## Figures

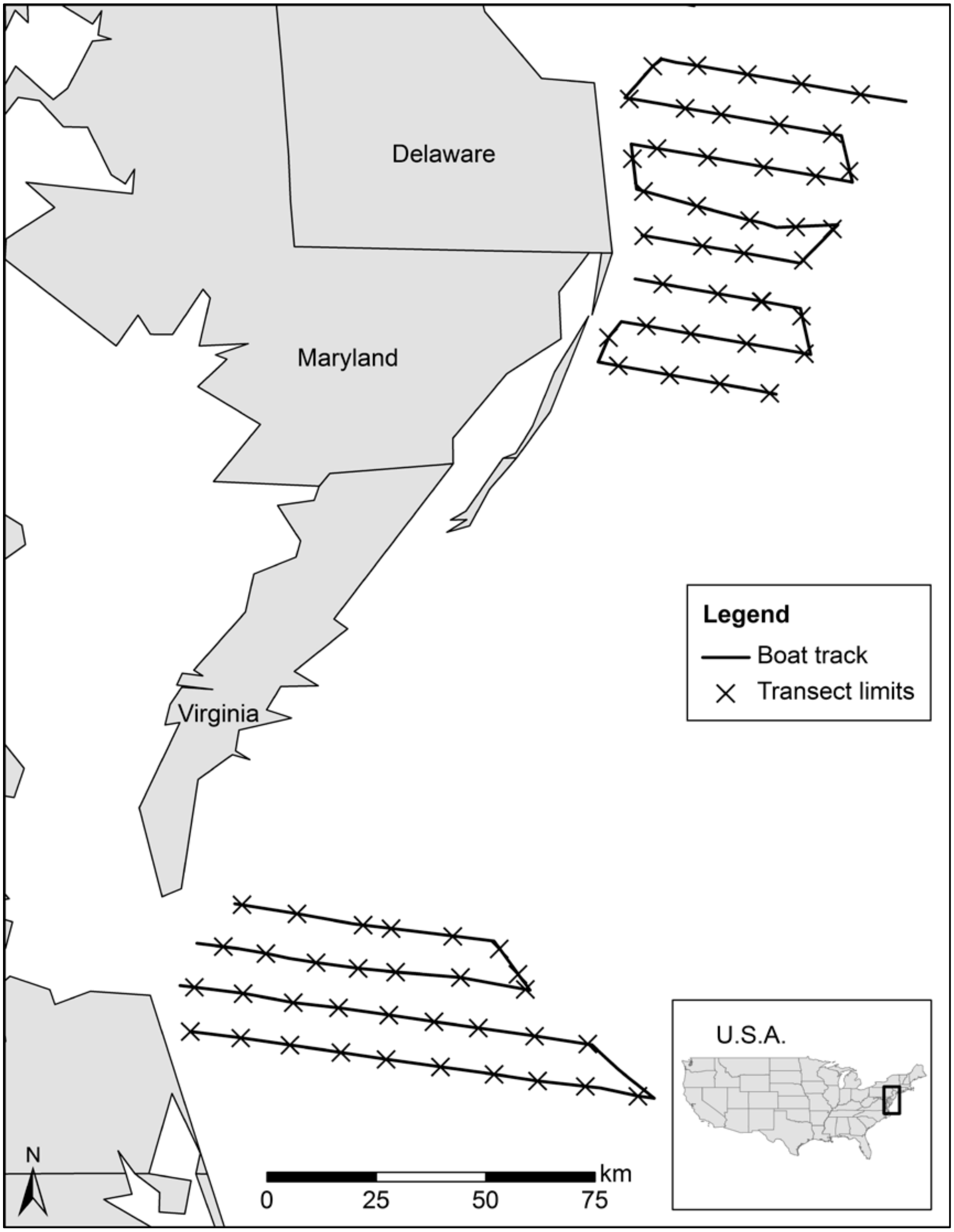

**Figure 11-1. Boat distance sampling transects implemented in April 2012 to survey seabirds, subdivided into 73 segments (x); inset map shows approximate location of the study area in the U.S.A.**

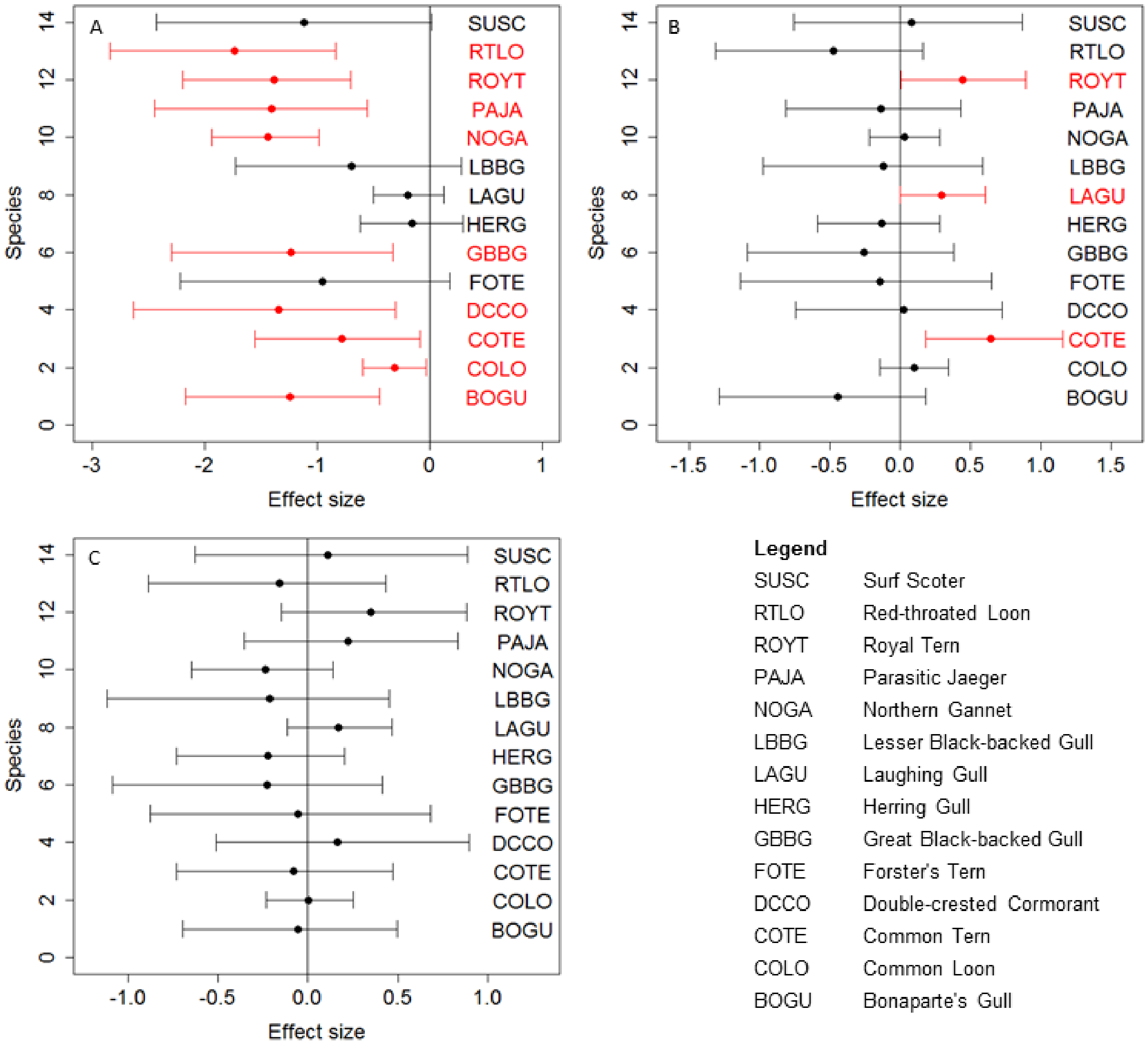

**Figure 11-2. Species specific effects of distance to shore (A), sea surface temperature (B) and prey density in the first 3-5 m of the water column (C) on seabird abundance, estimated from shipboard surveys off the coast of Delaware, Maryland and Virginia, using a community distance sampling model.** Red bars indicate species with effects significantly different from 0.

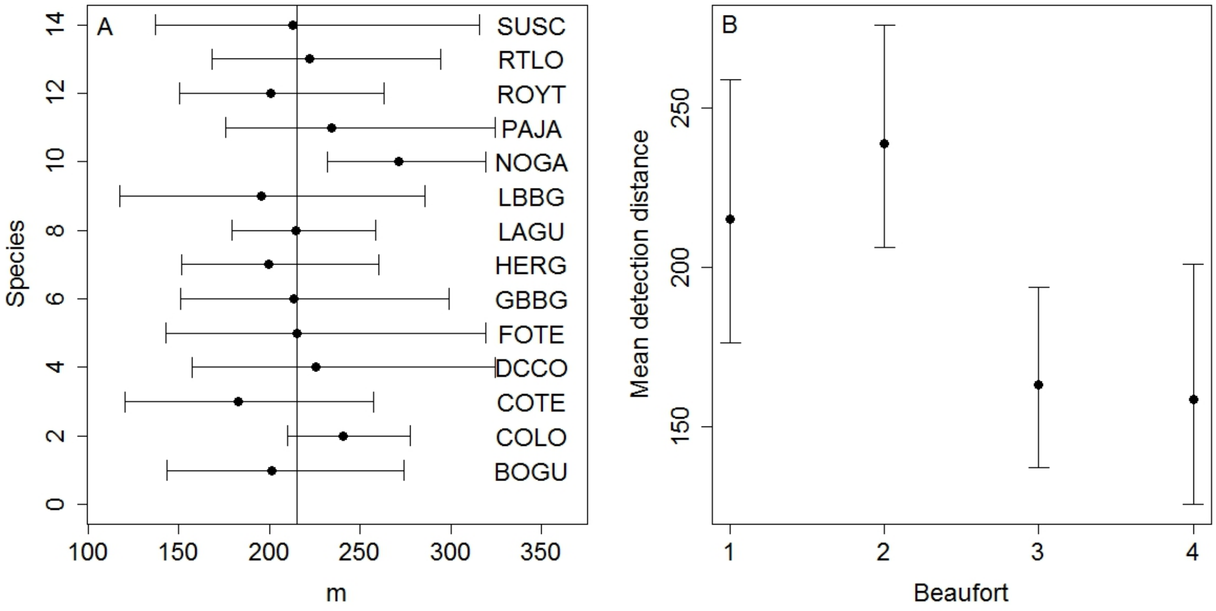

**Figure 11-3. Species specific detection parameters, *σ*, with vertical line representing community mean (A), and community mean *σ* (with posterior standard deviation) as a function of sea state (B).** For full species names, see Figure 2.

## Supplementary material

### Appendix 11A. Histograms of counts across distance classes

We plotted histograms of detection distance categories for data of all 14 seabird species (Table 11A-1), first combined (Figure 11A-1) and then for each species separately, to check for deviation from the prerequisite that detections decrease with distance (see main text for description of data set). Five species did not have the highest number of detections in the lowest distance category, but four of these species only had 1 – 3 records, so that these patterns likely arose by chance. The Red-throated Loon (Figure 11A-2) had 13 records spread across the first four distance categories without discernable pattern. This could be a spurious pattern due to low sample size, or indicate a wider shoulder in the detection function of this species. Since there was no strong indication of increasing detection with distance, evasive movements (a clear maximum at >1), or a multimodal detection function, we conclude that application of the half-normal model for detection was appropriate.

**Table 11A-1. Seabird species (common name, Latin name and four-letter code used in future tables and figures) and number of records (clusters) used in the present community distance sampling model; data collected during shipboard surveys off the coast of Virginia, Delaware and Maryland in April 2012.**

| Common name | Latin name | Code | Number of records |
|---|---|---|---|
| Bonaparte's Gull | *Larus philadelphia* | BOGU | 11 |
| Common Loon | *Gavia immer* | COLO | 282 |
| Common Tern | *Sterna hirundo* | COTE | 13 |
| Double-crested Cormorant | *Phalacrocorax auritus* | DCCO | 3 |
| Forster's Tern | *Sterna forsteri* | FOTE | 1 |
| Great Black-backed Gull | *Larus marinus* | GBBG | 6 |
| Herring Gull | *Larus argentatus* | HERG | 31 |
| Laughing Gull | *Larus atricilla* | LAGU | 91 |
| Lesser Black-backed Gull | *Larus fuscus* | LBBG | 3 |
| Northern Gannet | *Morus bassanus* | NOGA | 145 |
| Parasitic Jaeger | *Stercorarius parasiticus* | PAJA | 9 |
| Royal Tern | *Sterna maxima* | ROYT | 23 |
| Red-throated Loon | *Gavia stellata* | RTLO | 13 |
| Surf Scoter | *Melanitta perspicillata* | SUSC | 1 |

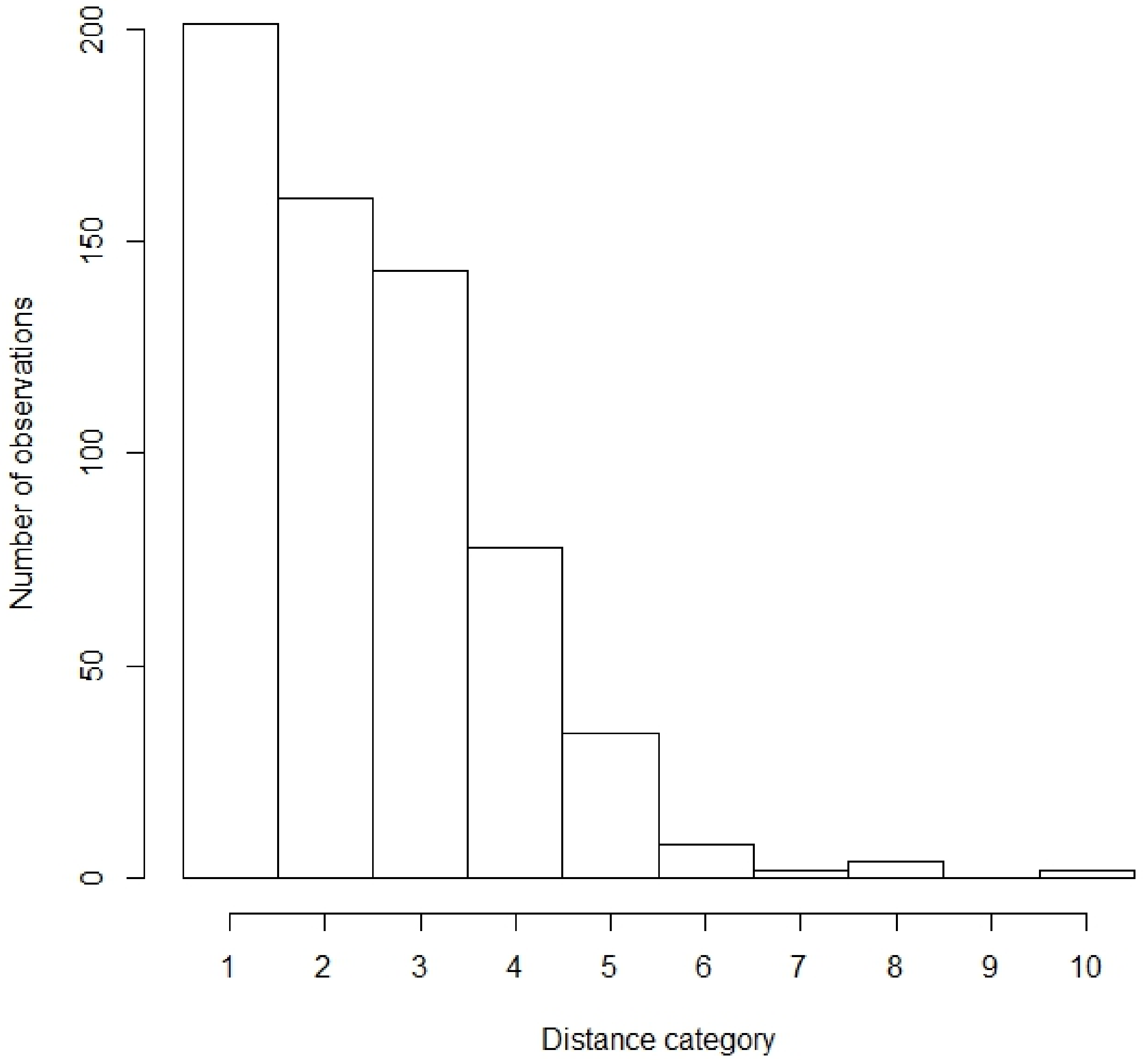

**Figure 11A-1. Histogram of observations in each distance class across all species and survey sites.** Each distance category is 100m in length.

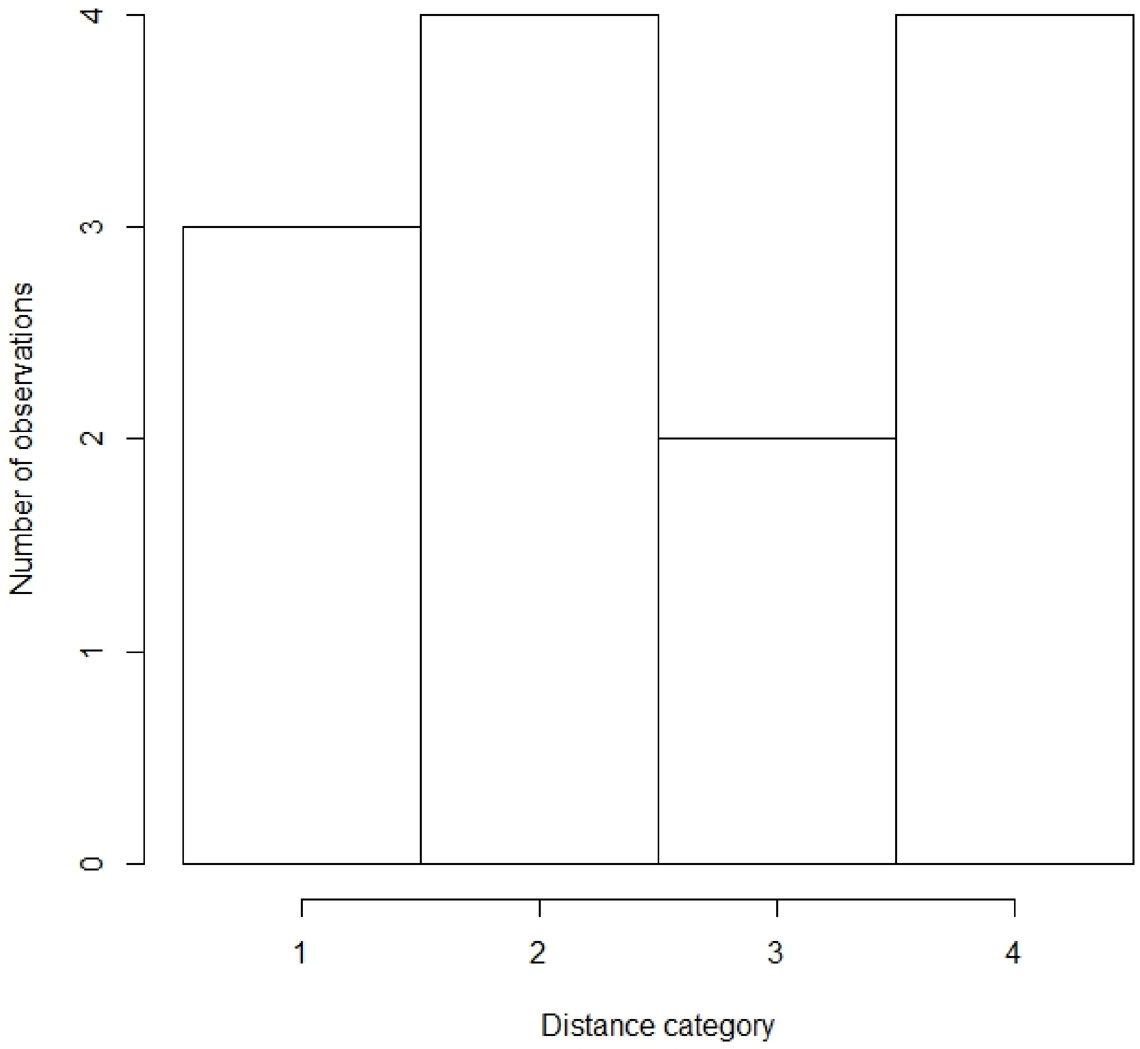

**Figure 11A-2. Histogram of observations per distance class for Red-throated Loons.** No observations were made in distance classes 5 – 10.

## Appendix 11B. Additional model results

We ran several community distance sampling models on the seabird data set (see main text for model and data set description), all of which included distance to shore (*DS*), sea surface temperature measured in situ (*TEMP*), and prey biomass density (*PD*) as covariates on abundance. The models differed in the detection covariates: visibility (*VIS*, binary, good or poor), Beaufort *(BEAU*, categorical, 1-4) or behavior (*BEHAV*, binary, on water or flying). Main results of the model including *BEAU* are presented in the main text. Detailed results of all models are summarized here.

### *Visibility as a covariate*

**Table 11B-1. Posterior summaries for community parameters (on log scale) from a community distance sampling model with visibility (1 = poor, 2 = good) as observation covariate, fitted to seabird observations collected off the shore of Maryland, Delaware and Virginia, USA. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles.** DTS = Distance to shore, TEMP = sea surface temperature, PD = prey biomass density.

| Component | Term | Mean | SD | 2.5% | 97.5% |
|---|---|---|---|---|---|
| Abundance | Intercept mean*; $\mu_{\alpha 0}$ | -3.135 | 0.574 | -4.309 | -2.028 |
| | Intercept SD*; $\sigma_{\alpha 0}$ | 1.978 | 0.469 | 1.279 | 3.099 |
| | DTS, mean*; $\mu_{\alpha 1}$ | -1.000 | 0.253 | -1.538 | -0.538 |
| | DTS, SD*; $\sigma_{\alpha 1}$ | 0.694 | 0.210 | 0.379 | 1.196 |
| | TEMP, mean*; $\mu_{\alpha 2}$ | -0.018 | 0.175 | -0.393 | 0.305 |
| | TEMP, SD*; $\sigma_{\alpha 2}$ | 0.474 | 0.162 | 0.233 | 0.860 |
| | PD, mean*; $\mu_{\alpha 3}$ | -0.006 | 0.158 | -0.336 | 0.291 |
| | PD, SD*; $\sigma_{\alpha 3}$ | 0.378 | 0.145 | 0.181 | 0.736 |
| | Negative binomial overdispersion; $r_N$ | 0.982 | 0.170 | 0.702 | 1.368 |
| Detection | Intercept mean*; $\mu_{\beta 0}$ | 5.319 | 0.164 | 5.001 | 5.647 |
| | Intercept SD*; $\sigma_{\beta 0}$ | 0.230 | 0.094 | 0.090 | 0.456 |
| | Visibility 2; $\beta_1$ | 0.015 | 0.075 | -0.136 | 0.161 |
| Cluster size | Mean, $\mu_C$ | 1.926 | 0.107 | 1.738 | 2.157 |
| | Negative binomial overdispersion; $\rho$ | 0.145 | 0.016 | 0.116 | 0.177 |

* Hyperparameters for random effects across species

**Table 11B-2. Posterior summaries for abundance of 14 seabird species within a 1000-m strip along 73 shipboard transect segments surveyed off the coast of Maryland, Delaware and Virginia, estimated using a community hierarchical distance sampling model with visibility as observation covariate.** See Table 1 for full species names and scientific names.

| Species | Mean | SD | Mode | 2.5% | 97.5% |
|---|---|---|---|---|---|
| BOGU | 91.365 | 33.394 | 77 | 43 | 171 |
| COLO | 1669.878 | 149.845 | 1631 | 1400 | 1987 |
| COTE | 141.090 | 47.837 | 119 | 72 | 257 |
| DCCO | 26.683 | 13.783 | 15 | 11 | 62 |
| FOTE | 10.085 | 10.364 | 1 | 1 | 38 |
| GBBG | 39.993 | 21.047 | 29 | 12 | 92 |
| HERG | 244.930 | 62.098 | 224 | 143 | 384 |
| LAGU | 573.472 | 88.748 | 547 | 418 | 764 |
| LBBG | 24.783 | 18.044 | 10 | 4 | 71 |
| NOGA | 1003.751 | 91.133 | 981 | 842 | 1198 |
| PAJA | 52.309 | 23.169 | 40 | 19 | 108 |
| ROYT | 174.250 | 44.400 | 163 | 103 | 275 |
| RTLO | 79.278 | 29.604 | 67 | 34 | 149 |
| SUSC | 10.231 | 10.508 | 1 | 1 | 39 |

### *Beaufort sea state as a covariate*

**Table 11B-3. Posterior summaries for community parameters (on log scale) from a community distance sampling model fitted to seabird observations collected off the shore of Maryland, Delaware and Virginia, USA. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles.** *DTS* = Distance to shore, *TEMP* = sea surface temperature, *PD* = prey biomass density, *BEAU* = Beaufort sea state.

| Component | Term | Mean | SD | 2.5% | 97.5% |
|---|---|---|---|---|---|
| Abundance | Intercept mean*; $\mu_{\alpha 0}$ | -3.122 | 0.572 | -4.271 | -2.036 |
| | Intercept SD*; $\sigma_{\alpha 0}$ | 1.980 | 0.474 | 1.286 | 3.119 |
| | *DTS*, mean*; $\mu_{\alpha 1}$ | -0.999 | 0.252 | -1.538 | -0.533 |
| | *DTS*, SD*; $\sigma_{\alpha 1}$ | 0.698 | 0.210 | 0.383 | 1.185 |
| | *TEMP*, mean*; $\mu_{\alpha 2}$ | -0.001 | 0.170 | -0.369 | 0.310 |
| | *TEMP*, SD*; $\sigma_{\alpha 2}$ | 0.462 | 0.159 | 0.228 | 0.844 |
| | *PD*, mean*; $\mu_{\alpha 3}$ | -0.018 | 0.159 | -0.349 | 0.286 |
| | *PD*, SD*; $\sigma_{\alpha 3}$ | 0.376 | 0.144 | 0.179 | 0.739 |
| | Negative binomial overdispersion; $r_N$ | 1.111 | 0.203 | 0.777 | 1.574 |
| Detection | Intercept mean*; $\mu_{\beta 0}$ | 5.372 | 0.098 | 5.172 | 5.558 |
| | Intercept SD*; $\sigma_{\beta 0}$ | 0.183 | 0.082 | 0.060 | 0.379 |
| | *BEAU* = 2; $\beta_1$ | 0.104 | 0.074 | -0.043 | 0.250 |
| | *BEAU* = 3; $\beta_2$ | -0.278 | 0.089 | -0.452 | -0.106 |
| | *BEAU* = 4; $\beta_3$ | -0.307 | 0.119 | -0.541 | -0.068 |
| Cluster size | Mean, $\mu_C$ | 1.927 | 0.107 | 1.739 | 2.155 |
| | Negative binomial overdispersion; $\rho$ | 0.145 | 0.016 | 0.116 | 0.177 |

* Hyperparameters for random effects across species

**Table 11B-4. Posterior summaries for abundance of 14 seabird species within a 1000-m strip along 73 shipboard transect segments surveyed off the coast of Maryland, Delaware and Virginia, estimated using a community hierarchical distance sampling model.** See Table A1 for full species names and scientific names.

| Species | Mean | SD | Mode | 2.5% | 97.5% |
|---|---|---|---|---|---|
| BOGU | 89.744 | 31.980 | 74 | 43 | 166 |
| COLO | 1739.703 | 155.662 | 1677 | 1460 | 2071 |
| COTE | 132.217 | 42.760 | 118 | 70 | 235.525 |
| DCCO | 28.214 | 14.704 | 17 | 11 | 66.525 |
| FOTE | 10.307 | 10.550 | 2 | 1 | 39 |
| GBBG | 40.968 | 21.026 | 29 | 12 | 92 |
| HERG | 229.144 | 57.698 | 225 | 135 | 357 |
| LAGU | 607.911 | 93.989 | 540 | 443 | 813 |
| LBBG | 24.517 | 17.211 | 11 | 4 | 68 |
| NOGA | 1039.370 | 96.975 | 1029 | 867 | 1244.525 |
| PAJA | 55.882 | 24.107 | 44 | 21 | 114 |
| ROYT | 183.932 | 46.341 | 168 | 109 | 286.525 |
| RTLO | 83.124 | 30.262 | 71 | 37 | 155 |
| SUSC | 10.461 | 10.455 | 2 | 1 | 40 |

## *Behavior as a covariate*

**Table 11B-5. Posterior summaries for community parameters (on log scale) from a community distance sampling model with behavior (on the water = 1, flying = 2) as observation covariate, fitted to seabird observations collected off the shore of Maryland, Delaware and Virginia, USA. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles.** DTS = Distance to shore, TEMP = sea surface temperature, PD = prey biomass density.

| Component | Term | Mean | SD | 2.5% | 97.5% |
|---|---|---|---|---|---|
| Abundance | Intercept mean, behavior = 1*; $\mu_{\alpha 0,1}$ | -4.439 | 0.636 | -5.781 | -3.251 |
| | Intercept SD, behavior = 1*; $\sigma_{\alpha 0,1}$ | 2.053 | 0.548 | 1.265 | 3.379 |
| | Intercept mean, behavior = 2*; $\mu_{\alpha 0,2}$ | -3.456 | 0.529 | -4.544 | -2.445 |
| | Intercept SD, behavior = 2*; $\sigma_{\alpha 0,2}$ | 1.792 | 0.433 | 1.144 | 2.821 |
| | DTS, mean*; $\mu_{\alpha 1}$ | -0.925 | 0.233 | -1.411 | -0.494 |
| | DTS, SD*; $\sigma_{\alpha 1}$ | 0.659 | 0.192 | 0.371 | 1.112 |
| | TEMP, mean*; $\mu_{\alpha 2}$ | -0.011 | 0.165 | -0.365 | 0.292 |
| | TEMP, SD*; $\sigma_{\alpha 2}$ | 0.456 | 0.152 | 0.227 | 0.816 |
| | PD, mean*; $\mu_{\alpha 3}$ | -0.020 | 0.156 | -0.344 | 0.275 |
| | PD, SD*; $\sigma_{\alpha 3}$ | 0.382 | 0.139 | 0.187 | 0.724 |
| | Negative binomial overdispersion; $r_N$ | 0.765 | 0.118 | 0.565 | 1.025 |
| Detection | Intercept mean*; $\mu_{\beta 0}$ | 5.265 | 0.111 | 5.033 | 5.471 |
| | Intercept SD*; $\sigma_{\beta 0}$ | 0.255 | 0.097 | 0.108 | 0.487 |
| | Behavior 2; $\beta_1$ | 0.089 | 0.068 | -0.043 | 0.222 |
| Cluster size | Mean, $\mu_C$ | 1.925 | 0.107 | 1.738 | 2.154 |
| | Negative binomial overdispersion; $\rho$ | 0.145 | 0.016 | 0.116 | 0.177 |

* Hyperparameters for random effects across species

**Table 11B-6. Posterior summaries for abundance of 14 seabird species within a 1000-m strip along 73 shipboard transect segments surveyed off the coast of Maryland, Delaware and Virginia, estimated using a community hierarchical distance sampling model with behavior (on the water or flying) as observation covariate.** See Table 11A-1 for full species names and scientific names.

| Species | Mean | SD | Mode | 2.5% | 97.5% |
|---|---|---|---|---|---|
| BOGU | 93.807 | 34.199 | 80 | 44 | 175 |
| COLO | 1670.765 | 150.078 | 1639 | 1401 | 1989 |
| COTE | 146.834 | 50.097 | 127 | 74 | 267 |
| DCCO | 29.537 | 15.110 | 19 | 12 | 68 |
| FOTE | 13.837 | 12.471 | 3 | 1 | 47 |
| GBBG | 42.777 | 22.378 | 32 | 13 | 98 |
| HERG | 246.100 | 62.362 | 228 | 144 | 386 |
| LAGU | 570.500 | 88.346 | 567 | 416 | 761 |
| LBBG | 28.785 | 19.987 | 15 | 5 | 80 |
| NOGA | 991.598 | 89.640 | 972 | 832 | 1183 |
| PAJA | 53.558 | 23.281 | 44 | 20 | 109 |
| ROYT | 172.406 | 43.738 | 160 | 102 | 271 |
| RTLO | 80.392 | 29.707 | 68 | 35 | 150 |
| SUSC | 14.050 | 12.708 | 3 | 1 | 48 |

**Table 11B-7. Percent of individuals estimated to be on the water, as opposed to flying, for 14 seabird species sampled along 73 shipboard transect segments off the coast of Maryland, Delaware and Virginia, estimated using a community hierarchical distance sampling model with behavior as observation covariate.** See Table 11A-1 for full species names and scientific names.

| Species | Mean | SD | 2.50% | 97.50% |
|---|---|---|---|---|
| BOGU | 0.295 | 0.137 | 0.078 | 0.597 |
| COLO | 0.773 | 0.044 | 0.679 | 0.850 |
| COTE | 0.164 | 0.102 | 0.027 | 0.414 |
| DCCO | 0.374 | 0.211 | 0.046 | 0.810 |
| FOTE | 0.281 | 0.232 | 0.006 | 0.817 |
| GBBG | 0.347 | 0.173 | 0.071 | 0.718 |
| HERG | 0.070 | 0.046 | 0.011 | 0.186 |
| LAGU | 0.092 | 0.036 | 0.037 | 0.176 |
| LBBG | 0.352 | 0.209 | 0.039 | 0.797 |
| NOGA | 0.310 | 0.064 | 0.195 | 0.445 |
| PAJA | 0.554 | 0.159 | 0.240 | 0.840 |
| ROYT | 0.322 | 0.111 | 0.133 | 0.559 |
| RTLO | 0.334 | 0.134 | 0.109 | 0.620 |
| SUSC | 0.286 | 0.233 | 0.006 | 0.822 |

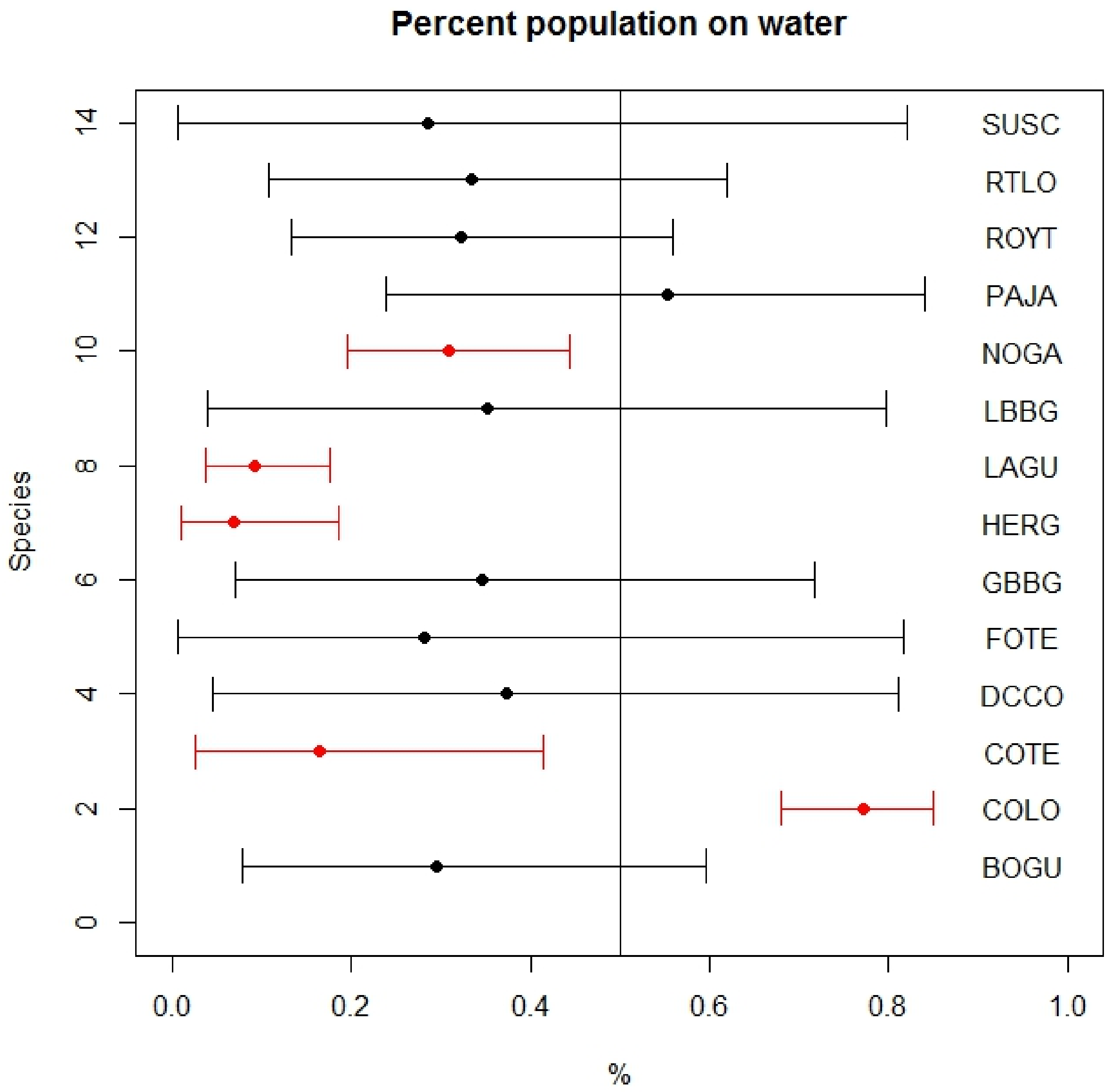

**Figure 11B-1. Percent of species population on water (as opposed to in the air), from behavioral model.** Red bars indicate a percentage on water significantly different than 50%.

## Appendix 11C. Model residuals and Bayesian p-values for the community distance sampling model

We assessed model fit using Bayesian p-values. For the calculation of the Bayesian p-value we used Freeman-Tukey residuals, *R*, of the general form

$$R(\mathbf{y},\boldsymbol{\theta}) = \sum \left(\sqrt{y} - \sqrt{E(y)}\right)^2,$$

where $\mathbf{y}$ is a collection of data, $\boldsymbol{\theta}$ are the parameters of the model describing $\boldsymbol{y}$ and E($\boldsymbol{y}$) is the expected value of $\boldsymbol{y}$. We calculated these residuals for the observed data and a new data set generated from the model under consideration. The Bayesian p-value is the percentage of time the residuals from the newly generated data are larger (or smaller) than those of the original data. We used this procedure to decide between a Poisson and a negative binomial distribution in the abundance component of the model (Table C1). For the community distance sampling model with the final set of covariates, we used this procedure to test fit of the three model components (see below; Table C2).

### *Abundance residuals*

Abundance *N* is estimated for each species *i* at each site *j*. We generated a new set of abundances from the model under consideration:

$$N.new_{ij} \sim NegBin(\lambda_{ij}, r)$$

$$\log(\lambda_{ij}) = \alpha_{0,i} + \beta 1_i DS + \beta 2_i TEMP + \beta 3_i PD,$$

where *DS* = distance to shore, *TEMP* = sea surface temperature measure in situ, and *PD* = prey biomass density derived from echo sounding.

We calculated residuals *R* (for $N_{ij}$) and *R.n* (for $N.new_{ij}$) at each iteration *t* of the Markov chain as

$$R = \sum_i \sum_j (\sqrt{N_{ij}} - \sqrt{\lambda_{ij}})^2$$

$$R.n = \sum_i \sum_j (\sqrt{N.new_{ij}} - \sqrt{\lambda_{ij}})^2.$$

We calculated the Bayesian p-value as

$$p = \sum_t R.n > R.$$

### *Observation model residuals*

In the Bayesian implementation of the model, we use a categorical distribution on the distance classes *k* of each individual observation *m*, rather than a multinomial distribution on the counts per distance class. We generated a new set of distances classes, *k.new*:

$$k.new_m \sim Categorical(\boldsymbol{\pi})$$

$$\boldsymbol{\pi} = \frac{\mathbf{p}}{\sum_k p_k},$$

where $p_k$ is defined by the half-normal detection function as described in the main text, and the detection parameter $\sigma$ is modeled as a function of species and sea state *BEAU*.

We calculated residuals *R* (for $k_m$) and *R.n* (for $k.new_m$) at each iteration *t* of the Markov chain as

$$R = \sum_m \left(1 - \sqrt{p_k}\right)^2$$

$$R.n = \sum_m \left(1 - \sqrt{p_{k.new}}\right)^2$$

We calculated the Bayesian p-value as

$$p = \sum_t R.n > R.$$

### *Cluster size model residuals*

Cluster size c is estimated for each observation m. We generated a new set of cluster sizes, *c.new*, as

$$c_m \sim zerotruncated\ NegBin(\gamma, \rho)$$

We calculated residuals *R* (for $c_m$) and *R.n* (for $c.new_m$) at each iteration *t* of the Markov chain as

$$R = \sum_m \left(\sqrt{c_m} - \sqrt{\gamma}\right)^2$$

$$R.n = \sum_m \left(\sqrt{c.new_m} - \sqrt{\gamma}\right)^2$$

We calculated the Bayesian p-value as

$$p = \sum_{t} R.n > R.$$

**Table 11C-1. Bayesian p-values for a community distance sampling model for seabirds without detection covariates using a Poisson and a negative binomial model to describe variation in abundance across sampling sites.** Values close to 0.5 indicate good model fit.

| Model component | p-value | |
|---|---|---|
| | Poisson | Negative binomial |
| Abundance | 0.013 | 0.379 |
| Detection | 0.574 | 0.568 |
| Cluster size | 0.587 | 0.579 |

**Table 11C-2. Bayesian p-values for the final community distance sampling model (see main text for model parameterization) for seabirds.** Values close to 0.5 indicate good model fit.

| Model component | p-value |
|---|---|
| Abundance | 0.462 |
| Detection | 0.613 |
| Cluster size | 0.582 |

# Chapter 12: Predicting the offshore distribution and abundance of marine birds from shipboard surveys, using a hierarchical community distance sampling model

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 12 Highlights

*Prediction of seabird densities across the study area by season, based on an incorporation of environmental data into a multi-species modeling approach*

### *Context*[1]

A broad geographic and temporal scale of analysis is required to assess exposure to wildlife from proposed development projects. Unlike several other chapters in Part IV of this report that utilize approaches for combining boat and digital aerial survey data, Chapter 12 focuses on using data from a single, well understood survey method to describe abundance patterns. Standardized boat-based surveys with distance estimation are a well-established method of obtaining density data for wildlife.

Project collaborators developed a community distance sampling (CDS) model for seabirds using data from the first boat survey (Chapter 11). Building on this novel multi-species approach, Chapter 12 analyzes data from 15 boat surveys and incorporates remotely-collected environmental covariate data into the hierarchical modeling structure. This approach accounts for imperfect detection to estimate "true" abundance, and predicts seabird distributions by season to help identify important habitat use areas and patterns.

### *Study goal/objectives*

Evaluate potential exposure of the marine bird community to offshore development by: 1) quantifying the detectability of 40 avian species to predict their seasonal abundance across the study area; and 2) identifying ecological drivers of distribution and abundance, both within and among species.

### *Highlights*

- Abundance and species composition varied across the study area, as well as by season.
- Distance to shore was generally the most common significant predictor of abundance.
- Estimated abundance was highest in the winter, and for most species was higher in the second (2013-14) than first (2012-13) winter of surveys. Species richness was also higher in the second winter.
- High species density and diversity also occurred in spring and fall, suggesting that migratory and overwintering species dominate the region's species composition.
- Although species abundance and richness was generally lower during the summer, members of some protected species were present during the summer, largely closer to the shore.

### *Implications*

Identifying areas more or less suitable for development involves prioritizing areas rich in abundant species, as well as important areas for species of concern (such as terns) that may be vulnerable even at low numbers.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

Proposed offshore wind energy development on the Atlantic Outer Continental Shelf has brought attention to the need for baseline studies of marine birds. We compiled line transect data from 15 shipboard surveys (June 2012 to April 2014), along with associated remotely-sensed habitat data, in the lower Mid-Atlantic Bight off the coast of Delaware, Maryland, and Virginia. We used observations from 40 marine bird species to inform a hierarchical community distance sampling model that estimated the seasonal detection and abundance of marine birds in the study area. We hypothesized that avian benthivores (bottom-feeders) respond more to static covariates that characterize seafloor variability, and that piscivores (fish-eaters) respond primarily to dynamic covariates that quantify surface productivity. Treating each season separately, we included six oceanographic parameters to estimate seabird abundance: three static (distance to shore, slope, sediment grain size), and three dynamic covariates (sea surface temperature, salinity, primary productivity). We compared the variation in species-specific and community-level responses to these habitat features, including for rare and protected species, and predicted the abundance for each species across the study area. Our hypothesis was partially supported by our results, but there was wide interannual, seasonal, and interspecies variation in habitat relationships. We found that abundance and diversity was highest for overwintering species. These results show the importance of quantifying detection and determining the ecological drivers of a community’s distribution and abundance, within and among species, for evaluating the potential exposure of marine birds to offshore development.

## Introduction

Proposed offshore energy development in the United States over the last decade has brought increased public attention to potential species-level impacts of anthropogenic activities on marine life (Caldow et al. 2015; Winiarski et al. 2014). We present a method of examining species- and community-level exposure of marine birds to potential development within wind energy areas (WEAs) in federal U.S. waters on the Atlantic Outer Continental Shelf. Identifying important habitat for marine communities of mammals, fish, and birds presents one of the most effective mitigation techniques for wind energy development's effects on wildlife: that is, avoiding, 'hotspots', defined as locations where high diversity and densities of sensitive species persist (Marques et al. 2014). Characterizing hotspots of seabird communities is important in assessing potential impacts from offshore development, particularly because as meso-predators, marine birds are useful indicators of environments that support high biodiversity (Lascelles et al. 2012).

The dynamic nature of pelagic marine communities is important to consider in siting offshore development, since marine predators locate prey in an environment characterized by exceptionally high spatial and temporal variability (Davoren et al. 2010; Fauchald et al. 2011). However, "enduring" features of the seafloor (e.g., shelf margins) can also drive the persistence or predictability of hotspots (Santora and Veit 2013). Our primary objective was to quantify the spatial and temporal variability of marine bird abundance and its relationship with habitat covariates in the offshore waters in and around the three WEAs located in the lower Mid-Atlantic Bight, off the coasts of Delaware (DE), Maryland (MD), and Virginia (VA; Figure 12-1).

We evaluated seasonal species abundance and community composition using two years of shipboard surveys and recently-developed hierarchical community distance sampling (HCDS) models (Chapter 11; Sollmann et al. in review). The high rates of identification in shipboard surveys make them a reliable method of documenting species richness for identifying important bird areas (Camphuysen et al. 2004; Smith et al. 2014). Increasing interest in quantifying species richness, as a measure of biodiversity, has spurred the development of community models in the field of ecology (Royle and Dorazio 2008). We use site-specific covariates in a hierarchical distance sampling model to estimate the abundance of multiple species (Royle et al. 2004), all within a single community model (Chapter 11; Sollmann et al. in review). Distance sampling accounts for imperfect detection to estimate 'true' (as opposed to relative) abundance (Buckland et al. 1993). In community models, certain parameters are shared and informed by all species, which improves the predictive power of rare species, because "borrowing strength" from the rest of the community renders the model robust to spurious covariate effects (Madon et al. 2013). Accurately representing the breadth of environmental variability across the study area is one of the most important factors in predicting the distribution and abundance of seabirds to unsampled areas, for assessing their potential post-construction displacement (Lapeña et al. 2011). Our approach enables us to incorporate infrequently detected species that may otherwise be excluded from modeling efforts, and thus we make use of the full shipboard survey dataset in analyzing species abundance and habitat relationships.

Seasonality in species richness or abundance is an important factor in determining when it is possible to minimize disturbance from the construction of wind facilities (Bailey et al. 2014). In our study area, breeders (e.g., pelicans, terns) and southern hemisphere winterers (e.g., storm-petrels) are generally present during the North Atlantic summer (see Table 12-1 for Latin names). Migratory and pelagic species that range throughout the region include ospreys, phalaropes, jaegers, fulmars and shearwaters. Overwintering, nonbreeding species in the region include northern breeders such as Northern Gannets, grebes, cormorants, gulls, loons, sea ducks, and alcids (e.g., murres).

Generally, these species fall into three feeding categories: piscivores (fish-eaters, e.g., Northern Gannets), planktivores (e.g., storm-petrels) or benthivores (bottom-feeding divers, e.g., sea ducks). Sea ducks such as scoters sometimes feed on fish and plankton, but primarily rely on more sessile benthic prey such as mollusks (Loring et al. 2014). The spatial and temporal patterns of marine birds at-sea are largely determined by these foraging ecologies, which factors into the cumulative impacts of disturbance, displacement, or collision risk from offshore wind energy development (for review, see Bailey et al. 2014; Langston 2013).

We hypothesized that habitat use would correspond to the foraging ecology of different species groups. We expected static seafloor characteristics to have a larger effect on benthivores (e.g., scoters), and dynamic sea surface characteristics (e.g., related to currents, etc.) to have a stronger effect on piscivores and planktivores (hereafter referred to as surface-feeders). Using the HCDS approach (Chapter 11; Sollmann et al. in review), we evaluate the relationships of species abundance with static and dynamic oceanographic parameters. The results of this study provide seasonal information on community composition and habitat use in the lower Mid-Atlantic Bight. We predict the distribution and abundance of seabirds for the purpose of minimizing effects to those populations from offshore wind energy development.

## Methods

### *Marine bird data collection*

From June 2012 to April 2014, we collected shipboard data on 15 surveys that lasted 4-5 days each. Two surveys were conducted in each year and season, defined as spring (Mar-May), summer (Jun-Aug), fall (Sep-Nov), and winter (Dec-Feb). We chartered a 55-ft vessel, which departed from the ports of Ocean City, MD and Virginia Beach, VA to transit 12 transects across the Atlantic Outer Continental Shelf (Figure 12-1). Two pairs of observers alternated 2-h shifts collecting standard line-transect data using distance sampling (Buckland et al. 1993). While the recorder entered data into the program dLOG (R.G. Ford Consulting, Inc.), and regularly updated changes in environmental conditions (Beaufort sea state, etc.), the observer scanned the horizon, focusing on one forward quadrant on either side of the vessel. We continuously recorded the species, count, distance, and angle to seabird observations (see Appendix 12A and Chapter 6 for more details on data collection methods).

### *Data analysis*

We implemented a set of HCDS models to estimate abundance and flock size while accounting for imperfect detection (Chapter 11; Royle et al. 2004; Sollmann et al. in review). Because HCDS requires spatial replication, we split the 12 tracklines for each survey into segments that averaged approximately 4 km, each of which is considered an individual 'site' in the model (Equation 1). We used seabird data observed up to one km perpendicular to the track line, beyond which there were few observations identified to species. We calculated mean habitat values per segment for six remotely-sensed covariates downloaded from online databases (Appendix 12A): three static (distance to shore, '$Dst$'; seafloor slope, '$Slp$'; sediment grain size, '$Grn$') and three dynamic (daily sea surface temperature [SST], '$Sst$'; daily salinity, '$Sal$', monthly chlorophyll anomaly, '$Chl$'). Sediment grain size ranged from fine to coarse sandy substrate, and is a proxy for variations in benthic prey assemblages (Loring et al. 2013). Chlorophyll anomaly is an index of high or low phytoplankton density, or extreme values of primary productivity at the sea surface (Santora and Veit 2013). Additional information on covariates may be found in Appendix 12A.

In a community model, multiple species are combined into one analysis that encompasses both abundant and uncommon species (Royle and Dorazio 2008). Here, we defined the marine bird community as a guild composed of species that are known to cohesively use marine habitat (we list those included in the community models in Table 12-1). Because scoters were largely identified to genus, as opposed to species, we removed them from the community model and treated them as a single group in a separate 'scoter' model (made up of White-winged Scoters, Black Scoters, and Surf Scoters; Table 12-2, Appendix 12A). We separated analyses by season to accommodate temporal changes in species composition resulting from migratory patterns, and to allow species-level covariate effects to vary independently by season for breeders and nonbreeders. Therefore, we present the results from one distance sampling model for scoters during the nonbreeding seasons when scoters were present in the area (first year: Nov 2012 – Mar 2013; second year: Oct 2013 – Apr 2014). We also present the results of one HCDS model for each of seven seasons (first year summer, fall, and winter, Jun 2012 – Jan 2013; second year spring, summer, fall, and winter, Mar 2013 – Feb 2014). There were at least five species with a single detection in each season of the second year (observed number of flocks = 1), which we removed to avoid problems with model convergence.

The sampling unit of analysis was an observation of a 'flock' containing one or more individuals. The model included two components that estimated (1) abundance of flocks (number of seabird clusters) based on distance sampling, and (2) flock size for each species to calculate total abundance (number of individuals). For the first component, we fit either a half-normal or negative exponential detection function on the observed distances to a flock, selecting the best-fitting distance function by computing Bayesian p-values using Freeman-Tukey fit statistics (Gelman et al. 2014). We also report this measure of goodness of fit for flock abundance and flock size.

Due to overdispersion, which is common in seabird counts (Zipkin et al. 2014), we assumed that the flock abundance, $N_{ij}$, of species $i$ at site $j$ followed a Negative Binomial distribution. We modeled the variation in mean abundance of flocks, $\lambda_{ij}$, as a function of the covariates such that:

$$N_{ij} \sim Negative\ Binomial(\lambda_{ij}, r)$$

$$\log(\lambda_{ij}) = \alpha_{0,i} + \log(site\ length_j) + \qquad (1)$$

$$\alpha_{1,i}Dst_j + \alpha_{2,i}Slp_j + \alpha_{3,i}Grn_j + \alpha_{4,i}Sst_j + \alpha_{5,i}Sal_j + \alpha_{6,i}Chl_j$$

where we included the log of the length of each segment as an offset in the model to standardize for slight variations in the true survey tracks (see Appendix 12A). Each parameter (e.g., $\alpha_{0,i}$…$\alpha_{6,i}$) was species-specific, governed by a hyperdistribution. For example, each species $i$ had an intercept $\alpha_{0,i}$, such that:

$$\alpha_{0,i} \sim Normal(\mu_{\alpha 0}, \sigma_{\alpha 0})$$

where the hyperparameters of these distributions, here $\mu_{\alpha 0}$ and $\sigma_{\alpha 0}$, are shared and informed by all species within the model. This allowed us to (1) retain species with few detections that would have otherwise been discarded from analysis, and (2) compare habitat use by each species to the overall mean community response. We modeled the observed flock sizes, $\boldsymbol{F}_i$, a vector of flock sizes for each species $i$, as an outcome of a zero-truncated Poisson – Negative Binomial mixture model, which allowed us to accommodate overdispersion, but with limits due to small sample sizes (Appendix 12A).

To predict to areas between and around the sampled transects, we first established a grid that contained the study area (Figure 12-1) based on the data layer with the coarsest spatial resolution (chlorophyll at 4 km). Daily covariate values made up the finest temporal resolution used in the model input, therefore, we used data from the midpoint of each season to predict overall abundance of flocks on that day (spring: 15 Apr, summer: 15 Jul, fall: 15 Oct, winter: 15 Jan). For example, we predicted the abundance for fall 2012 using the posterior mean parameter estimates and data from Oct 2012 for chlorophyll anomaly, and 15 Oct 2012 for SST and salinity. We implemented the HCDS models in a Bayesian framework using the package "rjags" to run the software JAGS (Plummer 2003) in program R version 2.15.3 (R Development Core Team 2013). We diagnosed convergence on three parallel chains that ran for 30,000 iterations (Gelman et al. 2014).

## Results

For the community models, we analyzed 40 marine bird species that fell into 11 taxonomic families (Table 12-1). Community composition differed between years (Table 12-3): there were 29 species observed in the first year (15 summer, 22 fall, 16 winter) and 35 observed in the second year (18 spring, 11 summer, 16 fall, 21 winter). The separate scoter group models included the three aforementioned species, White-winged, Black, and Surf Scoters, which were observed during the nonbreeding season (Table 12-2).

Overall patterns of estimated and predicted abundance for the entire community in most seasons reflect the influence of the shoreline, to which most species adhered closely (Figure 12-2, Figure 12-3). Exceptions to this pattern included several spring migrant species that were predicted in higher numbers offshore, such as Common Terns and Red Phalaropes, some wintering alcids (e.g., Dovekies), and Wilson's Storm-petrels in summer (Figure 12-4). Only in the fall of the first and second year did a covariate (grain size or distance to shore, respectively) have a strong effect on the entire community (Table 12-4), which was generally driven by the more abundant species (Table 12-3). Coefficient of variation (CV) maps (Figure 12-5) were calculated for the estimated number of flocks to show uncertainty relative to the predicted mean in flock abundance. In the case of scoters, the higher CV towards the edge of the Outer Continental Shelf was due to sparse data and estimated flock abundances close to zero in these areas (Figure 12-3).

Bayesian p-values (Table 12-5) indicated that the Negative Binomial distribution was a good fit for abundance for all species. Mean estimated flock sizes for each species corresponded closely to mean observed flock sizes (Table 12-1), although variation in the overdispersion of flocks produced poor fit statistics for a few of these models (Table 12-5), likely due to few observed flocks (small sample size) but large variation in the observed flock size that we could not adequately model. For the detection function, the half-normal distribution fit the first year summer community, while the negative exponential function fit the other seasons and the scoter observations (Table 12-5). As expected, we found that detection was significantly lower at higher Beaufort sea states (Table 12-4). Additionally, more conspicuous species such as Northern Gannets were detectable at farther distances than scoters (Appendix 12B).

To evaluate our hypothesis, we compared species (Figure 12-6, Figure 12-7) and community-level (Table 12-4) effects on the surface-feeding community to group-level habitat effects on scoters (benthivores, Figure 12-8, Table 12-6); responses were not consistent between species, seasons or years, as described below. In many cases, the community mean for the coefficient of distance to shore was not significantly different from zero (Table 12-4) but the species-specific parameter was significant (Figure 12-6, Figure 12-7). The three dynamic covariates (SST, salinity, and chlorophyll) were also significant predictors in many models, although their effects varied by species (Figure 12-7) and were much more important in some seasons (e.g., SST in the first fall) than others. During the fall of 2012, the surface-feeding community as a whole was associated with fine sediment grain size, which was driven by Royal Terns, Common Terns, Laughing Gulls, Northern Gannets, and Double-crested Cormorants; in fall 2013, the entire community was likely to be close to shore, driven by 13 of the 16 species (the main exception being Cory's Shearwater).

We focus primarily on winter models below, due to the high abundance and species diversity within the study area in this season. For details on the distribution and abundance of species in response to covariate effects in the spring, summer, and fall, see Appendix 12B and Figure 12-7.

### *Winter*

In the nonbreeding season across both years (2012-2014), scoter abundances had a significant relationship with distance to shore (a static covariate) and to high primary productivity (i.e., chlorophyll anomaly, a dynamic covariate; Figure 12-8). During the first year (2012-13), two static covariates (gentle slope and fine sediment) were strong predictors of scoter abundance but not of the wintering surface-feeding community. Additionally, scoter abundance was not associated with the dynamic covariate SST, but several wintering surface-feeders abundances were (Bonaparte's Gull, Manx Shearwater, Common Loon, Great Black-backed Gull, and Dovekie; Figure 12-6). In the second year (2013-14), scoter abundances were not related to those same two static covariates as in the year prior (slope and grain size), but they did associate with cold water, a dynamic covariate. During that same second year winter, surface-feeder abundances were not significantly correlated with sediment grain size but several surface-feeding species (Northern Gannets and three larids: Bonaparte's Gulls, Herring Gulls, and Ring-billed Gulls) were positively related with gentle slopes. The surface-feeders that associated with cold water in the second year winter were Northern Gannets, Herring Gulls, and Razorbills. Salinity was significantly lower in the second year summer, fall, and winter than in the first (Figure 12-9). Mean SST also contrasted sharply between winters; values used in model fit (i.e., along the sampled survey transects) were considerably warmer in the second year (mean 12.3 ± 2.8°C) compared to the first year (7.7 ± 2.8°C).

Among surface-feeders (Figure 12-6), Northern Gannets had higher estimated abundances close to shore (both years), as did Red-throated Loons; the same was true only for Year 2 for Common Loons. SST and primary productivity drove loon habitat partitioning in the first year, when Common Loons associated with higher SST, and Red-throated Loons associated with higher primary productivity. Alcids were observed farther from shore (i.e., closer to the continental shelf edge), particularly Atlantic Puffins (Year 1) and Razorbills (Year 2). Bonaparte's Gull abundances showed variable responses from the first to second year: in Year 1, they were associated with warm water and proximity to shore, while in Year 2 they associated with low salinity over gentle slope, further from shore. In the second year, Northern Gannets associated with gentle slope and cold water. Alcids also associated with cold water, specifically Dovekies (Year 1) and Razorbills (Year 2). With respect to chlorophyll anomaly (primary productivity), Dovekie abundances were negatively associated (Year 1), and Razorbills positively (Year 2). In the first year winter, Dovekie abundances had a positive relationship with cold water and low primary productivity, which resulted in higher winter 2013 predictions along the Atlantic Outer Continental Shelf (Figure 12-4). Horned Grebes also were estimated to have higher abundances in areas of higher primary productivity. Manx Shearwaters, which are northern breeders, had estimated higher abundance in warmer waters (Year 1).

## Discussion

Marine bird abundance estimates revealed that some species adhered closely to the shoreline (e.g., scoters), and were more common in the Delaware and Maryland WEAs, while some species showed pelagic distributions (e.g., during migration), and were more common in the Virginia WEA. By accounting for reduced detectability of scoters, which were present during the nonbreeding season, their estimated abundance was comparable to that of the more common surface-feeding species (e.g., Northern Gannet, Bonaparte's Gulls, and Common Loons in the spring, fall, or winter; Wilson's Storm-petrel, Laughing Gulls, Common Terns and Royal Terns in the summer). The HCDS model allowed us to include rare or elusive species, so as to directly compare habitat use in distinct seabird groups to the entire seabird community, and to document within- and between-species variability across seasons. The results show some consistencies with our hypothesis that the distribution of scoters would relate more to static covariates (distance to shore, slope, sediment grain size), compared to dynamic covariates (SST, salinity, chlorophyll anomaly), which we expected to drive the community of surface-feeders.

In line with our hypothesis, during the first year nonbreeding season (2012-2013), overwintering benthivores (scoters) showed significant relationships with static covariates characterizing seafloor variability (slope, sediment grain size), to which the wintering surface-feeders did not respond.

Furthermore, scoter abundances were not associated with the dynamic covariate SST, which was a significant predictor of the abundance of wintering surface-feeders. Scoters are known to adhere closely to the shoreline, where they have easier access to benthic prey at shallower depths (Loring et al. 2014). In our study area, primary productivity was high along the coast, which could explain the association between this dynamic covariate and high scoter abundance. However, during the second year winter season (2013-2014), scoter abundance was positively related to cold water (a dynamic covariate), and not significantly related to static covariates characterizing seafloor variability (slope, sediment grain size), unlike the first year. During that same second year winter, surface-feeders did not respond to sediment grain size, as we would expect, but a few species did respond to gentle slope. SST in the second year was significantly warmer compared to the first year, which could be due to eddies from the Gulf Stream off the Atlantic Outer Continental Shelf (Shealer 2001), or to variation in the North Atlantic Oscillation (Veit and Manne 2015), and may have influenced scoter selection of relatively colder water. This, along with the lack of an association with static seafloor characteristics, may also reflect dynamic movements of scoters in response to unstable sandy sediment (Dalyander et al. 2013) or ephemeral secondary productivity (zooplankton) and benthic prey resources in the second year (Loring et al. 2014).

Distance to shore dominated as one of the most consistent predictors of seabird distributions in our study area. Since it is an easily quantifiable metric for predicting abundance, distance to shore presents a useful foundation on which to base marine spatial planning efforts, but not to the exclusion of the other static and dynamic covariates that drive seabird abundance in this region. For example, northerly-migrating Common Tern abundance in the spring of 2013 had a positive relationship with warm water and low primary productivity, which led to predicted pre-breeding spatial distributions far from shore. Their positive association with fine sediment also resulted in a prediction of high Common Tern abundances at the center of the VA WEA in the spring (Figure 12-4). Considering that sediment grain size is a static covariate, we did not expect it to have a strong effect on the surface-feeding community, as occurred during the spring and first year fall. However, fine grain size correlated positively with proportion of sand, and terns are known to forage over sandy shoals that provide good habitat for high quality forage fish such as sandlance (*Ammodytes* spp.; Goyert 2015; Robards et al. 2000). Further research should investigate whether such a pattern in sediment grain size reflects the distribution of prey, and whether it is likely to persist during the migratory season from year to year, particularly in the WEAs.

We observed ‘hotspots’ around the mouth of the Delaware and Chesapeake Bay (for example, high richness and abundance of loons, razorbills, gannets, terns, gulls, scoters and others), which were likely driven by a salinity front and high primary productivity. This suggests that future efforts to assess the potential cumulative impacts of offshore wind energy development and shipping-channel traffic on seabird movements and populations may want to closely examine these regions (Chapter 1; Schwemmer et al. 2010). Productivity in our study area ranged from 1-5 mg $m^{-3}$, which corresponds to the lower end of the longer-term chlorophyll values that had strong positive effects on Common Loons in a study by Winiarski et al. (2013). Productivity relationships with loon abundance varied depending on the season. However, Red-throated Loons were consistently located closer to shore and in areas over colder water than Common Loons, which matched where productivity was generally higher in our study area (Powers and Cherry 1983). The fact that Dovekies associated with low primary productivity seems counterintuitive, but is likely a function of their distribution away from the highly productive coastline and over the outer edge of the shelf, where cold upwelled water can produce high concentrations of zooplankton (i.e., secondary productivity; Lieske et al. 2014; Veit and Guris 2009). Studies have shown that in the Northwest Atlantic, top-down forcing (negative predator-prey associations) occurs in subarctic waters under low productivity conditions, whereas bottom-up control (resource limitation inducing positive predator-prey relationships) dominates in waters off the east coast of the US where there is relatively higher primary or secondary productivity and species richness (Frank et al. 2007).

Observed species richness was highest in the second year winter and first year fall. High species diversity also extended to the spring, suggesting that migratory and overwintering species dominate the region's species composition. It is important that management considerations include the risk of displacement of nonbreeders that use this habitat while passing through the study area. For example, the procellarids and hydrobatids observed in our study were likely to be observed far from shore, associating with warm Gulf Stream water on the Atlantic Outer Continental Shelf (e.g., Wilson's Storm-petrels; Figure 12-4; Watson et al. 2013). Depending on climate patterns (e.g., the North Atlantic Oscillation), the region may continue to see increasing trends in the abundance of Cory's shearwaters, which reflects their northerly movement with increasingly warmer water along the US East Coast since 2009 (R.R. Veit, unpublished data).

While species abundance and richness was generally lower during the summer (breeding season for Northern Hemisphere species), some federally and state-listed Threatened or Endangered species were present in the region during that time of year (U.S. Fish & Wildlife Service, Delaware Division of Fish & Wildlife, Maryland Wildlife and Heritage Service, Virginia Department of Conservation and Recreation's Division of Natural Heritage). Examples include Roseate Terns (listed in DE, MD, VA, and federally in the USA), Least Terns (DE, MD), Common Terns (DE, MD), Forster's Terns (DE), and Royal Terns (MD). These species were primarily observed nearshore during summer months, while Common Terns were additionally abundant offshore in the spring, which corresponds to the pre-breeding migratory season (Appendix 12B). Studies have suggested that the foraging and breeding behavior of terns places them at risk of collision with offshore wind facilities (e.g., flying within rotor-height during repeated trips through facility footprints to feed chicks at the nest; Bradbury et al. 2014; Everaert 2014). The community distance sampling model enabled us to accommodate these relatively rare species. For example, in the fall, we had only 21 detections of Common Terns in the first year, and 6 in the second year, which might prohibit fitting a fully parameterized distance sampling model to those data. By combining data across species, we were able to estimate fall abundance for Common Terns and estimate their relationships with habitat features, improving our understanding of their distributions. This is particularly important because, while much focus on the exposure of terns to offshore wind energy development has been during the breeding season, we found their exposure to potential development within the Virginia WEA to be highest during the migratory period.

We also accounted for variation in detection, which is important in making comparisons between different species across time (Royle and Dorazio 2008). For example, Northern Gannets are large, white birds that contrast sharply against a deep blue ocean, and thus their detection probability is higher than less conspicuous species like smaller dark scoters. This results in differences between observed and estimated abundance that varies by species. After estimating detection and habitat relationships as well as abundance of marine birds in this study, future research should evaluate the types of risk that these populations face, as well as other conditions that were outside the sampling frame of the shipboard survey. For example, additional understanding of nocturnal movements and distributions of marine species under different weather conditions would be useful for informing further risk potential. In using our data to identify areas that may be more or less suitable for development, the decision-making process should prioritize further research within areas with high abundance and species richness, as well as areas with target species of concern (e.g., terns) that may be vulnerable even at low numbers.

In summary, species within the seabird community off the coasts of Delaware, Maryland, and Virginia show relatively high variability in their abundance and response to habitat covariates, which we were able to quantify reliably using HCDS. Although it has been suggested that a two-year study can capture much of the spatiotemporal variation in environmental conditions (Kinlan et al. 2012), our study had high variability across seasons from one year to the next. In planning for the potential construction of static structures (wind facilities) in a dynamic environment, it is important to consider that the distribution of hotspots is likely to change over a range of fine to coarse spatiotemporal scales. Considering that the operation of wind facilities can span decades, our study quantifies relatively short-term intra- and inter-annual volatility in the region. Further research is required to provide complementary information on the potential effects of long-term climatological cycles (e.g., North Atlantic Oscillation) or climate change on the exposure of marine animals to offshore energy development. Therefore, two years may provide baseline information on the seasonality of spatial trends, but it is likely not enough to quantify longer-term persistence, volatility, or vulnerability (Bailey et al. 2014).

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## Figures and tables

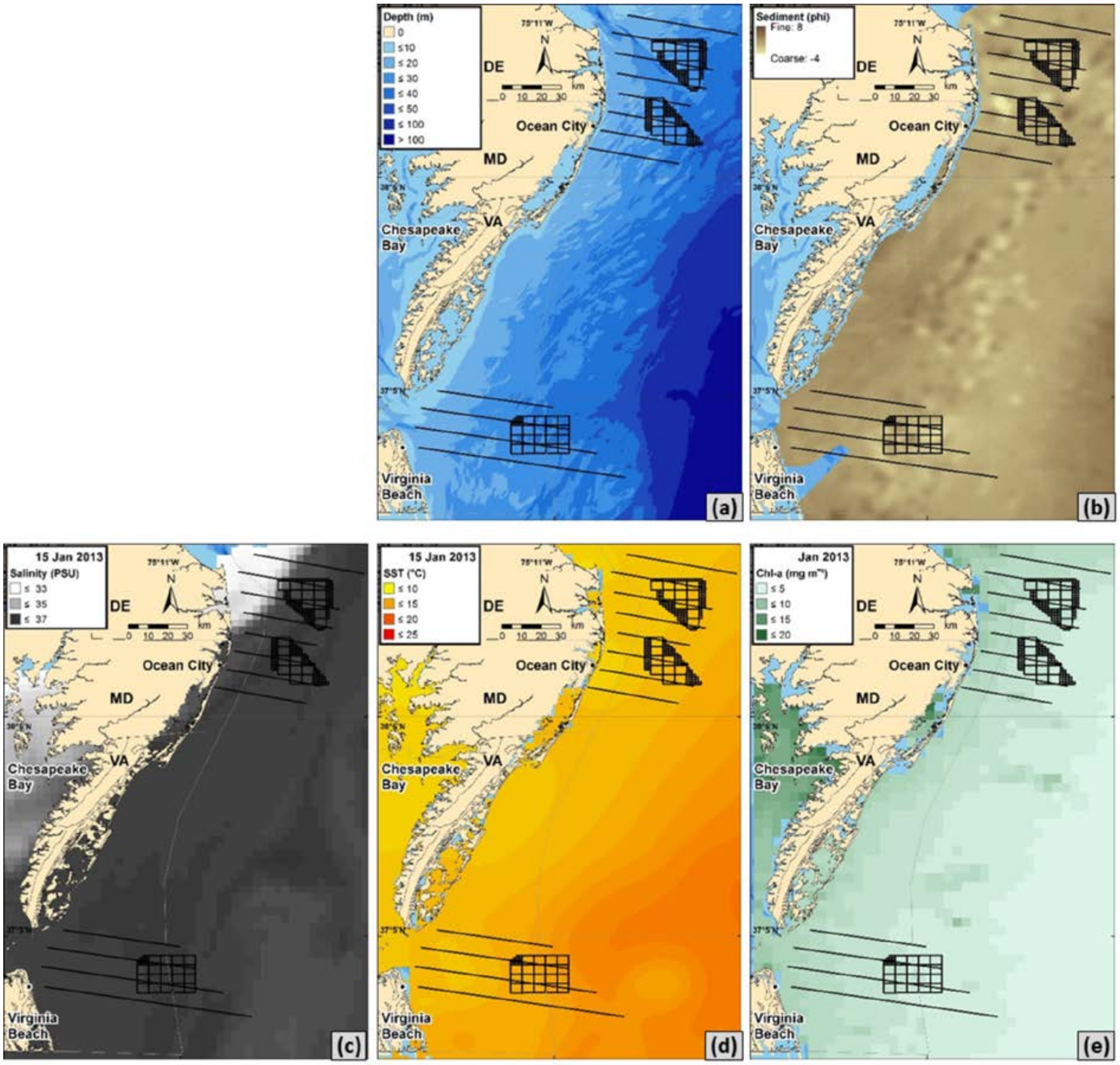


**Figure 12-1. Study area and example covariate data.** Transects were placed 10 km apart and ran perpendicular to the shoreline, covering federal waters greater than 5 km from the shore and extending out to a length of approximately 35-90 km. Black lines represent boat transects, black grids represent WEAs, and habitat covariates represent (a) bathymetry, distance to shore and slope, (b) sediment grain size (increases in phi units correspond to decreases in size; i.e., coarse to fine sand), (c) 15 Jan 2013 predictive salinity, (d) 15 Jan 2013 predictive sea surface temperature, and (e) Jan 2013 chlorophyll concentration used for model fit and predictions.

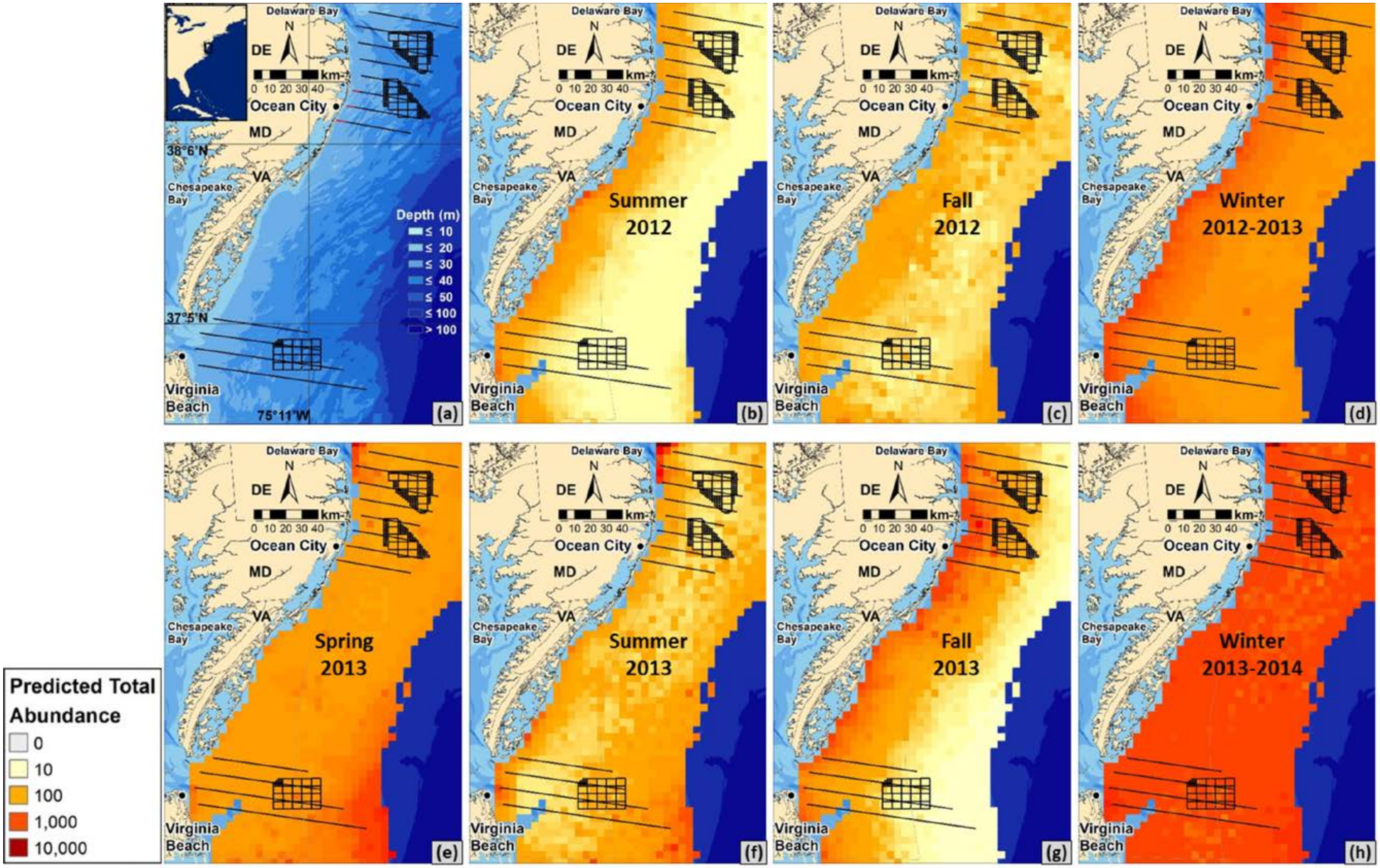


**Figure 12-2. Study area (a) and predicted total abundance maps for the first (top) and second (bottom) year in (b, f) summer, (c, g) fall, (d, h) winter (right column), and (e) spring.** Abundance maps (b-h) include all species in each seasonal community model (except scoters, which were modeled separately). Each map shows the posterior mean predicted total abundance across the study area: the expected number of flocks multiplied by flock size for each species, then summed across all species. Black lines represent boat transects, red transect segments in (a) delineate the MD extensions, black grids represent WEAs.

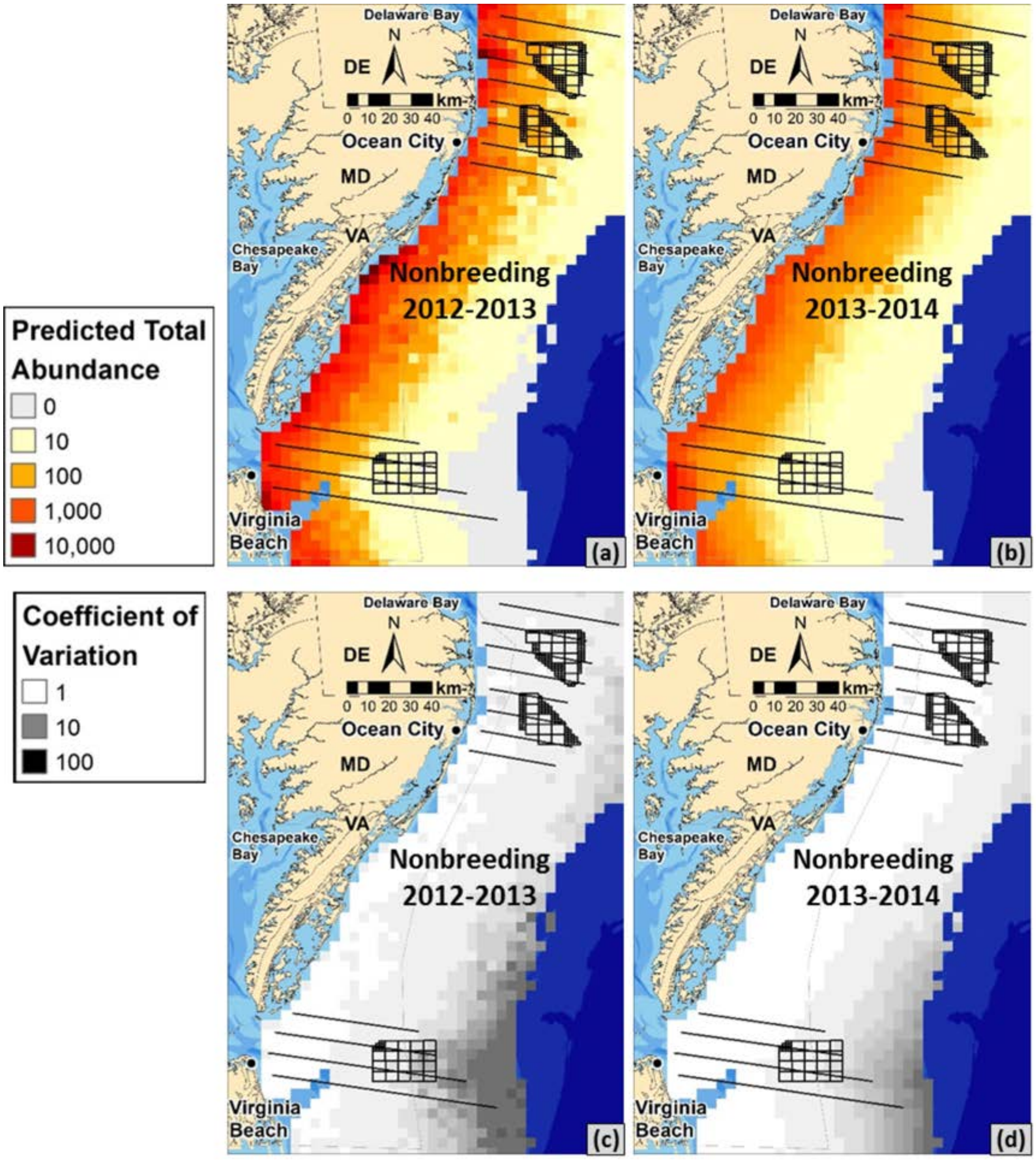


**Figure 12-3. Total abundance (top) for scoters during the nonbreeding season, predicted to 15 Jan 2013 (first year, left) or 15 Jan 2014 (second year, right). The coefficient of variation (CV) maps below were derived only for the abundance of flocks, not total abundance.** The higher CV towards the edge of the Outer Continental Shelf coincided with sparse data and estimated flock abundances close to zero in the areas farther away from the coastline. Black lines represent boat transects, black grids represent WEAs.


Predicted Total Abundance
0
10
100
1,000
10,000
Delaware Bay
N
DE
km
0 10 20 30 40
Ocean City
MD
Chesapeake Bay
VA
Virginia Beach
Summer 2012 Common Terns
(a)
Summer 2012 Royal Terns
(b)
Fall 2012 Bonaparte's Gulls
(c)
Fall 2012 Laughing Gulls
(d)
Winter 2012-2013 Razorbills
(e)
Winter 2012-2013 Dovekies
(f)
Winter 2012-2013 Common Loons
(g)
Winter 2012-2013 Red-throated Loons
(h)

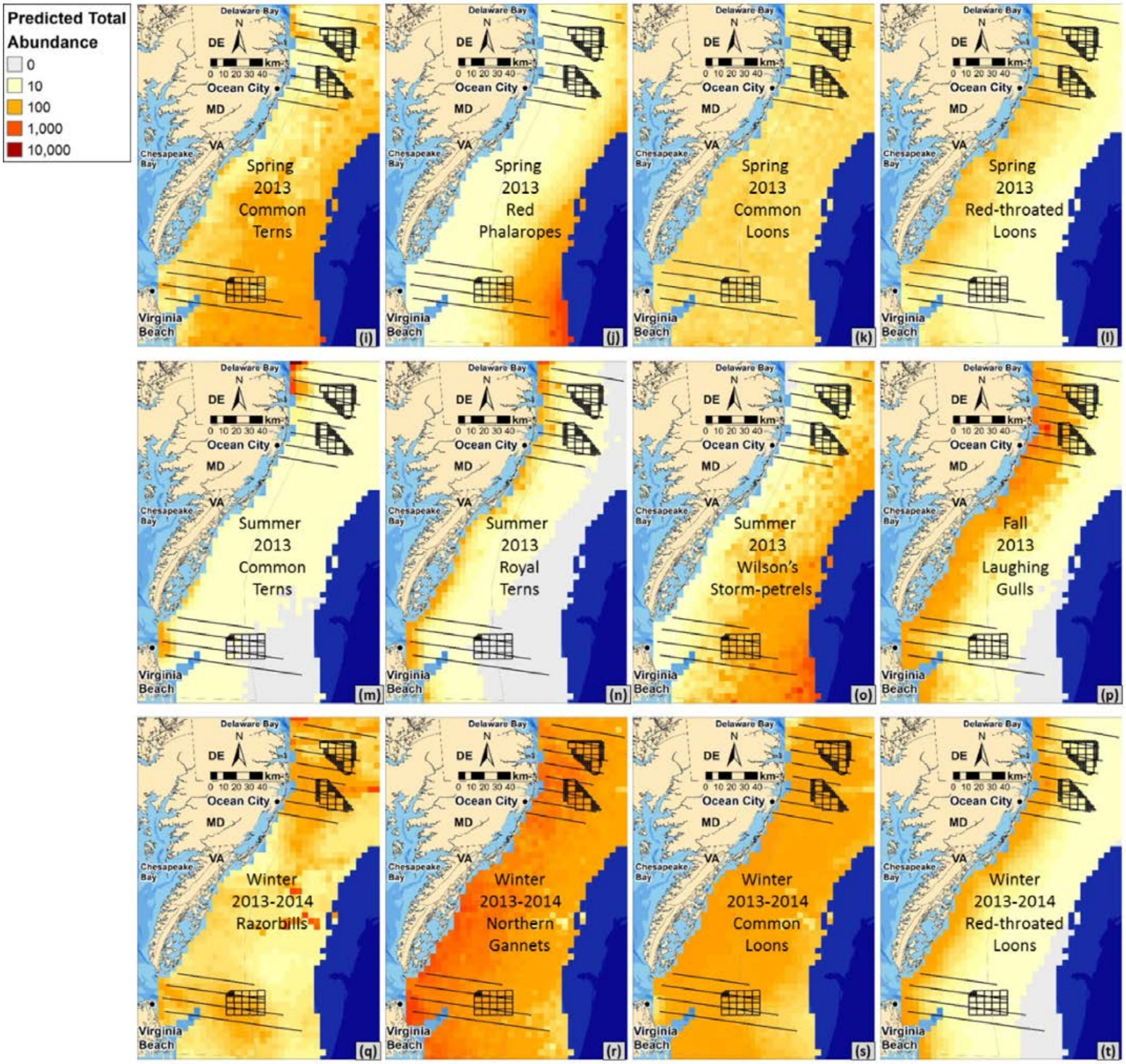


**Figure 12-4. Predicted total abundance for selected species per season.** The distribution of some of the most abundant species selected from each season, predicted to (a-b) summer 2012 (15 Jul), (c-d) fall 2012 (15 Oct), and (e-h) winter 2013 (15 Jan), as well as (i-l) spring 2013 (15 Apr), (m-o) summer 2013 (15 Jul), (p) fall 2013 (15 Oct), and (q-t) winter 2014 (15 Jan). Selected species include Common Terns (a, i, m), Royal Terns (b, n), Bonaparte's Gulls (c), Razorbills (e, q), Dovekies (f), Common Loons (g, k, s), Red-throated Loons (h, l, t), Red Phalaropes (j), Wilson's Storm-petrels (o), Laughing Gulls (d, p), and Northern Gannets (r).

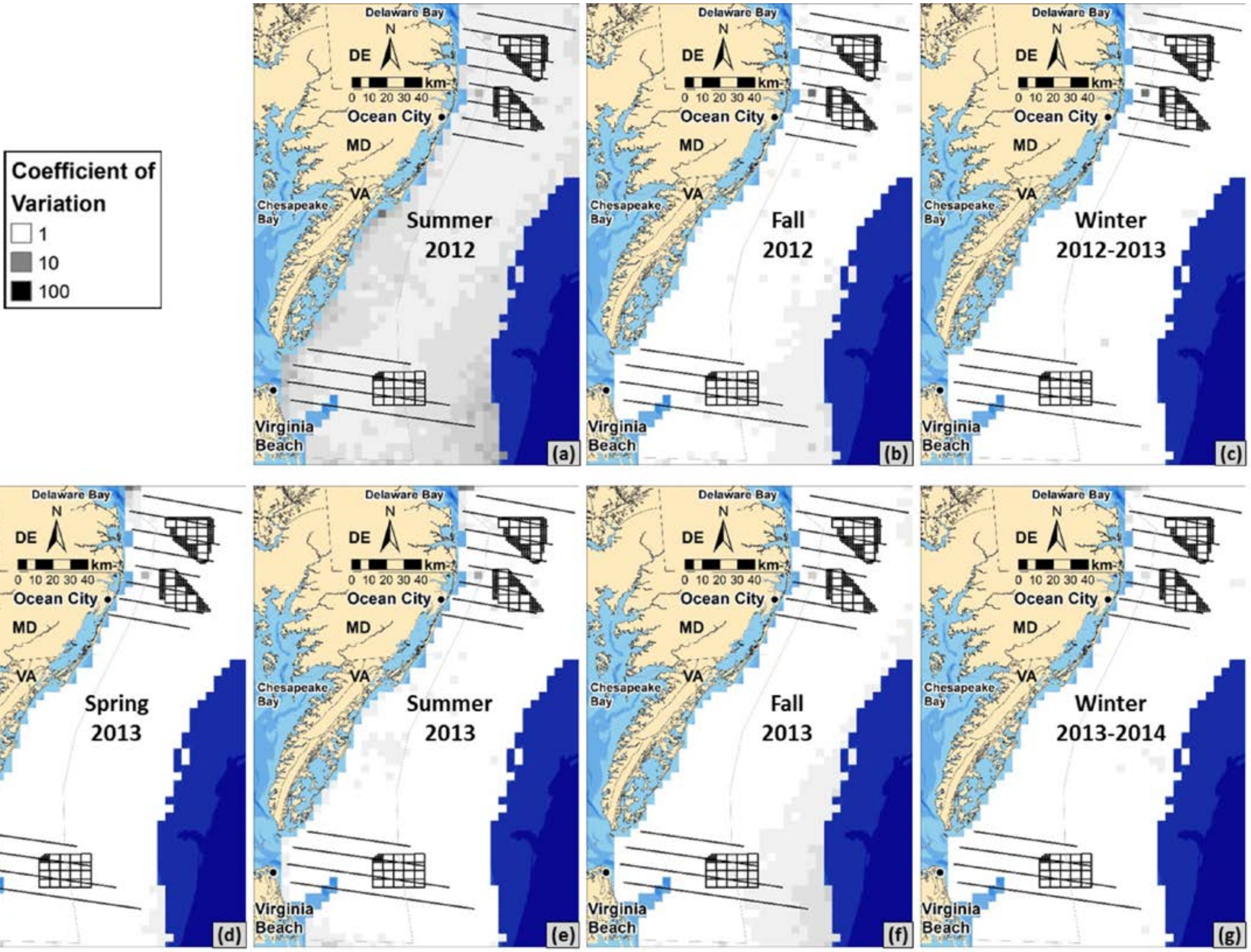
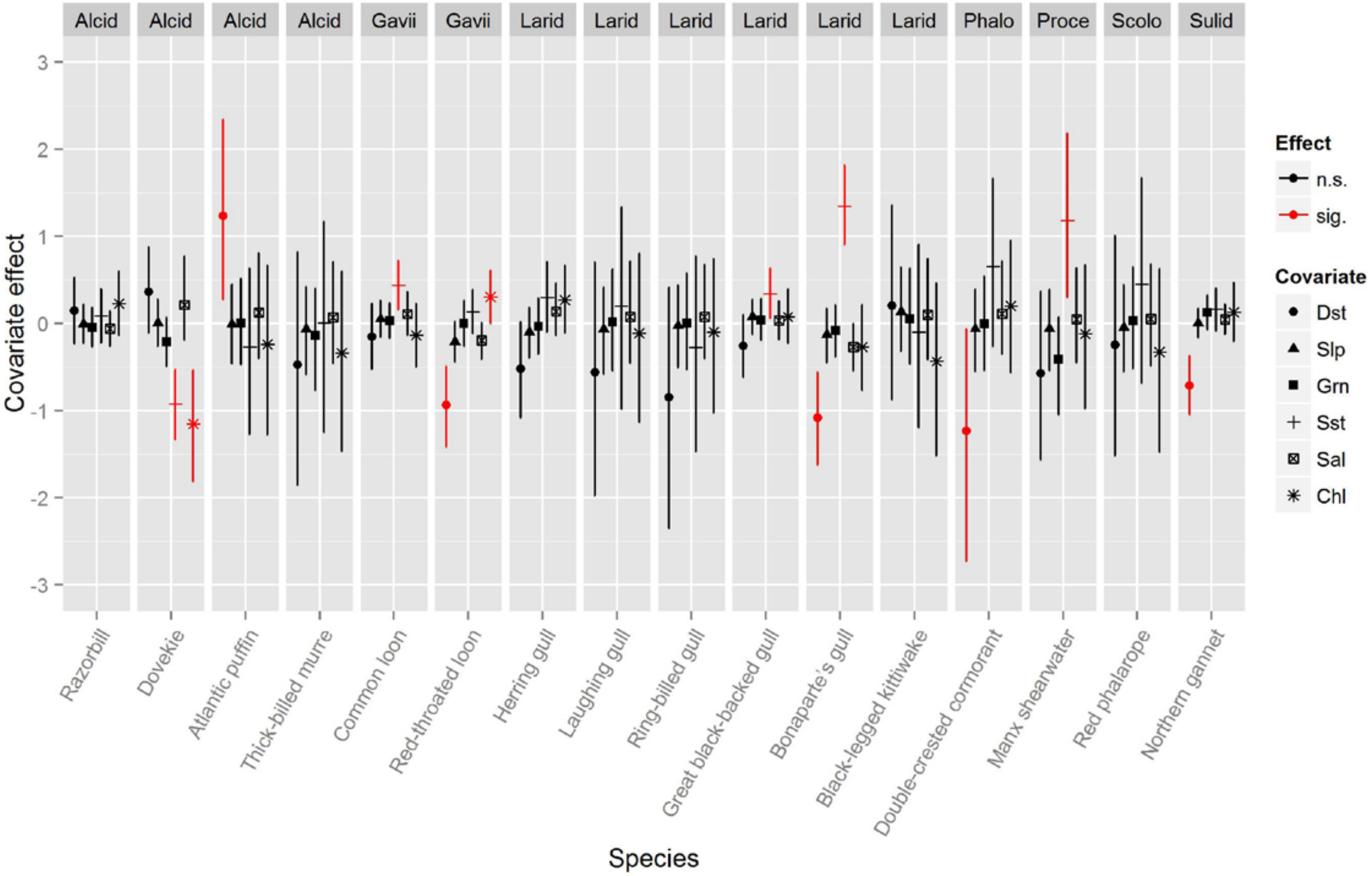
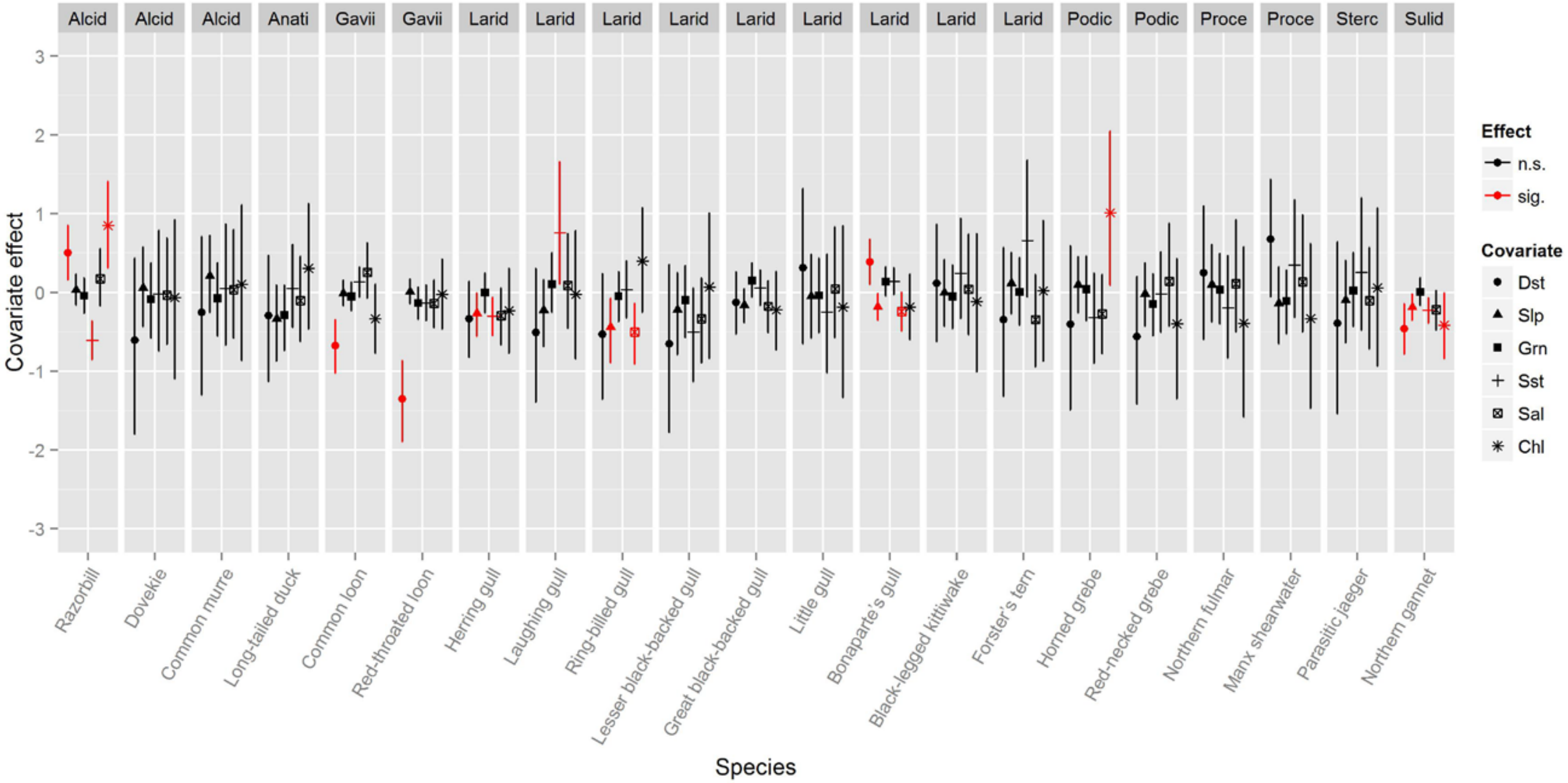


**Figure 12-5. Coefficient of variation (CV) maps for abundance of flocks in the first (top) and second (bottom) year (a, e) summer, (b, f) fall, (c, h) winter (right column), and (d) spring.** These figures include all species in each seasonal community model (to the exclusion of scoters, which were modeled separately) and predicted to the mid-point of the season as described in the text.

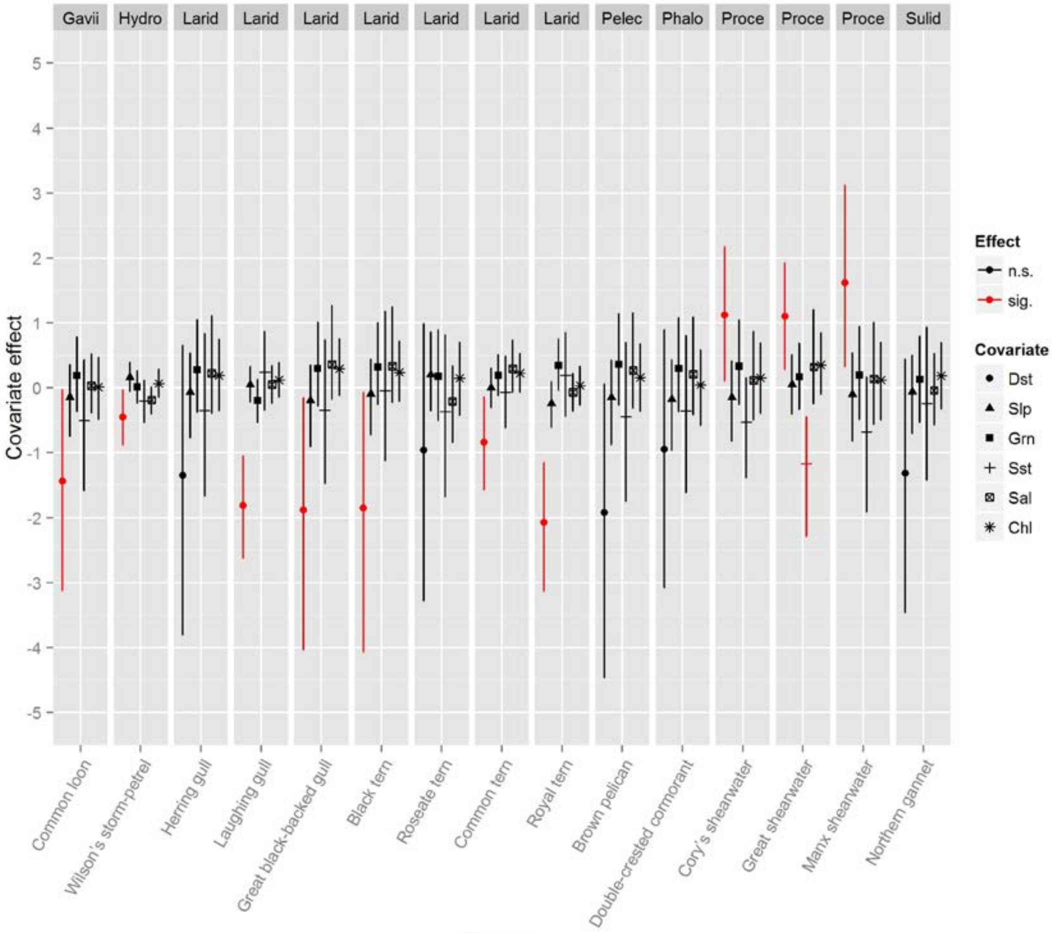

Alcid
Alcid
Alcid
Alcid
Gavii
Gavii
Larid
Larid
Larid
Larid
Larid
Larid
Phalo
Proce
Scolo
Sulid
Covariate effect
3
2
1
0
-1
-2
-3
Razorbill
Dovekie
Atlantic puffin
Thick-billed murre
Common loon
Red-throated loon
Herring gull
Laughing gull
Ring-billed gull
Great black-backed gull
Bonaparte's gull
Black-legged kittiwake
Double-crested cormorant
Manx shearwater
Red phalarope
Northern gannet
Species
Effect
n.s.
sig.
Covariate
Dst
Slp
Grn
Sst
Sal
Chl

**(a)**

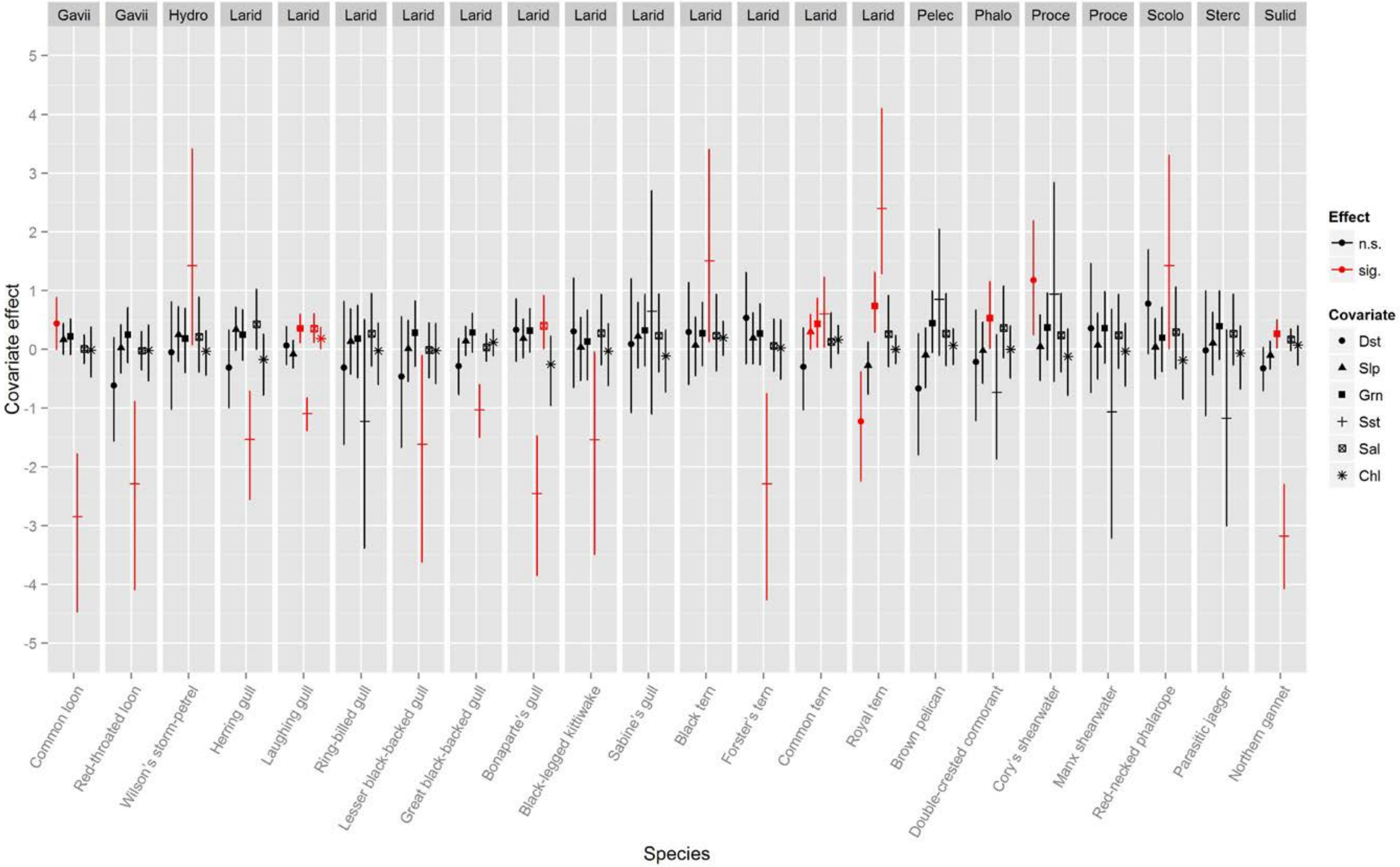


**(b)**

**Figure 12-6. First (a) and second (b) year effects of habitat covariates on each species in the winter community model (excluding scoters).** Error bars (Bayesian credible intervals) in red indicate which parameter estimates (on the log scale) were significantly different from zero; n.s. = not significant, sig. = significant, Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Chl = chlorophyll anomaly. Species are ordered by family (see Table 12-1 for abbreviations). Covariate effects are relative to study-specific habitat values, where negative responses indicate associations with proximity to shore, gradual slope, coarse sediment grain size, cold water, low salinity, and low primary productivity; positive responses indicate dependence on distance away from shore, steep slope, fine sediment grain size, warm water, high salinity, and high primary productivity.


Gavii
Hydro
Larid
Larid
Larid
Larid
Larid
Larid
Larid
Pelec
Phalo
Proce
Proce
Proce
Sulid
Covariate effect
5
4
3
2
1
0
-1
-2
-3
-4
-5
Effect
n.s.
sig.
Covariate
Dst
Slp
Grn
Sst
Sal
Chl
Common loon
Wilson's storm-petrel
Herring gull
Laughing gull
Great black-backed gull
Black tern
Roseate tern
Common tern
Royal tern
Brown pelican
Double-crested cormorant
Cory's shearwater
Great shearwater
Manx shearwater
Northern gannet
Species

**(a)**


Gavii
Gavii
Hydro
Larid
Larid
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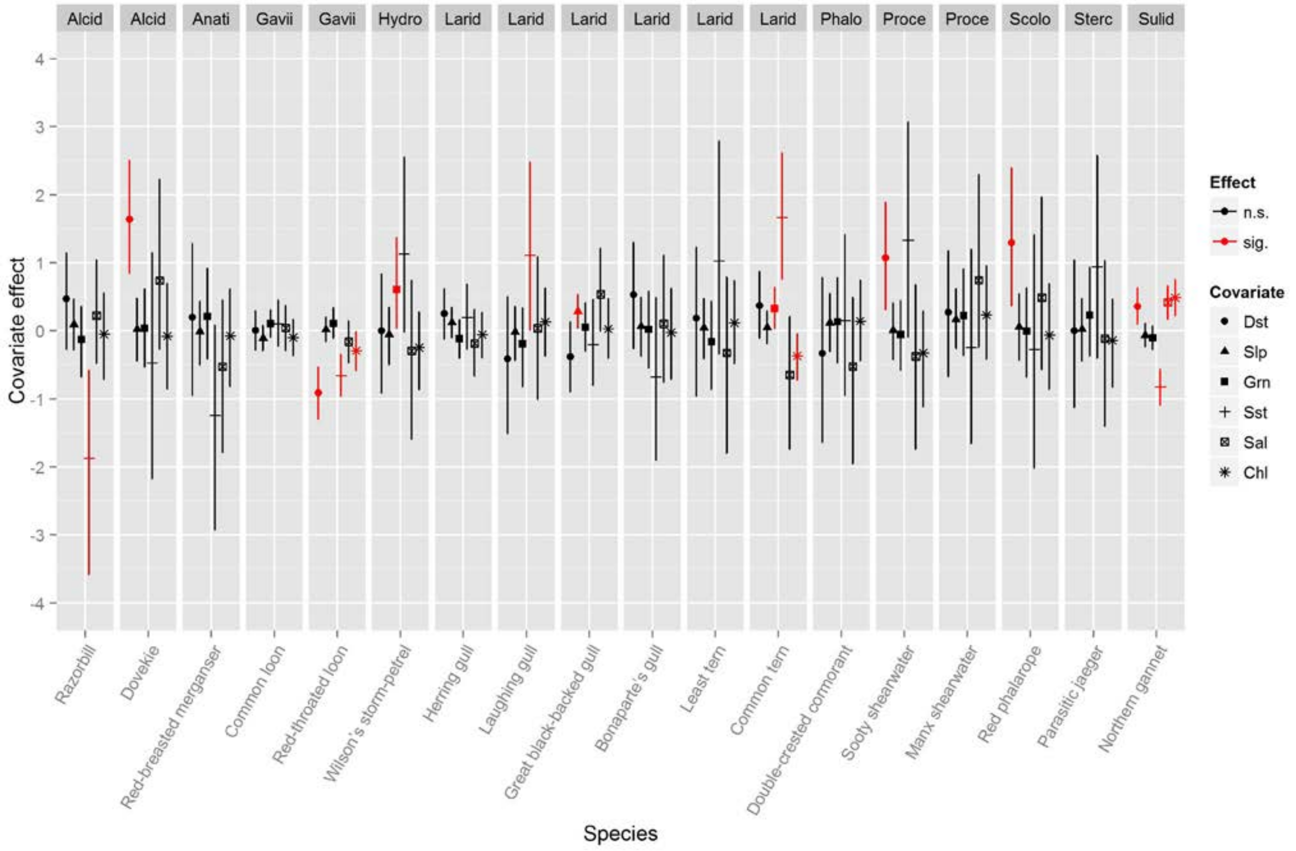
Larid
Larid
Larid
Larid
Larid
Larid
Larid
Larid
Pelec
Phalo
Proce
Proce
Scolo
Sterc
Sulid
5
4
3
2
1
0
-1
-2
-3
-4
-5
Covariate effect
Common loon
Red-throated loon
Wilson's storm-petrel
Herring gull
Laughing gull
Ring-billed gull
Lesser black-backed gull
Great black-backed gull
Bonaparte's gull
Black-legged kittiwake
Sabine's gull
Black tern
Forster's tern
Common tern
Royal tern
Brown pelican
Double-crested cormorant
Cory's shearwater
Manx shearwater
Red-necked phalarope
Parasitic jaeger
Northern gannet
Species
Effect
n.s.
sig.
Covariate
Dst
Slp
Grn
Sst
Sal
Chl

**(b)**


Alcid
Alcid
Anati
Gavii
Gavii
Hydro
Larid
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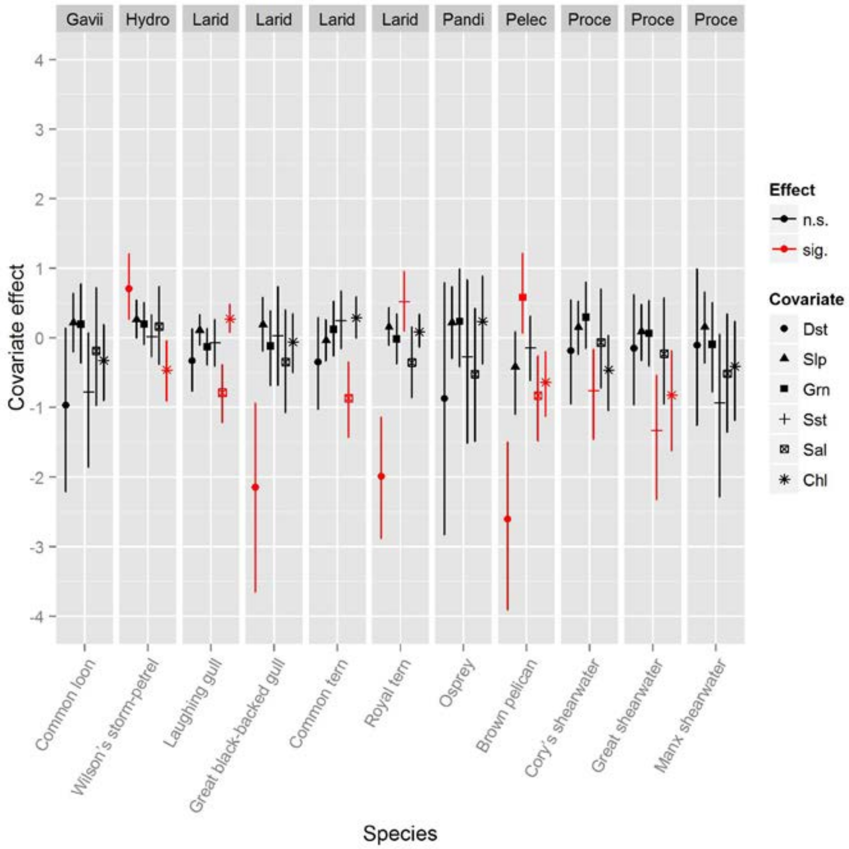
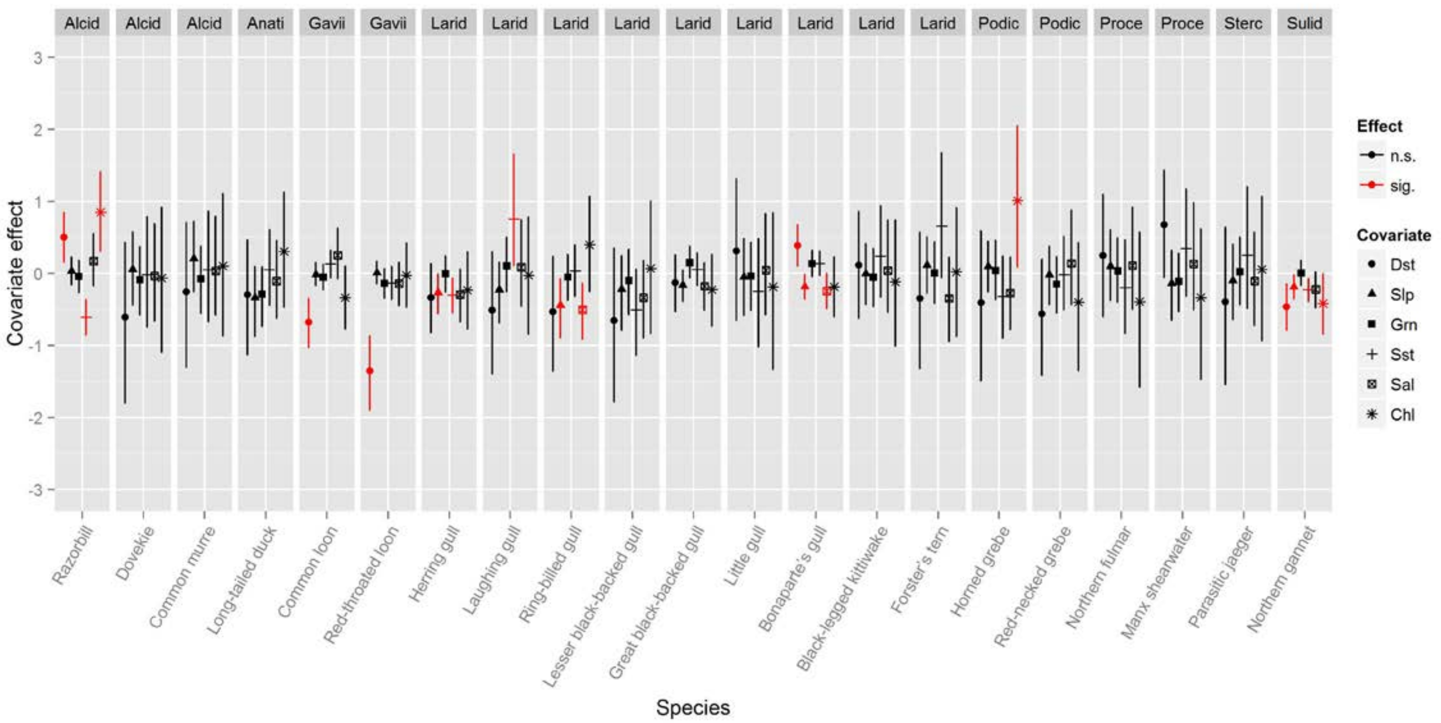
Larid
Larid
Larid
Larid
Phalo
Proce
Proce
Scolo
Sterc
Sulid
4
3
2
1
0
-1
-2
-3
-4
Covariate effect
Razorbill
Dovekie
Red-breasted merganser
Common loon
Red-throated loon
Wilson's storm-petrel
Herring gull
Laughing gull
Great black-backed gull
Bonaparte's gull
Least tern
Common tern
Double-crested cormorant
Sooty shearwater
Manx shearwater
Red phalarope
Parasitic jaeger
Northern gannet
Species
Effect
n.s.
sig.
Covariate
● Dst
▲ Slp
■ Grn
+ Sst
⊠ Sal
✳ Chl

**(c)**


Gavii
Hydro
Larid
Larid
Larid
Larid
Pandi
Pelec
Proce
Proce
Proce
Covariate effect
4
3
2
1
0
-1
-2
-3
-4
Common loon
Wilson's storm-petrel
Laughing gull
Great black-backed gull
Common tern
Royal tern
Osprey
Brown pelican
Cory's shearwater
Great shearwater
Manx shearwater
Species
Effect
n.s.
sig.
Covariate
Dst
Slp
Grn
Sst
Sal
Chl

**(d)**



**(e)**

**Figure 12-7. First year summer (a), and fall (b), and second year spring (c), summer (d), and fall (e) effects of habitat covariates on each species in the community models.** Error bars (Bayesian credible intervals) in red indicate which parameter estimates (on the log scale) were significantly different from zero; n.s. = not significant, sig. = significant, Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Chl = chlorophyll anomaly. Species are ordered by family (see Table 12-1 for abbreviations). Covariate effects are relative to study-specific habitat values, where negative responses indicate associations with proximity to shore, gradual slope, coarse sediment grain size, cold water, low salinity, and low primary productivity; positive responses indicate dependence on distance away from shore, steep slope, fine sediment grain size, warm water, high salinity, and high primary productivity.

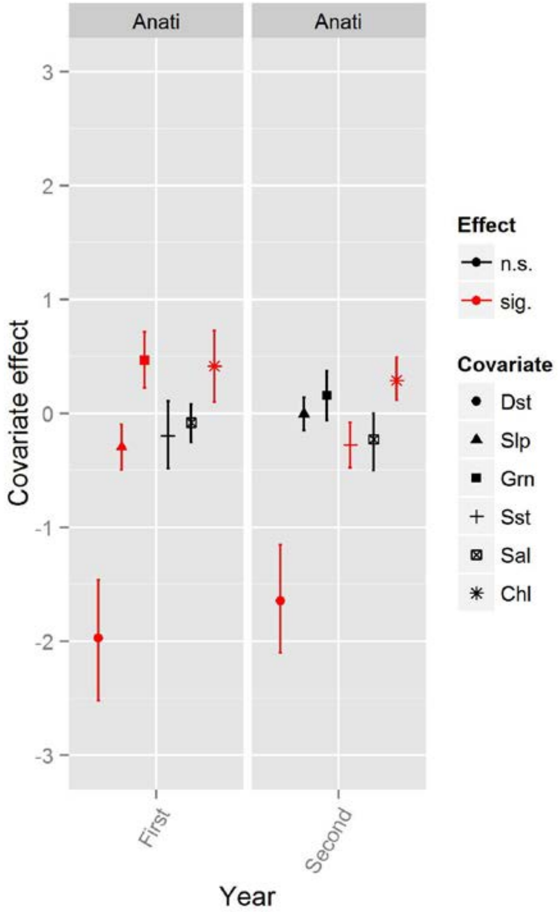


**Figure 12-8. Habitat effects for scoters during the nonbreeding season (Nov 2012 - Mar 2013, and Oct 2013 – Apr 2014).** Error bars (Bayesian credible intervals) in red indicate which parameter estimates (on the log scale) were significantly different from zero; n.s. = not significant, sig. = significant, Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Chl = chlorophyll anomaly, Anati = anatid family. Covariate effects are relative to study-specific habitat values, where negative responses indicate associations with proximity to shore, gradual slope, coarse sediment grain size, cold water, low salinity, and low primary productivity; positive responses indicate dependence on distance away from shore, steep slope, fine sediment grain size, warm water, high salinity, and high primary productivity.

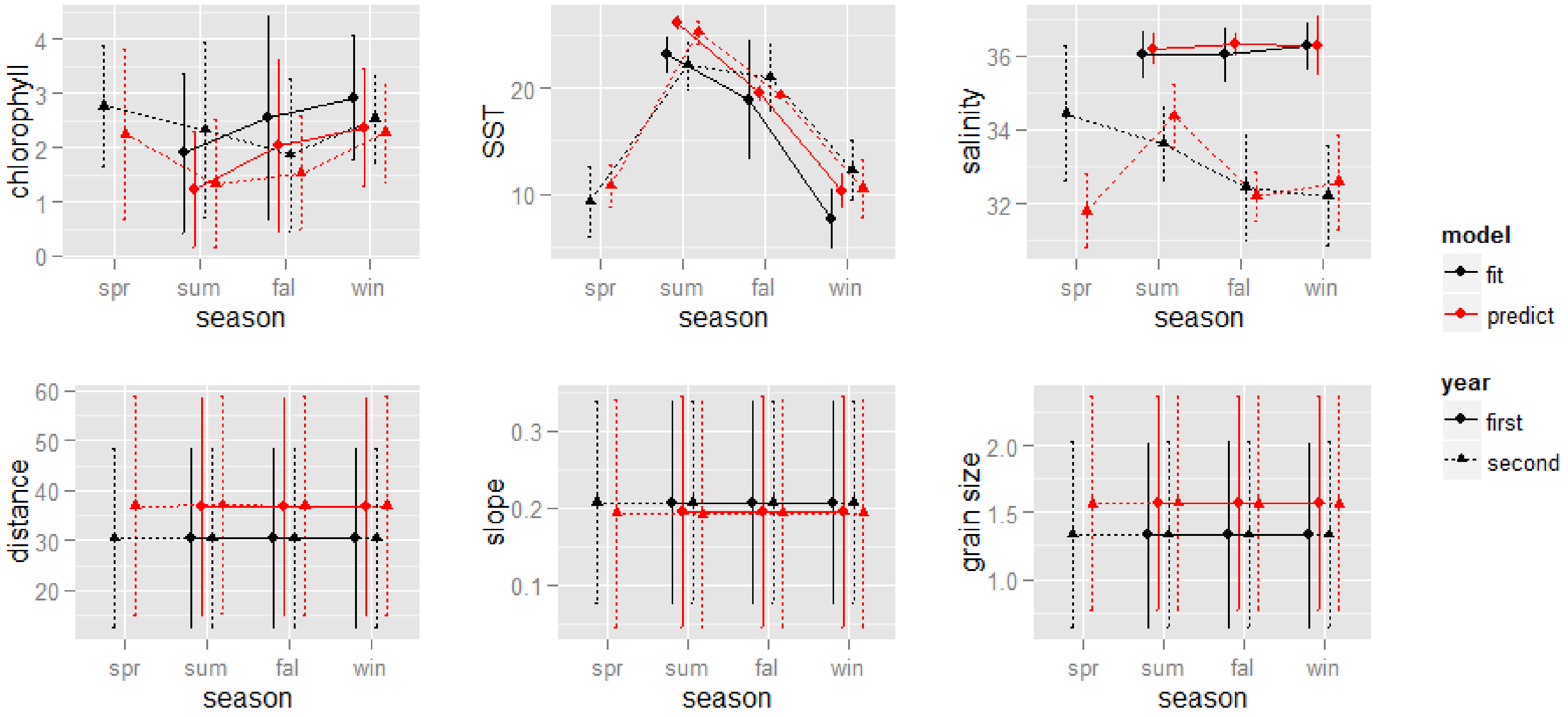


**Figure 12-9. Comparison of mean covariate values and standard deviations (bars) used in model fit (along transects) and prediction (entire study area) for the first and second year, across each season (spr = spring, sum = summer, fal = fall, win = winter).** Dynamic covariates are shown in the top row and static covariates in the bottom row.

**Table 12-1. Seasonal flock size for each species in the community models, including Latin names and taxonomic family.** "Observed" and "estimated" refer to the sampled area along transects, by season (across two surveys), where "Obs." is the mean of the observed flock sizes, and "Estim." is the estimated posterior mean for flock size.

| Common Name | Latin Name | Family | 2012 Summer | | 2012 Fall | | 2012-13 Winter | | 2013 Spring | | 2013 Summer | | 2013 Fall | | 2013-14 Winter | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | **Obs.** | **Estim.** | **Obs.** | **Estim.** | **Obs.** | **Estim.** | **Obs.** | **Estim.** | **Obs.** | **Estim.** | **Obs.** | **Estim.** | **Obs.** | **Estim.** |
| Razorbill | *Alca torda* | Alcidae | | | | | 2.5 | 2.5 | 2.1 | 2.1 | | | | | 3.1 | 3.2 |
| Dovekie | *Alle alle* | Alcidae | | | | | 1.9 | 2.0 | 1.0 | 1.0 | | | | | 1.0 | 1.2 |
| Atlantic Puffin | *Fratercula arctica* | Alcidae | | | | | 1.8 | 1.8 | | | | | | | | |
| Common Murre | *Uria aalge* | Alcidae | | | | | | | | | | | | | 1.3 | 1.4 |
| Thick-billed Murre | *Uria lomvia* | Alcidae | | | | | 1.0 | 1.6 | | | | | | | | |
| Long-tailed Duck | *Clangula hyemalis* | Anatidae | | | | | | | | | | | | | 2.7 | 2.7 |
| Red-breasted Merganser | *Mergus serrator* | Anatidae | | | | | | | 8.0 | 7.8 | | | | | | |
| Common Loon | *Gavia immer* | Gaviidae | 1.0 | 1.2 | 2.0 | 2.1 | 1.5 | 1.5 | 1.5 | 1.5 | 2.0 | 2.0 | 1.2 | 1.5 | 5.0 | 5.4 |
| Red-throated Loon | *Gavia stellata* | Gaviidae | | | 1.5 | 1.5 | 1.3 | 1.3 | 1.3 | 1.3 | | | | | 1.6 | 1.6 |
| Wilson's Storm-petrel | *Oceanites oceanicus* | Hydrobatidae | 1.8 | 2.1 | 1.0 | 1.1 | | | 1.0 | 1.0 | 1.6 | 1.7 | | | | |
| Herring Gull | *Larus argentatus* | Laridae | 1.0 | 2.0 | 1.1 | 1.1 | 1.2 | 1.3 | 1.5 | 1.6 | | | 1.7 | 1.8 | 2.4 | 2.5 |
| Laughing Gull | *Leucophaeus atricilla* | Laridae | 1.4 | 1.5 | 1.3 | 1.3 | 1.0 | 1.6 | 1.3 | 1.3 | 1.6 | 1.7 | 3.5 | 3.5 | 1.3 | 1.4 |
| Ring-billed Gull | *Larus delawarensis* | Laridae | | | 1.0 | 1.2 | 1.0 | 1.4 | | | | | 1.0 | 1.0 | 1.6 | 1.6 |
| Lesser Black-backed Gull | *Larus fuscus* | Laridae | | | 1.0 | 1.1 | | | | | | | | | 1.3 | 1.4 |
| Great Black-backed Gull | *Larus marinus* | Laridae | 1.2 | 1.4 | 1.2 | 1.2 | 1.3 | 1.3 | 1.5 | 1.5 | 1.1 | 1.2 | 1.5 | 1.5 | 1.4 | 1.4 |
| Little Gull | *Hydrocoloeus minutus* | Laridae | | | | | | | | | | | | | 1.0 | 1.2 |
| Bonaparte's Gull | *Chroicocephalus philadelphia* | Laridae | | | 9.1 | 8.6 | 3.0 | 2.9 | 1.0 | 1.0 | | | | | 15.4 | 14.9 |
| Black-legged Kittiwake | *Rissa tridactyla* | Laridae | | | 1.0 | 1.1 | 1.0 | 1.3 | | | | | 1.0 | 1.0 | 1.4 | 1.4 |
| Sabine's Gull | *Xema sabini* | Laridae | | | 1.0 | 1.2 | | | | | | | | | | |
| Black Tern | *Chlidonias niger* | Laridae | 1.5 | 1.8 | 1.2 | 1.2 | | | | | | | | | | |
| Least Tern | *Sternula antillarum* | Laridae | | | | | | | 1.3 | 1.4 | | | | | | |
| Caspian Tern | *Hydroprogne caspia* | Laridae | | | | | | | | | | | 1.5 | 1.5 | | |
| Roseate Tern | *Sterna dougallii* | Laridae | 1.0 | 2.0 | | | | | | | | | | | | |
| Forster's Tern | *Sterna forsteri* | Laridae | | | 3.7 | 3.6 | | | | | | | | | 7.4 | 7.1 |
| Common Tern | *Sterna hirundo* | Laridae | 4.7 | 5.5 | 3.6 | 3.7 | | | 4.3 | 4.4 | 1.9 | 2.0 | 6.0 | 6.0 | | |
| Royal Tern | *Thalasseus maximus* | Laridae | 1.5 | 1.6 | 2.0 | 2.0 | | | | | 1.3 | 1.4 | 1.6 | 1.7 | | |
| Osprey | *Pandion haliaetus* | Pandionidae | | | | | | | | | 1.0 | 1.5 | | | | |
| Brown Pelican | *Pelecanus occidentalis* | Pelecanidae | 2.0 | 2.5 | 2.0 | 2.0 | | | | | 2.0 | 2.1 | 1.5 | 1.5 | | |
| Double-crested Cormorant | *Phalacrocorax auritus* | Phalacrocoracidae | 5.0 | 6.0 | 24.4 | 23.1 | 1.0 | 1.3 | 53.7 | 52.2 | | | 88.6 | 87.6 | | |
| Horned Grebe | *Podiceps auritus* | Podicipedidae | | | | | | | | | | | | | 1.1 | 1.1 |
| Red-necked Grebe | *Podiceps grisegena* | Podicipedidae | | | | | | | | | | | | | 1.1 | 1.1 |
| Northern Fulmar | *Fulmarus glacialis* | Procellariidae | | | | | | | | | | | | | 1.4 | 1.4 |
| Cory's Shearwater | *Calonectris diomedea* | Procellariidae | 1.0 | 1.2 | 1.3 | 1.4 | | | | | 1.5 | 1.6 | 1.0 | 1.0 | | |
| Great Shearwater | *Puffinus gravis* | Procellariidae | 1.1 | 1.2 | | | | | | | 1.1 | 1.2 | | | | |
| Sooty Shearwater | *Puffinus griseus* | Procellariidae | | | | | | | 6.0 | 6.0 | | | | | | |
| Manx Shearwater | *Puffinus puffinus* | Procellariidae | 1.0 | 1.5 | 1.0 | 1.2 | 2.3 | 2.2 | 1.0 | 1.0 | 1.6 | 1.7 | 1.0 | 1.0 | 2.3 | 2.3 |
| Red Phalarope | *Phalaropus fulicaria* | Scolopacidae | | | | | 3.0 | 2.4 | 74.7 | 72.7 | | | | | | |

| Common Name | Latin Name | Family | 2012 Summer | | 2012 Fall | | 2012-13 Winter | | 2013 Spring | | 2013 Summer | | 2013 Fall | | 2013-14 Winter | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Obs. | Estim. | Obs. | Estim. | Obs. | Estim. | Obs. | Estim. | Obs. | Estim. | Obs. | Estim. | Obs. | Estim. |
| Red-necked Phalarope | *Phalaropus lobatus* | Scolopacidae | | | 2.2 | 2.2 | | | | | | | 2.3 | 2.3 | | |
| Parasitic Jaeger | *Stercorarius parasiticus* | Stercorariidae | | | 1.0 | 1.1 | | | 1.0 | 1.0 | | | 1.0 | 1.0 | 1.0 | 1.2 |
| Northern Gannet | *Morus bassanus* | Sulidae | 1.0 | 1.5 | 2.8 | 2.9 | 3.6 | 3.7 | 1.9 | 2.0 | | | 2.8 | 2.9 | 7.4 | 7.5 |

**Table 12-2. First and second year abundance of scoters during the nonbreeding season (8 surveys).** "Observed" and "estimated" refer to the sampled area along transects, by season (across four surveys), where "Obs." is the raw count of individuals, and "Estim." is the estimated posterior mean for total abundance. "Predict." is the posterior mean for the total abundance predicted to the 15 Jan 2013 (first year) or 15 Jan 2014 (second year), over the entire study area (including unsampled areas). Three species make up the 'scoter' group: White-winged Scoters *Melanitta. fusca*, Black Scoters *M. nigra,* and Surf Scoters *M. perspicillata*.

| **Scoters** | **Nonbreeding abund.** | | | **Flock size** | |
|---|---|---|---|---|---|
| | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** |
| **First year (Nov 2012 – Mar 2013)** | 11990 | 90545.3 | 706723.8 | 24.4 | 24.7 |
| **Second year (Oct 2013 – Apr 2014)** | 4906 | 36572.1 | 305488.7 | 14.6 | 14.9 |

**Table 12-3. Seasonal abundance for each species in the community models (excluding scoters).** "Observed" and "estimated" refer to the sampled area along transects, by season (across two surveys), where "Obs." is the raw count of individuals, and "Estim." is the estimated posterior mean for total abundance. "Predict." is the posterior mean for total abundance predicted to a single day at the midpoint of each season, over the entire study area (including unsampled areas; see text for more details). Species are listed in order of decreasing total mean estimated abundance, averaged across the seven seasons. The five most abundant (estimated) species in each season are in bold. In the second year, there were at least five species with a single detection in each season (observed number of flocks = 1), which we removed to avoid problems with model convergence related to sparse observations.

| Species | 2012 Summer | | | 2012 Fall | | | 2012-13 Winter | | | 2013 Spring | | | 2013 Summer | | | 2013 Fall | | | 2013-14 Winter | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. | Obs. | Estim. | Predict. |
| **Northern Gannet** | 2 | 8.6 | 274.4 | 1227 | **5728.1** | 6308.5 | 2790 | 12503.9 | **213891.6** | 1041 | **5768.1** | 24305.9 | NA | NA | NA | 678 | **3998.3** | 47755.1 | 4148 | **21795.7** | 383298.0 |
| **Bonaparte's Gull** | NA | NA | NA | 372 | **2200.1** | 11285.2 | 282 | 1458.2 | 44000.0 | 6 | 65.2 | 798.1 | NA | NA | NA | NA | NA | NA | 5471 | **38203.7** | 704669.7 |
| **Common Loon** | 6 | 49.1 | 599.8 | 208 | **1550.2** | 3805.0 | 421 | 2520.1 | **62373.2** | 313 | **3205.9** | 57157.5 | 16 | 132.8 | 1134.6 | 27 | 142.2 | 2485.9 | 1586 | **10688.2** | 152887.8 |
| **Laughing Gull** | 106 | **394.4** | 11549.0 | 292 | **2264.1** | 26044.9 | 1 | 15.0 | 496.6 | 10 | 191.0 | 2518.2 | 289 | **1847.5** | 87989.4 | 804 | **5862.7** | 77448.1 | 20 | 176.8 | 1591.7 |
| **Double-crested Cormorant** | 5 | 62.5 | 1564.5 | 122 | 705.8 | 12127.5 | 4 | 31.1 | 921.5 | 161 | 2067.2 | 48545.5 | NA | NA | NA | 797 | **5761.1** | 84029.7 | NA | NA | NA |
| **Razorbill** | NA | NA | NA | NA | NA | NA | 375 | 3839.4 | **63221.3** | 23 | 530.9 | 1509.4 | NA | NA | NA | NA | NA | NA | 494 | **3141.5** | 89158.5 |
| **Common Tern** | 252 | **1042.1** | 19105.0 | 76 | **852.9** | 13205.1 | NA | NA | NA | 376 | **3835.1** | 105664.4 | 93 | **797.9** | 134278.6 | 36 | 469.7 | 2591.6 | NA | NA | NA |
| **Red-throated Loon** | NA | NA | NA | 19 | 147.7 | 484.1 | 277 | 1792.9 | **34540.6** | 259 | **2541.1** | 31648.4 | NA | NA | NA | NA | NA | NA | 404 | **2461.2** | 37633.2 |
| **Great Black-backed Gull** | 6 | 75.5 | 1715.0 | 71 | 516.5 | 4961.8 | 243 | 1641.3 | 33454.0 | 58 | 727.8 | 5260.7 | 14 | 100.4 | 1965.3 | 148 | **1207.2** | 18366.8 | 179 | 1263.7 | 20451.3 |
| **Wilson's Storm-petrel** | 319 | **2041.4** | 20385.2 | 5 | 63.3 | 699.9 | NA | NA | NA | 9 | 135.5 | 4027.9 | 290 | **3120.8** | 115828.5 | NA | NA | NA | NA | NA | NA |
| **Herring Gull** | 1 | 12.3 | 557.0 | 20 | 173.7 | 1668.6 | 73 | 592.5 | 10446.9 | 104 | 1106.2 | 25948.1 | NA | NA | NA | 101 | **941.2** | 12227.5 | 235 | 1707.9 | 30256.5 |
| **Red Phalarope** | NA | NA | NA | NA | NA | NA | 3 | 46.0 | 1023.9 | 224 | **3559.0** | 79819.2 | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| **Dovekie** | NA | NA | NA | NA | NA | NA | 204 | 2681.1 | **38556.3** | 9 | 117.2 | 1860.9 | NA | NA | NA | NA | NA | NA | 2 | 19.6 | 400.8 |
| **Royal Tern** | 86 | **422.6** | 17971.6 | 86 | 502.5 | 4260.0 | NA | NA | NA | NA | NA | NA | 85 | **730.2** | 18995.6 | 57 | 384.1 | 2948.4 | NA | NA | NA |
| Ring-billed Gull | NA | NA | NA | 1 | 13.9 | 181.3 | 2 | 23.4 | 461.6 | NA | NA | NA | NA | NA | NA | 6 | 81.7 | 1375.1 | 62 | 612.3 | 12388.2 |
| Sooty Shearwater | NA | NA | NA | NA | NA | NA | NA | NA | NA | 60 | 642.1 | 45266.5 | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| **Brown Pelican** | 4 | 34.2 | 1024.6 | 14 | 92.1 | 1402.2 | NA | NA | NA | NA | NA | NA | 79 | **472.4** | 11108.1 | 6 | 39.7 | 435.3 | NA | NA | NA |
| Forster's Tern | NA | NA | NA | 26 | 200.5 | 1035.8 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 52 | 381.2 | 3465.8 |
| Manx Shearwater | 2 | 13.4 | 488.8 | 1 | 15.8 | 311.6 | 16 | 163.9 | 5136.0 | 4 | 52.3 | 380.2 | 8 | 63.3 | 429.2 | 2 | 30.6 | 449.7 | 21 | 186.5 | 5783.7 |
| Cory's Shearwater | 5 | 60.7 | 1561.8 | 4 | 55.9 | 1946.4 | NA | NA | NA | NA | NA | NA | 29 | 245.8 | 2313.3 | 11 | 101.4 | 1004.8 | NA | NA | NA |
| Red-necked Phalarope | NA | NA | NA | 11 | 157.4 | 3104.7 | NA | NA | NA | NA | NA | NA | NA | NA | NA | 14 | 205.7 | 1198.5 | NA | NA | NA |
| Red-breasted Merganser | NA | NA | NA | NA | NA | NA | NA | NA | NA | 16 | 255.6 | 3383.7 | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Long-tailed Duck | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 35 | 246.9 | 3902.5 |
| Great Shearwater | 20 | **92.3** | 377.2 | NA | NA | NA | NA | NA | NA | NA | NA | NA | 21 | 101.9 | 431.7 | NA | NA | NA | NA | NA | NA |
| Black-legged Kittiwake | NA | NA | NA | 3 | 3NA | 342.4 | 3 | 32.3 | 996.3 | NA | NA | NA | NA | NA | NA | 3 | 24.8 | 361.2 | 14 | 100.8 | 1751.9 |
| Black Tern | 6 | 52.7 | 3289.2 | 6 | 58.0 | 617.2 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Parasitic Jaeger | NA | NA | NA | 2 | 17.6 | 266.9 | NA | NA | NA | 3 | 44.3 | 1071.5 | NA | NA | NA | 2 | 24.7 | 478.9 | 2 | 20.4 | 361.6 |

| **Species** | **2012 Summer** | | | **2012 Fall** | | | **2012-13 Winter** | | | **2013 Spring** | | | **2013 Summer** | | | **2013 Fall** | | | **2013-14 Winter** | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** | **Obs.** | **Estim.** | **Predict.** |
| Red-necked Grebe | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 14 | 101.9 | 1615.3 |
| Lesser Black-backed Gull | NA | NA | NA | 3 | 29.0 | 211.7 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 8 | 66.7 | 1610.3 |
| Atlantic Puffin | NA | NA | NA | NA | NA | NA | 14 | 94.6 | 3006.9 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Least Tern | NA | NA | NA | NA | NA | NA | NA | NA | NA | 4 | 75.8 | 1871.8 | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Horned Grebe | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 11 | 73.7 | 11049.9 |
| Northern Fulmar | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 7 | 58.2 | 1446.8 |
| Caspian Tern | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 3 | 38.8 | 289.0 | NA | NA | NA |
| Common Murre | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 4 | 36.2 | 652.5 |
| Osprey | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 2 | 21.7 | 50046.9 | NA | NA | NA | NA | NA | NA |
| Little Gull | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | 2 | 18.1 | 507.7 |
| Thick-billed Murre | NA | NA | NA | NA | NA | NA | 1 | 14.7 | 458.6 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Sabine's Gull | NA | NA | NA | 1 | 13.4 | 320.6 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA |
| Roseate Tern | 1 | 5.7 | 194.1 | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA | NA |

**Table 12-4. First and second year posterior summaries for the community-level parameters by season.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Chl = chlorophyll anomaly, NB overdisp. is the Negative Binomial overdispersion parameter, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). Parameters are presented on the log scale, and the posterior mean for covariates where the 95% Bayesian credible interval does not overlap zero are in bold italics; all SD terms (shaded) are greater than zero by necessity.

| Component | Term | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD | Mean | SD |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Abundance** | **Intercept**; $\mu_{\alpha 0}$ | ***-4.8*** | 0.7 | ***-4.1*** | 0.5 | ***-2.6*** | 0.7 | ***-3.0*** | 0.6 | ***-3.2*** | 0.6 | ***-3.7*** | 0.5 | ***-2.5*** | 0.5 |
| | **Intercept SD**; $\sigma_{\alpha 0}$ | 2.3 | 0.6 | 1.7 | 0.4 | 2.6 | 0.6 | 2.1 | 0.4 | 1.8 | 0.5 | 1.7 | 0.4 | 2.0 | 0.4 |
| | **Dst, mean**; $\mu_{\alpha 1}$ | -0.9 | 0.5 | 0.0 | 0.2 | -0.4 | 0.3 | 0.3 | 0.2 | -0.8 | 0.4 | ***-1.4*** | 0.3 | -0.3 | 0.2 |
| | **Dst, SD**; $\sigma_{\alpha 1}$ | 1.5 | 0.4 | 0.7 | 0.2 | 0.8 | 0.2 | 0.8 | 0.2 | 1.2 | 0.4 | 0.7 | 0.2 | 0.6 | 0.2 |
| | **Slp, mean**; $\mu_{\alpha 2}$ | -0.1 | 0.1 | 0.1 | 0.1 | 0.0 | 0.1 | 0.0 | 0.1 | 0.1 | 0.1 | 0.1 | 0.1 | -0.1 | 0.1 |
| | **Slp, SD**; $\sigma_{\alpha 2}$ | 0.3 | 0.1 | 0.3 | 0.1 | 0.2 | 0.1 | 0.2 | 0.1 | 0.3 | 0.1 | 0.2 | 0.1 | 0.3 | 0.1 |
| | **Grn, mean**; $\mu_{\alpha 3}$ | 0.2 | 0.2 | ***0.3*** | 0.1 | 0.0 | 0.1 | 0.1 | 0.1 | 0.1 | 0.2 | -0.2 | 0.1 | 0.0 | 0.1 |
| | **Grn, SD**; $\sigma_{\alpha 3}$ | 0.3 | 0.1 | 0.3 | 0.1 | 0.3 | 0.1 | 0.3 | 0.1 | 0.4 | 0.1 | 0.3 | 0.1 | 0.2 | 0.1 |
| | **Sst, mean**; $\mu_{\alpha 4}$ | -0.3 | 0.3 | -0.6 | 0.4 | 0.2 | 0.2 | 0.1 | 0.3 | -0.3 | 0.3 | -0.3 | 0.3 | 0.0 | 0.1 |
| | **Sst, SD**; $\sigma_{\alpha 4}$ | 0.6 | 0.2 | 1.8 | 0.4 | 0.7 | 0.2 | 1.2 | 0.3 | 0.7 | 0.3 | 1.0 | 0.3 | 0.5 | 0.1 |
| | **Sal, mean**; $\mu_{\alpha 5}$ | 0.1 | 0.2 | 0.2 | 0.1 | 0.0 | 0.1 | 0.0 | 0.2 | -0.4 | 0.2 | 0.1 | 0.2 | -0.1 | 0.1 |
| | **Sal, SD**; $\sigma_{\alpha 5}$ | 0.4 | 0.2 | 0.3 | 0.1 | 0.3 | 0.1 | 0.6 | 0.3 | 0.5 | 0.2 | 0.4 | 0.2 | 0.3 | 0.1 |
| | **Chl, mean**; $\mu_{\alpha 6}$ | 0.2 | 0.1 | 0.0 | 0.1 | -0.1 | 0.2 | 0.0 | 0.1 | -0.2 | 0.2 | -0.3 | 0.2 | 0.0 | 0.2 |
| | **Chl, SD**; $\sigma_{\alpha 6}$ | 0.3 | 0.1 | 0.3 | 0.1 | 0.5 | 0.2 | 0.4 | 0.1 | 0.5 | 0.2 | 0.4 | 0.2 | 0.6 | 0.2 |
| | **NB overdisp.**; $r_N$ | ***0.5*** | 0.1 | ***0.6*** | 0.1 | ***0.6*** | 0.0 | ***0.9*** | 0.1 | ***0.5*** | 0.1 | ***1.0*** | 0.1 | ***0.9*** | 0.1 |
| **Detection** | **Intercept**; $\mu_{\beta 0}$ | ***5.1*** | 0.2 | ***5.0*** | 0.1 | ***4.9*** | 0.1 | ***4.5*** | 0.1 | ***5.0*** | 0.1 | ***4.9*** | 0.2 | ***5.2*** | 0.1 |
| | **Intercept SD**; $\sigma_{\beta 0}$ | 0.5 | 0.2 | 0.4 | 0.1 | 0.4 | 0.1 | 0.4 | 0.1 | 0.4 | 0.1 | 0.4 | 0.2 | 0.2 | 0.1 |
| | **Beaufort 3-6**; $\beta_1$ | ***-0.5*** | 0.1 | ***-0.3*** | 0.1 | ***-0.2*** | 0.0 | ***-0.1*** | 0.1 | ***-0.3*** | 0.1 | -0.1 | 0.1 | ***-0.4*** | 0.0 |

**Table 12-5. First and second year Bayesian p-values for the abundance and detection components of the models.** Values close to 0.5 indicate good model fit. Value in bold indicate which detection function we selected to model abundance. In the case of the second-year spring and fall, the low fit statistic was due to our fitting a Poisson distribution to the flock size of two species with sparse data (< 20 observations) that had large variance-to-mean ratios: Double-crested Cormorants (fall, spring), Common Terns (fall) and Red Phalaropes (spring). We recalculated the fit statistic without those species, to evaluate the impact of just a few species with few observations, and found that this improved the Bayesian p-value (from the reported 0.0 to a value of 0.3, which is closer to the ideal of 0.5). HN = Half Normal, NE = Negative Exponential functions.

| Year | Model | Abundance | Detection | | Flock |
|---|---|---|---|---|---|
| | | | HN | NE | size |
| **First** | Community summer | 0.50 | **0.49** | 0.37 | 0.52 |
| | Community fall | 0.51 | 0.89 | **0.51** | 0.11 |
| | Community winter | 0.50 | 0.92 | **0.39** | 0.61 |
| | Scoter nonbreeding | 0.49 | 0.89 | **0.48** | 0.75 |
| **Second** | Community spring | 0.52 | 0.72 | **0.34** | 0.01 |
| | Community summer | 0.50 | 0.73 | **0.42** | 0.44 |
| | Community fall | 0.48 | 0.93 | **0.49** | 0.00 |
| | Community winter | 0.49 | 0.90 | **0.32** | 0.60 |
| | Scoter nonbreeding | 0.51 | 0.78 | **0.47** | 0.63 |

**Table 12-6. First and second year posterior summaries of the parameters for scoters during the nonbreeding season (4 surveys, Nov 2012 - Mar 2013 or Oct 2013 – Apr 2014).** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Chl = chlorophyll anomaly, NB overdisp. is the Negative Binomial overdispersion parameter, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). Parameters are presented on the log scale, and the posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **Nonbreeding scoters** | | **First year** | | | | **Second year** | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept**; $\alpha_0$ | **-2.5** | 0.2 | -3.0 | -2.1 | **-2.3** | 0.2 | -2.7 | -2.0 |
| | **Dst**; $\alpha_1$ | ***-2.0*** | 0.3 | -2.5 | -1.5 | ***-1.6*** | 0.3 | -2.1 | -1.1 |
| | **Slp**; $\alpha_2$ | ***-0.3*** | 0.1 | -0.5 | -0.1 | 0.0 | 0.1 | -0.1 | 0.1 |
| | **Grn**; $\alpha_3$ | ***0.5*** | 0.1 | 0.2 | 0.7 | 0.2 | 0.1 | -0.1 | 0.4 |
| | **Sst**; $\alpha_4$ | -0.2 | 0.2 | -0.5 | 0.1 | ***-0.3*** | 0.1 | -0.5 | -0.1 |
| | **Sal**; $\alpha_5$ | -0.1 | 0.1 | -0.2 | 0.1 | -0.2 | 0.1 | -0.5 | 0.0 |
| | **Chl**; $\alpha_6$ | ***0.4*** | 0.2 | 0.1 | 0.7 | ***0.3*** | 0.1 | 0.1 | 0.5 |
| | **NB overdisp.**; $r_N$ | ***0.5*** | 0.1 | 0.4 | 0.7 | ***0.7*** | 0.1 | 0.5 | 1.0 |
| **Detection** | **Intercept**; $\beta_0$ | **5.0** | 0.1 | 4.9 | 5.2 | **5.0** | 0.1 | 4.9 | 5.2 |
| | **Beaufort 3-6**; $\beta_1$ | ***-0.3*** | 0.1 | -0.3 | -0.3 | ***-0.2*** | 0.1 | -0.1 | -0.2 |

# Supplementary material

## Appendix 12A. Supplementary information on methods

### *Covariate data collection*

We considered eight habitat covariates as explanatory variables for variation in abundance. Five of the habitat covariates were static: distance to shore, bathymetry, and three seafloor features. We calculated distance to shore (km) as the distance to the nearest Delmarva shoreline (North or South regions[2]). We extracted a bathymetry data layer from the National Oceanic and Atmospheric Administration's (NOAA) National Geophysical Data Center (NGDC) Coastal Relief Model (3-second, or 30-m resolution), using the spatial extent -76.1°W, 36.5°N to -74.4°W, 38.9°N[3]. Seafloor feature characteristics were derived by NOAA/NOS National Centers for Coastal Ocean Science (NCCOS, Kinlan et al. 2013) at a 370-m resolution and included: (1) seafloor slope (% rise) (2) predicted surficial sediment mean grain size ($\phi$ = -log2[mean grain diameter in mm]), and (3) predicted surficial sediment percent sand (%). On rare occasions (<0.01% of sites), sediment data contained missing values, which we imputed using the average from neighboring sites.

We used three dynamic habitat covariates: daily SST and salinity, and monthly chlorophyll. We downloaded the two daily data layers (1) one km SST (°C) from the Group for High-Resolution Sea Surface Temperature (GHRSST[4]) and (2) three km salinity (Practical Salinity Units, PSU) from the Global Navy Coastal Ocean Model (NCOM[5]). We downloaded monthly composites of 4-km chlorophyll concentration (mg/cubic m) from the Marine Geospatial Ecology Tools (Roberts et al. 2010) in ArcGIS 10.2, which accesses the OceanColor Level 3 Standard Mapped Image, via the National Aeronautics and Space Administration's (NASA) Goddard Space Flight Center (GSFC) Aqua satellite. We used monthly chlorophyll data because coastal satellite interference produced too many missing chlorophyll values at finer temporal resolution. Instead of using chlorophyll concentration, we calculated chlorophyll anomaly, by centering on the monthly mean of all sites, or effectively standardizing this covariate to a mean = 0 and a standard deviation < 2.

We conducted a preliminary analysis to select which covariates, at what spatiotemporal scale, were needed to quantify habitat suitability. This involved a tradeoff in selecting enough covariates to maximize variation, yet minimize overparameterization and co-linearity (Dormann et al. 2013). We evaluated co-linearity of these eight covariates by calculating Pearson's correlation coefficients and generalized variance inflation factors (GVIF, Zuur et al. 2010). As a result, this reduced our parameter space to include six covariates: three static (distance to shore, seafloor slope, sediment grain size) and three dynamic (SST, salinity, chlorophyll anomaly).

Due to slight survey-specific variation in the course-made-true (e.g., at the ends of each transect), we included the length of each segment as an offset in the model to standardize abundance by effort for

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[2] pubs.usgs.gov/of/2010/1119/data_catalog.html
[3] maps.ngdc.noaa.gov/viewers/wcs-client/
[4] coastwatch.pfeg.noaa.gov/erddap/griddap/jplG1SST.html
[5] http://edac-dap3.northerngulfinstitute.org/erddap/griddap/US_East_3D_agg.html

each of the 1206 transect segments (each of which was considered an individual ‘site’ in the model). Eight of the 15 surveys (from March 2013 to February 2014) also included extensions of three transects farther west into Maryland state waters, with a total additional transect length of approximately 12 km per survey. Where track line segments (sites) crossed over multiple values of each habitat covariate, we calculated the mean value per segment. Therefore, we modeled seabird observations in each 4-km segment by fitting them to the corresponding segment-level mean SST and salinity value on their day of observation, and mean chlorophyll from their month of observation (e.g., Figure 12-1).

### *Modeling*

Some seabird groups were only present during two surveys (e.g., during the summer or winter), which is one of the reasons why we analyzed each season separately in the community models; there were at least 5 species with a single detection in each season of the second year (observed number of flocks = 1), which we removed to avoid problems with model convergence related to lack of data. Scoters, on the other hand were present for the four surveys that comprised the nonbreeding season for Northern Hemisphere breeders. Therefore, the scoter models included the same surveys from the two winter seasons in the community models(Dec-Feb), but also included an adjacent survey from either shoulder season (fall and spring). We separated scoters from the community models for three reasons: (1) of all observations not identified to species in the dataset, > 90% were scoters, and 75% of all scoter observations were identified to the scoter genus, *Melanitta* spp., or to “non-White-winged Scoter”, but not to species; (2) their flock sizes were larger than any other species in the community; (3) they are benthivores, whereas the rest of the community is composed primarily of surface-feeding piscivores and planktivores. Analyzing scoters separately allowed for estimates of their flock sizes and habitat responses to remain independent of the surface-feeding community, and avoid influences acting upon or arising from other species.

As described in Chapter 11, the sampling unit of analysis was an observation of a seabird ‘flock’, consisting of one or more individuals. First, we used observed distances to a flock to estimate the detection function that describes decline in detection probability with distance from the transect. To do so, we binned the observed distances into $k$= 10 distance categories of $w = 100$ m each, where $b$ corresponds to the break points (we truncated the data at a maximum perpendicular distance of 1-km from the boat). Let $p_{ijk}$ be the detection probability of species $i$ at site $j$ in distance bin $k$. Then, under a Gaussian (or half-normal) detection function,

$$p_{ijk} = \frac{\int_{b_k}^{b_{k+1}} \exp\left(-\frac{x^2}{2\sigma_{ij}^2}\right) dx}{w}.$$

We allowed the scale parameter $\sigma_{ij}$ in the detection function to vary by species $i$ and a binary indicator of sea state at site $j$. We classified Beaufort state as a 0 if the mean Beaufort state was 0-2 for a segment (calm seas), and as a 1 if mean Beaufort state was 3-6 for a segment (rough seas). . For example, a sea state of 6 represents wind velocities that reach up to 27 knots (38 mph or 14 m/s), thus the weather conditions were variable during sampling. These high wind periods also fall within the range at which offshore wind turbines can operate at maximum rated power (Jonkman et al. 2009). Accounting for sea state accommodated reduced visibility due to increased wave height and occasions when observers switched platforms between calm and rough seas, following equipment safety protocol (Chapter 6).

We modeled the observed number of flock detections of species $i$ at site $j$, $n_{ij}$, as an outcome of a Binomial random variable where $N_{ij}$ is the true abundance of species $i$ at site $j$, and $p.t_{ij}$ is the total detection probability ($p.t_{ij} = \sum_k p_{ijk}$) such that:

$$n_{ij} \sim Binomial(N_{ij}, p.t_{ij})$$

See text in Chapters 11 and 12 for further details on estimating flock abundance as a function of covariates. To model observed flock sizes, $\boldsymbol{F}_i$ (a matrix of $\sum_j N_{ij}$ by $i$ flock sizes for each species $i$) we used a Poisson – Negative Binomial mixture model to accommodate overdispersion, but with limits due to small sample sizes. Through data exploration, we found that there was overdispersion of flock sizes for many species; however, there were also often very small sample sizes. Therefore, we set the threshold of the mixture to be 20 observed detections for each species in each season, and we fitted flock sizes to a Poisson distribution when those detections fell below this threshold, or a Negative Binomial distribution otherwise.

$$\boldsymbol{F}_i \sim \begin{cases} \text{zt Poisson}(\mu_i) & < 20 \text{ detections} \\ \text{zt Negative Binomial}(\mu_i, \rho_i) & \geq 20 \text{ detections} \end{cases}$$

Note that $\boldsymbol{F}_i$ is partially observed for each species, i.e., known for observed clusters and unknown for $\sum_j (N_{ij} - n_{ij})$ unobserved clusters of species $i$. We implemented a hyperparameter on mean flock size and on the dispersion parameter for all seasons except the summer of the first year, when flocks were more consistent in size. .

Using the package “rjags” (Plummer 2014) in program R version 2.15.3 (R Development Core Team 2013), we ran the software JAGS. We standardized the covariates for analysis to center them on a mean = 0, with a variance close to 1. We initialized three parallel Markov chains at different values and ran them for 30,000 iterations following a burn-in of 1,000 iterations. We checked for chain convergence visually (posterior density and trace plots), and quantitatively using the Gelman-Rubin statistic; this R-hat statistic indicated that chains converged as a measure of among-chain versus between-chain variance (R-hat < 1.1; Gelman et al. 2014).

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## Appendix 12B. Supplementary information on results

High levels of spatiotemporal variability were observed between species, for both detection and abundance. Highly conspicuous Northern Gannets, for example, had a higher detection probability than scoters (in rough seas during the first year winter/nonbreeding season, detection probability, $p = 0.33 \pm 0.012$ SD for Northern Gannets and $p = 0.23 \pm 0.015$ for scoters). Seasonal patterns of distribution and abundance by season are described in further detail below; winter patterns are described in the main text.

### *Spring*

In the spring (2013, Figure 12-7c), Northern Gannets showed consistency with the second year winter associations (2013-14), as they were likely to be observed over high primary productivity, high salinity, and cold water areas, and were more likely to venture farther away from shore than in other seasons. With respect to alcids, Dovekies were observed far away from shore, and Razorbills associated with cold water, as in the second year winter. As for gaviids, Red-throated Loons were likely to be close to shore (as in the winter), over low primary productivity (unlike the first year winter) and cold water (unlike Common Loons in the first year winter). The procellarids and hydrobatids we observed during the spring were southern ocean breeders: Sooty Shearwaters were likely to stay far from shore, while Wilson's Storm-petrels associated with warm water and fine grain size. Red Phalaropes were observed away from the coast. Common Terns are larids that migrate northerly during the spring, and they were predicted to have higher abundances in low primary productivity and high SST (as in other offshore studies, Amorim et al. 2009; Goyert et al. 2014), as well as to fine sediment grain size (see discussion).

### *Summer*

During the summer (2012-2013, Figure 12-7a, d), SST was significantly warmer than the other seasons. Warm water positively influenced the distribution of Royal Terns (Year 2), and negatively affected Great Shearwaters (both years) and Cory's Shearwaters (Year 2). The procellarids and hydrobatids we observed were likely to be far from shore (Year 1), and included southern breeders (Wilson's Storm-petrel, Great Shearwater), northern breeders (Manx Shearwater), and East Atlantic breeders (Cory's Shearwater). In the second year, Wilson's Storm-petrels were again far from shore, and associated with steep slope and low primary productivity. Low primary productivity also had a strong effect on Great Shearwaters (Year 2). Terns adhered closely to the shoreline, particularly Common Terns (Year 1), which are northern breeders, and Royal Terns (both years), which are local breeders. In the second year, Common Terns additionally associated with high primary productivity (similarly to a northerly nearshore study, Goyert 2014) and low salinity. Laughing Gulls were likely to be observed close to shore (Year 1), over high primary productivity (Year 2) and low salinity (Year 2), which led to high density predictions around the Delaware Bay. Brown Pelicans, which breed locally, were likely to be close to shore, over low primary productivity, low salinity, and fine sediment.

### *Fall*

During the fall (Figure 12-7b, e) of the first year (2012), the surface-feeding community as a whole were positively associated with fine sediment grain size (Table 12-4), which was driven by Royal Terns, Common Terns, Laughing Gulls, Northern Gannets, and Double-crested Cormorants (Table 12-3). In the second year (2013), the entire community was likely to be close to shore, driven by 13 of the 16 species

(Table 12-4, Figure 12-7); distance to shore had no strong effect on Cory's Shearwater, which was the only species to respond differently from the rest of their community (its Bayesian credible interval did not overlap the community mean effect). Cory's Shearwaters and Common Loons were likely to be observed far from shore in the first year fall. In addition to their response to fine sediment, Royal Terns associated with proximity to shore (both years), high salinity (Year 2), and warm water (Year 2). In the first year, SST strongly affected many species in the community both positively and negatively. Wilson's Storm-petrels, Common Terns and Black Terns associated with warm water. Across both years, the species that associated with cold water were Northern Gannets (as in the winter and spring), Common Loons (unlike in the first year winter), Herring Gulls and Great Black-backed Gulls. Other larids and gaviids that associated with cold water in the first year were: Laughing Gulls, Black-legged Kittiwakes, Lesser Black-backed Gulls, Bonaparte's Gulls, Forster's Terns, and Red-throated Loons. In the second year fall, several species associated with low primary productivity: Northern Gannets (unlike in the spring), Laughing Gulls, Common Loons (like Red-throated Loons in the spring), Red-necked Phalaropes, and Cory's Shearwaters. Gaviid abundances were negatively related to low primary productivity, specifically Common Loons in the second year fall and Red-throated Loons in the spring (in contrast to the first year winter, when they associated with higher primary productivity). Primary productivity was lowest in the summer and second year fall.

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# Introduction to Part IV
# Integrating data across survey platforms

## Report structure

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure I).

Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure I. In Parts II (Chapters 3-6) and III (Chapters 7-12), we describe methods and results for high resolution digital video aerial surveys and boat surveys, respectively. Part IV of this report (Chapters 13-19) combines data from both survey approaches to develop a comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area. Part V (Chapters 20-25) focuses on individual movements and habitat use of focal avian species, tracked via satellite telemetry; and Part VI (Chapters 26-27) focuses on population-level migratory movements over the oceans, using several approaches for studying nocturnal avian migration. An additional study effort, which further explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

## Part IV: Integrating data across survey methods

High resolution digital video aerial surveys are a relatively new method for collecting distribution and abundance data on animals (Thaxter and Burton 2009, Buckland et al. 2012), and ours was the first study to use this method on a broad scale in the U.S. The technology used in this study, developed by HiDef Aerial Surveying, Ltd., based in the U.K., is one of several digital aerial survey methodologies currently in use around the world. Digital aerial survey approaches have largely replaced visual aerial surveys for offshore wind energy research in Europe. Their greater aircraft speed and much higher flight altitude makes them safer to conduct than visual aerial surveys, and reduces or eliminates disturbance to wildlife as compared to visual aerial or boat survey approaches. They also produce archivable data, which allow for a robust quality assurance and audit process. They are a relatively new technology, however, and methodological and analytical processes for collecting and analyzing these data are still being addressed in the scientific literature.

Standardized boat-based surveys with distance estimation are a widely used and well-established method of obtaining density data for birds, sea turtles, and marine mammals. This survey method allows for the development of more detailed behavioral data than is possible with digital aerial approaches, and also provides excellent identification rates for most species (though identifications are generally not verified, either during or after the fact, which can be problematic in certain cases; Hobbs and Waite 2010, Conn et al. 2013). Detection bias is a known issue for boat-based surveys, but it is also an issue that is relatively well understood, and can be addressed in part with established analytical approaches (Buckland et al. 2001).

There are seven chapters in Part IV of this report, focused on the comparison and integration of data from boat surveys and digital video aerial surveys to examine wildlife distributions and relative abundance in the mid-Atlantic:

Chapter 13. A standardized comparison study of boat-based and digital video aerial surveys for marine wildlife in the U.S.

Chapter 14. A general comparison of results from boat surveys and digital video aerial surveys in the mid-Atlantic (2012-2014).

Chapter 15. Density modeling with environmental covariates for marine mammals and turtles.

Chapter 16. Modeling species assignment in strip transect surveys with uncertain species identification.

Chapter 17. Integrating data across survey methods: persistent hotspots and temporal changes in observed abundance.

Chapter 18. Comparison of boat and aerial models of abundance with environmental covariates for seabirds.

Chapter 19. Integrating aerial and boat data with environmental covariates to develop joint predictions of abundance for seabirds.

Several chapters focus on contrasting boat and digital video aerial survey approaches (Chapters 13-14, 18). In some cases, data from one survey approach are used independently to analyze wildlife distributions and relative abundance (e.g., in the case of sea turtles, Chapters 15 and 17, or Bottlenose Dolphins, *Tursiops truncatus*, Chapter 15). In other cases, digital video aerial survey data and boat survey data are used jointly (Chapters 16-17 and 19) to describe distributions and abundance of animals across the study area.

### *Comparisons of the two survey approaches*

In order to test the utility of high resolution digital video aerial surveys in the U.S., and to integrate new digital aerial survey data with historical data, we compared the digital aerial data to boat-based surveys using experimentally controlled methods (Chapter 13). This comparison indicated largely complementary strengths of the two survey approaches, though it also highlighted their respective weaknesses (namely, the need for additional analytical development for digital surveys, and the issue of disturbance to wildlife populations caused by the boat during surveys). The two survey methods found similar distribution patterns for scoters (sea ducks; *Melanitta* spp.), but were poorly correlated for highly mobile Northern Gannets (*Morus bassanus*), which at the density of transects in the comparison study were not adequately surveyed by the plane’s relatively narrow transect strip width.

In addition to this formal comparison of methods, project collaborators also pursued other methods of comparing and contrasting the two survey datasets (Chapter 14). Species identification rates, as well as detection rates, varied considerably between methods for some taxa. Aquatic species, such as sea turtles, rays, sharks, and fishes, were observed in much higher numbers in the aerial data than the boat data. While some of these animals were also observed in the boat survey, the aerial surveys provided an excellent platform for detecting and identifying animals within the upper reaches of the water column. In particular, higher counts and species diversity of sea turtles and mammals were detected on the aerial surveys than from the boat. A similar efficiency in detecting and identifying sea turtles and marine mammals from high resolution digital aerial platforms (as compared to visual aerial or boat surveys) has also been observed elsewhere (Normandeau Associates Inc. 2013). In contrast, boat survey observers detected larger numbers of more species of birds than the aerial survey, which may be partially due to differences in detectability between the two survey types. Northern Gannets and larger gulls, for example, were visible at great distances from the boat survey, as observers could look from the vessel all the way to the horizon. Reviewers of aerial survey data, in contrast, could only see animals present in the narrow strip of the transect onscreen, and aerial survey speed was roughly 13.5 times that of the boat, potentially limiting onscreen appearances by highly mobile animals (Chapter 13; Figure II).

Rates of identification of animals to species were lower for many taxa in digital video aerial surveys than boat surveys. The exhaustive quality assurance and audit protocol followed by aerial video reviewers, as well as characteristics inherent to the video review process itself (such as the use of multiple levels of “certainty” criteria in identifications), ultimately led to fewer definitive identifications than were afforded by direct observational approaches. This may not be as much of a disadvantage of the digital video aerial survey approach as it seems; in some cases, species misclassification in visual surveys may actually lead to less reliable density estimates than classifying animals as “unknown” (Conn et al. 2013). The detailed and exhaustive quality assurance process applied to digital video aerial survey data (Chapter 4) recognized the inherent uncertainty in the identification process, which is generally under-recognized in visual surveys, as it can be difficult to measure. The limitation of many aerial identifications to the family or genus level is also likely due in part to image quality, however. The current generation of cameras being used in Europe have much higher resolution and color rendition than the cameras used in this study, with better identification rates as a result (A. Webb pers. comm.). In the mid-Atlantic, gulls and terns (Laridae), loons (Gaviidae), and auks (Alcidae) all had much higher identification rates to the species level from the boat surveys than in aerial video (Chapter 14). Aerial video observers were better at identifying the most common avian family, Anatidae (scoters, ducks, and geese), to species than were boat observers, and observers from both survey types had similarly high identification rates of shearwaters (Procellariidae). Identification rates of toothed whales (Odontoceti) were higher on boat surveys, but baleen whales (Mysticeti) had higher rates of identification from aerial surveys.

In addition to these general comparisons of survey results, project collaborators compared the estimated effects of habitat on seabird abundance using the boat and digital video aerial datasets. Chapter 18 presents a preliminary analysis of data from four seabird groups (terns, gannets, loons, and alcids), in which remotely-collected environmental data were incorporated into the models. Boat data were analyzed similarly to other chapters (Chapters 11-12), but focused on single species rather than seabird communities; aerial data were analyzed similarly to the approach used for sea turtles in Chapter 15, but utilizing Generalized Linear Models (GLMs) rather than Generalized Additive Models (GAMs). Slightly different formulations of models were used than in these other chapters to facilitate comparison between the two survey approaches. Results were compared to determine if the two sampling methods detected similar patterns in seabird abundance, with the goal of determining how best to combine boat and digital aerial survey data for a joint analysis. Boat vs. aerial survey data did indicate some differences in species-habitat relationships, which suggested that joint modeling approaches that incorporated both sources of data could prove fruitful for describing species distributions, relative abundance, and habitat use throughout the study area.

### *Integrated analyses of boat and digital video aerial survey data*

The best methodological approach for surveys of offshore wildlife will depend on the specific characteristics of each study area and on project goals (Camphuysen et al. 2004), and may involve a combination of complementary survey methods. It is important to understand how to successfully integrate data from different survey platforms, in order to ensure compatibility among studies, maintain a continuous historical record, and enable the assessment of long-term changes in wildlife distributions and abundance. The differences in detectability, species identification, field of view, and species-habitat relationships between survey approaches provides an opportunity to create higher-quality end products, by incorporating complementary data streams from both survey approaches. In addition, there is a need to further the development of analytical approaches for digital aerial surveys. Because the cameras are pointed down towards the water's surface (Figure II), digital aerial surveys avoid the common problem of distance bias; but, to date, other types of detection bias have not been addressed for digital aerial surveys. Collecting these data alongside traditional boat survey data provides an opportunity to explore new approaches for understanding and analyzing digital video aerial survey data for wildlife.

On a small scale, this has led to the publication of a scientific paper on Eastern Red Bat (*Lasiurus borealis*) migration in the offshore environment of the mid-Atlantic (Hatch et al. 2013; Chapter 17). Collaborators also used the two datasets to identify temporal and spatial patterns of species presence and relative abundance in the study area, including the identification of "persistent hotspots," or geographic areas with consistently high numbers of animals or species through time (Chapter 17). These persistent hotspots of abundance and species richness could indicate important habitat use areas

(Santora and Veit 2013). Temporal patterns of observations of different species and groups within the study area can also be used to determine potential exposure to offshore development activities at different times of year (Chapter 17).

A broad geographic and temporal scale of analysis may be required to fully assess exposure to wildlife from proposed development projects, however, including the examination of locations which were not directly surveyed. The incorporation of environmental covariates into modeling efforts allowed for the prediction of relative densities across the study area for many taxa (Chapters 12, 15-16, and 18-19), with one or both survey datasets used to describe populations of interest. In some cases, one survey method was significantly better than the other for surveying a particular taxon (for example, digital aerial surveys for sea turtles; Chapter 15), while in other cases, the two datasets could be combined using recently developed joint modeling frameworks (Chapter 19). Common Loons (*Gavia immer*) and Red-throated Loons (*G. stellata*), which proved difficult to distinguish in aerial video, provided a test case for using boat-based species identifications to inform aerial models and developing spatially explicit species-specific estimates of relative abundance (Chapter 16). In Chapter 19, project collaborators developed an integrated modeling approach in which predictions of marine bird abundance and distribution were jointly informed by aerial surveys (which encompassed a large geographic area), and boat surveys (which allowed for estimation of detection probability). Building on Chapters 12 and 18, Chapter 19 incorporated remotely collected environmental covariate data into the hierarchical modeling structure and produced a single prediction of abundance and distribution across the study area that utilized data from both survey approaches. Integrated models for the four taxa examined (terns, alcids, loons, and gannets) predicted taxon-specific hotspots that generally concurred with the results from Chapters 12 and 17, and in some cases performed better than models developed using data from a single survey approach. While additional exploration and model development is needed, these results indicate that joint modeling approaches may be a fruitful avenue of continued research.

### *Implications*

Our application of these methods in the mid-Atlantic is expected to be useful for understanding wildlife populations and minimizing impacts to those populations from offshore wind energy development in several ways:

- First, this study has explored technological advancements and assessment methods that could be used in future monitoring efforts. Comparisons of high resolution digital video aerial surveys to boat-based surveys allow us to better understand the potential uses of high resolution digital video aerial surveys in relation to offshore development in U.S. waters, and to understand when and where each survey approach may be best suited to meet the monitoring needs of regulators, resource managers, and developers.
- Second, we identify species that are likely to be exposed to offshore wind energy development activities in the mid-Atlantic study area, along with their important habitat use or aggregation areas and temporal variation in distribution patterns. By combining data from two quite different survey approaches, we can develop a better view of wildlife populations and distribution patterns than either survey method could provide alone. This information can be helpful for:

  - Informing the siting of future projects, by incorporating wildlife patterns into marine spatial planning and decision making, and by using exposure data as a first step towards defining relative risk by location;
  - Informing the permitting process for projects, by contributing data towards National Environmental Protection Act (NEPA) and other regulatory requirements, and by helping to define target taxa or research priorities on which to focus on during site-specific pre- and post-construction monitoring studies; and
  - Informing mitigation efforts and construction and operations plans, by presenting temporal data on community composition, distributions, and abundance that can be used to time certain activities to coincide with reduced potential for exposure of certain populations.
- Third, digital aerial surveys have some considerable advantages over traditional visual observation approaches, most notably in relation to survey speed and safety, but they also require some different analytical approaches than traditional surveys, which the scientific community is still in the process of developing. We explore statistical models aimed at improving our utilization of digital video aerial survey data, particularly in combination with boat data, to understand wildlife patterns.

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**Figure I. Organization of chapters within this final report.**

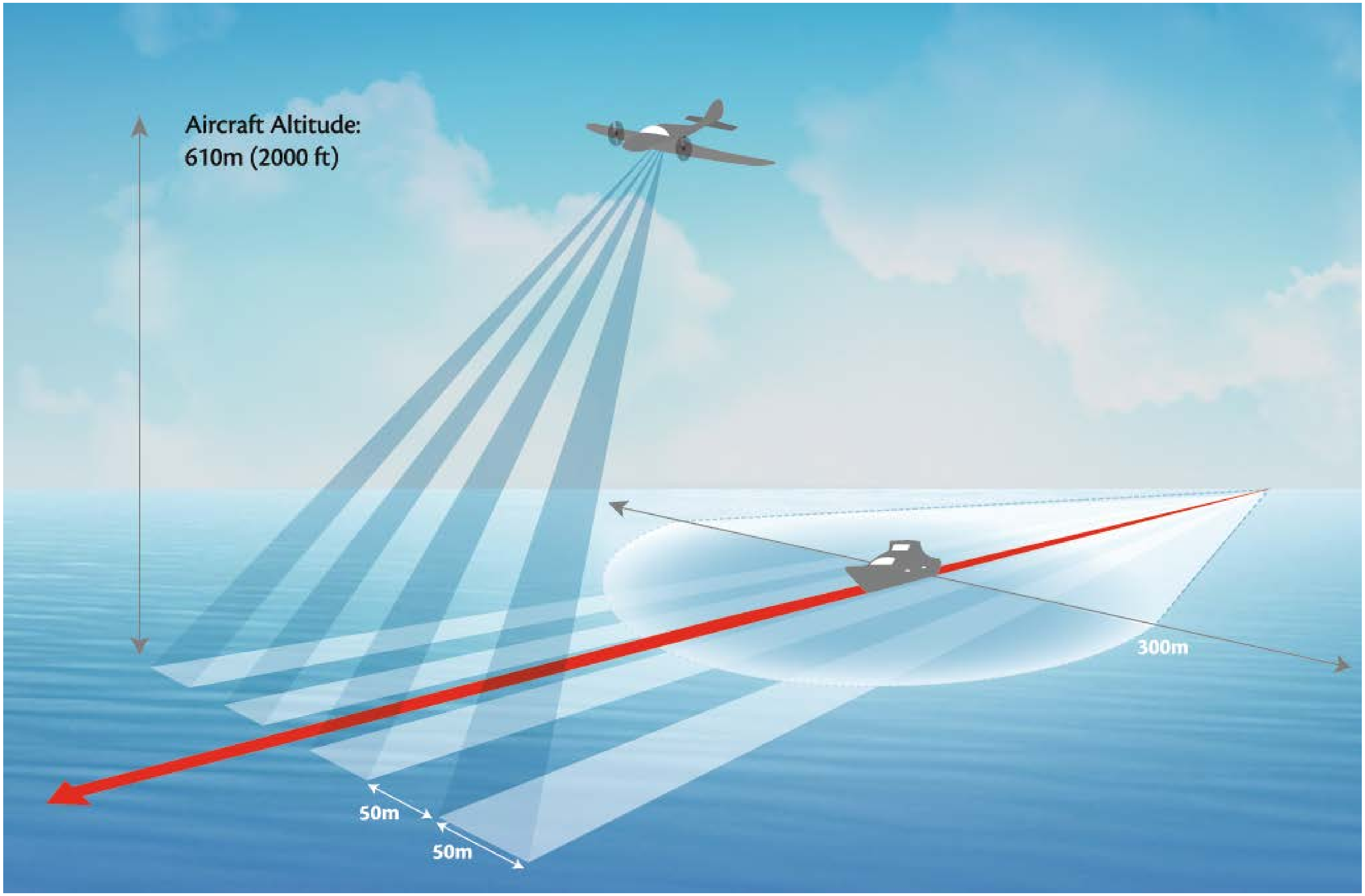

**Figure II. Diagram showing the fields of view available during boat surveys and digital video aerial surveys.** The combined strip width for the four video cameras is 200 m; the boat transect has an intended minimum strip width of 300 m, although observations of animals were made up to 1,000 m from the vessel. (Note that apart from the experimental comparison described in Chapter 13, the survey boat and planes followed different transect lines; see Chapters 3 and 7).

# Chapter 13: Integrating novel and historical survey methods: a comparison of standardized boat-based and digital video aerial surveys for marine wildlife in the United States

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 13 Highlights

*A standardized comparison of boat-based and digital video aerial surveys for marine wildlife*

### *Context*[1]

All field survey methods have strengths and weaknesses and understanding the nuances of a new method can be challenging. Recent advances in aerial survey methods in Europe use digital video and photography to collect distribution and abundance data for wildlife in the offshore environment. In Part II of this report, we discuss the first broad-scale application of high resolution digital video aerial surveys in North America. Part III of this report focuses on the use of standardized boat-based surveys with distance estimation, a well-established method of obtaining density data for wildlife. In Part IV, we focus on comparing and integrating data from these two survey approaches. With the help of project collaborators (HiDef Aerial Surveying, Ltd. and the City University of New York), BRI conducted an experimental comparison of boat-based survey and high resolution video aerial survey methods in 2013. A more general comparison of the two methods (using two full two years of survey data, but with boat and aerial surveys conducted at different times and locations), is presented in Chapter 14.

### *Study goal/objectives*

We compare two alternative survey methods for assessing the distribution and abundance of wildlife offshore, and explore specific challenges faced in implementing digital video surveys in the U.S.

### *Highlights*

- Compared results from simultaneous boat-based and digital video aerial surveys on transects off the mid-Atlantic coast of the U.S., using experimentally controlled methods.
- Most taxa were identified to species more often from the boat than in the video aerial data. An exception was scoters (*Melanitta* spp.), which were more often identified to species from the air.
- Northern Gannets (*Morus bassanus*) showed no significant effect of disturbance from the survey vessel, but 21% fewer scoters were observed in areas recently surveyed by the boat.
- Abundance estimates using boat data were higher than those from aerial data, likely in part because boat data were corrected for distance bias, as well as the poorer spatial coverage and greater velocity of the plane in this particular study.

### *Implications*

These two methods each have complementary strengths and notable weaknesses, and the optimal survey approach will vary based on location, species, and study goals. Despite a short-term displacement of some species by the survey vessel, boats will continue to provide a useful survey platform. The archivable and auditable nature of the digital survey data may be attractive to developers and regulators, particularly as the limitations of this method are ameliorated by technological advances.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

Introducing a novel field methodology based on advancing technology can be desirable when it improves the quality of data or simplifies data collection methods, but it can also complicate assessment of long-term changes to habitats or populations. Though it has recently become a common survey methodology for offshore wind energy development in Europe, surveying offshore wildlife distributions using high resolution videography is a new technique in North America. To assess its effectiveness and experimentally compare the results of this survey method with traditional boat-based surveys, we conducted a boat-based survey off the coast of Virginia in 2013, while a survey aircraft using high resolution video repeatedly flew the same transect lines over the same time period. Rates of species identification varied widely between survey methods; boat-based observers were better able to identify most animals to species level than were aerial video reviewers, with the exception of scoters (*Melanitta* spp.), a sea duck family. Though Northern Gannets (*Morus bassanus*) did not appear to be displaced by the survey vessel, an estimated 21% fewer scoters were observed in the aerial survey of a given transect segment after the boat had passed through it, suggesting substantial disturbance to surveyed populations of this species. Population size was estimated for scoters and Northern Gannets from both survey platforms; these estimates were reasonably well-correlated for scoters (Spearman's correlation =0.68), but were not well-correlated by location for Northern Gannets. The inexact temporal overlap between methodologies, due to differing survey speeds, likely contributed to this poor correlation for highly mobile Northern Gannets. For both scoters and Northern Gannets, population size estimates developed from the boat data were higher than aerial estimates in almost all locations; differences may have been partially related to the poorer spatial coverage and greater velocity of the plane in this study.

Boat-based surveys and digital aerial methods have complementary strengths, and also both have notable weaknesses; the optimal survey approach will vary based on location, species, and study goals. Several weaknesses of the digital aerial surveys used in this comparison (low availability for highly mobile species, poor species identification rates for some taxa, and a lack of analytical methods for addressing detection bias) could be ameliorated by technological or analytical advances in this field. This study provides key comparative data for integrating digital video aerial approaches into the suite of survey methodologies used in North America, and maintaining continuous historical records on seabird and marine mammal distributions and abundance.

## Introduction

Introducing a new field technique into an established discipline requires identifying the technique's advantages (and disadvantages) relative to more established methods, developing appropriate analytical approaches, and determining how best to integrate newly collected information with existing bodies of historical data. Though it is often desirable to introduce new methods that can improve data quality or simplify data collection, this presents a particular challenge for maintaining long-term datasets that can be used to track population- or landscape-level changes in the environment. Based on the efficiencies of cost on a broad scale, for example, it has been suggested that digital methods could largely replace visual surveys for offshore wildlife (Buckland et al. 2012). Understanding the nuances of a new method can be extremely challenging, however, as can acceptance of new methods and the integration of data collected via new methodologies into existing, long-running bodies of knowledge.

Gathering accurate information on the abundance and distribution of marine wildlife in space and time is increasingly necessary to assess the effects of environmental and ecological stressors on marine ecosystems, and to inform marine spatial planning and conservation efforts. Estimating spatial patterns in relative abundance in the offshore environment can be difficult, as these systems are extremely dynamic, animals tend to show high degrees of spatial autocorrelation or aggregative behaviors, and surveys are logistically challenging and more expensive than terrestrial equivalents. In the past century, offshore distributional surveys have mostly been carried out by direct visual observation of wildlife from boats or aircraft. Standardized methods using strip or line transects are common for monitoring marine species on both boat-based surveys (Tasker et al. 1984, Camphuysen and Garthe 2004, Camphuysen et al. 2004, Gjerdrum et al. 2012) and aerial surveys (Camphuysen et al. 2004, Certain and Bretagnolle 2008), and have been refined over the last few decades to achieve more accurate estimates of population size (Buckland et al. 1993, Buckland et al. 2001, Strindberg and Buckland 2004, Certain and Bretagnolle 2008, Evans and Hammond 2004, Kaschner et al. 2012).

In this study, we compare a new technique for monitoring offshore animals in North America, high resolution digital video aerial surveys (also "digital video aerial surveys"), to a more traditional approach. Digital aerial surveys using video and stills have been developed and successfully deployed in Europe to assess marine wildlife populations in relation to offshore wind energy development (e.g., Groom et al. 2013, Buckland et al. 2012). Very little work has been published comparing these new surveys to traditional distance sampling methods of aerial survey in that region, and results regarding the precision and magnitude of resulting relative abundance estimates are contradictory (Burt et al. 2009, Buckland et al. 2012, Webb and Hawkins unpubl. data). Such work is particularly important for species thought to be most vulnerable to offshore wind energy development; European vulnerability assessments for seabirds have prioritized several taxa that are also present in North America, including loons (*Gavia* spp.), scoters (*Melanitta* spp.), and Northern Gannets (*Morus bassanus*; Garthe and Huppöp 2004, Furness et al. 2013).

In the western hemisphere, we have no significant prior experience with this method, particularly with how detection rates, identification rates, and estimates of population size may differ between digital video aerial surveys and more traditional survey methods. The only published study to use the technique on a broad scale in the western hemisphere described an offshore movement of bats in 2012 (Hatch et al. 2013). The only relevant comparison of survey methods that has occurred to date in the United States employed a methodology for digital aerial surveys that has not been used elsewhere (Normandeau Associates, Inc., 2013), and varied in several ways from the approach presented in this study. Thus, this technique is new for North American species, and studies comparing it to traditional techniques in the peer-reviewed literature across the globe have been few and limited in scope. Such work must be taxonomically specific, as different survey strip widths and speeds can interact with animal movement behavior (primarily through the process of availability to the survey) to create species-specific bias in detection rates (Certain and Bretagnolle 2008, Spear et al. 2004).

There are some distinct advantages to high resolution digital surveys over their more traditional counterparts (Buckland et al. 2012) that could cause researchers to switch to these methods consistently in the Americas. First, they are considered safer than traditional aerial surveys, with plane speeds above 100 knots (185 km/hr) and survey altitudes above 500 m (as compared to common flight altitudes for visual aerial surveys, which can range from 60-180 m; Camphuysen et al. 2004, Certain and Bretagnolle 2008). Second, their comparatively high survey altitude limits disturbance on the wildlife being surveyed, unlike both boat-based and traditional aerial surveys (Mosbech and Boertmann 1999, Schwemmer et al. 2011). Aerial surveys also do not attract wildlife being surveyed, a phenomenon that could bias abundance estimates from boat-based surveys for certain species, such as scavenging seabirds (e.g., Hyrenbach 2001, Votier et al. 2013). Third, these surveys can be cost effective at large spatial scales, so much so that digital aerial surveys are expected to largely replace visual surveys, by either boat or aircraft, in the offshore environment around the U.K. and other western European countries (Buckland et al. 2012). Fourth, a digital record of all surveys is kept so that they can be reanalyzed or reassessed at later points in time, which may be of particular importance for phased infrastructure development scenarios or for species with changes to their conservation status. But such advantages come with potential trade-offs for digital video aerial surveys, including a relatively narrow and defined strip width and thus more variable encounter rate (Burt et al. 2009), decreased detection or identification rates for some taxa, less detailed behavioral information, and infrastructure challenges associated with the management of large quantities of video data (though it should be noted that recent technological advances have increased strip width and identification rates substantially; Webb and Hawkins 2013).

To better understand the application of digital video aerial surveys generally, and the specific challenges faced in implementing the technique in North America, we compared results from simultaneous boat-based and digital video aerial surveys on transects off the mid-Atlantic coast of the U.S., using experimentally controlled methods. Due to the survey design used in this comparison, we were also able to examine evidence of disturbance by the survey boat. For taxa with sufficient observations, we compared identification rates and population size estimates between the two methods. This comparison provides the first information of its kind for North America, improves our understanding of the relative utility of boat and video aerial surveys, and will assist in integrating data derived from new survey technologies into historical databases. Such integration of new and traditional methodologies is essential in order to maintain consistent data on marine wildlife distributions and relative abundance, and enable detection of long-term trends.

## Methods

The comparison survey was conducted on March 22, 2013, off the coast of Virginia, USA (Figure 13-1). Two parallel transects were located 10 km apart and between 5.5 and 64.6 km from shore (5.3-36.9 m water depth). Total combined transect distance was 109.6 km. Weather conditions were conducive to surveys using both platforms, with light winds and no low cloud cover, mist, or fog.

### *Boat-based survey*

The boat-based survey was carried out by the City University of New York and the Biodiversity Research Institute, with experienced observers working from the deck of a 17 m long charter vessel. The survey was conducted in "passing mode", where the vessel stayed on a predetermined transect line and at a constant survey speed (10 knots, 18.5 km/h), except when avoiding other vessel traffic or complying with National Marine Fisheries Service (NMFS) rules about approaching marine mammals. This method was largely compliant with European Seabirds At Sea (ESAS; Camphuysen et al. 2004) and Eastern Canadian Seabirds at Sea (ECSAS; Gjerdrum et al. 2012) standards for boat-based surveys and was comparable to many other boat-based surveys conducted in the United States and elsewhere.

During surveys, teams of two observers alternated two-hour observation periods and used line transect methods to observe and record animals. The two observers were stationed on the flying bridge in most circumstances, and moved to the pilot house when wind speeds increased to the point that salt spray could damage the computer used to record observations. A continuous watch was maintained by one observer, who counted all animals within at least a 300 m bow-to-beam arc to one side of the boat. The second observer recorded the observations on a laptop computer and also watched outside the strip transect for cetaceans and sea turtles. Each record included data on species identification, number of animals present, behavior, radial distance from the boat and degree of the animal's angle to the bow of the boat (from the location of first detection), direction of movement, and, where possible and appropriate, age and plumage/molt state. The second observer was consulted in cases of uncertain identifications. Radial distance and angle data allowed for the use of detection functions to estimate densities (Buckland et al. 2001). The computer program dLOG3, specifically designed for seabird and marine mammal surveying (Ford 2009), was linked to a GPS and used to record the vessel track; location data were recorded approximately every 5 seconds, and each animal observation was individually georeferenced. Sea state was recorded hourly using the Beaufort Wind Force Scale.

### *Digital video aerial survey*

The aerial survey was carried out by HiDef Aerial Surveying, Limited, a technology development company based in the United Kingdom. The survey was conducted from a multi-engine Cessna 300 series aircraft equipped with the company's first generation camera system, consisting of four super high resolution digital video cameras facing forward or backward (depending upon time of day, to reduce glare) at 30-45° from vertical, in a specially designed and patented air frame secured to the lower fuselage. Surveys were flown at an average speed of 250 km/h and 610 m above sea level, according to standard protocols (Hatch et al. 2013). The four cameras were set to 2 cm Ground Spatial Resolution

(GSR) and had non-overlapping 50 m strip widths, for a total strip width of 200 m. Cameras captured up to 15 frames per second. At one second intervals, position data for the aircraft were captured on a Garmin GPSMap 296 (Garmin International, Inc., Olathe, KS) receiver with differential GPS enabled to give 1 m precision for each position. Due to the height at which surveys were flown, there was little risk of affecting the behavior of animals at or near the water's surface, and no permits were required from the National Marine Fisheries Service (NMFS). Flights complied with all Federal Aviation Administration (FAA) regulations. Recorded images were stored on heavy duty disk drives or solid state recording devices for subsequent review and analysis.

Digital video data were manually reviewed to identify segments containing objects (including wildlife, boats, etc.), in line with HiDef's typical quality control procedures (Chapter 4). The review process was audited by experienced staff using a 20% blind sampling audit methodology. Once this process was complete, video footage was examined by trained biologists to identify objects to the lowest possible taxonomic level (either species or species grouping). Observers used a three tiered system to rate certainty in their identifications (Hatch et al. 2013). Twenty percent of object identifications were independently reviewed to determine the rate of agreement among analysts; a third reviewer examined all objects for which the original analyst and the second reviewer disagreed. Completed analysis provided data on the number of target organisms in the video, the species or species grouping of organisms, the approximate flight height for flying animals (after Hatch et al. 2013), and geospatial data for all biota.

### *Comparing survey types*

Due to the difference in survey speeds between the two platforms, when the plane reached the end of the first transect line (after passing directly over the boat), it turned around and repeated the transect in the opposite direction. In all, the plane travelled the length of each transect six times in the time it took the boat to traverse it once, and these replicates were treated as replicates of the same transect in analysis. Given the distance between transects, and the narrow strip width involved, the likelihood of animals moving between transects (and the risk of double-counting individuals) was considered negligible in aerial surveys. While the boat moved between transects, the plane landed to refuel, and then repeated the process on the second transect. In total, the plane surveyed for three hours and 35 minutes (9:15-11:15 a.m. and 1:10-2:45 p.m.), while the boat surveyed for six hours and 54 minutes (8:45 a.m.-3:39 p.m.). While exact spatial and temporal overlap between the two survey types was not possible, except on twelve brief occasions when the plane was directly above the boat, the close spatial and temporal proximity afforded by this design allowed the boat and aerial surveys to sample approximately the same population of animals over the same time period.

To compile boat and aerial datasets for analysis, objects identified with >50% confidence in the aerial dataset were included at the specified identification, while objects identified with <50% confidence were considered at the next lowest taxonomic level (e.g., a "possible Black Scoter," *Melanitta americana,* became an "Unidentified Scoter" for analysis purposes; Hatch et al. 2013). Boat-based surveys had no confidence rankings for identifications, and all observations were taken at 100% certainty.

### *Species identification rates: all taxa*

We used the raw counts from each survey to examine identification rates derived from the two survey platforms (Appendix 13A). Within each family (Alcidae, Anatidae, Gaviidae, Laridae, Sulidae, and Delphinidae), the proportion of observations in which animals were identified to the species level (e.g., "Common Loon", *Gavia immer*) vs. the group level (e.g., "unidentified loon sp.") was compared between survey methods.

### *Population size estimates derived from boat vs. aerial surveys: scoters and gannets*

Survey transects were divided into 2.5 km segments for analysis ($n$ = 25; Figure 13-1). For each segment, we developed estimates of population size for the two species groups with sufficient data for analysis: scoters (including Black Scoters; Surf Scoters, *Melanitta perspicillata*; and unidentified scoters) and Northern Gannets. Boat-based survey estimates of total number of individuals per segment were developed using the 'distsamp' function in Package 'unmarked' in the R Statistical Environment (Royle et al. 2004, Fiske and Chandler 2011, R Core Team 2014). Distance bands were set to 0-50 m, 50-100 m, 100-200 m, 200-300 m, 300-500 m, and 500-800 m. Wind speed (m/s) from a nearby weather station (Figure 13-1) and visibility (a categorical variable recorded by boat observers) were used as covariates to detection. No covariates to abundance were included in the model, as there was insufficient data to parameterize the model to this degree.

For video aerial surveys, it was unclear whether there was a variable like distance from observer that affected detection. Analysis of digital aerial survey data in the published and gray literature has acknowledged variable encounter rates, due to diving animals and clumped distributions, but has generally assumed perfect detection (e.g., Burt et al. 2009, Buckland et al. 2012). Thus, to quantify variance within the raw aerial survey data in an initial analysis, we used the six aerial replicates to bootstrap segment-level estimates of average numbers of individuals per segment; 500,000 simulations were run in package 'bootstrap' in R (Tibshirani and Leisch 1993). While this approach provided error estimates, it did not allow for explicit estimation of detectability. Prior to comparisons of segment-level population size estimates between survey methods, all estimates were standardized to density/km$^2$ to eliminate the effect of differences in effort (e.g., strip width or effective strip width) between survey platforms.

### *Disturbance to wildlife populations from the survey vessel: scoters and gannets*

In addition to estimating abundance (or variance around counts in the case of aerial data), we conducted a separate analysis for the same two species groups (scoters and Northern Gannets) focused on the effect of disturbance on raw count data. We specified a generalized linear mixed model (using package 'lme4' in R version 1.1-7; Bates et al. 2014) that assumed a Poisson distribution for raw aerial survey counts and used a Laplace approximation to calculate likelihood. A negative binomial distribution was also examined, but did not improve model fit (likely because of limited zero-inflation in the data). Segment was included as a random effect to control for unidentified spatial variation. Wind speed and survey replicate (categorical) were added to the model as fixed effects; wind speed was averaged for each segment of each replicate. A categorical 'disturbance' variable was also added to the model; each replicate of a given segment was categorized as 'disturbed' if the survey boat had passed through it <1 hr prior to the passage of the plane, or 'undisturbed' if the boat had not yet entered the segment or had passed through the segment >1 hr prior to the plane. A single model with all abovementioned covariates was tested in this analysis and the importance of boat disturbance for influencing counts (either negatively or positively) was evaluated by a z-test of the parameter estimate from the model.

## Results

A total of 3,484 birds and aquatic animals were observed from the boat's single replicate of the two transects. During the six aerial survey replicates, a total of 2,711 birds and aquatic animals were observed (Figure 13-2), with an average of 451 (±75 SE) animals observed per replicate. The six aerial surveys allowed us to estimate coefficients of variation (CV) for each species group in each transect segment. Scoters had the lowest CV at 0.35; Northern Gannets had a CV of 0.58; and loons had the highest level of variation between replicates, at 0.69.

### *Species identification rates: all taxa*

Rates of identification to species varied by taxon and by survey platform. Overall, 11.6% of observations from the boat were at the species level, due to the large number of unidentified scoters in the boat-based survey; 45.7% of observations were identified to species from the aerial survey (Figure 13-2). Excluding scoters, observations from the boat were identified to species 100% of the time, while 51.2% of aerial observations were identified to species (Figure 13-2). Two auks (Alcidae) were observed in the aerial survey, but were not identified to species; none were observed from the boat (Figure 13-3). Video reviewers also had difficulty identifying gulls and loons to species (with identification rates of 16.7% and 0%, respectively), though 100% of observations from these families were identified to species by boat observers (Figure 13-3). In addition, 3.1% of aerial observations were not identified to the family level (i.e., were classified as "unidentified birds" or unidentified aquatic animals). Both survey platforms proved efficient in identifying Northern Gannets to species (100% identification rate in both surveys) and delphinids (81.8% identification to species from the air, and 100% from the boat).

### *Population size estimates derived from boat vs. aerial survey methods: scoters and gannets*

Population size estimates for scoters were much larger for nearshore segments than for areas farther offshore, particularly for segments near the mouth of Chesapeake Bay. Northern Gannets were more evenly distributed across the study area, and were present in much smaller numbers. In almost all cases, modeled population size estimates using boat data were higher than aerial estimates for the same locations (Figure 13-4). Boat and aerial estimates of scoter relative abundance were correlated by segment (Spearman's correlation = 0.68), but Northern Gannet abundance estimates were not well correlated between the two survey platforms (Spearman's Rho = 0.17; Figure 13-4).

### *Disturbance to wildlife populations from the survey vessel: scoters and gannets*

Ten percent more Northern Gannets were observed in 'disturbed' segments than 'undisturbed' segments in the aerial footage, though the effect was not statistically significant (Beta = 0.09; 95% CIs: -0.332, 0.519, $z = 0.43$, $p = 0.66$). Scoters showed a significantly negative relationship to disturbance, with 21% fewer scoters present in 'disturbed' segments (Beta = -0.23; 95% CIs: -0.289, -0.117, $z = -4.2$, $p < 0.0001$).

## Discussion

The traditional boat-based survey method and the high resolution digital video aerial survey method each had clear strengths and weaknesses, with overall counts and identification rates varying considerably by taxon as well as survey platform. Abundance estimates derived from both survey platforms were much higher for scoters located in segments near the mouth of Chesapeake Bay, and were well-correlated between the two survey platforms. Northern Gannets were more widely dispersed across the study area, but relative abundance estimates between the two survey platforms were less well correlated for this species. The boat provided better species identification capabilities for many species groups than did the aerial video, but the boat also caused substantial disturbance for some taxa, potentially complicating both identification efforts and abundance estimation for scoters, the most abundant species group in the survey. Digital video aerial surveys, on the other hand, are considered to be less affected by observer biases than visual surveys (Normandeau Associates, Inc. 2013). In their assessment of a slightly different digital aerial technology in North America, Normandeau Associates, Inc. (2013) concluded that “aerial high-resolution digital imaging is likely to produce superior animal detection, density calculation, and taxonomic identification accuracy compared with conventional visual observer surveys from either boat or aircraft.” While not all of these predictions were borne out by our results in this study, there are distinct advantages to high resolution digital surveys over their more traditional counterparts (Buckland et al. 2012) that could encourage the widespread adoption of these methods in the Americas, particularly with further technological and analytical advances.

### *Identification rates: all taxa*

We found substantial differences in species identification rates between the two survey platforms. One observer participated in both the boat-based survey and video review for this comparison study, and two of the other three video reviewers participated in previous boat-based surveys in the region. Rather than inter-observer differences, we suggest that the disparity in species identification rates was partially due to differences in the quality assurance protocols that are applied to the two datasets. Observational data collected from the boat are not replicable, and the near-instantaneous species identifications required of observers can seldom be verified after the fact. The exhaustive quality assurance and audit protocol followed by aerial video reviewers, as well as characteristics inherent to the video review process itself (such as the use of multiple levels of “certainty” criteria in identifications), ultimately lead to fewer definitive identifications than do observational approaches. This is not to discount the differences in observation acuity between the two survey types; video aerial surveys have lower fidelity than the human eye, and do not allow for extended observation periods, which can facilitate the incorporation of behavior into identification protocols. As a result, the likelihood of identifying certain individuals to species in the video is almost certainly lower than from the boat. However, it is also a clear recognition of the inherent uncertainty in the identification process, which can be difficult to account for in unrecorded visual surveys. This uncertainty is generally under-recognized or ignored, as it can be difficult to measure, but in some cases species misclassification in visual surveys may actually lead to less reliable density estimates than classifying animals as “unknown” (Conn et al. 2013), as was more frequently done in digital aerial surveys in this study. Identification rates in digital aerial surveys have also continued to improve with technological advances in the field; a new generation of cameras is now being used in Europe, which have much higher resolution and enhanced color rendition, with improved identification rates as a result (95% on average; A. Webb pers. comm.).

Northern Gannets and delphinids were highly identifiable using both survey methods, due to their large size and distinctive coloration. Scoters were the one taxonomic group in this study for which identification to species was significantly easier from the aerial video than from the boat, and we suggest that this may be partly due to the effects of disturbance from the survey vessel. Previous studies (as well as this study) have found that scoters and other sea ducks show variable flush distances by species, as well as variable recovery times to pre-disturbance behaviors or locations (Schwemmer et al. 2011, Mosbech and Boertmann 1999). We suggest that the low rate of identification of scoters to species from the boat in our study was largely due to the fact that 84% of scoters were observed >300 m from the boat; in fact, over 70% were in the 500-800 m distance band, and it can be difficult to differentiate species at this distance. The larger rafts of scoters in this survey were located close to shore, beyond the end of our survey transects; however, disturbance from the boat likely also affected the distance at which scoters were observed.

In contrast, all loons (Gaviidae) in this comparison study were identified to species by boat observers, and none were identified to species in the aerial video. This difficulty in differentiating loon species has not occurred to the same degree in digital aerial surveys in Europe to date (Hexter 2009). There is a substantial overlap in body size among loon populations using the mid-Atlantic coast of the United States during winter, however (Gray et al. 2014, Barr et al. 2000), which often prevented video aerial observers from distinguishing the two species. Offshore wind energy development in Europe is known to have caused displacement of Red-throated Loons (*Gavia stellata,* one of the few species for which consistent post-construction effects have been seen at multiple project sites; Halley and Hopshaug 2007, Petersen and Fox 2007, Langston 2013). As Red-throated Loons are a species of interest to U.S. regulatory agencies (USFWS 2008), the inability of aerial video data to reliably differentiate Red-throated Loons from Common Loons was a shortcoming of this survey approach in this study. Where both survey types are coincident, using the boat-based data to inform aerial species identification could provide an analytical solution to this problem in the future (Chapter 16; Johnston et al. 2014). Recent advances in camera technology would likely also ameliorate the issue to an extent.

### *Correlations in relative abundance estimates between survey platforms: scoters and gannets*

Relative abundance estimates for scoters were well-correlated between the two survey platforms, but less well correlated for Northern Gannets. Since Northern Gannets were highly identifiable to species and scoters to genus using both survey methods, identification bias appears to be an unlikely explanation for these differences. Gannet attraction to the survey vessel is a possibility, as this species is known to change behaviors relative to fishing boats from up to 11 km away (Votier et al. 2010, Bodey et al. 2014), but no statistically significant attraction to the boat was observed during this study. We suspect that there were three main factors at work that led to this difference: (1) during non-breeding periods and in daylight hours, Northern Gannets are generally observed in flight, and are highly mobile (Mowbray 2002), (2) exact temporal and spatial overlap between survey methodologies was not possible, due to substantial differences in survey speed, and (3) the study design utilized in this comparison yielded relatively low encounter rates for Northern Gannets in the aerial survey. Due to the species' mobility and the inexact temporal overlap between methodologies, it is likely that individuals were observed in slightly different locations between the boat and aerial surveys, particularly for aerial replicates that were conducted well before or well after the boat's passage through a given segment. In addition, the aerial transect strip width is narrower than the effective strip width for the boat-based survey (which is intended to have a minimum of 300 m, but for gannets and ducks was estimated to be between 379 and 571 m in this study). At the low density transect spacing in this study, it seems likely that aerial surveys were simply inconsistent at observing highly mobile species (such as Northern Gannets) between replicates. A similar issue has been found for highly mobile gull species using digital aerial survey approaches in the United Kingdom (Burt et al. 2010). Encounter rate variability has been noted to affect precision in abundance estimates in digital aerial surveys in the North Sea, and previous studies have recommended careful survey design, including high transect densities for digital aerial surveys, to reduce uncertainty and ensure sufficient statistical power to detect change (Burt et al. 2009, Buckland et al. 2012). This type of availability bias was likely less of an issue for scoters in this study, which were observed in large, more geographically stable aggregations. This same issue of availability, however, explains why distance-corrected boat-based estimates of seabird population size were consistently higher than bootstrapped aerial estimates for the same locations. Newer generations of camera systems have increased the strip width for aerial surveys from 200 m to 500 m to help address this issue (Webb and Hawkins 2013).

### *Disturbance from the survey vessel: scoters and gannets*

We expected that animal distributions could be disturbed by the boat survey vessel, as displacement by and attraction to survey vessels has the potential to bias estimates of population size made using a boat-based platform (Buckland et al. 2012). The boat did appear to affect distributions of scoters in the short term. A displacement effect has been observed in other studies of scoters, and the degree of geographic or temporal displacement appears to vary by species (Schwemmer et al. 2011, Mosbech and Boertmann 1999). This has the potential to lower detection rates of scoters and could negatively bias estimates of scoter population size, though we saw no clear evidence of such in this study. However, boat disturbance probably did play a role in the poor species identification rates for scoters. Northern Gannets showed no significant attraction or displacement in this study, though they are known to be attracted to fishing vessels under some circumstances (Votier et al. 2010). Attraction is possible for other taxa observed in this study as well, including dolphins, which were infrequently observed but were more common in the boat-based survey than in any single aerial replicate.

### *The future of offshore wildlife surveys*

The best methodological approach for studies of offshore wildlife will depend on the specific characteristics of each study area and project goals (Camphuysen et al. 2004), and may involve a combination of complementary survey methods. Digital aerial survey approaches have largely replaced visual aerial surveys for offshore wind energy research in Europe, as they are safer to conduct than visual aerial surveys, reduce or eliminate disturbance to wildlife as compared to visual aerial or boat survey approaches, and produce archivable data, which allows for a robust quality assurance and audit process. Boat-based surveys also have methodological strengths, including excellent (although generally unverifiable) identification rates for most species, and the ability to obtain more detailed behavioral data than is possible with digital aerial approaches. Detection bias is a known issue for boat-based surveys, but it is also an issue that is relatively well understood, and can be addressed in part with established analytical approaches (Buckland et al. 2001).

While digital aerial approaches were developed in Europe, this application of these technologies in North America demonstrates clear avenues for additional research and development. The species composition of ecological communities in the western Atlantic varies considerably in some cases from what is present in the North and Baltic Seas. Early indications suggest that digital aerial surveys may have distinct advantages over visual aerial or visual boat surveys for sea turtles, for example, a taxon of considerable interest in North America but one that rarely occurs in Europe (Chapters 14 and 15, Normandeau Associates, Inc., 2013). Even pan-Atlantic species may possess different characteristics in North America than in Europe, as evidenced by the large range of body sizes for Common Loons that winter in the mid-Atlantic United States (Gray et al. 2014, Barr et al. 2000), and the resulting difficulty in differentiating Red-throated Loons and Common Loons by body size in aerial video in this study. Additional exploration of species identification capabilities—for example, by conducting test flights over known-species flocks—could aid the future application of this technology in the U.S.

In addition, there is a need to further the development of analytical approaches for digital aerial surveys. Because the cameras are pointed down towards the water’s surface, digital aerial surveys avoid the common problem of distance bias; but, to date, other types of detection bias have not been addressed for digital aerial surveys. Further examination of detection rates (in relation to taxon, weather, sea state, time of day, and other factors) could be a fruitful avenue for methodological development. Existing audit processes for object location in aerial video could be easily modified to incorporate a double observer approach and lead to more statistically rigorous, accurate, and reliable estimates of abundance for North American populations.

There will continue to be a role for boat surveys in North America, as they can provide more detailed information on behaviors (and in some cases, species identifications), and can be cost-effective in smaller geographic areas, particularly close to shore. However, at the proposed geographic scale of offshore wind energy build out in the United States (USDOE 2011), it will also be essential to explore more efficient survey alternatives. The key will be to successfully integrate data from these newer survey platforms, in order to ensure compatibility among future studies, maintain a continuous historical record, and enable the assessment of long-term changes in wildlife distributions and abundance. Boat survey data from historical databases such as the Compendium of Avian Information, maintained by the U.S. Fish and Wildlife Service, are being used to assess baseline wildlife distributions (O’Connell et al. 2009, Kinlan et al. 2012), but those baselines will only be relevant for assessing future change if they can be interpreted in combination with data derived from more recent, technologically advanced approaches. This issue has remained largely unaddressed in European studies to date, in part due to the low precision of early abundance estimates developed from digital approaches (Burt et al. 2010). The technology has advanced dramatically in recent years, however, and the development of calibration factors will be essential to ensure comparability between survey datasets and facilitate the deployment of this technology in North America.

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## Figures

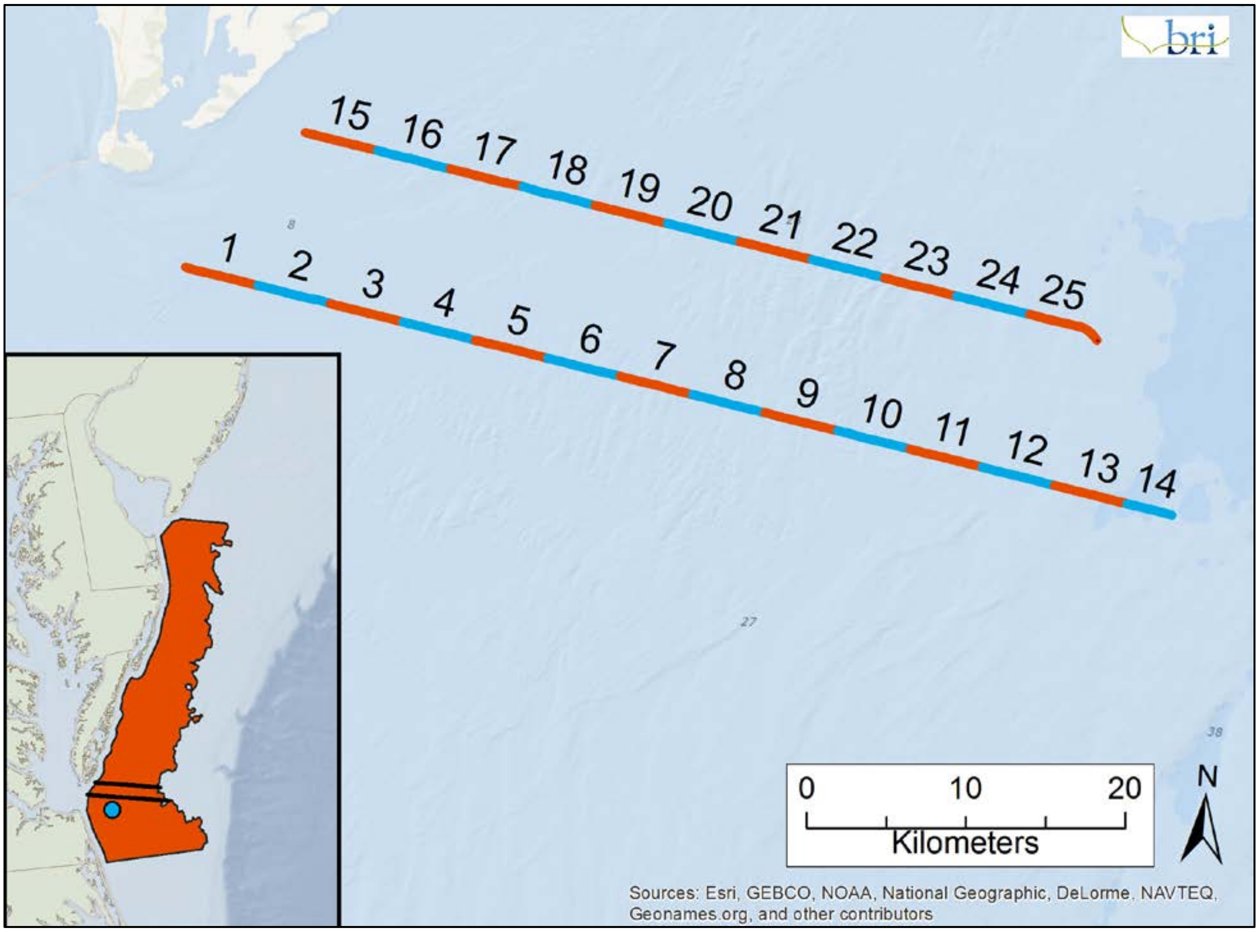

**Figure 13-1. The comparison study occurred on two transects, 5-66 km offshore of Virginia.** Transects were subdivided into 25 segments 2.5 km long. The inset map shows the broader project study area, the two transect strips (in black), and the location of the Chesapeake Light weather station (blue circle).

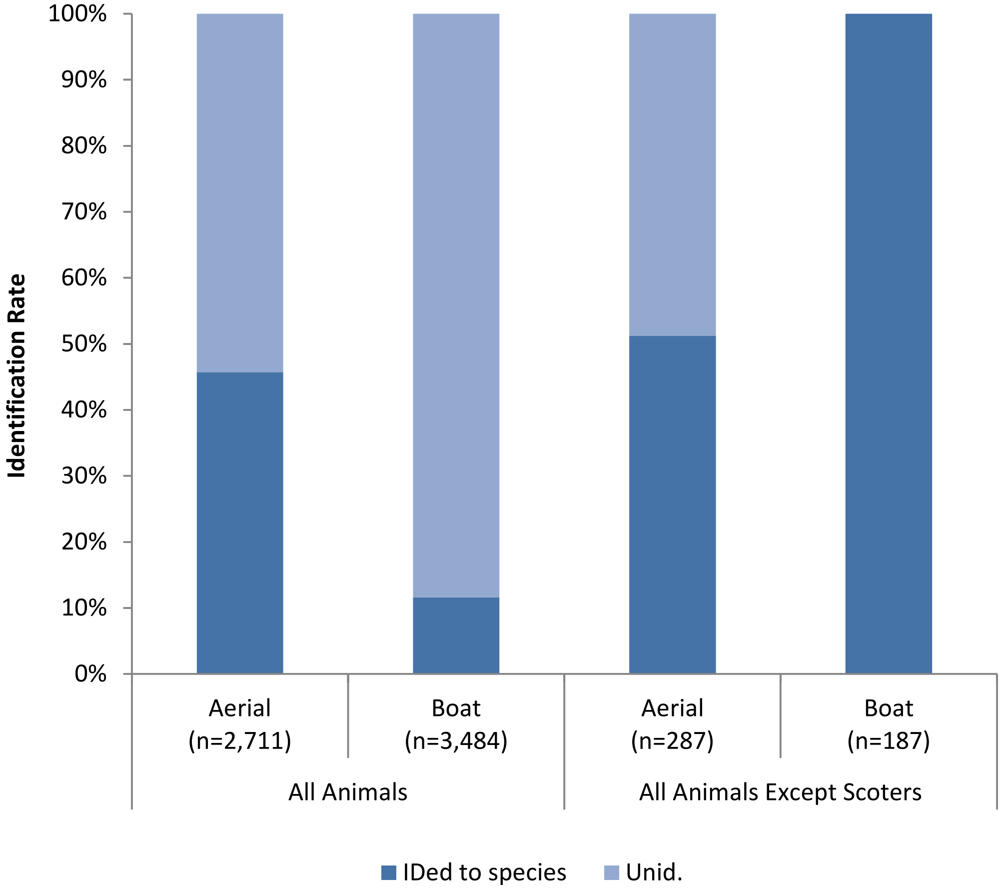

**Figure 13-2. Rate of identification for all animals (with and without scoters) in the boat and digital video aerial surveys.** Sample sizes are noted below each category.

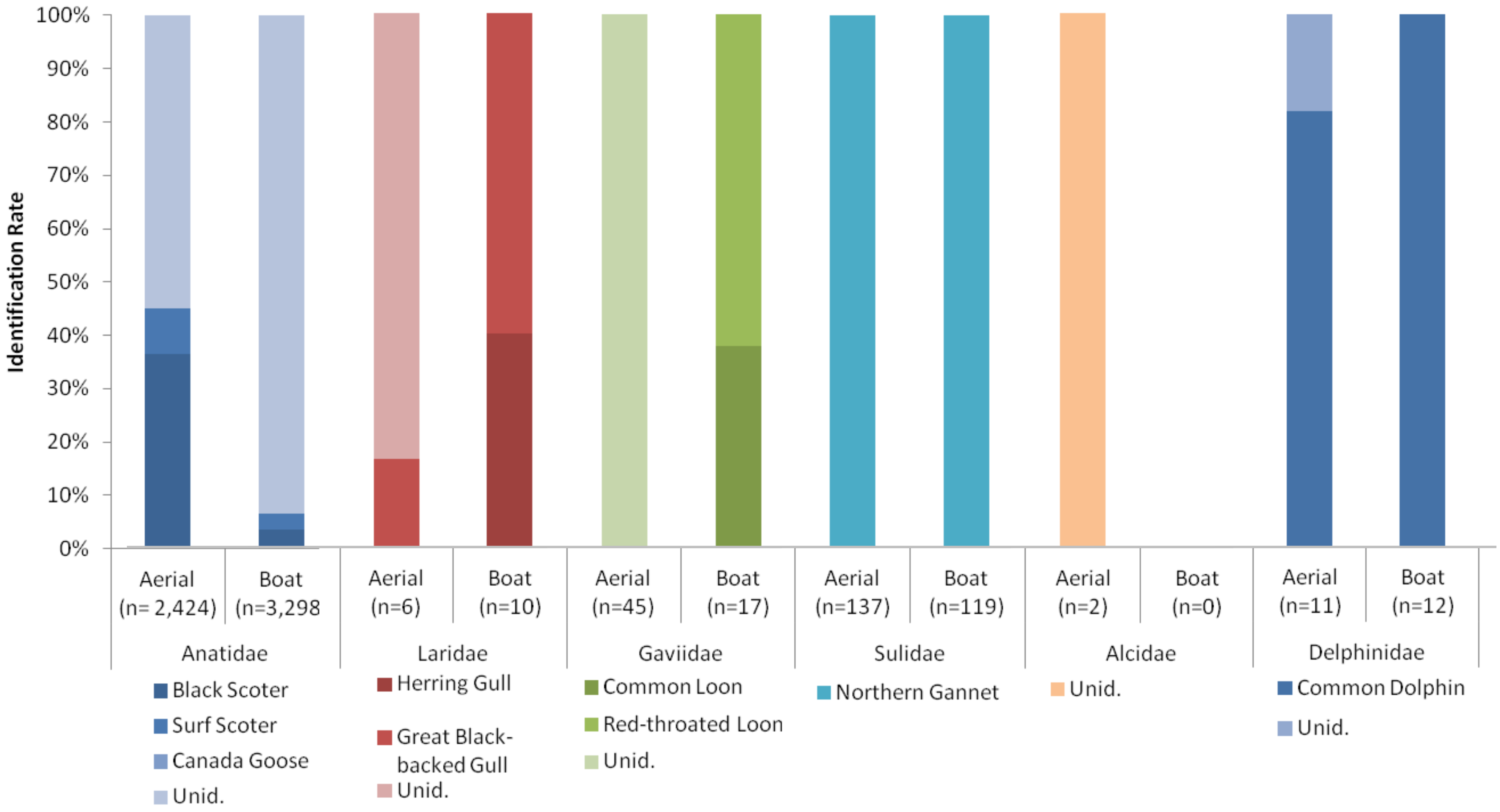

**Figure 13-3. Rate of identification for animal groups for the boat and aerial surveys.** Sample sizes and species included in counts are noted for each category.

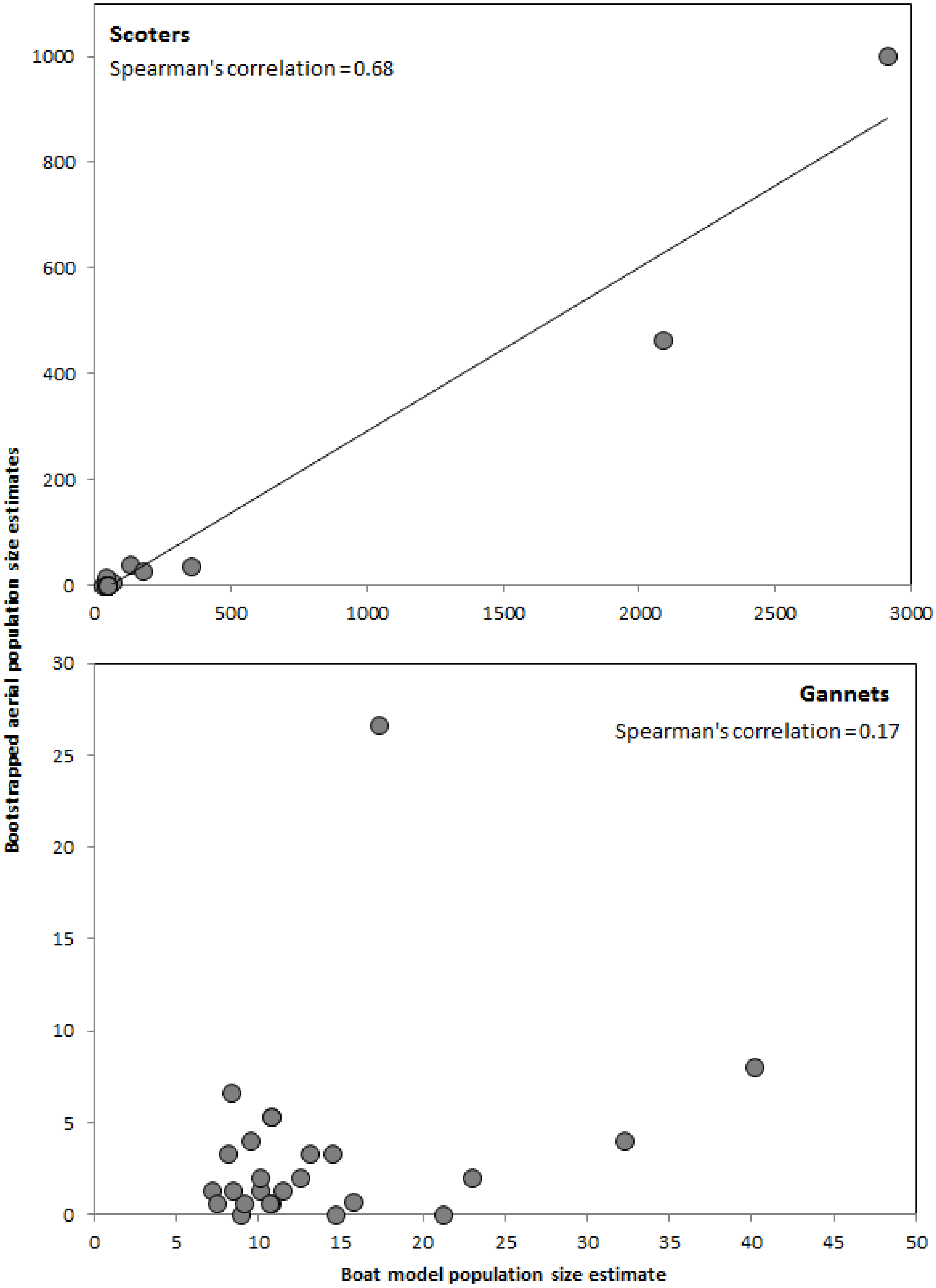

**Figure 13-4. Correlations between boat model population size estimates and bootstrapped aerial survey population size estimates, by transect segment, for scoters and Northern Gannets.**

# Supplementary material

## Appendix 13A.

**Table 13A-1**. **Boat-based and digital aerial survey counts of wildlife by transect segment and family.** All six aerial replicates are included in counts. Counts are not corrected for transect strip width.

| Segment | Alcidae | | Anatidae | | Delphinidae | | Gaviidae | | Laridae | | Sulidae | | Unidentified birds | | Unidentified aquatic animals | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat | Aerial (all reps.) | Boat |
| **1** | 1 | 0 | 707 | 1,246 | 0 | 0 | 5 | 7 | 1 | 0 | 46 | 6 | 9 | 0 | 0 | 0 |
| **2** | 0 | 0 | 1,520 | 1,732 | 0 | 0 | 2 | 8 | 1 | 2 | 9 | 2 | 17 | 0 | 0 | 0 |
| **3** | 0 | 0 | 56 | 178 | 0 | 0 | 2 | 3 | 2 | 0 | 6 | 6 | 7 | 0 | 0 | 0 |
| **4** | 0 | 0 | 12 | 17 | 0 | 0 | 3 | 6 | 0 | 1 | 1 | 2 | 4 | 0 | 0 | 0 |
| **5** | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 4 | 1 | 0 | 8 | 2 | 5 | 0 | 0 | 0 |
| **6** | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 1 | 0 | 5 | 0 | 0 | 0 | 0 |
| **7** | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 2 | 5 | 5 | 2 | 0 | 0 | 0 |
| **8** | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 1 | 0 | 0 | 6 | 3 | 3 | 0 | 4 | 0 |
| **9** | 0 | 0 | 0 | 0 | 0 | 4 | 1 | 2 | 0 | 0 | 2 | 5 | 1 | 0 | 0 | 0 |
| **10** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 2 | 0 | 1 | 0 | 0 | 0 |
| **11** | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 1 | 0 | 0 | 3 | 6 | 1 | 0 | 0 | 0 |
| **12** | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 1 | 0 | 1 | 1 | 2 | 0 | 2 | 0 |
| **13** | 0 | 0 | 0 | 0 | 1 | 1 | 4 | 1 | 0 | 1 | 7 | 8 | 1 | 0 | 1 | 0 |
| **14** | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 1 | 0 | 0 | 13 | 32 | 1 | 0 | 0 | 0 |
| **15** | 0 | 0 | 63 | 47 | 0 | 0 | 1 | 3 | 0 | 1 | 5 | 0 | 5 | 0 | 0 | 0 |
| **16** | 1 | 1 | 41 | 71 | 0 | 0 | 2 | 1 | 0 | 1 | 10 | 1 | 4 | 1 | 0 | 1 |
| **17** | 0 | 0 | 22 | 7 | 0 | 0 | 3 | 1 | 0 | 0 | 1 | 8 | 4 | 0 | 0 | 0 |
| **18** | 0 | 0 | 1 | 0 | 0 | 0 | 2 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **19** | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 2 | 2 | 9 | 0 | 0 | 0 |
| **20** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 1 | 3 | 2 | 0 | 0 | 0 | 0 |
| **21** | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 0 |
| **22** | 0 | 0 | 0 | 0 | 7 | 3 | 1 | 0 | 0 | 0 | 1 | 2 | 2 | 0 | 0 | 0 |
| **23** | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 |
| **24** | 0 | 0 | 0 | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 3 | 12 | 0 | 0 | 0 | 0 |
| **25** | 0 | 0 | 2 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 3 | 1 | 0 | 0 | 0 |
| **Totals** | **2** | **1** | **2,424** | **3,298** | **11** | **12** | **45** | **45** | **6** | **10** | **137** | **120** | **79** | **1** | **7** | **1** |

# Chapter 14: Summary of boat and aerial datasets: comparison between survey methods

### Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 14 Highlights

*Examining differences in observations and species identifications between the boat-based and digital video aerial survey datasets*

### *Context*[1]

Digital video aerial surveys and boat surveys were both used to collect data on marine animal abundances and distributions in the mid-Atlantic study area, and the results of each study provide complimentary information about the ecology of the region. Digital video aerial surveys are described in detail in Part II, with information on the methods used to collect and analyze the survey data (Chapters 3 and 4), and the results of the surveys (Chapter 5). Part III describes the boat surveys, with methods outlined in the protocol (Chapter 7), and a summary of the results (Chapter 8). Subsequent chapters in Part III analyze the boat survey data in greater detail using statistical models.

Part IV of this report examines ways to integrate the two survey datasets, using a variety of methods. Chapter 13 details an experimentally controlled comparison study completed in March 2013, where the boat and plane surveyed the same transect lines on the same day, allowing for a direct comparison of the two methods. The spatial and temporal coverage of this comparison were limited, however, as were the numbers of species observed, so this chapter compares the overall results of the two study methods across the two years of surveys to increase our understanding of how each of the methods can best be used to examine the marine environment. Subsequent chapters in Part IV use the two datasets together to develop more integrated views of wildlife distributions and abundance (Chapters 15-19).

### *Study goal/objectives*

Examine the differences between data collected during two years (2012-2014) of boat-based and digital video aerial surveys in the mid-Atlantic.

### *Highlights*

- More birds and more bird species were observed in the boat surveys, and birds made up a higher proportion of boat observations (98%) compared to digital video aerial surveys (43%).
- Scoters were the most abundant avian group observed using both study methods.
- Rays were the most abundant animal observed in the digital video aerial surveys, but were rarely observed in the boat study.
- More sea turtles were observed in the digital video aerial surveys, with more species observed; in both methods, turtles were most abundant in warmer months (spring through fall).

### *Implications*

Both survey methods have distinct strengths and weaknesses, though they showed similar overall patterns for avian species. Digital video aerial surveys appear to be particularly good for observing sea turtles and other aquatic animals, while boat surveys generally had higher identification rates.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

High resolution digital video aerial surveys are a relatively novel method for collecting information on marine wildlife distributions and abundances, and this study is the first to use these methods on a broad scale in the United States. In contrast, standardized boat based surveys are widely used to collect information on marine animals, and biases inherent to this survey approach are well understood. Our study focused on collecting marine bird, mammal, and sea turtle data within the mid-Atlantic study area using both methodologies in 2012-2014. More birds, and more species of birds, were observed in the boat surveys, while much higher numbers of aquatic animals were observed in the digital video aerial surveys; birds made up a much higher proportion of the animals observed in boat surveys as compared to digital video aerial surveys. Similar avian species were found to be abundant in the study area according to both methods, but digital video aerial surveys observed more rays and turtles than the boat surveys, where the most abundant aquatic animals were toothed whales. Identification rates were notably different between study methods, with higher rates of animals identified to species level on the boat surveys. Sea turtles provided an interesting case study for comparing the study methods. Taken together, the two methodologies provide complementary information on marine animal abundances and distributions in the mid-Atlantic study area.

## Introduction

The mid-Atlantic region is an extremely important area for a broad range of marine wildlife species throughout the year. This is largely due to a relatively high level of productivity as compared with the rest of the western North Atlantic, and to the region’s geographic location on the eastern edge of the continent (Chapter 1). The mid-Atlantic area supports large populations of marine wildlife in the summer; some breed along the coastline west of the project study area, including some tern species, while others visit from the southern hemisphere in their non-breeding season, such as shearwaters. In the fall, many of the summer residents migrate south and are replaced by species that have travelled from their northern breeding grounds to winter in the mid-Atlantic. Additionally, many pelagic, coastal, and terrestrial species make annual migrations up and down the eastern seaboard and travel directly through the region in spring and fall. Thus, many species use or funnel through the mid-Atlantic region each year, resulting in a complex ecosystem where the community composition is constantly shifting and the temporal and geographic patterns are highly variable.

In this study, we aimed to produce the data required to inform siting and permitting processes for offshore wind energy development in the mid-Atlantic. We collected information on bird, sea turtle, and marine mammal abundances and movements over a two-year period (2012-2014) using a variety of technologies and methods to examine spatial patterns and trends, while simultaneously testing a new technology for the first time in the United States, high resolution digital video aerial surveys (hereafter digital video aerial surveys or digital aerial surveys). Digital video aerial surveys are a relatively new method for collecting distribution and abundance data on animals in the marine ecosystem (Thaxter and Burton, 2009). Although digital video aerial surveys have become common practice for offshore wind energy planning and monitoring in Europe, these baseline wildlife studies in the mid-Atlantic (funded by the Department of Energy and the state of Maryland) are the first projects to use these methods on a large scale in the United States. We also conducted boat surveys for wildlife within the study area on the

Outer Continental Shelf to accompany and compare with the data from the digital aerial surveys. Standardized boat-based surveys are a widely used method of obtaining density data for birds, sea turtles, and marine mammals (Camphuysen and Garthe, 2004; Gjerdrum et al., 2012; Tasker et al., 1984). A focused comparison study of the two methods was conducted in March of 2013 (Chapter 13), but we present a more general examination of the full datasets here, to provide details on a broader range of animals observed by boat and by digital aerial surveys throughout the study area over the two-year survey period. While the experimental comparison study in Chapter 13 was focused on comparing results of the two survey methodologies from the same location and time period, the diversity and number of animals observed during this experimental comparison was limited, and comparing the two full datasets can provide further insight into the relative utility and strengths of the two survey methodologies.

We examine the differences in observations and identification rates between the two study methods, with a particular focus on sea turtles. All five species of sea turtle present in the mid-Atlantic study area are listed as threatened or endangered under the Endangered Species Act. Fisheries bycatch affects Loggerhead Sea Turtles (*Caretta caretta*) and Leatherback Sea Turtles (*Dermochelys coriacea*) in the mid-Atlantic and directly negatively impacts their populations' survival (Murray and Orphanides, 2013). Turtles are also vulnerable to vessel collisions, particularly at higher ship speeds (Hazel et al., 2007). Relatively little is known about sea turtle hearing capabilities, or the effects of noise on these species, but the hearing range of the Leatherback overlaps with all noise-generating activities conducted during offshore wind development, and they can detect and react to noises of the same frequencies as those emitted during offshore wind construction (Dow Piniak et al., 2012; Lenhardt et al., 1983; Read, 2013), and the noise generated during offshore wind construction is thought to be a potential concern for this taxon (Michel, 2013).

## Methods

### *Data collection*

Details on data collection methods used in both the aerial and the boat surveys can be found in Chapters 3-4, and 7. Between March 2012 and May 2014, 15 digital video aerial surveys and 16 boat surveys were conducted in the mid-Atlantic study area (Figure 14-1). In the second year of surveys (March 2013 – May 2014), funding from the state of Maryland led to the addition of 747 km of high density aerial survey transects to the west and south of the Maryland wind energy area (WEA), and a total of approximately 12.5 km of additional boat survey transect at the western edges of three existing transect lines off of Maryland (Figure 14-1). Analyses below include data collected under both the Department of Energy-funded and Maryland-funded survey efforts.

Aerial observers indicated a degree of certainty for each object identified (Chapter 3). For the summaries below, all aerial identifications were taken at face value (e.g., an identified "possible Black Scoter [*Melanitta americana*]" was considered to be a Black Scoter, rather than an "Unidentified Scoter"; see Chapter 4 for additional information on certainty levels and identification criteria).

### *Observation rates*

Digital aerial survey data were easily effort-corrected to present observations per square kilometer surveyed, as aerial transects had a defined strip width (Chapter 3). For purposes of comparison with boat-based survey data, aerial count data were taken at face value, and were not corrected for distance bias or other potential biasing factors, apart from variation in survey effort (Chapter 13; Buckland et al., 2012). Boat surveys were designed to have a strip width of at least 300 m, but the *effective* strip width varied by taxon (Chapter 7). Detection of objects in boat surveys is known to vary with distance from the observer (Thomas et al., 2010), and thus species that were readily detected large distances away from the boat had a larger surveyed area, or effective strip width, than species that were generally only detectable near the boat. We calculated effective strip half widths for the four avian taxa where data were sufficient to parameterize a null distance model in package 'unmarked' in the R Statistical Computing Environment (R Core Team, 2014). These groups included Sulidae (gannets), Laridae (gulls and terns), Gaviidae (loons), and Anatidae (scoters, ducks, and geese; Figure 14-2). Effective strip width was calculated in 'unmarked' by applying distance-based detection functions (half-normal distributions) to species groups during distance modeling, and integrating the area underneath the distance curve. Because of specific properties of distance detection curves, this number is equal to the distance at which there is a 50% chance of detecting an object (Royle et al., 2004). Because we surveyed on both sides of the ship, this effective strip half-width was multiplied by two to obtain the full effective strip width for each species group. This value was multiplied by the total linear distance of the survey to estimate the effective boat survey area for each species group. For species groups with insufficient boat observations to fit a distance curve, we used the median observation distance as a proxy for the effective half strip width, as the two values appeared to be comparable for the species where we could calculate both values.

### *Identification rates*

We used the naïve counts from each survey to calculate identification rates for the data collected on the two survey platforms. Within each of the most commonly observed family groups, including Anatidae, Sulidae, Laridae, Gaviidae, Alcidae (alcids, including puffins, murres, and others), Procellariidae (shearwaters and fulmars), Odontoceti (toothed whales, including dolphins and porpoises), Testudines (sea turtles), and Mysticeti (baleen whales), the proportion of observations in which animals were identified to the species level vs. the group level (e.g., Common Tern, *Sterna hirundo*, vs. "Unidentified Tern") was compared between survey methods.

## Results

Our assessments of the boat-based and aerial survey data indicated that the two methods differed in their abilities to detect and identify certain taxa. We discuss these results in detail below.

### *Observation rates*

Boat survey observers detected larger numbers of birds per unit effort and more species of birds than the digital video aerial survey observers (Figure 14-3), while the digital video aerial surveys appeared to be better at detecting certain aquatic animals (Figure 14-4, Table 14-1). Birds made up a large proportion of the animals observed on the boat survey (98%) compared to the digital video aerial survey (43%; Figure 14-5). Scoters were the most abundant avian group identified during boat-based surveys

(35% of birds) and digital video aerial surveys (46% of birds; Figure 14-5). Gannets, loons, and gulls and terns were also commonly observed in both datasets, though the relative abundance of each taxon varied between the two survey methods (Figure 14-5).

Boat survey observers detected larger numbers of more species of birds than the aerial survey observers, while the digital aerial surveys appeared to be better at detecting certain aquatic animals (Figure 14-4, Table 14-1), including sharks, fish, and rays. While some of these animals were also observed in the boat survey, the aerial surveys provided an excellent platform for detecting and identifying animals within the upper reaches of the water column. In particular, higher counts and species diversity of sea turtles and mammals were detected on the aerial surveys (Chapter 5) than from the boat (Chapter 8).

Of the non-avian digital video aerial observations, the bulk of detections were rays, with many fish, toothed whales, and some turtles observed as well; in contrast, the most commonly observed aquatic species group in the boat data was toothed whales (dolphins and porpoises, Figure 14-5). Major migrations of Cownose rays (*Rhinoptera bonasus*) were observed in the aerial surveys but went undetected in the boat surveys; almost 48,000 rays were observed in aerial surveys in all, and 200 times as many rays were observed from the aerial surveys as from the boat surveys (Figure 14-4, Chapter 5). Both surveys detected bats, specifically Eastern Red Bats (*Lasiurus borealis*), though more bats were detected in the aerial survey, and the bats observed from the aerial platform were flying higher than could be detected from the boat (Hatch et al., 2013). Many schools of baitfish were observed in the aerial data, some spanning hundreds of meters, with peak observations occurring in July-September 2013. Schools of small fish were not measured nor individuals enumerated, but a total of 7,501 schools of fish of varying sizes were observed on the aerial surveys, while 50 were counted on the boat surveys. Baitfish schools were observed primarily in nearshore areas, and many were observed in the high density transect extensions offshore of Maryland in the second year of surveys, as well as in western extents of the sawtooth transects (Figure 14-6).

### *Identification rates*

There appeared to be differences in observers' ability to identify animals between the aerial and boat-based surveys in some cases (Figure 14-7). More than twice as many bird species were definitively detected in the boat surveys than from the air (Table 14-1), with many more aerial observations limited to the family or genus level of identifications. Gulls and terns, loons, and alcids all had much higher identification rates to the species level from the boat surveys than from the aerial surveys (Figure 14-7). Aerial observers were better at identifying scoters, ducks, and geese to species, however, which is likely due to boat observers' difficulty in differentiating large flocks of Black Scoters and Surf Scoters (*M. perspicillata*) at a distance (Figure 14-7, see Chapter 13 for a more detailed discussion). Observers from both survey types had similarly high identification rates of shearwaters and fulmars (Procellariidae).

As fish were not a focal taxon for research in this study, neither platform identified fish to species, aside from Ocean Sunfish (*Mola mola*); the aerial observers detected 168 sunfish, while the boat observers detected three. Identification rates of toothed whales were higher on boat surveys, but baleen whales had higher rates of identification from aerial surveys (Figure 14-8), and each method observed a few species that were missed by the other (Chapters 5 and 8). While we saw more large (baleen and unknown) whales in the boat-based surveys ($n$ =35) than in the digital video aerial surveys ($n$ =16), for example, we saw eight North Atlantic Right Whales (*Eubalaena glacialis*) in the aerial surveys versus one in the boat surveys (Chapters 5 and 8).

### *Case study: sea turtles*

Much higher counts and species diversity of sea turtles were detected on the aerial surveys than on the boat surveys (Figure 14-9). While there were higher identification rates of turtles on the boat survey (Figure 14-8), only two species of turtles were identified (Loggerhead and Leatherback Sea Turtles). Despite difficulties with differentiating some subsurface turtles in the aerial footage, video observers were able to identify three additional species of turtles (Kemp's Ridley, *Lepidochelys kempii*; Hawksbill, *Eretmochelys imbricate*; and Green, *Chelonia mydas*).

Turtle distributions shift according to temperature, as they are poikilotherms and are limited to certain water temperature ranges (Gardner et al., 2008; Chapter 15). There were times of year when turtles were far less abundant in the study area; as shown in Chapter 15 and the figures below, sea turtles had highest abundances from May through October. Overall, turtles were more abundant in the southerly survey transects, especially near the Virginia WEA (Figure 14-10 - Figure 14-12). Seasonal distributions varied between species groups, however. In the spring, Loggerhead Sea Turtles were found predominantly off the coast of Virginia, with a few individuals observed on the sawtooth transects further up the coast (Figure 14-10). One Leatherback and a few Green Sea Turtles were seen in Virginia and Maryland further offshore, and Kemp's Ridley Sea Turtles were seen mostly in the south (Figure 14-10). More Leatherbacks were observed in the summer compared to the other seasons, and while observations occurred mostly in the south, some were seen as far north as Delaware (Figure 14-11). Loggerheads were found further north in the summer as well, but all Kemp's Ridley and Green Sea Turtle observations were made off of Virginia (Figure 14-11). Turtles were much more evenly distributed up the coast in the fall than during earlier seasons; all five species were observed in the Virginia and Maryland WEAs during fall surveys. The only sightings of Hawksbill Sea Turtles occurred in the fall, in the Virginia and Maryland WEAs (Figure 14-12). There were only two turtle sightings in winter, one Kemp's Ridley and one unidentified to the species level, both located off of Virginia.

## Discussion

Overall, there were substantial similarities between the species groups detected via the two study methods. Scoters were the most abundant bird group detected in both studies, with gannets, loons, and gulls observed in high numbers from both platforms as well. Both platforms detected similar species of the different taxa observed. Chapters 18-19 continue to explore these similarities, with the goal of developing an integrated model that uses data from both survey platforms to yield more information about the study area than would have been possible through the use of either survey method alone.

However, there were notable differences in observation and identification rates between the two survey methods as well, which point towards differing strengths and weaknesses of the two methods (Figure 14-13). For example, there were more birds and more species of birds observed in the boat surveys, while aerial surveys detected many more aquatic animals. A similar efficiency in detecting and identifying sea turtles and marine mammals from high resolution digital aerial platforms (as compared to visual aerial or boat surveys) has also been observed elsewhere (Normandeau Associates Inc., 2013). Some of the discrepancies in observations point towards potential differences in detectability between the two survey types; for example, Northern Gannets (*Morus bassanus*) and larger gulls were visible at great distances from the boat survey, as observers could look from the vessel all the way to the horizon. Reviewers of aerial survey data, in contrast, could only see animals present in the narrow strip of the transect onscreen, and aerial survey speed was roughly 13.5 times that of the boat, potentially limiting onscreen appearances by highly mobile animals (Chapter 13). Boat surveys are also known to affect animal behavior, and possibly detections as a result. Gulls are often attracted to boats as potential sources of food, while scoters are sensitive to disturbance by boats (Buckland et al., 2012; Schwemmer et al., 2014), which we found to be the case in our comparison study (Chapter 13). Marine mammals are also known to be attracted to or disturbed by boats (Buckland et al., 2012; Mattson et al., 2005), and were potentially biased by the influence of the vessel's presence. Differences in identification abilities between survey methods may have also played a role in explaining lower detections for many avian taxa, as some aerial observations of birds were recorded as "Unidentified Bird" or "ID Impossible" due to poor image quality (Chapters 4-5).

Identification rates for birds observed in digital video aerial surveys were low compared to boat surveys. Low rates of aerial species identification were not altogether surprising for alcids and terns, given their small size and subtle differences between species. However, higher identification rates had been expected for loons based on results from European studies. Aerial video reviewers faced difficulties in differentiating the two loon species that use the mid-Atlantic during the non-breeding season, due to the high degree of suspected size overlap (particularly for birds sitting at the water's surface) in this time period and region of the U.S. (Gray et al., 2014). Additionally, the aerial results were analyzed using defined confidence level criteria, and were audited following an exhaustive quality assurance protocol. Both of these processes increase the amount of scrutiny given to identifications, which could result in lower identification rates. In contrast, boat observations are generally unverifiable and unable to be audited. The quality assurance and quality control protocol followed during analysis of digital video aerial survey data recognizes the inherent uncertainty in the identification process, which is generally under-recognized in visual surveys, as it is difficult to measure. At the same time, some of the lower identification rates in aerial surveys were, in the opinion of reviewers, clearly due to image quality, and this issue limits the utility of the digital video aerial surveys for describing the distributions of some taxa. The next generation of cameras being used in Europe have higher resolution and color rendition than the cameras used in this study, however, with increased identification rates as a result (A. Webb pers. comm.), so technological advances in the field may largely ameliorate this issue.

Detailed examinations of the cost differences between the two survey methods are beyond the scope of this comparison, but the cost effectiveness of either method over the other would depend greatly on the specific study design. Relative costs would vary based on distance from shore, the size of the area to be studied, the taxa to be targeted in the study, and key questions to be addressed with the survey data. The greatly increased speed at which the aerial surveys can be conducted (13.5x faster than the boat) means that they become more cost effective relative to boat-based surveys in situations that require longer times on the water, for example, in areas that are further from shore and/or over larger survey areas (Chapter 13; Buckland et al., 2012).

We examined sea turtles more closely to compare observers' abilities to detect and identify aquatic animals using the two study methods, and found the digital video aerial surveys to be particularly good for observing and identifying sea turtles in the mid-Atlantic. High resolution digital aerial platforms have previously been shown to be efficient means to detect sea turtles as compared to visual aerial or boat survey platforms (Normandeau Associates Inc., 2013). Looking directly down on the surface of the water likely allows for clearer views of submerged or partially submerged animals, and video capture allows for in-depth examination of the animals for key characteristics. The animals are also not disturbed in the same way that they would be by a boat or a low-flying airplane (Hazel et al., 2007; Normandeau Associates Inc., 2013). Given that all species of sea turtle in the mid-Atlantic are federally listed and are of conservation concern, more accurate counts and distribution data for these species (even if many of the observations are recorded as simply non-Leatherback unidentified turtles), are still extremely useful for resource managers. As mentioned above, cameras used in this study have already been replaced by better models in Europe, and continued technological improvements are likely to dramatically increase identification rates for this key taxon.

Given the seasonal distribution patterns found for sea turtles, it is clear that there is overlap between these species' observed distributions and the locations of planned offshore wind energy development (WEAs), in part because sea turtles in the mid-Atlantic display a generally more offshore distribution pattern (thus placing them in areas of potential exposure to development activities in federal waters; Chapter 15). Offshore wind construction is the development period with the most risk to sea turtles, due to noise from pile driving and other activities, as sea turtles can detect and react to low-frequency sounds of the same frequencies as those emitted by seismic airguns, offshore drilling, sonar, pile driving, ships, and operational wind turbines (Chapter 15; Dow Piniak et al., 2012; Lenhardt et al., 1983; Read, 2013) Sea turtles are also vulnerable to collisions with ships, particularly those moving at higher speeds (Hazel et al., 2007). Turtles can be displaced from operating offshore wind facilities due to turbine or vessel noise, or artificial reef effects could lead to turtles aggregating around turbine foundations(Read, 2013).

It may be possible to minimize potential effects of offshore wind energy development on sea turtles in the mid-Atlantic by planning offshore wind energy construction activities for periods in which turtles are not present (e.g., winter), though conducting construction activities during winter can be difficult or impossible. Since it is likely that turtle presence and construction will overlap, the development of techniques to avoid or reduce interactions between sea turtles and development activities should be a priority (Chapter 15). Restricting vessel speeds within areas and times of year when turtles are present could also help prevent negative impacts and/or mortalities of sea turtles (Hazel et al., 2007). Aerial video surveys appear to be an effective means to document sea turtle distributions, and we would suggest that future studies of sea turtles strongly consider digital aerial survey methodologies in order to obtain the best possible data for conservation and mitigation purposes.

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## Figures and tables

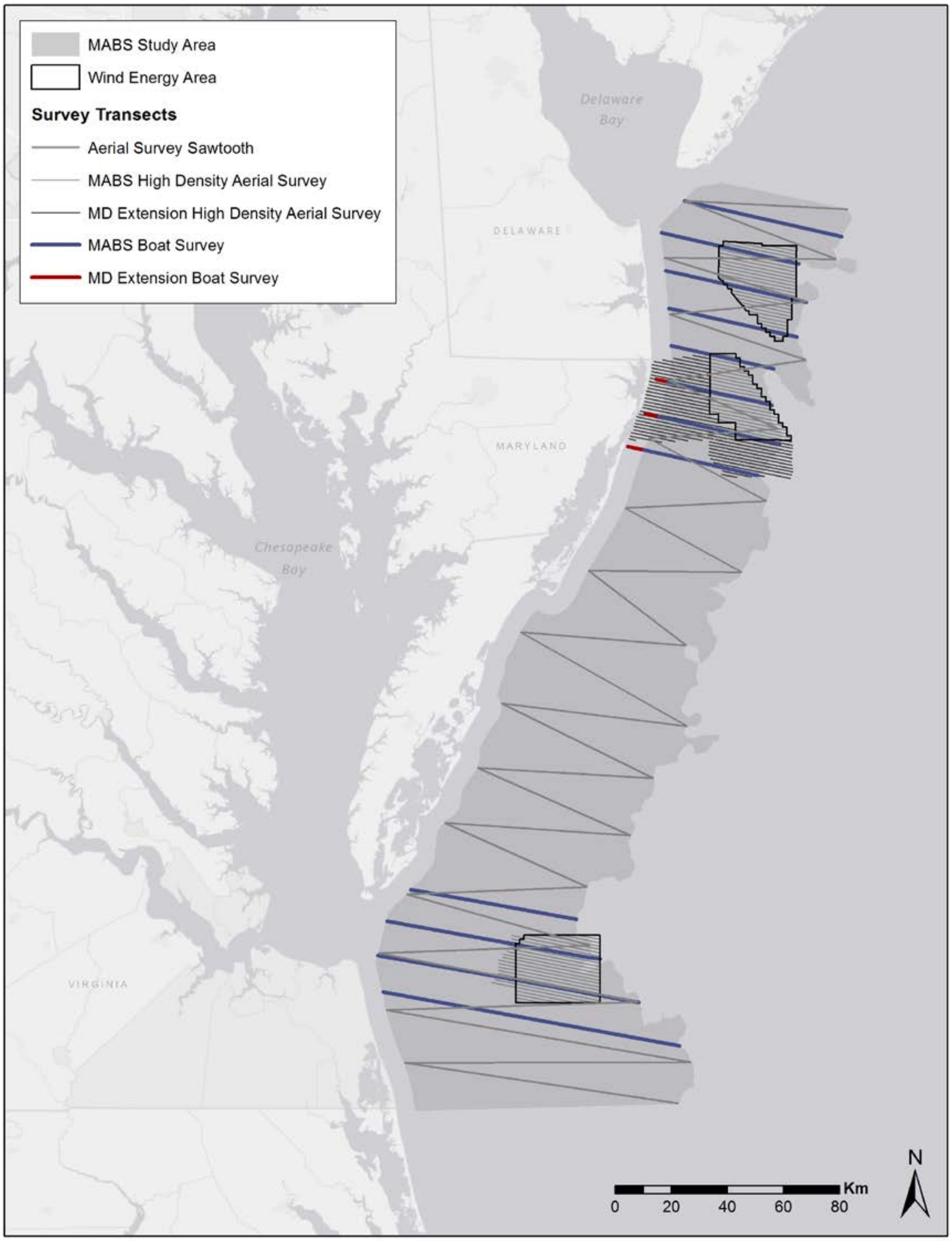


**Figure 14-1. Map of transects for the digital video aerial surveys and the boat surveys for the Department of Energy (DOE)-funded Mid-Atlantic Baseline Studies Project (2012-2014) and the state of Maryland-funded Maryland Project (2013-2014).** DOE-funded aerial transects are shown in light gray. High-density Maryland extension transects are shown in dark gray. DOE-funded boat surveys are shown in blue, and Maryland extension transects in state waters in red.

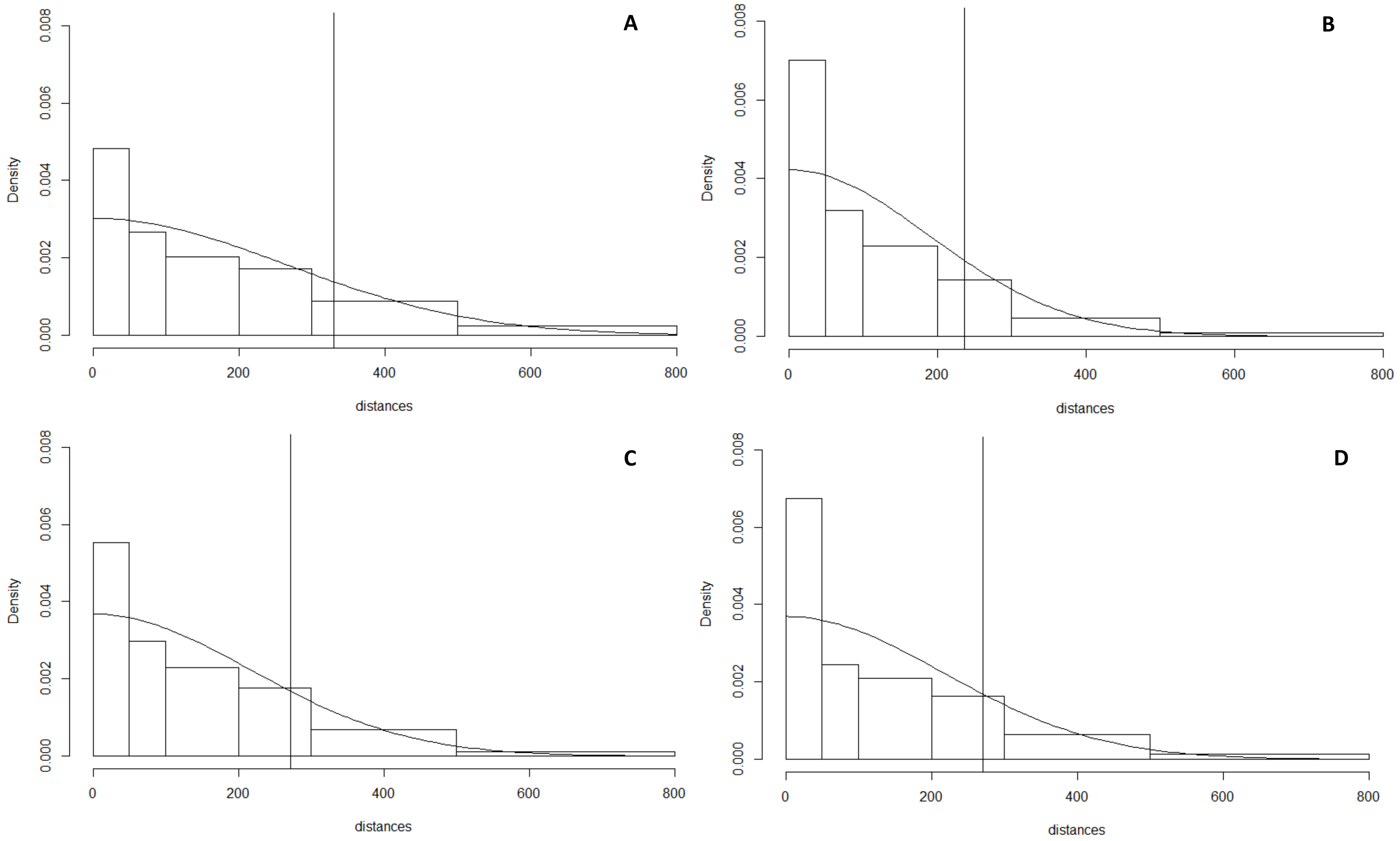


**Figure 14-2. Distance functions for a) Sulidae, b) Laridae, c) Gavidae, and D) Anatidae from the boat survey data.** Effective strip half-widths, or the distance from the boat at which there is average detection probability, are indicated by the vertical line in each chart (330m for Sulidae, 236m for Laridae, 272m for Gavidae, and 271m for Anatidae).

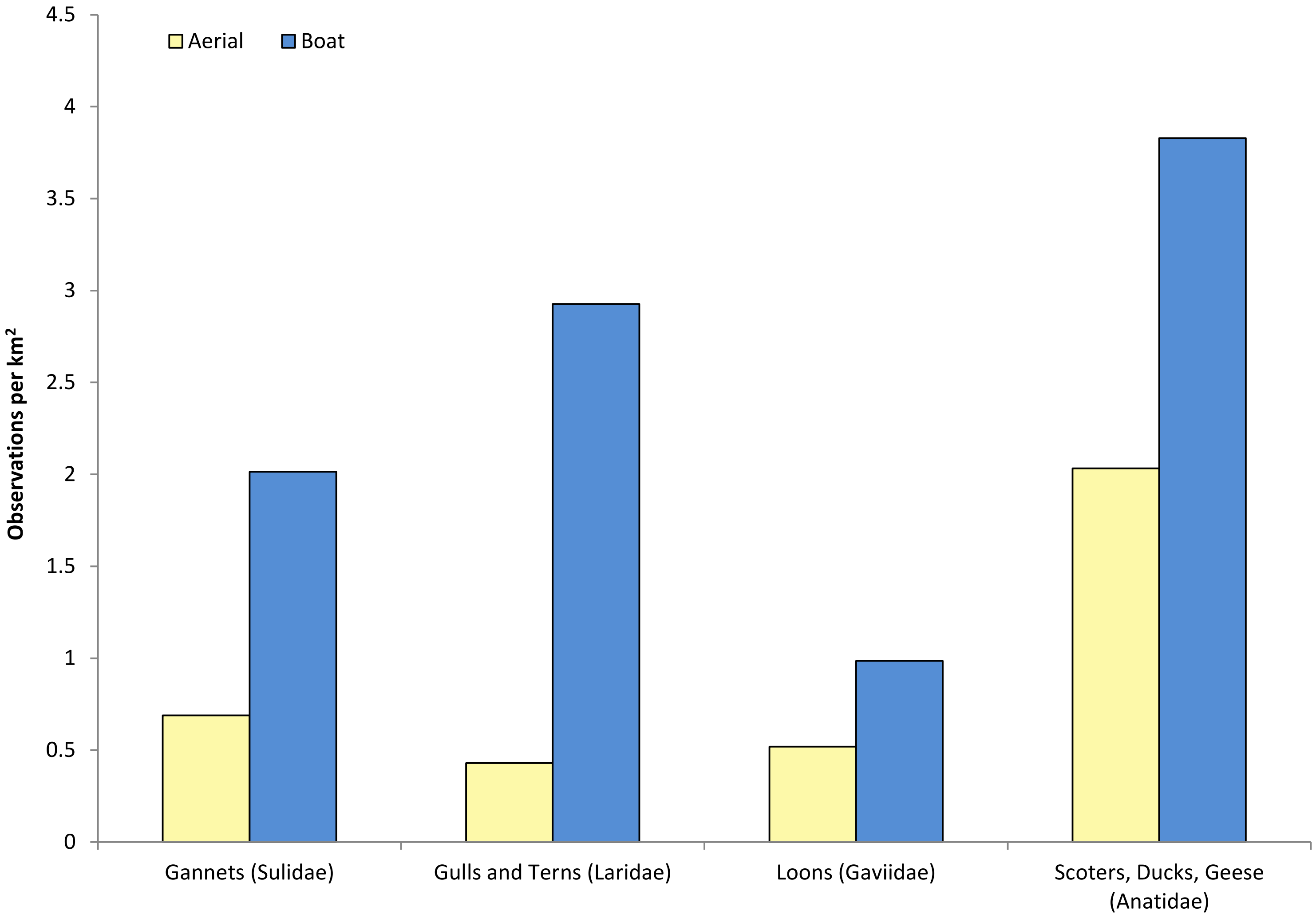


**Figure 14-3. Comparison of total effort-corrected boat and aerial survey counts by taxon for all surveys.** Densities are calculated by the total number of counts divided by the total survey area. Aerial data have transect widths of 200 or 300 meters (Chapter 2). Effective boat transect strip widths were calculated for each group based on the effective half strip width (see **Figure 14-2**).

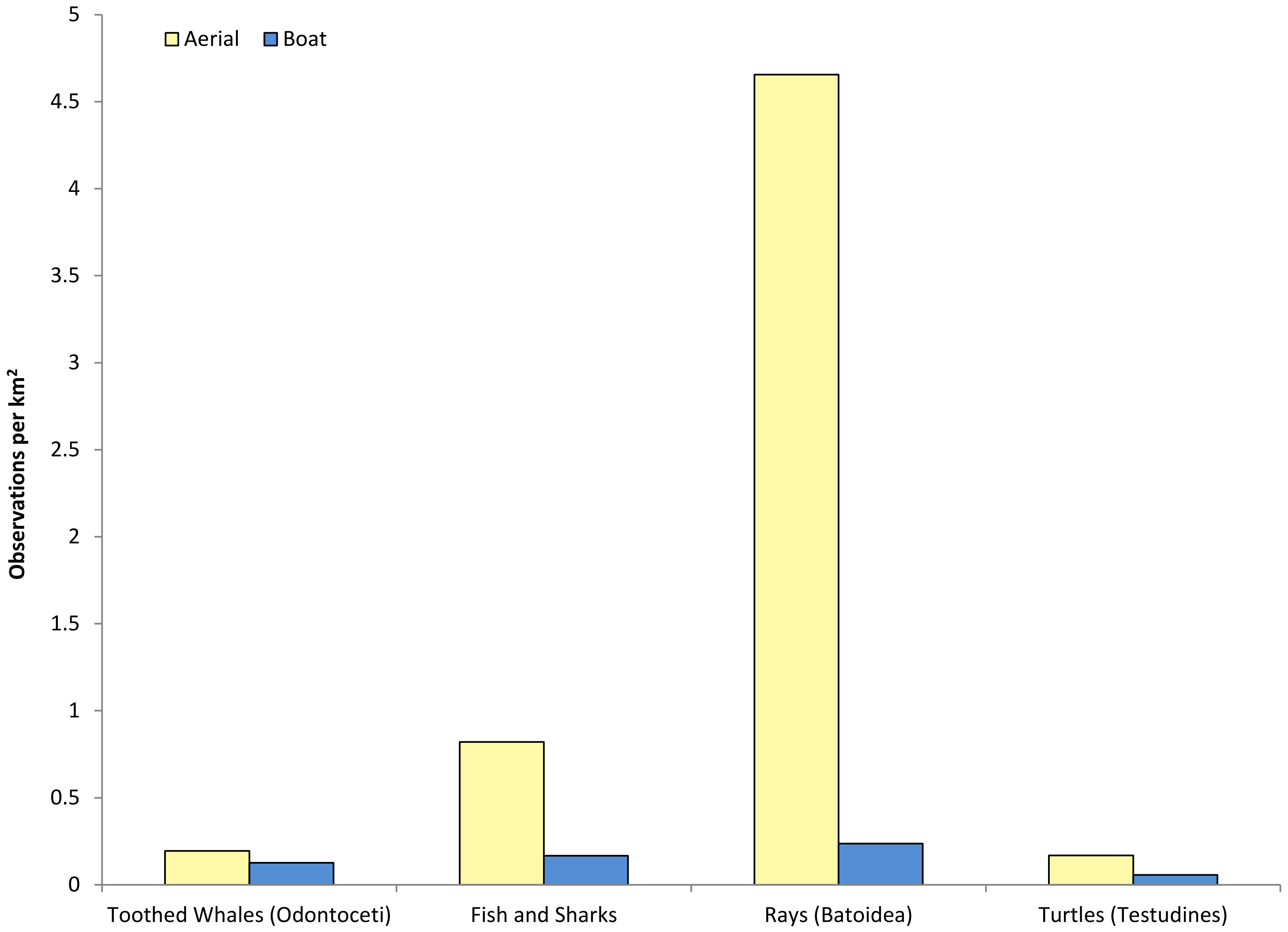


**Figure 14-4. Comparison of total effort-corrected boat and aerial survey counts by taxon for all surveys.** Densities are calculated by the total number of counts divided by the total survey area. Aerial data have transect widths of 200 or 300 meters (see Chapter 3). Boat data transect widths were based on the median distance of observations from the boat, in meters (Odontoceti, 300; Fish/Sharks, 50; Batoidea, 7.5; Testudines, 100 meters). Observations of groups that were not individually counted or identified (e.g., bait balls, ray schools) are excluded from this figure (see Chapter 5 for more information).

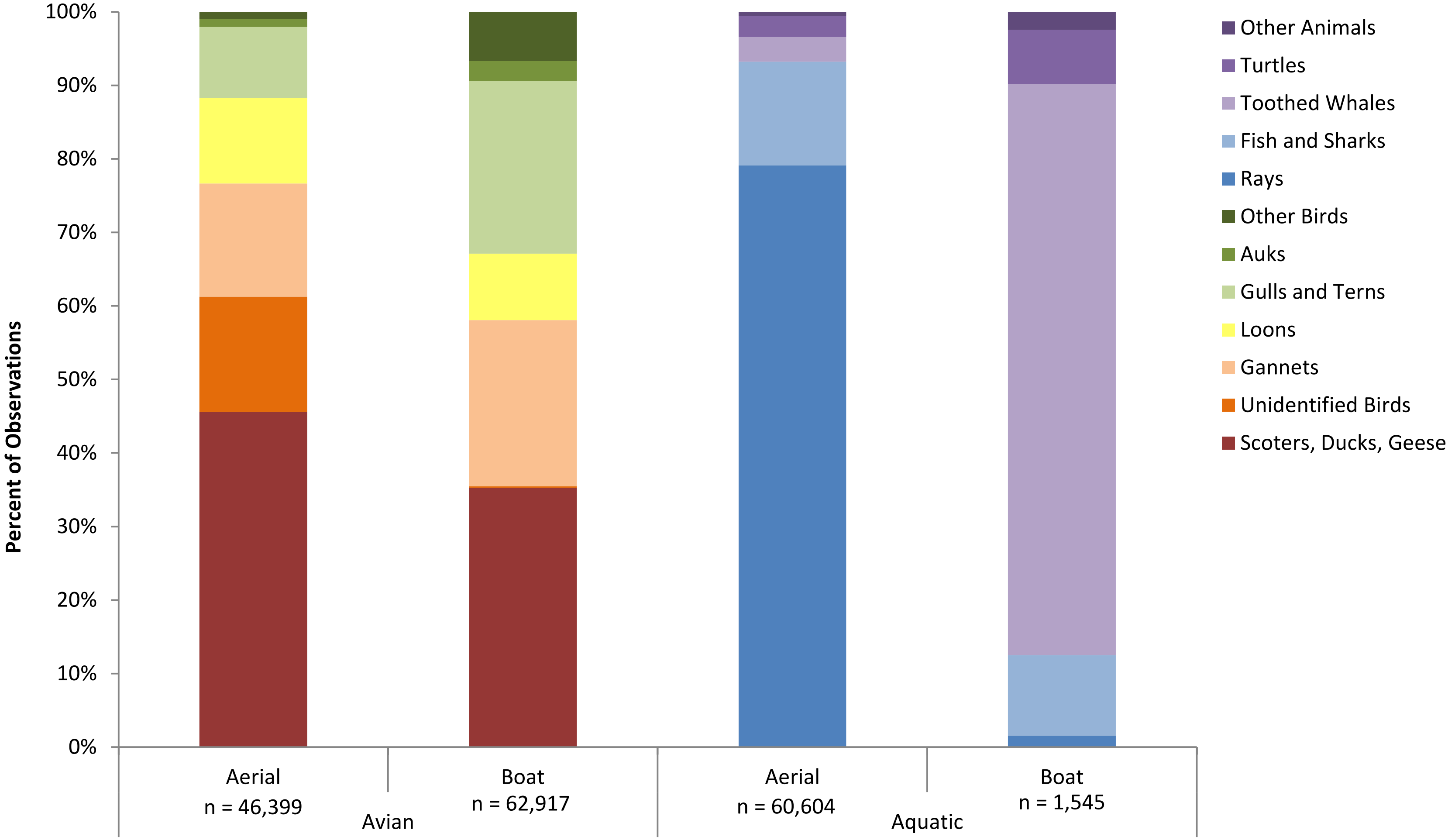


**Figure 14-5. Avian (left) and aquatic (right) animals observed in the digital video aerial surveys and boat-based surveys.** The sample size for each group is noted on the x-axis. Taxonomic categories shown for avian observations are scoters, ducks, and geese (Anatidae); unidentified birds (birds not identified to lower taxonomic levels); gannets (Sulidae); loons (Gaviidae); gulls and terns (Laridae); alcids (Alcidae); and other birds (additional less common bird groups, see Chapters 5 and 8 for animals observed). Aquatic taxa include rays (Batoidea); fish and sharks (Chordata); toothed whales (Odontoceti); turtles (Testudines); and other animals (additional less common animal groups, see Chapters 5 and 8 for animals observed).

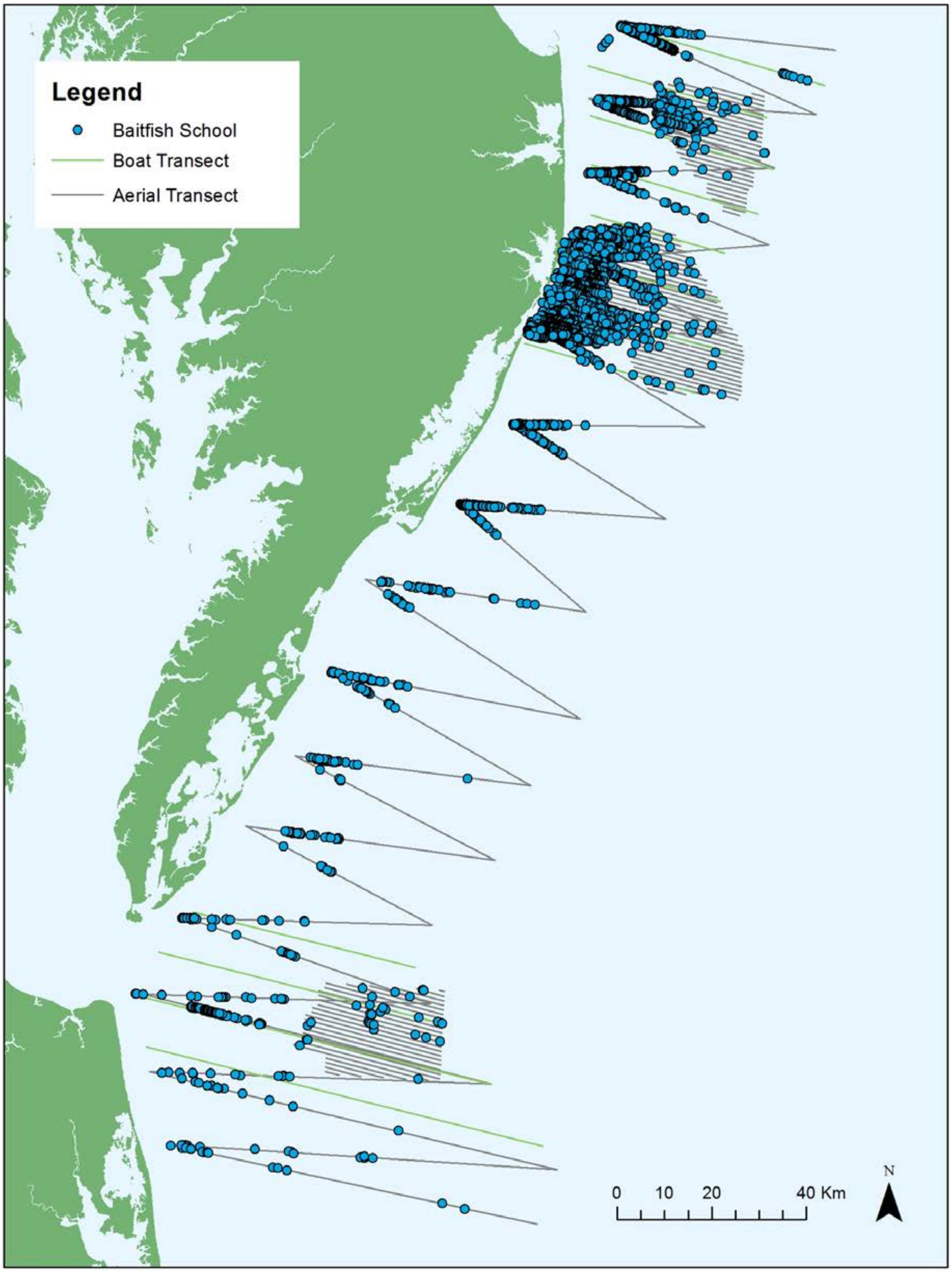


**Figure 14-6. Schools of baitfish (forage fish) observed in boat and high resolution digital video aerial surveys.**

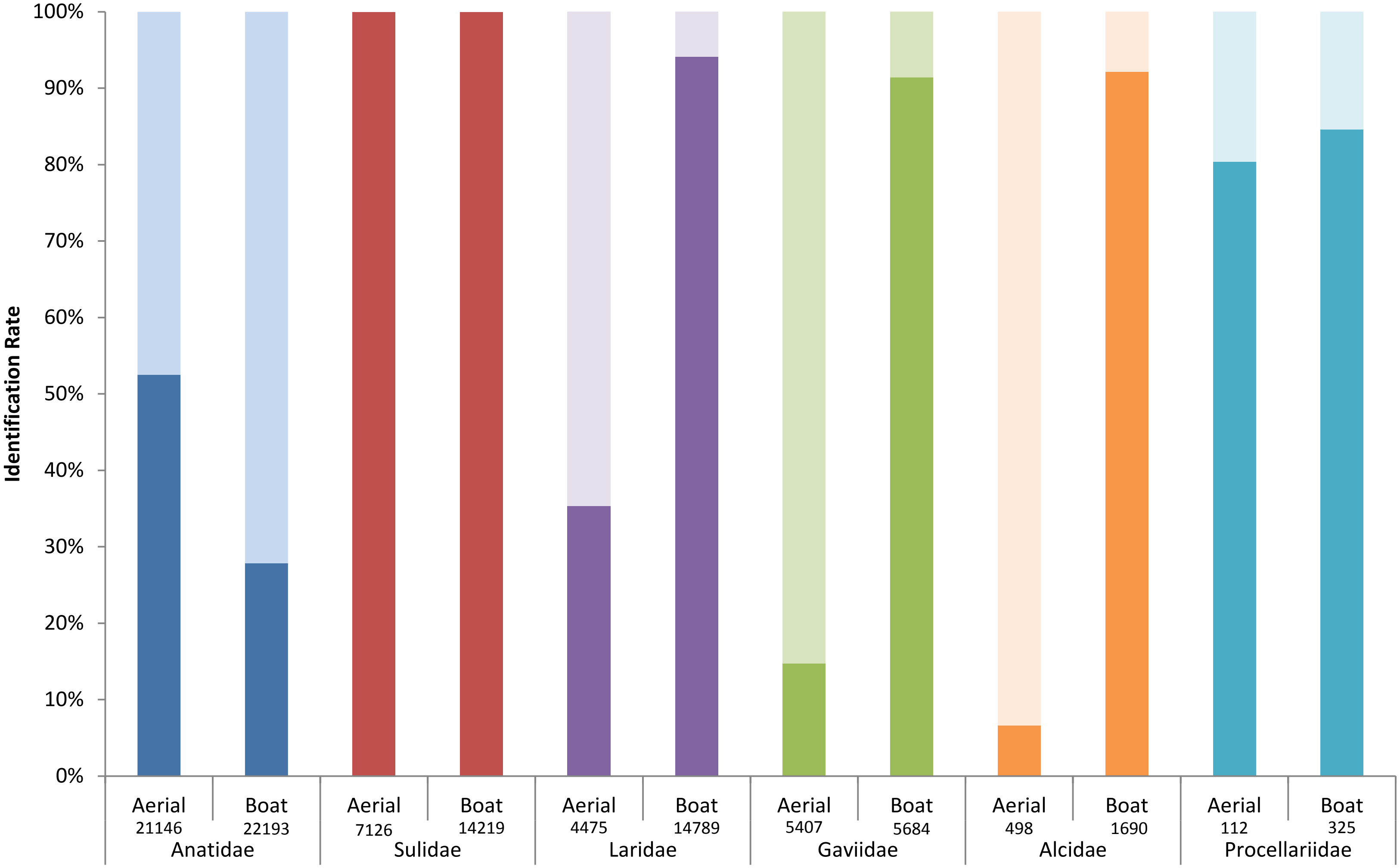


**Figure 14-7. Identification rates for common bird taxa observed in boat and high resolution digital video aerial surveys, in order of abundance.** Darker colors indicate animals identified to species, and lighter colors indicate animals identified to higher taxonomic levels. Sample sizes are noted in the x-axis. Details on species sighted within each taxonomic group can be found in Chapters 5 and 8. The most common avian families observed in surveys were scoters, ducks, and geese (Anatidae); gannets (Sulidae); gulls and terns (Laridae); loons (Gaviidae); alcids (Alcidae); and fulmars and shearwaters (Procellariidae).

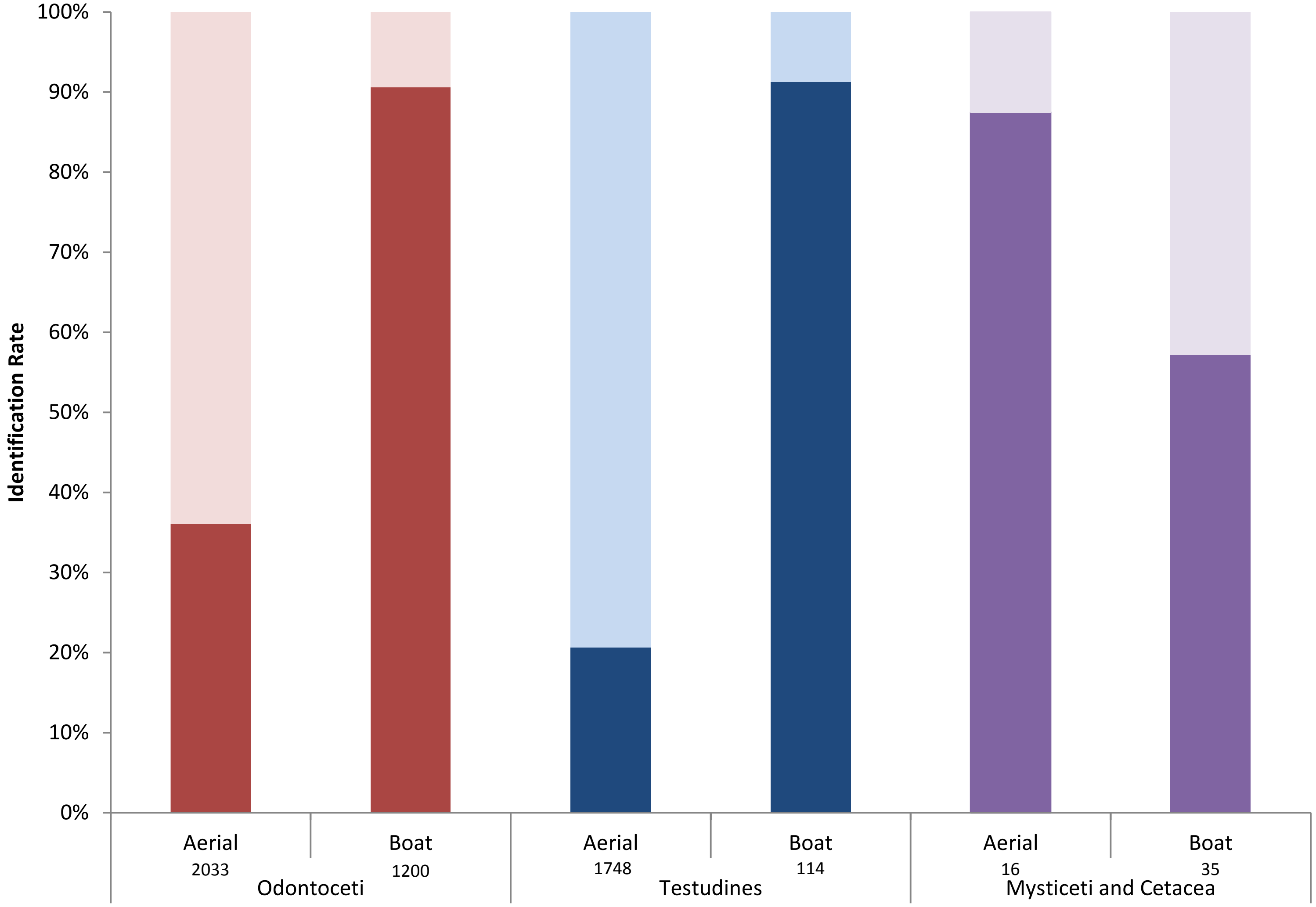


**Figure 14-8. Identification rates of mammals and turtles observed during boat-based and high resolution digital video aerial surveys.** Darker colors indicate animals identified to species, and lighter colors indicate animals identified to higher taxonomic levels. Sample sizes are noted in the x-axis. Details on species included in each taxonomic group can be found in Chapters 5 and 8. Groups are toothed whales (Odontoceti); sea turtles (Testudines); and large whales (Mysticeti and unidentified large Cetaceans).

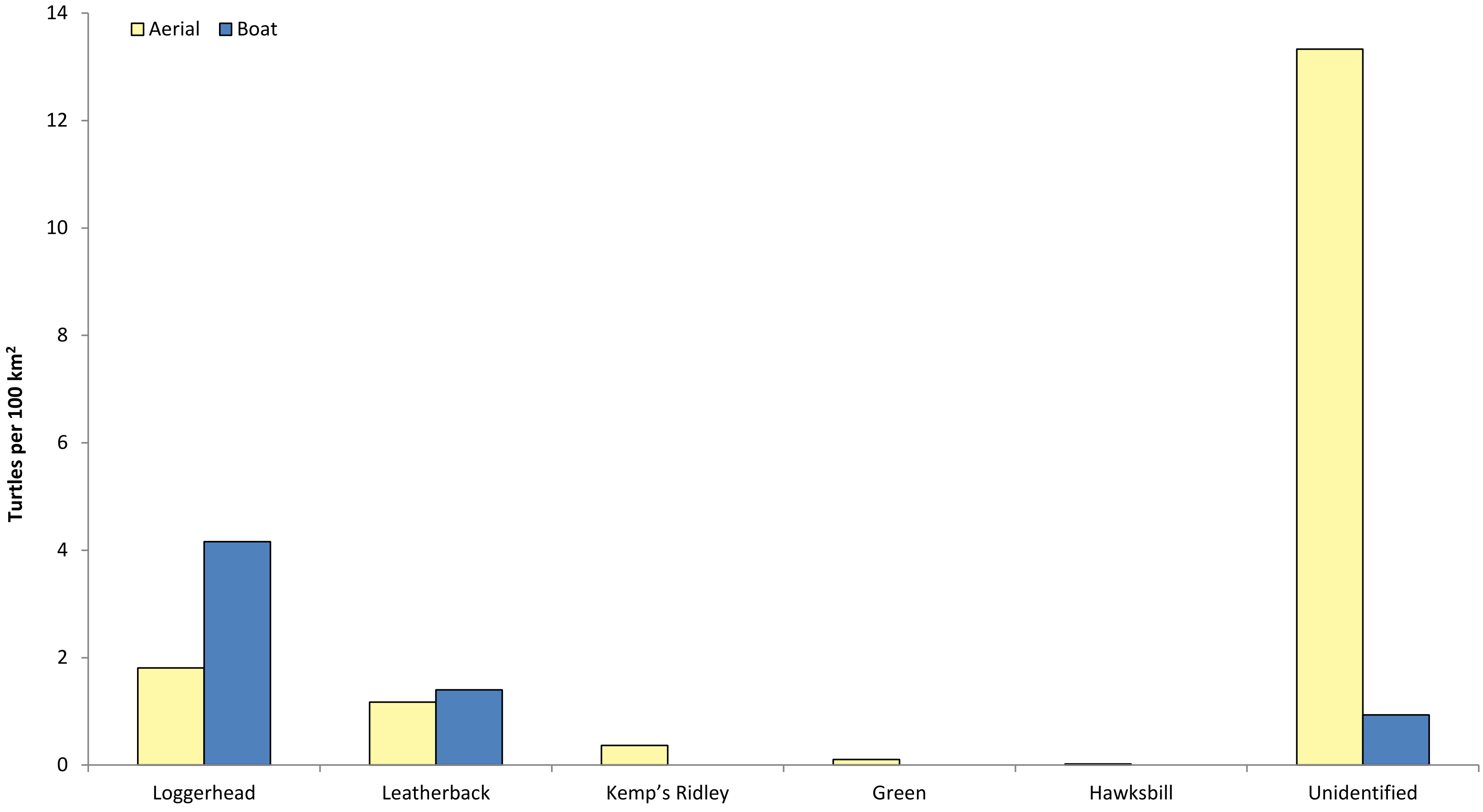


**Figure 14-9. Comparison of total effort-corrected boat and aerial survey counts of sea turtles.** Densities were calculated by the total number of counts divided by the total survey area across all surveys, and standardized to 100 square km. Aerial surveys had transect strip widths of 200 or 300 m (Chapter 3). Boat transect strip widths were based on the median distance of observations from the boat, in meters (Loggerhead 100, Leatherback 50, Unidentified 50 m), and multiplied by two to account for observations made on both sides of the boat.

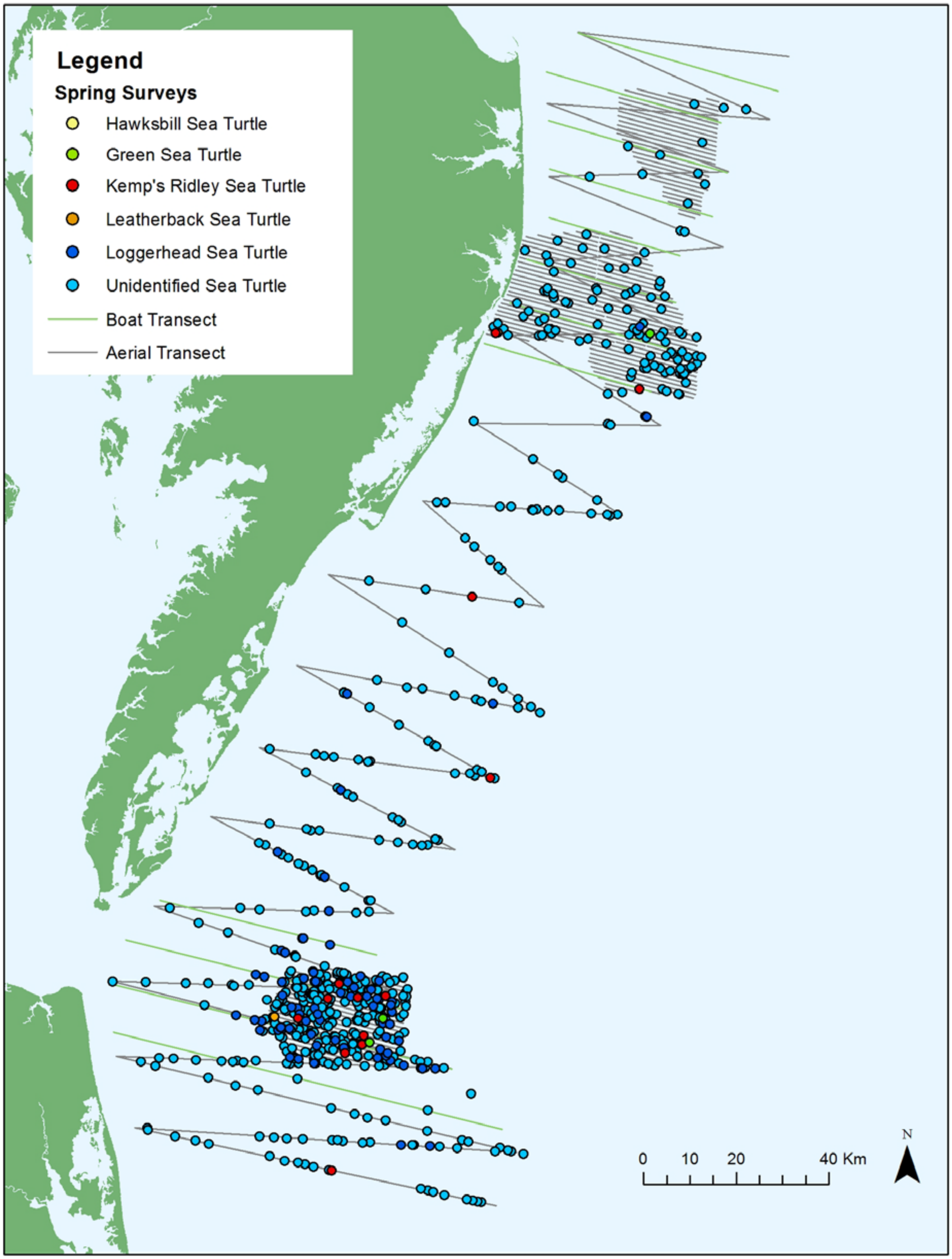


**Figure 14-10. Turtles observed in the spring (Mar.-May, 2012-2014) in boat and high resolution video aerial surveys.** Unidentified sea turtles are any turtles not identified to species, and could represent any of the four smaller turtle species present in the study area (excluding Leatherback Sea Turtles).

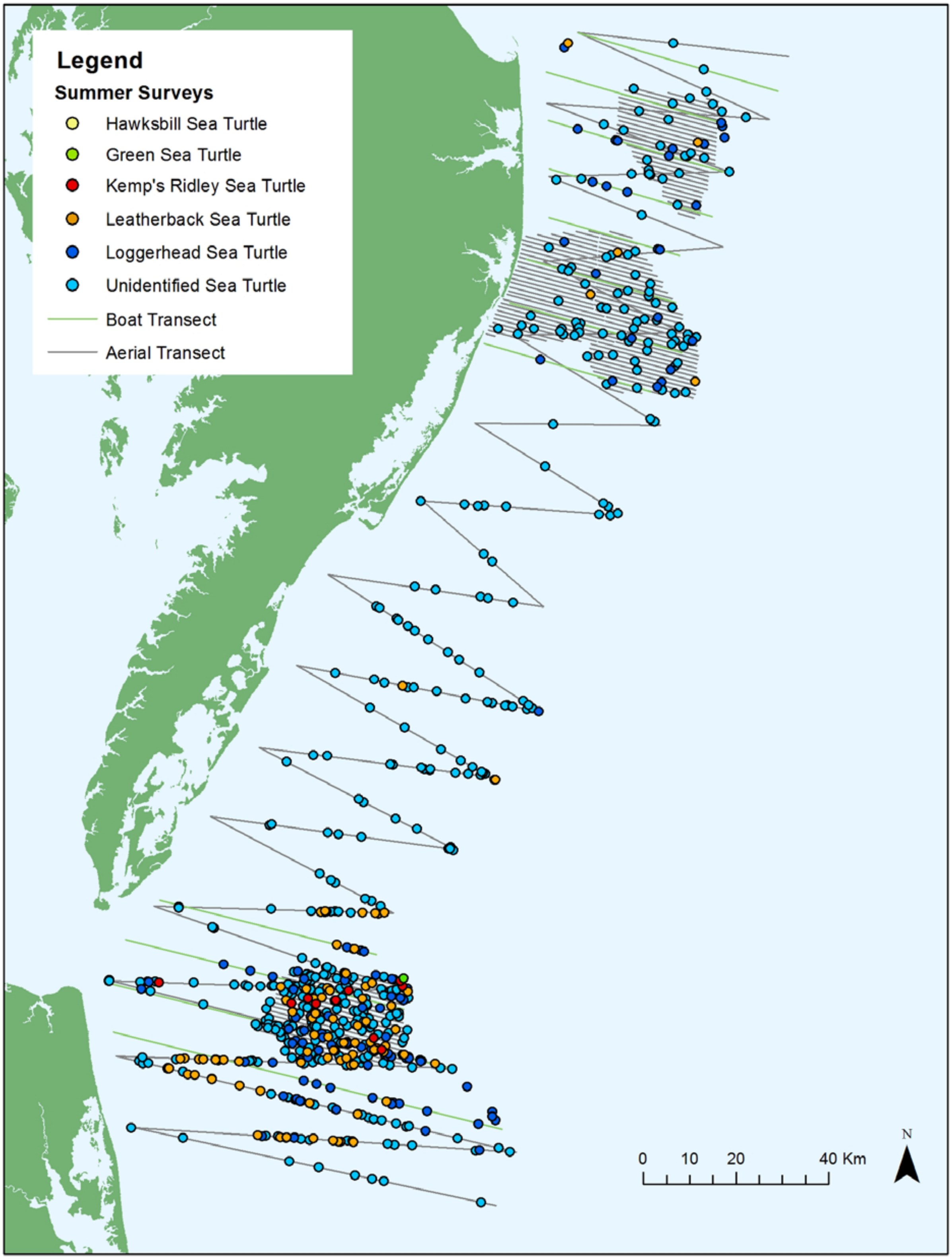


**Figure 14-11. Turtles observed in the summer (Jun.-Aug., 2012-2013) in boat and high resolution video aerial surveys.** Unidentified sea turtles are any turtles not identified to species, and could represent any of the four smaller turtle species present in the study area (excluding Leatherback Sea Turtles).

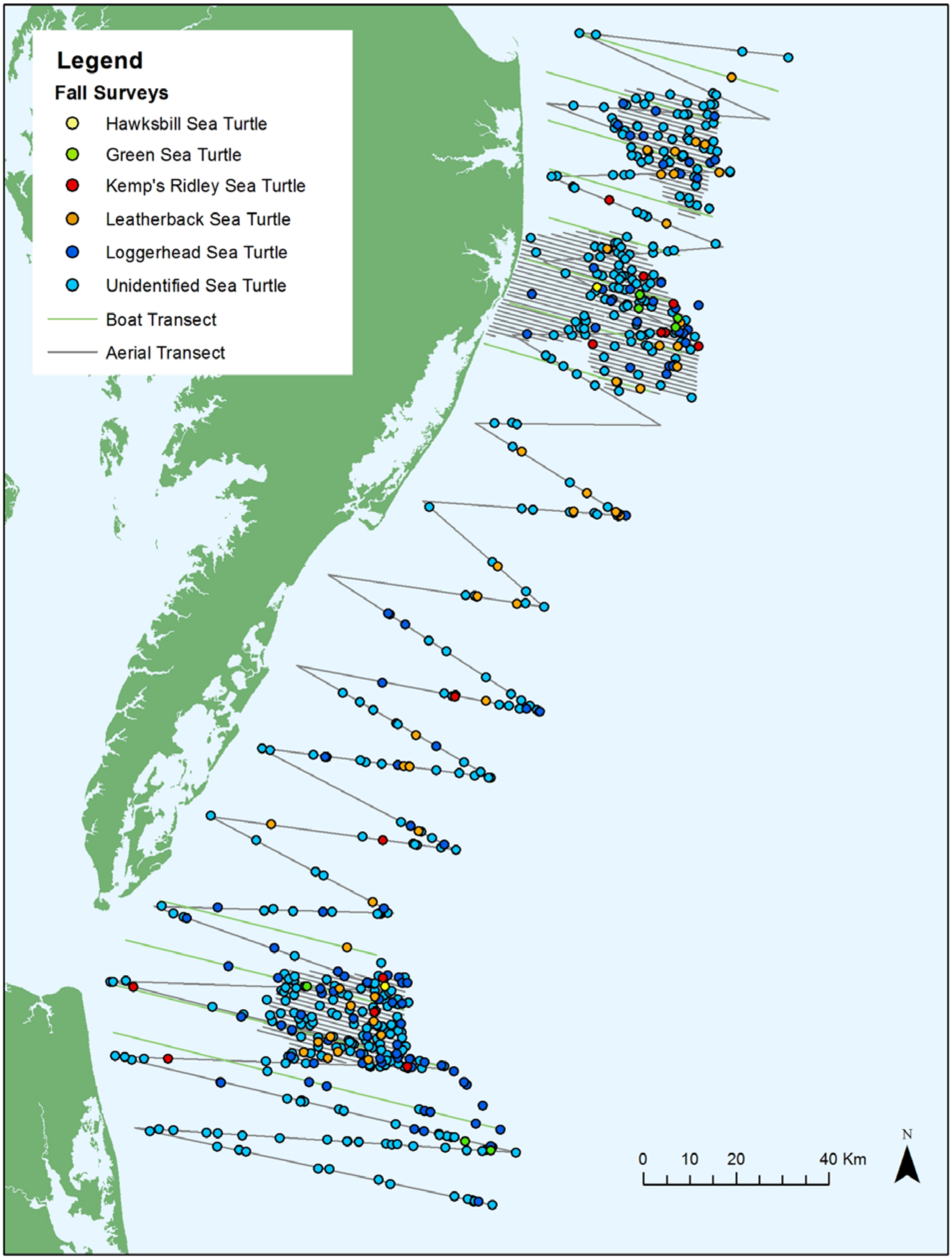
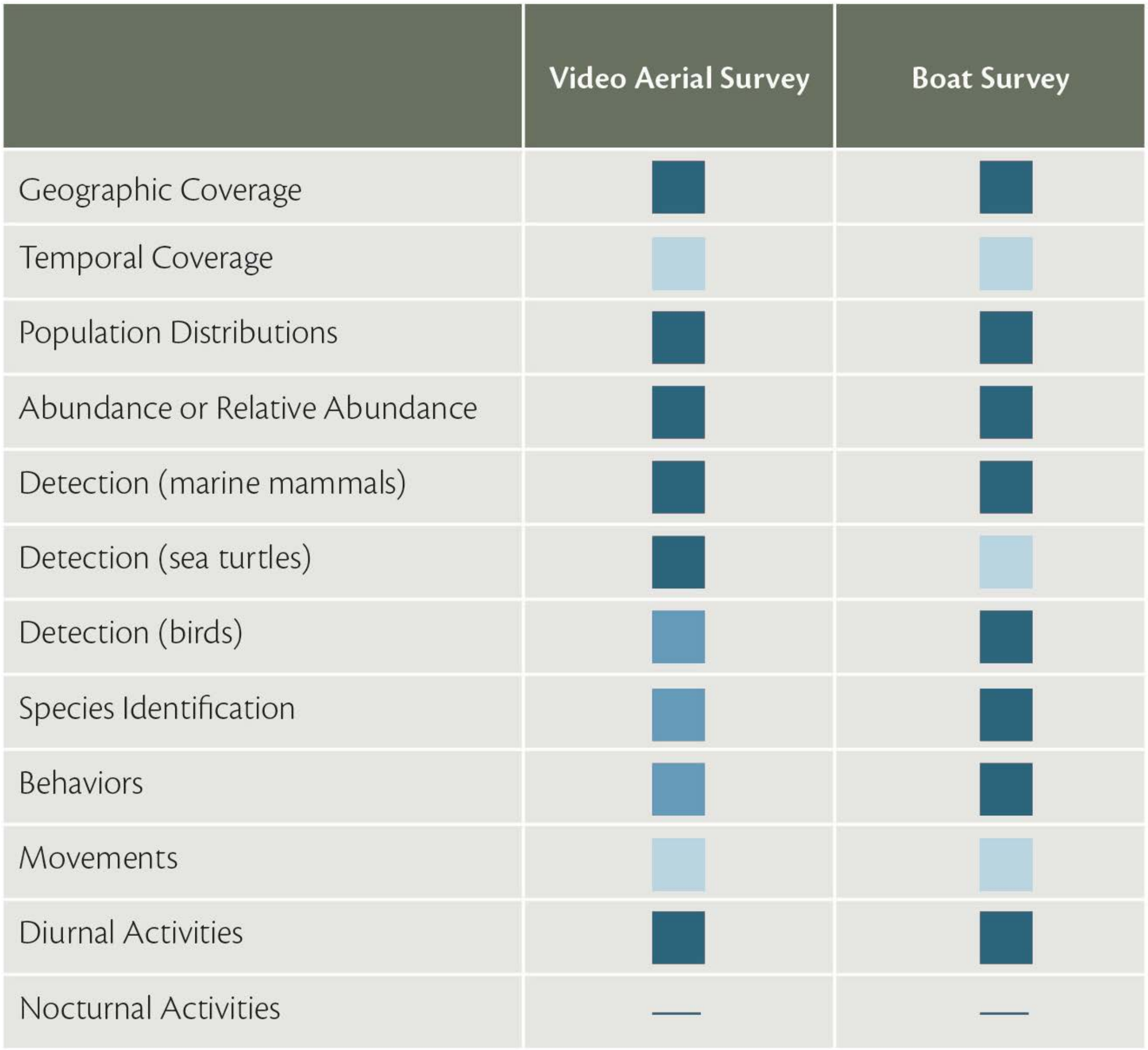


**Figure 14-12. Turtles observed in the fall (Sep.-Nov., 2012-2013) in boat and high resolution video aerial surveys.** Unidentified sea turtles are any turtles not identified to species, and could represent any of the four smaller turtle species present in the study area (excluding Leatherback Sea Turtles).



| | Video Aerial Survey | Boat Survey |
|---|---|---|
| Geographic Coverage | ■ | ■ |
| Temporal Coverage | □ | □ |
| Population Distributions | ■ | ■ |
| Abundance or Relative Abundance | ■ | ■ |
| Detection (marine mammals) | ■ | ■ |
| Detection (sea turtles) | ■ | □ |
| Detection (birds) | ▨ | ■ |
| Species Identification | ▨ | ■ |
| Behaviors | ▨ | ■ |
| Movements | □ | □ |
| Diurnal Activities | ■ | ■ |
| Nocturnal Activities | — | — |

**Figure 14-13. Methods for surveying offshore wildlife in this study.** Relative strengths and weaknesses of video aerial and boat surveys in this study are indicated by depth of color (dark blue = good, medium blue = fair, light blue = poor). A dash indicates that data are not available from this survey method. Values are subjective; for example, while detection bias was not quantified for aerial surveys, detection of avian species in our boat surveys appeared to be better than digital video aerial surveys in many cases, at least after correction for distance bias in boat data. Thus, boat surveys were categorized as “good” for this type of data, while digital video aerial surveys were considered “fair”.

**Table 14-1. Total number of individuals observed, species observed, and survey effort for the boat-based and high resolution digital video aerial surveys (2012-2014).** Aerial transect width was 200 meters, with the partial exception of the first three surveys (when the sawtooth transect width was 300 meters). Boat data were collected at varying distances from the transect line, depending on the taxon, but the effective transect width for the survey likely falls between 300 and 500 meters for most taxa (and these two numbers are used to present an approximate range of total area covered by the boat surveys in the table below). See Chapters 3 and 7 for more details on data collection methods.

| **Survey** | **Avian Animals** | | **Non-Avian Animals** | | **Effort** | |
|---|---|---|---|---|---|---|
| | Number observed | Species | Number observed | Species | Linear km | Area ($km^2$) |
| **Aerial** | 46,399 | 47 | 60,604 | 19 | 49,576 | 10,403 |
| **Boat** | 59,336 | 94 | 1,439 | 12 | 10,698 | 3,209-5,349 |

# Chapter 15: Density modeling for marine mammals and sea turtles with environmental covariates

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** www.briloon.org/mabs

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# Chapter 15 Highlights

*Density modeling for marine mammals and sea turtles*

### *Context*[1]

Part IV of this report contains several instances in which boat survey and digital video aerial survey datasets were modeled with environmental covariates to describe populations of interest (Chapters 15-16, 18-19). Cetaceans and sea turtles are taxa of regulatory and conservation concern in the mid-Atlantic region. By combining boat and aerial survey data for these taxa with remotely sensed environmental data, we can use spatial-temporal modeling methods to estimate habitat influences on distributions and relative abundance, and explore potential overlap with offshore human interests, including Wind Energy Areas (WEAs). In some cases, one survey method was significantly better than the other for surveying a particular taxon, as with digital video aerial surveys for sea turtles. Both boat and aerial surveys were suspected to inaccurately estimate group size for cetaceans, so models were developed to identify patterns of occurrence of delphinid pods, rather than abundance of individual animals.

### *Study goal/objectives*

Describe the distributions of cetaceans and sea turtles across the mid-Atlantic Outer Continental Shelf using boat and aerial survey data.

### *Highlights*

- At least five different species of dolphins and porpoises were observed in surveys. Five species of baleen whales were also observed, including nine North Atlantic right whales.
- Bottlenose dolphins were observed primarily in more nearshore areas in spring through fall. Primary productivity and sea surface temperature were also important predictors; models suggest minimal presence of the species in mid-Atlantic WEAs during cooler months.
- Common dolphins were most frequently observed in offshore areas in winter and early spring.
- Five species of sea turtles were observed in boat and aerial surveys.
- Turtles were much more frequently observed in digital aerial surveys than in boat surveys.
- Sea turtles were most abundant from May to October. In addition to water temperature, primary productivity and distance from shore were important influences, and sea turtles were primarily distributed offshore. There was substantial overlap with the Virginia WEA in the spring and overlap with the more northern WEAs when turtles were more broadly distributed in the summer and fall.

### *Implications*

Small sample sizes made modeling difficult for some taxa, but results suggest that there may be seasonal overlap between cetacean and sea turtle distributions and WEAs in the mid-Atlantic. Given the protected status of these species, additional research on their distributions may be indicated, as well as the development of potential approaches for mitigating the effects of wind power development.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

Marine mammals and sea turtles are often of management and conservation concern, and effective management of these large marine vertebrates requires reliable information on distribution, abundance, and trends in habitat use. This chapter utilizes observation data of cetaceans and sea turtles from boat and digital video aerial surveys to describe the distributions of these taxa across the mid-Atlantic Outer Continental Shelf, determine the habitat or environmental drivers of these distributions, and identify the locations and timing of potential overlap with Wind Energy Areas (WEAs). Dolphin, porpoise, whale, and sea turtle observations from boat and aerial surveys were assessed for species composition, relative numbers, and geographic and temporal distributions. Relative density estimates were produced for sea turtles (using digital video aerial survey data) and bottlenose dolphins (using boat survey data) using both general linear and general additive models (GLMs and GAMs, respectively). For both bottlenose dolphins and sea turtles, GAMs proved more effective at modeling the density of these animals with relation to spatial covariates then their counterpart GLMs. Bottlenose dolphins were observed primarily in more near shore areas in spring, summer, and fall, in areas with high levels of primary productivity and higher sea surface temperatures. There were few observations of the species during cooler months. Sea turtles were also most abundant from May to October, and their densities were correlated warmer water temperatures and farther distances from shore. There was substantial overlap between sea turtle distributions and WEAs, particularly in the southern part of the study area. There was also overlap between WEAs and predicted habitat usage of bottlenose dolphins and other delphinids, although the degree of this overlap was difficult to discern with the datasets used in this analysis.

## Introduction

Marine mammals and sea turtles are often of management and conservation concern, as their large home ranges and habitat requirements tend to overlap and conflict with human activities such as offshore development and commercial fishing (Trites et al. 1997). Offshore explorations for oil and gas development have boomed in the United States since the 1970s (Boesch and Rabalais 1987). In recent years, it has become clear that the United States has a huge domestic resource for offshore wind energy as well. However, as these explorations have progressed, we have come to realize that these proposed development areas often occur in a very “busy” medium, as these energy resources are often located in areas where many other offshore uses occur (including important commercial fisheries, shipping lanes, recreational areas, and military areas, as well as areas of ecological importance).

Cetaceans (whales, dolphins, and porpoises) and sea turtles represent a particular challenge for population monitoring, due to their vast ranges and cryptic behaviors, resulting in only small portions of the animals’ bodies being visible (Hammond et al. 2002). However, the conservation and management of these large marine vertebrates requires reliable information on distribution, abundance, and trends in habitat use, and quantitative research is essential for overcoming these challenges. Acoustic disturbance has been recently identified as a primary concern for marine mammals and sea turtles within the marine environment (Dow Piniak et al. 2012; Bergström et al. 2014). This includes such noises as shipping, seismic surveys, blasting, pile driving, and operational wind turbines. The severity of avoidance and displacement effects appear to vary with a variety of factors, including the species being exposed as well as the frequency, intensity, and duration of noise (Goold 1996; McCauley et al. 2000; Madsen et al. 2002). These disturbances may not only deter marine species from development areas, but have the potential to be detrimental to the animals in other ways as well, including a variety of behavioral, acoustical, and physiological effects (Nowacek et al. 2007; Southall et al. 2007; Tyack et al. 2011). Current mitigation practices include “exclusion zones” around activities such as the operation of naval sonar that may cause physiological stress or other responses, to address the potential for non-displacement effects. It has been suggested that larger exclusion zones may be needed for some activities, locations, or populations, particularly for beaked whales (Wright et al. 2011). The expansion of these exclusion zones during certain development activities may be one tool marine construction operators could use to mitigate such effects.

The mid-Atlantic Outer Continental Shelf (OCS) is of key importance to many large marine species during both breeding and nonbreeding periods. This region also acts as a key migration route for one of the most sensitive and protected marine mammals, the North Atlantic Right whale (Kenney et al. 2001). The most recent marine mammal stock assessment reports (SARs) for the North Atlantic place 13 cetacean and three pinniped species within the OCS study area, all of which are protected under the Marine Mammal Protection Act (Waring et al. 2011; Waring et al. 2013). It is also important to note that sound travels long distances underwater, and just to the east of the study area (over the shelf break), a whole new range of deep diving cetaceans such as sperm and beaked whales that are highly sensitive to marine noise may also be exposed to development noise from the study area (Mate et al. 1994; Cox et al. 2006). Furthermore, five of the seven extant species of sea turtle occur in the mid-Atlantic OCS, and all five are protected under the Endangered Species Act. The abundance of large marine megafauna within the mid-Atlantic OCS makes it a potentially sensitive and challenging location for offshore development.

Given the difficulties associated with estimating animal abundance (or occurrence) based on count data from large-scale surveys (Royle et al. 2007), modeling spatial and temporal distributions of animals can help to determine areas of high and low use and inform decisions for development (Garthe and Hüppop 2004; Kinlan et al. 2012). However, distributions of animals in the offshore environment can be highly variable, and are driven by environmental and biophysical factors working at a variety of temporal and spatial scales (O’Connell et al. 2009; Zipkin et al. 2010). By combining boat and aerial survey data with oceanographic habitat and climatological data, we can use spatial-temporal modeling methods to estimate these habitat influences on the distributions and relative abundances of a species of interest, and explore potential overlap with offshore human interests. Accurately assessing such relationships is essential for predicting spatial distributions and the potential shifts that could occur in these geographic distributions. In this study, we quantify sea turtle and marine mammal densities seasonally throughout the study region; develop models to examine spatial patterns and trends based on interactions with environmental conditions; and help identify species at potential risk from turbine construction and operation due to their movements, behavior, or migration strategies.

## Methods

### *Survey methods*

Standardized boat-based surveys are a widely used method of obtaining density estimates for birds, sea turtles, and marine mammals (Thompson and Harwood 1990). In our boat-based surveys, transects extend perpendicularly to the coastline, from three nautical miles offshore to the 30 m isobath or the eastern extent of the mid-Atlantic WEAs, whichever was furthest. Boat transects were spaced 10 km apart and extend at least one transect north and south of each WEA (Figure 15-1 to Figure 15-3). We conducted eight surveys per year on a scheduled basis as the weather allowed, between April 2012 and April 2014. Eight of the 16 surveys (from March 2013 to February 2014) also included extensions of three transects farther west into Maryland state waters. Two pairs of observers alternated 2-h shifts collecting standard strip- and line-transect data using distance sampling (Buckland et al. 1993). While the recorder entered data into the program dLOG (R.G. Ford Consulting, Inc.), and regularly updated changes in environmental conditions (sea state, etc.), the observer scanned the horizon, focusing on one forward quadrant on either side of the vessel. We continuously recorded the species, count, distance, and angle to the observation (see Chapter 7 for more details on data collection methods). Cetaceans were photographed when possible. Photos were submitted for individual identification using the established North Atlantic fin whale (*Balaenoptera physalus*), humpback whale (*Megaptera novaeangliae*), and North Atlantic right whale (*Eubalaena glacialis*) catalogues. Surveys were conducted in “passing mode,” meaning that the boat stayed on transect and at constant survey speed (10 knots) except when complying with National Marine Fisheries Service (NMFS) rules about approaching marine mammals, including rules regarding vessel speed and encounters with endangered North Atlantic right whales.

Fifteen high resolution digital video aerial surveys were conducted by HiDef Aerial Surveying, Ltd. (hereafter HiDef) between March 2012 and May 2014 (Chapter 3). Each survey was completed using two small commercial aircraft, allowing complete coverage of the study area in two to three days (weather permitting). Aerial transects were flown at high densities within the Delaware, Maryland, and Virginia WEAs to obtain accurate abundance estimates within these specific footprints; additional high density transects were added adjacent to the Maryland WEA in the second year of surveys. The remainder of the study area was surveyed on an efficient sawtooth transect path to provide broad-scale context for the intensive WEA surveys (Figure 15-4 to Figure 15-6). Recorded images were stored on heavy duty disk drives or solid state recording devices for subsequent review and analysis. Wildlife locations, taxonomic identities, and behaviors were determined from the video footage (Hatch et al. 2013).

### *Data preparation*

Boat-based and aerial survey observations of marine mammals and sea turtles are summarized in Table 15-1 and Table 15-2. All animals not identified to the species level were combined into an “unidentified” category. Due to the lack of data at the species level for the aerial surveys, sea turtle observations were grouped as a single taxon for all further analyses by season (Spring: Mar.-May, Summer: June-Aug., Fall: Sept.-Nov., Winter: Dec.-Feb.). The number of whale sightings from both surveys (n=51) was not sufficient to produce descriptive models, and thus this taxon was excluded from further analysis.

Effort and species observation data were modeled using the “count” method (Hedley et al. 1999). Boat and aerial survey track lines were divided into segments approximately 5 km in length. Start and end locations of these segments were calculated using the COGO proportions function in ArcMap 10.2 (ESRI 2011). The location of the midpoint of each segment (latitude and longitude) was calculated using the feature to point command in ArcMap 10.2.

Oceanographic processes were evaluated as spatial covariates to predict marine mammal and sea turtle location and density. Sea surface temperature (SST) and chlorophyll a (Chl a), were extracted using the Marine Geospatial Ecology Toolbox (MGET) data products function in order to provide spatial coverage across the study area (Roberts et al. 2010). SST and Chl a data were extracted a monthly average, for all twelve months, at a 4 km spatial resolution. The monthly averages were then averaged by season. Additionally, the distance from shore (DFS) from each transect segment’s midpoint to the nearest coastline was calculated (ESRI 2011).

### Modeling detection probability

A conventional stratified analysis was conducted on the boat-based survey data in program DISTANCE to estimate the probability of detecting delphinids within a 5% truncation of the trackline (Laake 1994). In standard distance sampling a truncation limit of the largest distances, generally 5%, is set to avoid a size bias and increase the estimation of the detection function. Detection probability of bottlenose dolphin (*Tursiops truncatus*) encounters across seasons was modeled at the species level as a smooth function of perpendicular distance. Common dolphin (*Delphinus delphis*), unidentified dolphin, and sea turtle sightings were not included in this distance analysis due to the lack of total individuals. The sightings included in the line transect distance analysis only included those within the front 180 degree observation window from the boat, and thus effective strip width was used to calculate relative density of bottlenose dolphin encounters. Encounters of bottlenose dolphin groups (rather than individuals) were modeled due to uncertainty in group size estimates arising from “passing mode” surveys. Relative density modelling was stratified by season for spring, summer and fall (Figure 15-7 to Figure 15-9), as there were not enough sightings of bottlenose dolphins in winter. Candidate forms for the detection function were the half-normal model and hazard rate function with a cosine smoothing term (Buckland et al. 2001). Sea state, as recorded by observers on the Beaufort scale, was not included as a candidate covariate as no plausible detection functions were produced. Models were selected using Akaike’s Information Criterion (AIC; Akaike 1973).

Aerial transects were treated as strip transects, whereby density was determined as the number of sightings per transect length of 5km and strip width of 200m, and detection was assumed to be perfect (Buckland et al. 2005). Relative density estimates from the aerial transects were only produced for sea turtles by season for spring, summer and fall, as there were not enough sea turtle sightings during the winter aerial surveys. Species-specific aerial density estimates for marine mammals were not modelled due to the small sample size of individuals identified at the species level. Furthermore, a general “all delphinids” model was not run due to the challenges that arise by lumping multiple species that have distinct behaviors.

### *Building descriptive models*

The covariates for each 5 km boat and aerial transect midpoint were joined with their corresponding density and input into R for model fitting. For the purposes of this study, both general additive models, or GAMs (using R package mgcv), and general linear models, or GLMs (using the built-in glm R function), were used in model development following a negative binomial family fitting response (Wood 2006; Dobson and Barnett 2011; R Core Team 2014). Both model outputs were a result of different combinations of covariates. Seven different models were used for each model type (Table 15-4 to Table 15-9). The selection of the best model was based according to the AIC score and the percent of deviance from the null model that the model explained (Table 15-4 to Table 15-9). The higher the percent of deviance explained from the null model, the better that particular model fits the input data. In cases where the AIC values of two models were very similar, the percent deviance was solely used as the deciding criterion for model selection.

Once a model had been chosen according the selection criterion above, a 4 km square gridded data set was created for each season to act as the predicting platform for the model results. This platform extended 25-30 km east of the WEAs, 30 km south of the VA WEA and 75 km north of the DE WEA. Every grid cell centroid was assigned a distance to shore value, as well as SST and Chl a values extracted from seasonal climatologies using the MGETs data products toolbox in ArcGIS (Roberts et al. 2010). The seasonal prediction grids were then passed to the chosen descriptive model for bottlenose dolphins and sea turtles using the predict.gam command in R. The estimated encounter rates from the bottlenose dolphin detection functions and the calculated relative density of sea turtles per strip segment were used as the model response variables. The output of the model was an estimate of the predicted relative density at 1 km$^2$ at the center of each grid cell according to the variables used in the chosen model. These predicted densities were scaled according to the 16 km$^2$ prediction grid, imported in ArcMap 10.2 as a raster data set, and smoothed using the point to raster conversion function (ESRI 2011). Missing (white) cells in the interpolated relative density maps indicate areas where no covariate data were available or the prediction grid limits ended.

## Results

A total of 374 marine mammal and sea turtle sightings were reported in boat-based surveys, representing 1,349 individuals. Of these, 1,211 individuals were identified to the species level (Table 15-1). Of all observed marine mammal and turtle individuals, 1,200 were dolphins, 35 were whales, and 114 were sea turtles (Table 15-1). A total of 3,808 individual marine mammals and sea turtles were observed during the aerial surveys (Table 15-2). Of these, 2,036 were dolphins, 3 were porpoises, 16 were whales, 5 were unidentified cetaceans, and 1,748 were sea turtles (Table 15-2). Locations of whales, dolphins, and sea turtles observed on the boat survey are presented in Figure 15-1 to Figure 15-3. Locations of whales, dolphins, and sea turtles observed on the aerial surveys are presented in Figure 15-4 to Figure 15-6. Aerial survey observations were highest in May and July (Table 15-3). Humpback whales were the most common large whales observed, and five species were observed overall (Table 15-1 and Table 15-2). Bottlenose dolphins were the most common of the four delphinid species observed, and were observed mainly inshore (Table 15-1, Table 15-2, Figure 15-2, Figure 15-5). Common dolphins were the next most abundant species, and were more commonly observed offshore

(Figure 15-2, Figure 15-5). Sea turtle distributions were primarily offshore (Figure 15-3, Figure 15-6), and loggerhead sea turtles (*Caretta caretta*) were the most abundant of the five species observed (Table 15-1, Table 15-2).

In all cases, GAMs outperformed GLMs (Table 15-4 to Table 15-9). The encounter rate (number of sightings per kilometer squared) model for bottlenose dolphins in the spring predicted a strong nearshore-oriented density gradient within the prediction area, and the corresponding density map correlated well with the bottlenose sighting data spatially. The highest predicted encounter rates were at the mouth of the Delaware Bay (Figure 15-10), as well as near the western edges of the Delaware and Maryland WEAs. The encounter rate model for bottlenose dolphins in the summer predicted very strong nearshore-oriented, northerly concentrated density gradient in and around the mouth of the Delaware Bay (Figure 15-11), including a density of encounters near the western edges of the Delaware and Maryland WEAs. Like the spring model, the encounter rate model for bottlenose dolphins in the fall predicted a strong nearshore-oriented density gradient along the prediction area, with the highest densities seen farther south at the mouth of the Chesapeake Bay (Figure 15-12). The fall model predicted no substantial encounter overlap with any of the WEAs.

The relative density model for sea turtles in the spring predicted a very strong off shore-oriented, southern density gradient (Figure 15-13), including high densities within the Virginia WEA. The density model for sea turtles in the summer predicted a less dense gradient across the southeastern portions of the study area, including areas near and within the Virginia WEA (Figure 15-14). In the summer density map, the relative density of sea turtles also begins to migrate north. The predicted density model for sea turtles in the fall predicted a less dense, latitudinally uniform density gradient (Figure 15-15). The corresponding fall density map predicted high densities within all three WEAs.

## Discussion

Effective conservation plans require precise assessments of the spatial distributions and densities of the species they are trying to protect. With such information, policy makers, regulators, and managers can predict how a species’ distribution may respond to change within their environment, including naturally occurring fluctuations and human activities. Species distribution modeling can provide a measure of a species’ spatial density over a desired region. Our primary goal was to quantify sea turtle and marine mammal densities seasonally throughout the study area by developing models to examine spatial patterns and trends based on interactions with environmental conditions, in hopes of identifying species that could be exposed to future turbine construction and operation. By applying spatial modelling techniques to line transect boat-based survey data and high resolution digital video aerial survey footage, we produced relative density estimates of sea turtles and relative encounter rate estimates for bottlenose dolphins (as dolphin sightings may represent either an individual or a pod) across the study area by correlating species abundance to spatial and environmental covariates. One of the possible advantages gained by utilizing a spatial model to estimate density is an enhancement in the estimated precision, as deviation in density can be explained by relatively few spatial covariates (Hedley et al. 1999).

The combined effort of both surveys did not yield enough whale sightings to investigate potential density relationships with environmental covariates. An examination of publicly available whale data outside the study was conducted, but there were still insufficient sightings within the last ten years to allow for parameterization of a model (the 10-year temporal limit was set to avoid any variation in sighting patterns that could be caused by climate change). It is still important to note, however, that large whales were observed across the survey area during both surveys, including within each of the proposed WEAs. Of particular importance, nine North Atlantic right whales were observed during surveys. While these data were insufficient to identify geographic patterns, they corroborate data from previous studies indicating that the mid-Atlantic region is in the path of North Atlantic right whale annual migratory movements. Currently, North Atlantic right whales are among the most endangered whales in world, with an estimated 455 individuals left in the western North Atlantic (Fisheries 2015). They are protected under the United States Endangered Species Act (ESA) as well as the Marine Mammal Protection Act (MMPA). Vessel strikes and entanglement in fishing gear account for nearly half of all North Atlantic right whale mortality since 1970 (Knowlton and Kraus 2001). Considering hearing is a sensory modality for these animals, it is important to understand the potential increase in underwater noise posed by construction of offshore wind energy facilities. A study published in 2012 discovered that a decrease in underwater noise was associated with a decrease in baseline levels of stress-related hormones, such as glucocorticoids and cortisol, which are associated with chronic stress, and if not produced at proper levels can hinder the processes of a successful birth and even lead to adult mortality (Rolland et al. 2012). A recent passive acoustic study showed that North Atlantic right whales were present off the coasts of North Carolina and Georgia in all seasons, with peaks in abundance in autumn and winter, when they were not expected to be present (Hodge et al. 2015). We encourage further data collection in the region to better understand the distribution and the timing of presence of large whales, in particular the North Atlantic right whale, in relation to environmental covariates and the position of the WEAs.

For both bottlenose dolphins and sea turtles, GAMs proved more effective at modelling the density of these animals with relation to spatial covariates then their counterpart GLMs. This is due to GAMs' capacity to model the non-linear nature of ecological data and produce complex response curves (Guisan et al. 2002; Venables and Dichmont 2004). It is also important to look at the effectiveness of each models capacity to model temporal trends. GLMs are generally used in modelling long-term trends, such as annual outcomes, while GAMs are better at modelling short term responses, such as across seasons (Cheng and Gallinat 2004). However, it is also important to note that if not used carefully, GAMs can seriously over-fit data, and thus have low predictive power. GAMs also do not allow for the depiction of the interaction of two or more spatial covariates.

The relative density of bottlenose dolphin encounters within the study area during the spring was explained by Chl a and DFS, the summer model was best explained by only SST and Chl a, and the fall model was best explained by SST and DFS. The relationship with SST may be attributed to bottlenose dolphins' migratory behaviors, as the species generally moves south as temperatures decline (Barco et al. 1999; Natoli et al. 2005). It is also probable that there are permanent residents, transients, and seasonal migrants of this species that occupy estuarine, coastal, and offshore waters from Florida to

New Jersey (Urian et al. 2009). North of Cape Hatteras, North Carolina, bottlenose dolphins display a bimodal distribution with coastal and offshore components (Kenney 1990), and the mid-Atlantic study area likely contains several different coastal morphotypes at different times of year, including both Northern Migratory and Southern Migratory stocks (Waring et al. 2013). The relationship between bottlenose dolphin encounter rates and DFS in this study is likely due to the inshore distribution of the coastal ecotype of bottlenose dolphins during the spring, summer and fall seasons (Kenney 1990; Gannon and Waples 2004). It is possible that during the spring and fall months, resident coastal ecotype dolphins were surveyed more often, thus producing the very nearshore density gradient observed in this study. In summer, however, the influx of transient populations may have produced a more robust density gradient from west to east. The association with high areas of Chl a may be attributed to delphinids' capacity to utilize areas of high primary productivity for feeding, particularly in and around the mouths of the Chesapeake Bay and Delaware Bay (Young and Phillips 2002). It is important to note that the development of the bottlenose dolphin models excluded dolphins lumped into the "unidentified" category, of which some proportion were likely bottlenose dolphins.

The relative density of sea turtles during the spring was best explained by SST and DFS. The relative density of sea turtles during the summer was best explained by SST and Chl a, while the fall model was best explained by only DFS. Past aerial surveys have shown that loggerhead sea turtles, in particular, migrate into mid-Atlantic coastal waters at depths of 60 meters or less as the water warms during the spring (Shoop 1980). This would explain the higher density of sea turtles predicted in the spring, as roughly 60% of the identified sea turtles from both surveys were loggerhead sea turtles. In general, there was a decreasing trend in density from spring to fall. The most common sea turtles observed in the aerial survey were loggerhead and leatherback sea turtles. Prime nesting for these two species occurs from March to September along the east coast of the United States (Miller et al. 2003; Rabon Jr et al. 2003). As nesting of female sea turtles occurs on sandy beaches, we would expect the sexually mature females to be closer to shore during the nesting season. It is possible that the aerial survey did not efficiently survey the nesting population during the summer, as surveys did not extend within 5.5 km of shore in most locations; this could explain the lower predicted densities than in spring. Furthermore, the northern migration of predicted densities during the summer and fall could be a result of the mixing of the northern Labrador and Gulf Stream currents. The complete mixing of the these currents around the survey region occurs in late summer and early fall (Talley and McCartney 1982; Rossby and Benway 2000). The delayed uniform mixing of these currents has the potential to hinder the northern migration of these turtles. This is also likely why so few turtles were observed in the winter, as bottom temperatures in the mid-Atlantic drop to 10°C or less by mid-December, a known lethal thermal limit for some species of sea turtles (Schwartz 1978; Lutcavage and Musick 1985; Hawkes et al. 2007). It is also possible that this delayed mixing accounts for the greater number of turtles observed in the summer as it is estimated that turtles within our study area spend about 25% of the time at the surface basking during the spring (cooler water temps), as opposed to about 5% of the time during the summer and fall when current mixing has occurred (Barco et al. 1999).

### *Sea turtles and offshore wind energy development in the mid-Atlantic*

Five of the seven extant species of sea turtle occur in the mid-Atlantic study area, and all five are protected under the Endangered Species Act. As such, they are likely to be priority species for regulators during the offshore wind environmental permitting process. Sea turtles are uncommon in European waters, so no information is available about their interactions with offshore wind facilities. Sea turtles could potentially be affected by offshore wind energy development in several ways, however, including noise from seismic surveys, construction, and operations; electromagnetic fields; vessel collisions; and changes to habitat caused by artificial reef effects (Read 2013).

Construction of offshore wind facilities has been identified as the development period with the most potential risks for sea turtles, due to noise from pile driving and other activities, though the potential for injury remains largely unknown (Michel 2013). Sea turtles can detect low-frequency sounds (Lenhardt et al. 1983; Dow Piniak et al. 2012), and the frequencies emitted by seismic airguns, offshore drilling, low-frequency and mid-frequency sonar, pile driving, cargo vessels, and operational wind turbines are all within the underwater hearing range of Leatherback sea turtles (Dow Piniak et al. 2012). Sea turtles have exhibited startle responses when exposed to low frequency sounds and vibrations in a laboratory setting (Lenhardt et al. 1983), and laboratory and *in situ* studies on seismic airguns for offshore oil and gas exploration have showed changes in sea turtle swimming pattern and orientation (O'Hara and Wilcox 1990) and a range of avoidance behaviors up to at least one kilometer away from the source (O'Hara and Wilcox 1990; McCauley et al. 2000). Sea turtles are known to collide with vessels, and are also displaced from areas with vessel traffic, though observed responses to boat noise have varied with species (Samuel et al. 2005; Lester et al. 2013).

During operations of offshore wind facilities, sea turtles may be displaced due to turbine or vessel noise, or may aggregate around turbine foundations due to artificial reef effects, which change local habitats (Read 2013). Similar aggregation patterns have been observed around oil rigs in the Gulf of Mexico (Continental Shelf Associates 2004). The degree to which turbines may aggregate sea turtles will likely vary by location and other factors, and the effects on individuals or populations are unclear. Likewise, past studies have shown that electromagnetic fields (EMF) and heat signatures associated with offshore turbines have the potential to affect species such as sea turtles that use geomagnetic cues during migration (Lohmann et al. 2008). Data on the effects of EMF on sea turtles are generally lacking, however (Read 2013), and we know of no studies to date that have assessed whether EMF emissions from subsurface cables at operational facilities influence navigational decisions of turtles.

Overall, our results indicate that there is overlap between predicted habitat usage of sea turtles and the placement of proposed WEAs in the mid-Atlantic. Chesapeake Bay and the coastal waters of Virginia are known to serve as a key summer developmental habitat for juvenile sea turtles, particularly loggerheads and Kemp’s ridley sea turtles, thus placing the Virginia WEA in a potentially sensitive location (Lutcavage and Musick 1985). Winter is the time period where the likelihood of interactions between offshore construction and sea turtles is lowest, but winter is a difficult time for offshore construction, and most development activities are likely to occur in the other three seasons. During spring, summer, and fall, the relative density of sea turtles did not change drastically, though the distribution of turtles across the study area varied substantially (Figure 15-13 to Figure 15-15). As such, and given the group’s conservation status, the development of techniques to avoid or reduce interactions between sea turtles and construction activities, vessel traffic, and other development activities should be considered a priority. The development of taxon-specific effects data is also a key area for additional research.

### *Bottlenose dolphins and offshore wind energy development in the mid-Atlantic*

All cetaceans that occur in the U.S. are protected under the Marine Mammal Protection Act. Cetaceans use sound for communication, and some, like dolphins, also use echolocation to navigate through their environment and hunt for prey. Acoustic disturbance has been recently identified as the primary concern for marine mammals with regards to offshore wind development in Europe (Bergström et al. 2014). This may include noise from seismic surveys, blasting, pile driving, and operational turbines. The severity of avoidance and displacement effects appear to vary with a variety of factors, including the frequency, intensity, and duration of noise, as well as species and time of year (Goold 1996; McCauley et al. 2000; Madsen et al. 2002). European studies have indicated that Harbor Porpoises could hear pile driving noise over 80 km from the source, and showed displacement up to 20 km away during construction (Thomsen et al. 2006; Teilmann and Carstensen 2012). Results of operational displacement studies in Denmark and the Netherlands have varied (Scheidat et al. 2011; Teilmann and Carstensen 2012). There has been little or no detectable avoidance during operations at some facilities, while in at least one instance, even nine years after construction had been completed, porpoise acoustic activity levels were at only 29% of pre-construction levels (Teilmann and Carstensen 2012). Prey availability may be an important factor affecting porpoise behavior around operational wind facilities (Teilmann and Carstensen 2012).

Overall, our results indicate that there is overlap between predicted habitat usages of bottlenose dolphins and the placement of proposed WEAs in the mid-Atlantic, although the relationship between dolphin distributions and these areas of potential offshore wind energy development may be somewhat difficult to interpret from this particular data set. Our models suggest minimal presence of bottlenose dolphins within WEAs during cooler months. However, it is important to note that other species of delphinids, such as common dolphins, are more cold-tolerant than bottlenose dolphins. Common dolphin observations increased in both the boat-based and aerial surveys during winter and early spring. Thus, it is possible that delphinids will be present in some numbers in WEAs during all seasons. Efforts to mitigate the effects of construction activities, in particular, will be important as offshore wind energy development proceeds in the mid-Atlantic.

### *Caveats, considerations and next steps*

Conservationists and policy makers must remember that models are simply an approximation of a species' potential distribution and density. Modeling the density and distributions of marine mammals and sea turtles in the present study was challenging due to the methods employed during surveys, the limited number of sightings generated during surveys, and the difficulties of merging aerial and boat-based survey data. "Passing mode" surveys, where the research vessel does not deviate from the transect line, present significant challenges in determining species identifications and group size. Many marine mammals will form multi-species groups that often become apparent only after close approach, and the movements and dive behavior of these animals make judging group size from a distance difficult. As a result, we chose to model encounter rates (with one or more delphinid) rather than predict numbers of individual animals. Clearly, in applying any analytical technique to ecological data, tradeoffs are often involved to meet certain assumptions. Traditional distance sampling, in particular, assumes all objects on or near the transect line are detected with 100% certainty; that the animals are detected at their initial location; and that recorded distances and angles made by the observer are exact, without measurement error or bias (Thomas et al. 2002). The marine environment and the general physiology (diving behavior) of these animals make it very difficult to meet these assumptions. As previously mentioned, marine mammals were not modelled using any of the aerial data due to the small sample size of individuals identified at the species level. Furthermore, a general “all delphinids” model would not have been useful as lumping multiple species that have distinct behaviors would have likely been problematic and uninformative. Many species of marine mammals can be highly clustered in space and time, leading to difficulties in merging datasets collected under disparate methods, both of which contained methodological and technological shortcomings.

Future boat survey assessments of marine mammals in this region should be designed to best address issues associated with species identification and group size estimation, ideally using a “closing mode” approach, whereby the research vessel would deviate from the transect line to more accurately describe a sighting by allowing more time for each encounter. A dedicated dual observer approach would also be warranted, as observers searching for both birds and marine mammals must maintain an extremely high level of vigilance to achieve appropriate survey effort. Clearly, aerial surveys pose a challenge to marine mammal surveys due to behaviors such as fast surface intervals as well as species identification success. However, the aerial survey did prove useful in sea turtle relative density estimates, where unlike marine mammals, the number of species observed in our study area was limited, as is the diversity in species-specific behaviors.

Finally, small sample sizes pose challenges to any statistical analyses, and result in diminished analytical potential as compared to models developed with more data (McPherson et al. 2004). As sample size increases, accuracy and predictive power also increase, at least until reaching a maximum accuracy potential (Hernandez et al. 2006). Future surveys designed specifically for marine mammals will help address this issue and improve our understanding of marine mammal distributions and habitat use in the mid-Atlantic United States.

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## Figures and tables

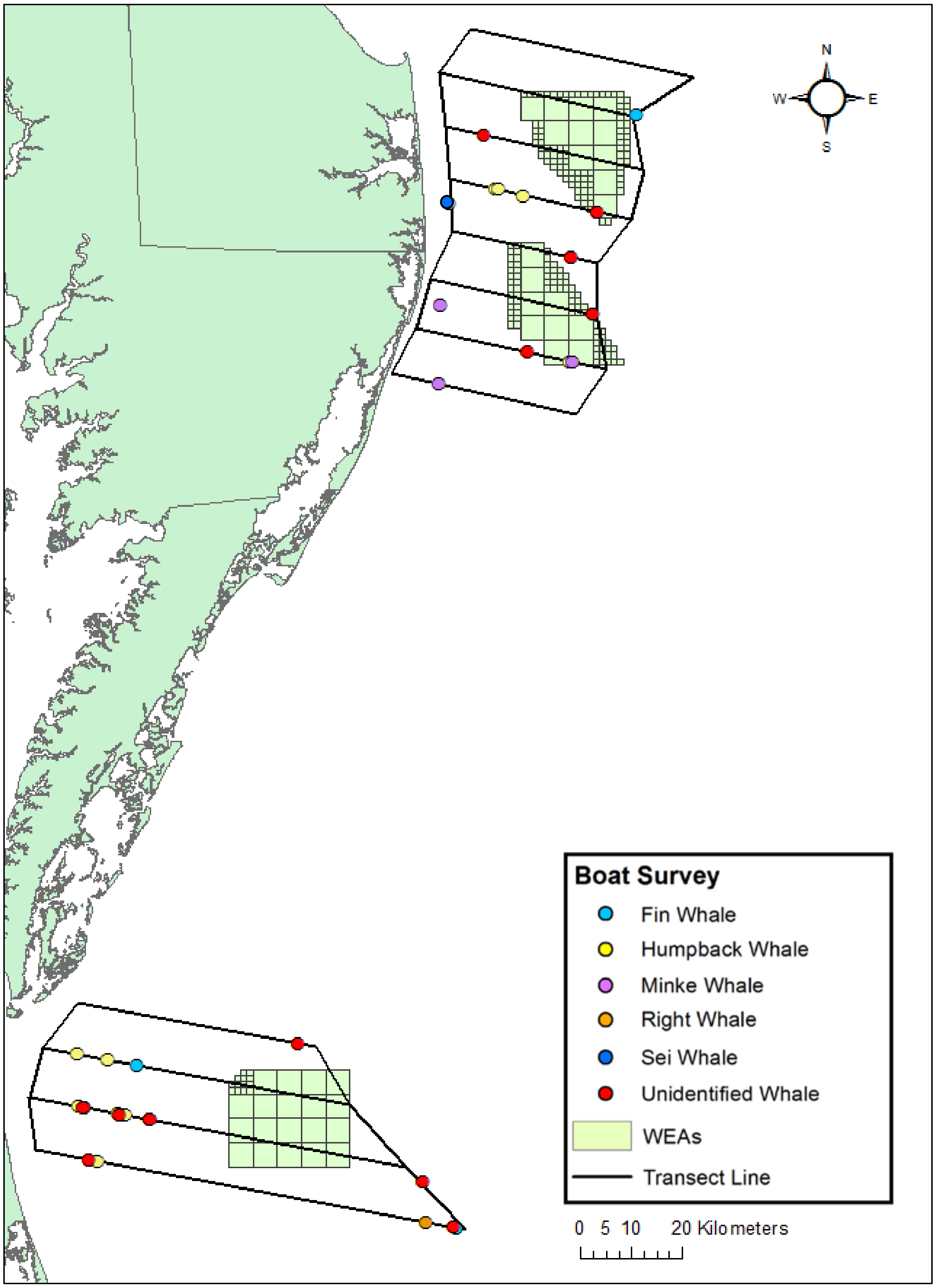

**Figure 15-1. Whale sightings from boat survey transects (all surveys, 2012-2014).**

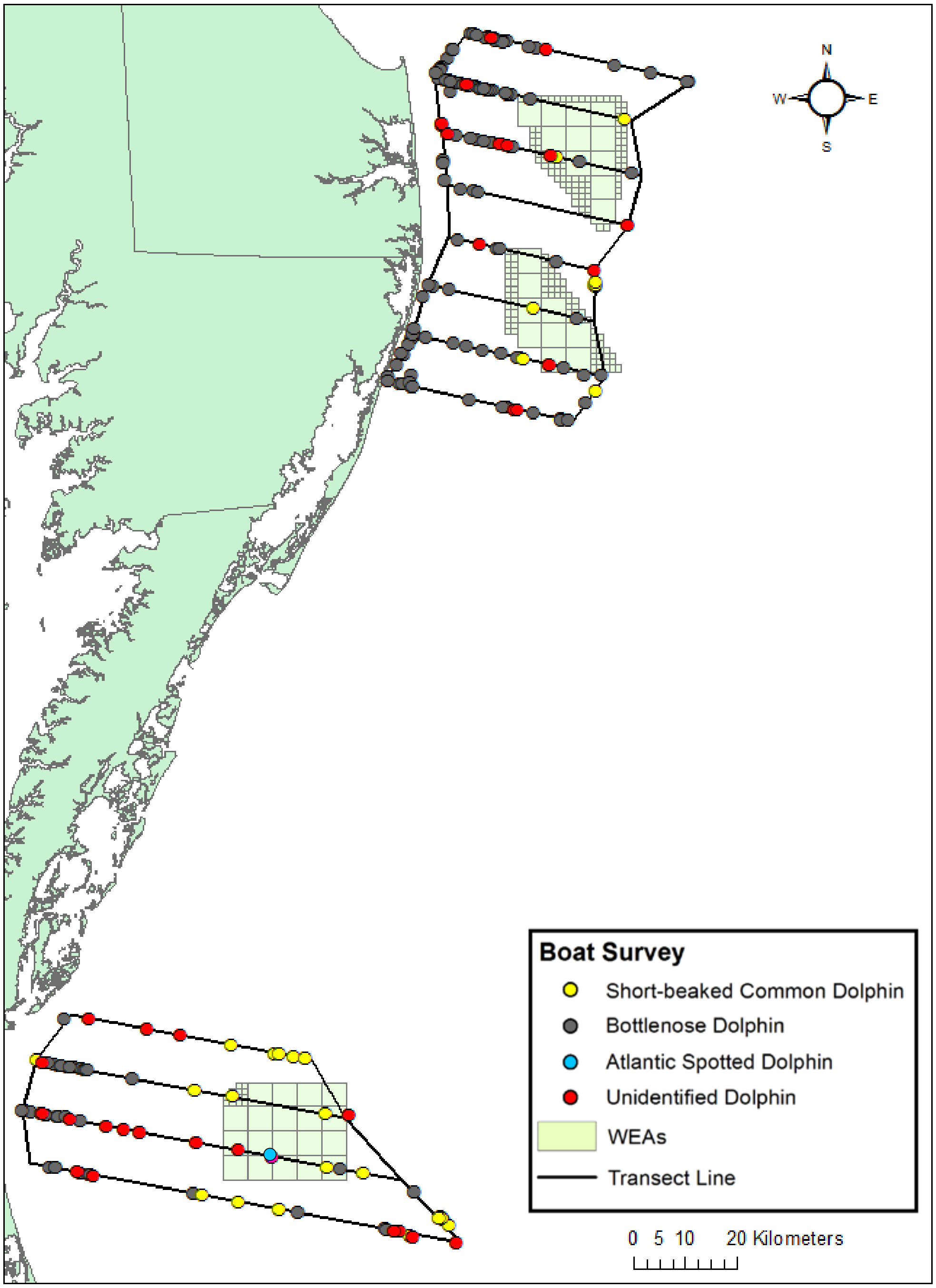

**Figure 15-2. Delphinid sightings from boat survey transects (all surveys, 2012-2014).**

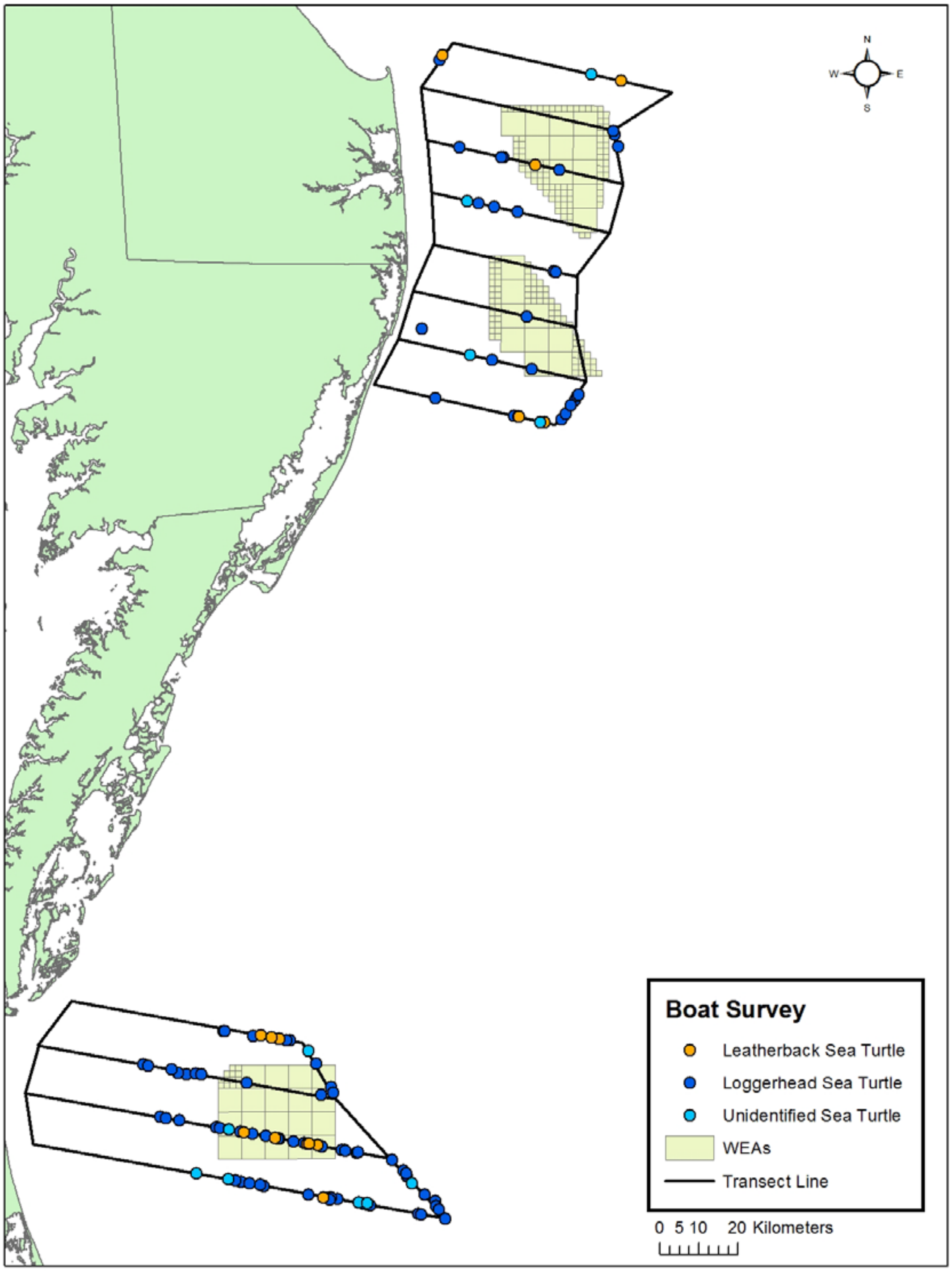

**Figure 15-3. Sea turtle sightings from boat survey transects (all surveys, 2012-2014).** Unidentified sea turtles are non-Leatherback Sea Turtles that were not definitively identified to species.

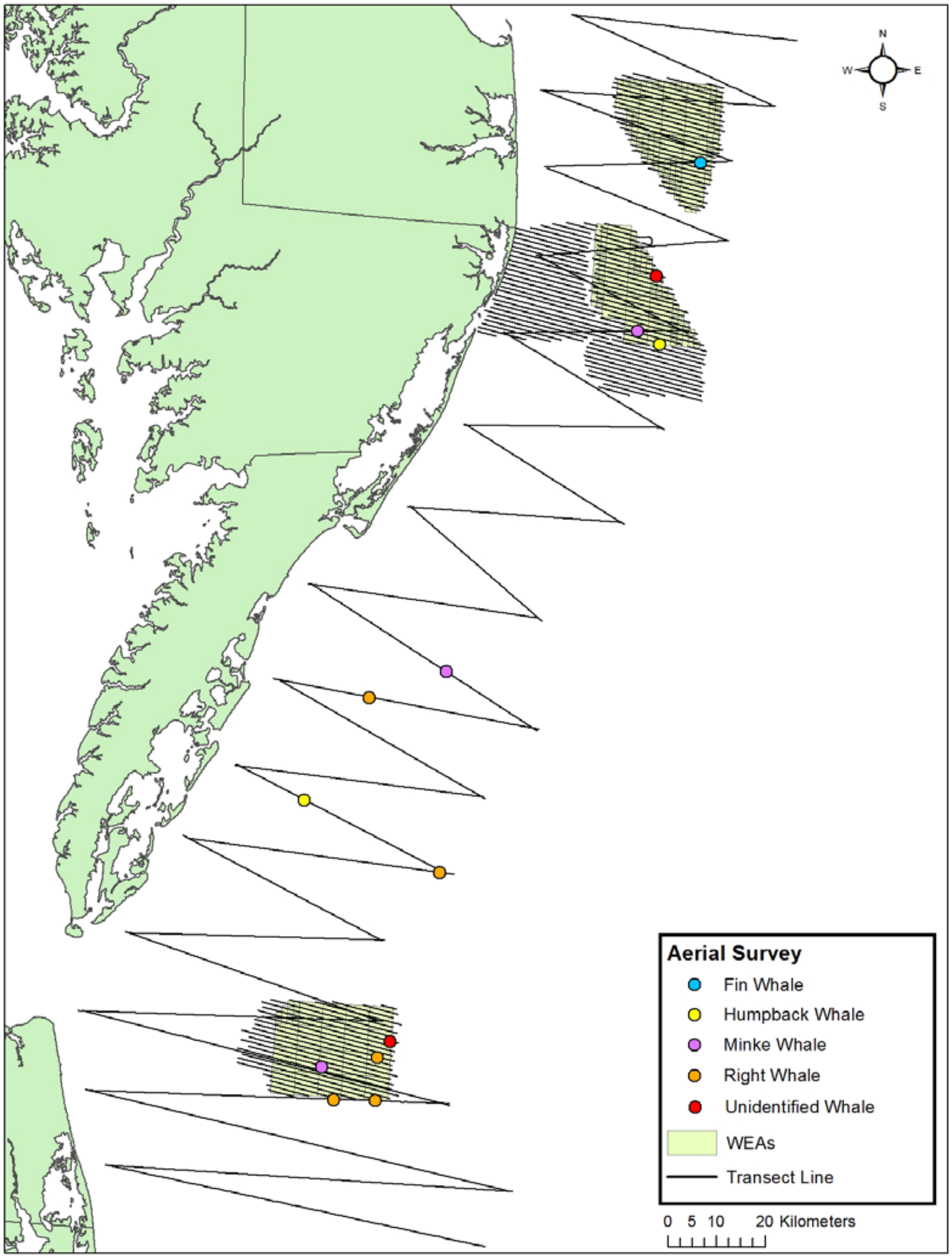

**Figure 15-4. Whale sightings from aerial survey transects (all surveys, 2012-2014).**

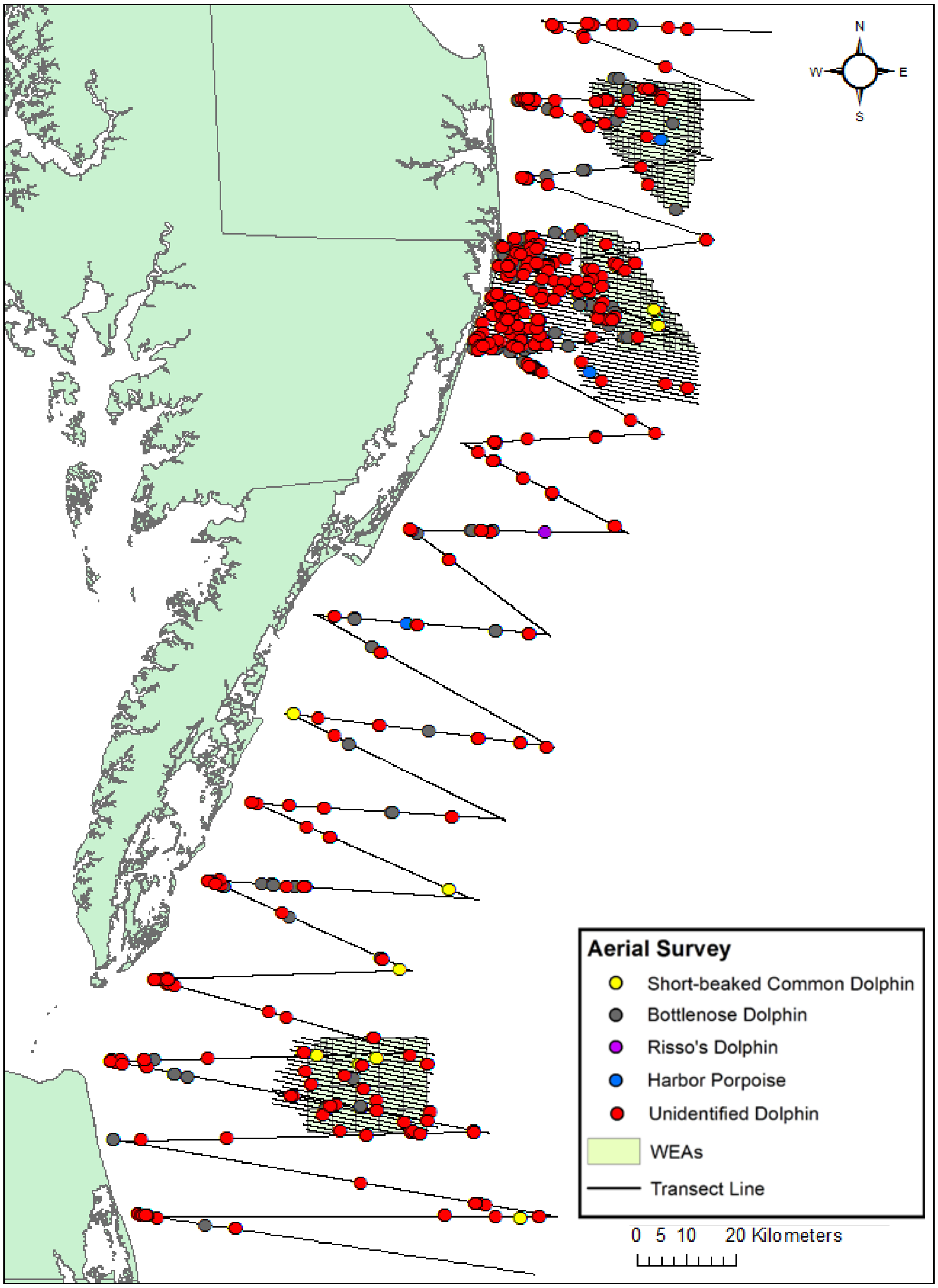

Figure 15-5. Delphinid and porpoise sightings from aerial survey transects (all surveys, 2012-2014).

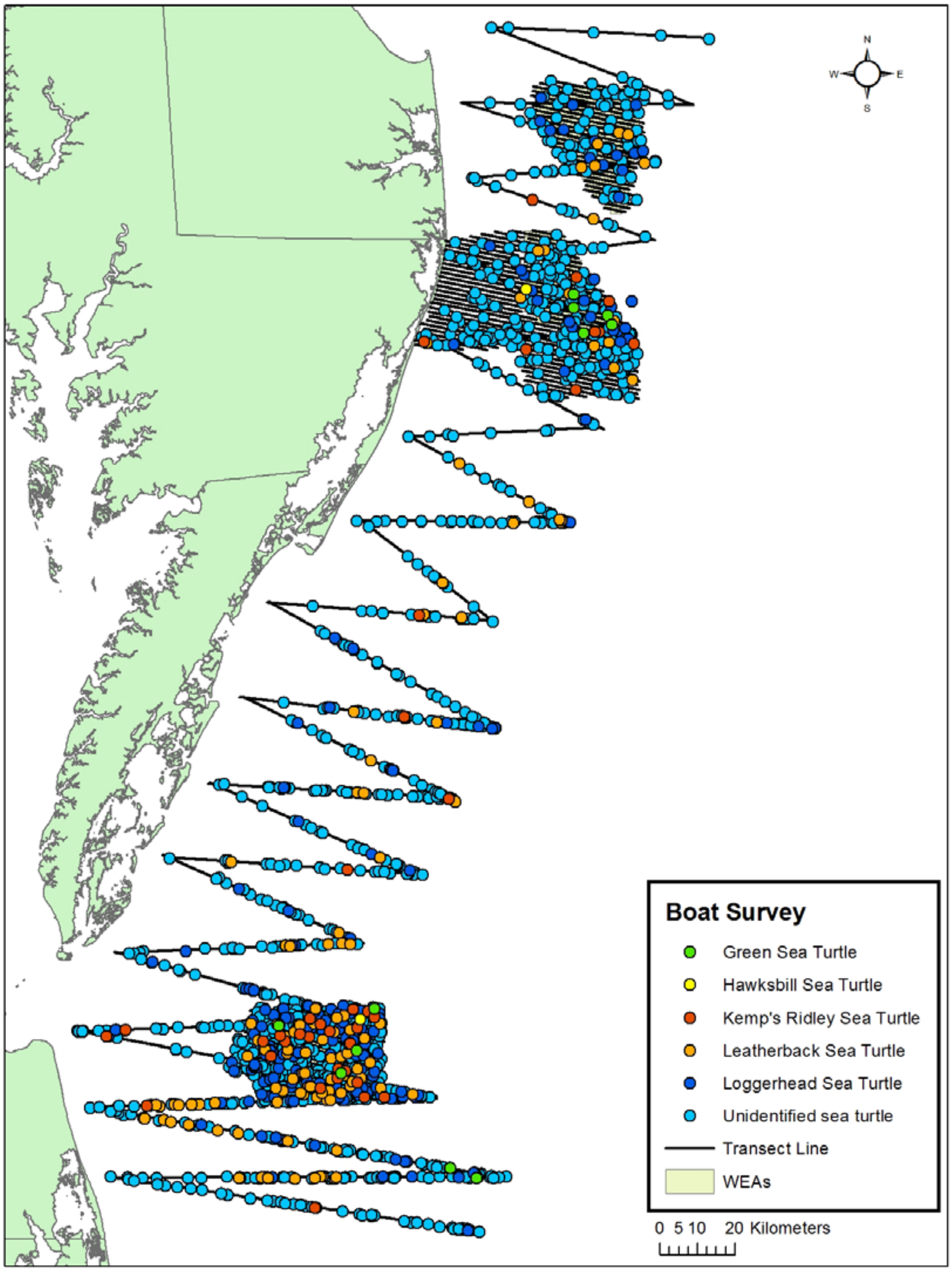

**Figure 15-6. Sea turtle sightings from aerial survey transects (all surveys, 2012-2014).** Unidentified sea turtles are non-Leatherback Sea Turtles that were not definitively identified to species.

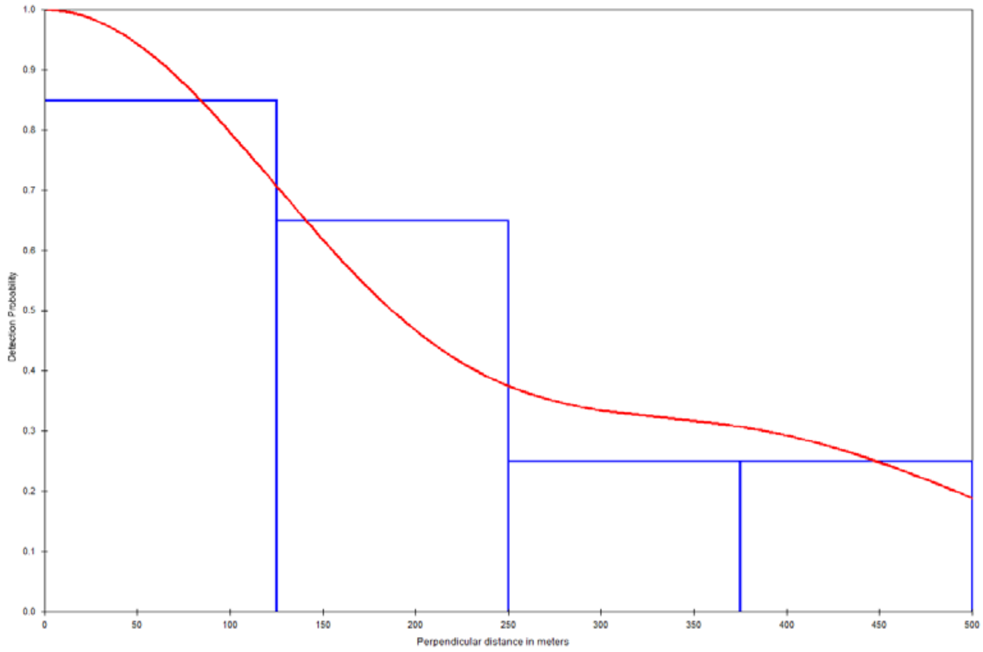

Figure 15-7. Spring global detection function used in boat survey bottlenose dolphin line transect distance density analysis.

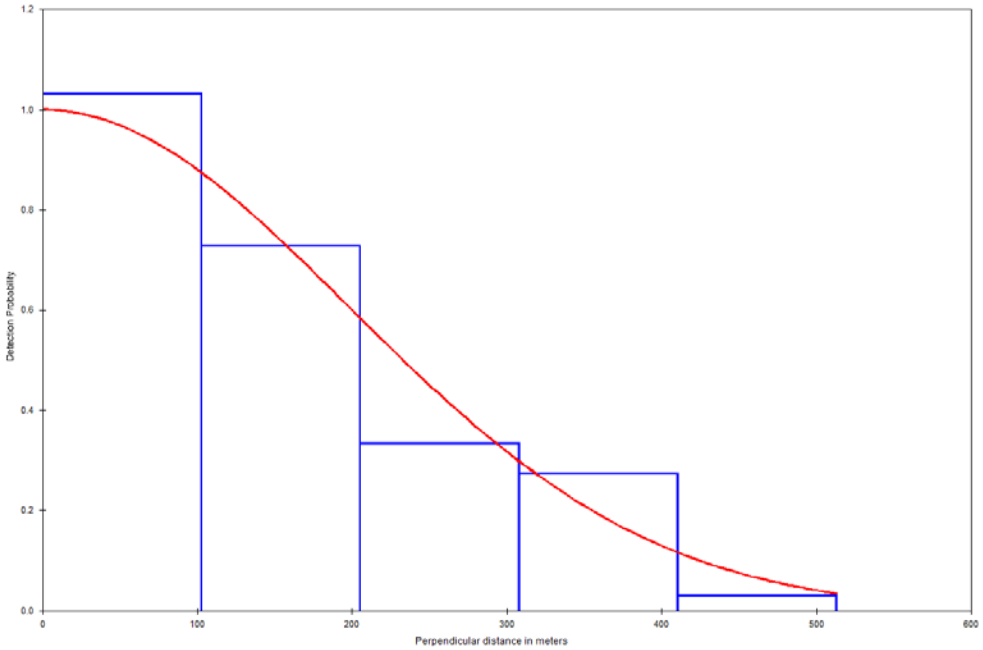

Figure 15-8. Summer global detection function used in boat survey bottlenose dolphin line transect distance density analysis.

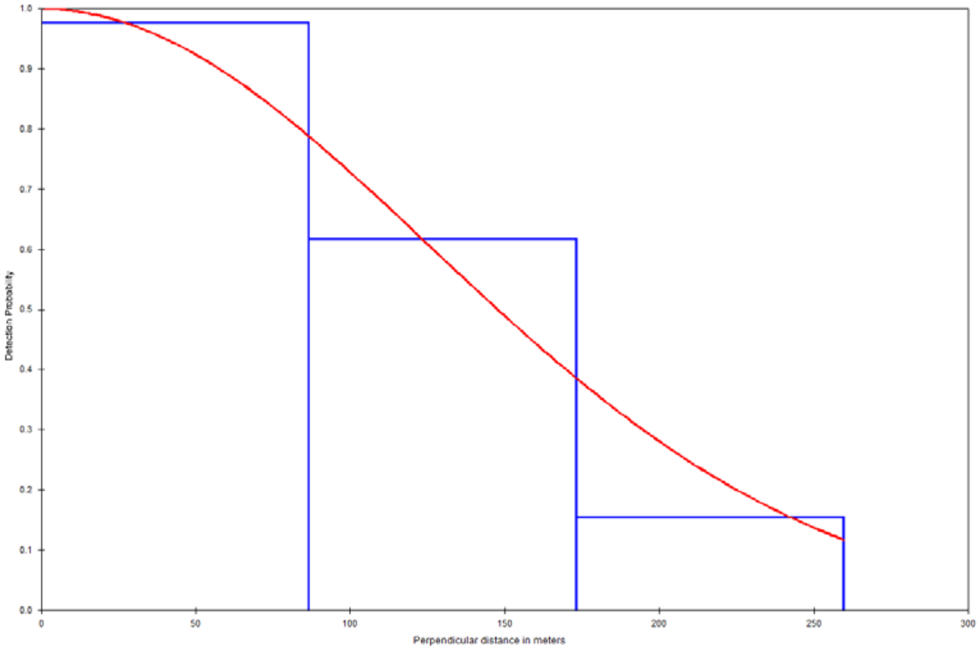

**Figure 15-9. Fall global detection function used in boat survey bottlenose dolphin line transect distance density analysis.**

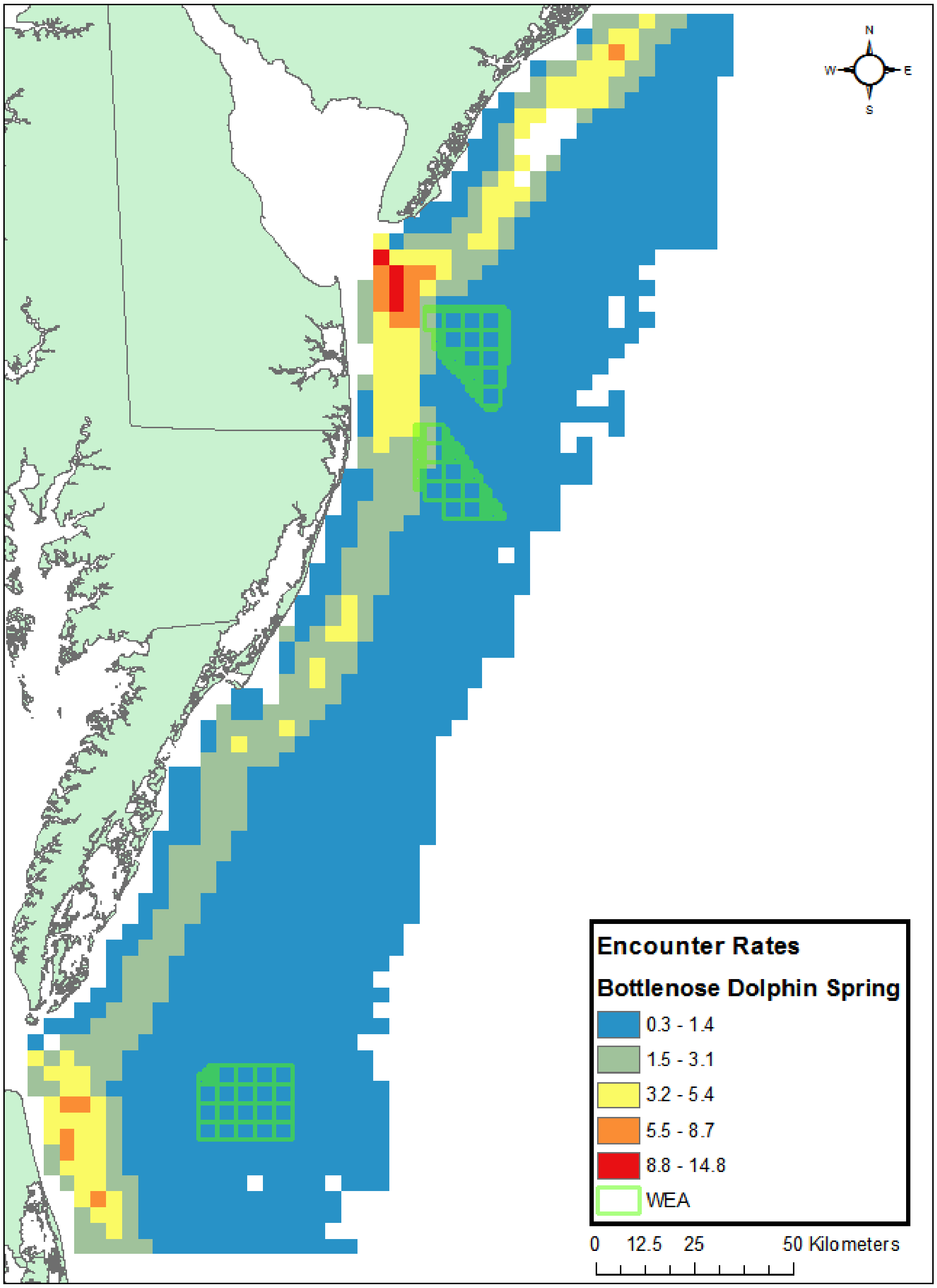

**Figure 15-10. Interpolation of encounter rates of bottlenose dolphins in the study area during the spring (Mar.-May), based on two years of boat survey data (2012-2014).**

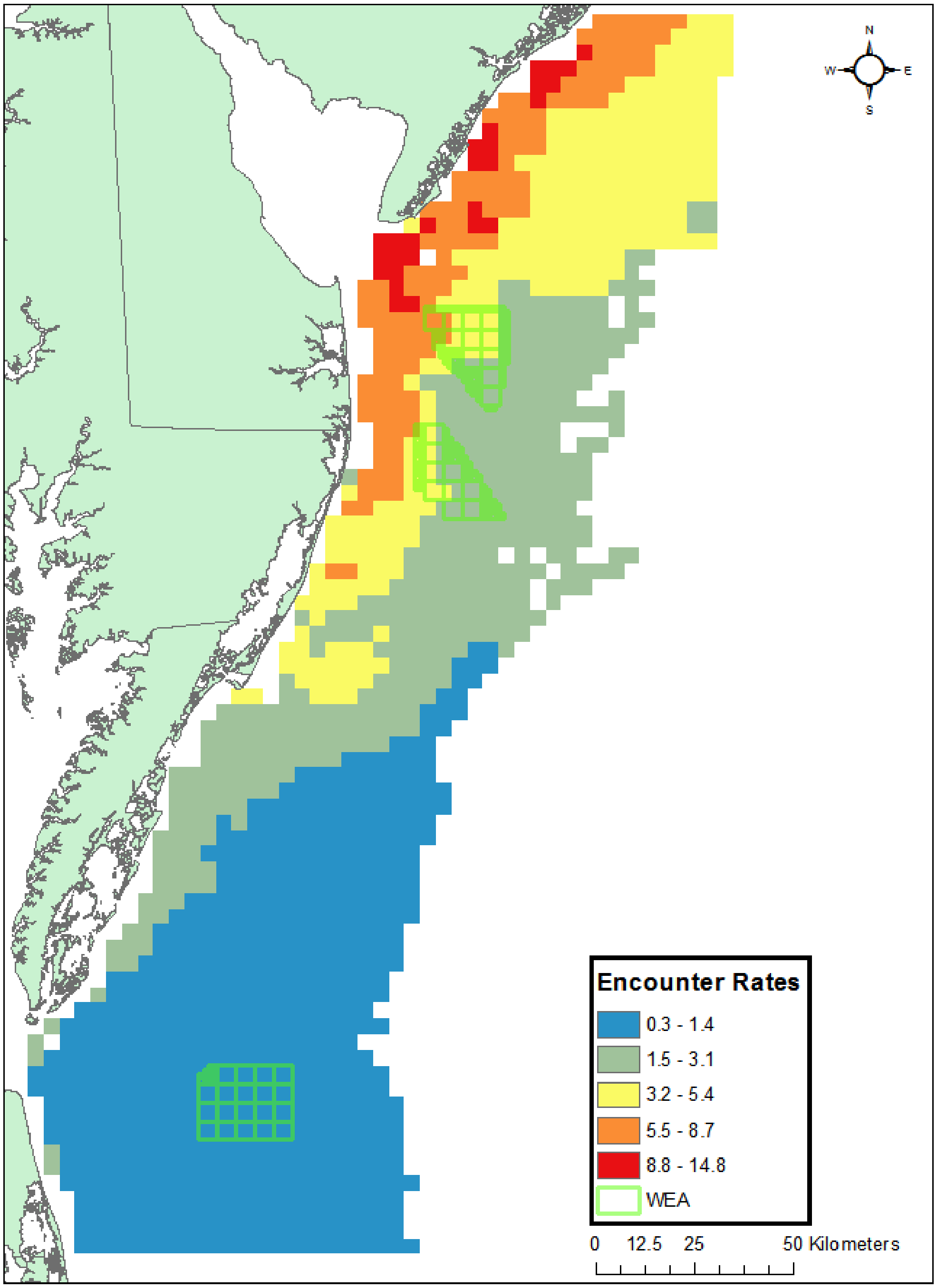

**Figure 15-11. Interpolation of encounter rates of bottlenose dolphins in the study area during the summer (Jun. –Aug.), based on two years of boat survey data (2012-2014).**

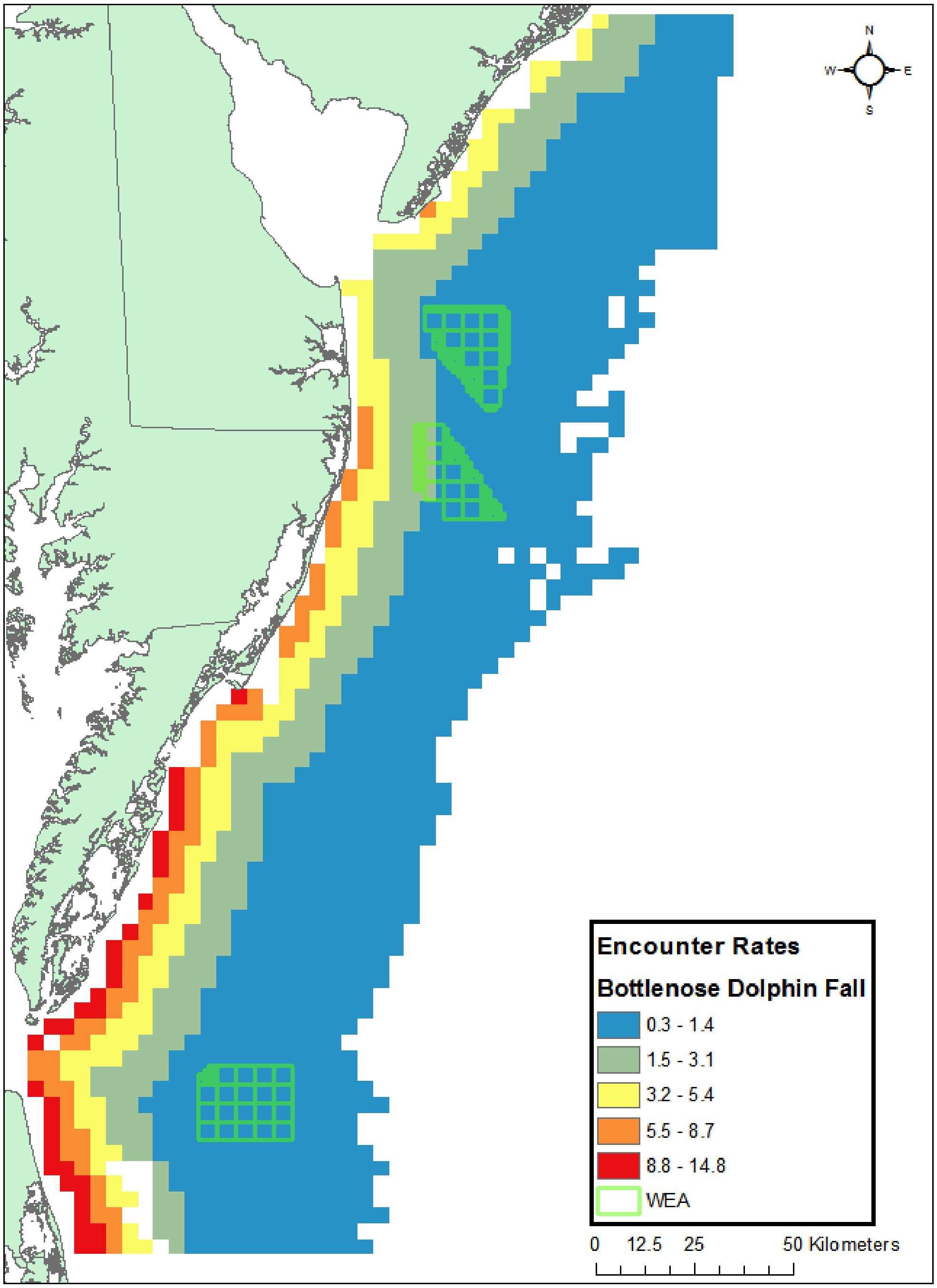

**Figure 15-12. Interpolation of encounter rates of bottlenose dolphins in the study area during the fall (Sep.-Nov.), based on two years of boat survey data (2012-2014).**

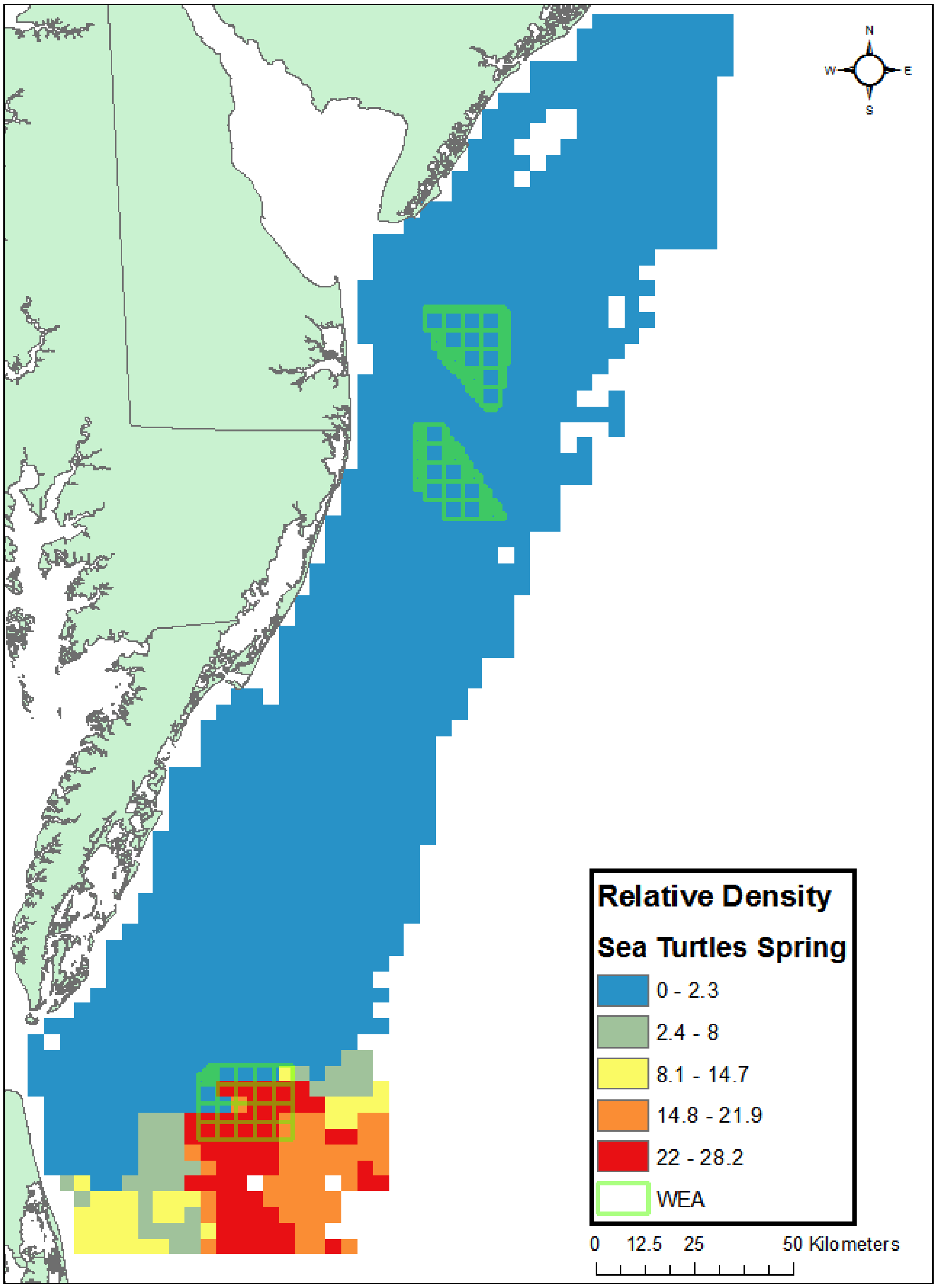

**Figure 15-13. Interpolation of predicted relative density of sea turtles in the study area during the spring (Mar.-May), based on two years of aerial survey data (2012-2014).**

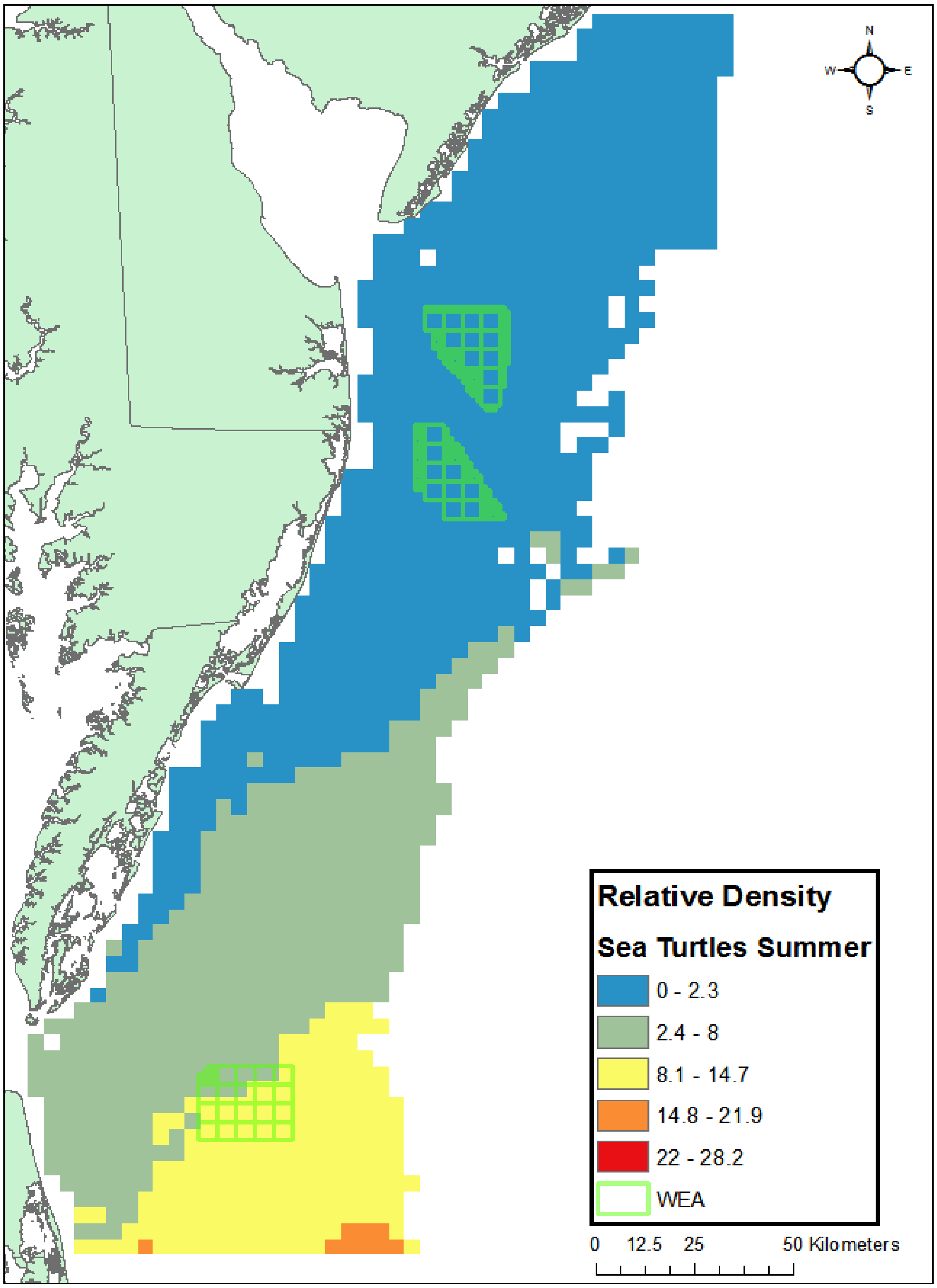

**Figure 15-14. Interpolation of predicted relative density of sea turtles in the study area during the summer (Jun.-Aug.), based on two years of aerial survey data (2012-2014).**

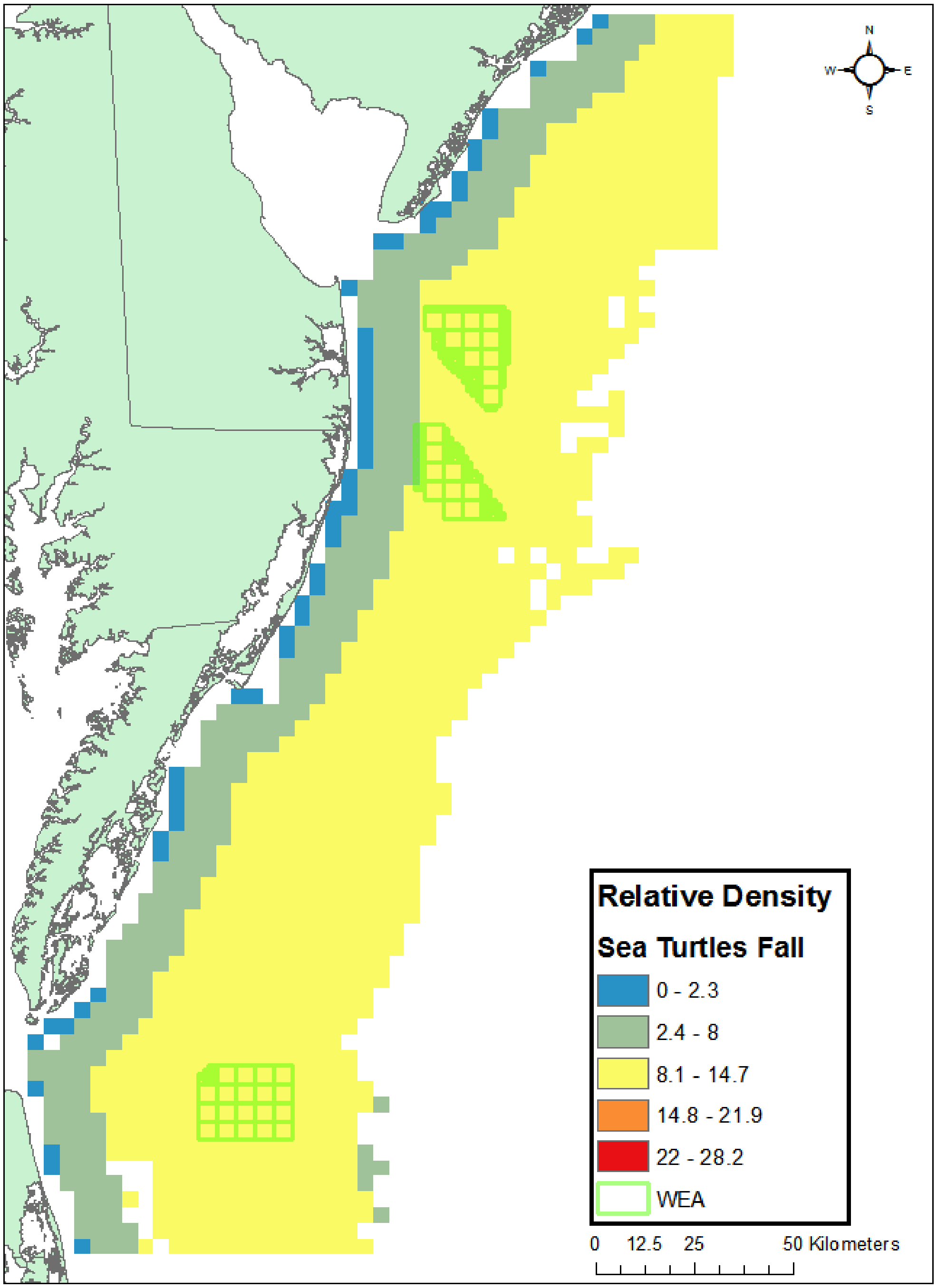

**Figure 15-15. Interpolation of predicted relative density of sea turtles in the study area during the fall (Sep.-Nov.), based on two years of aerial survey data (2012-2014).**

**Table 15-1. Summary data for boat surveys by season.** (Spring: March - May; Summer: June - August; Fall: September - November; Winter: December – February). Counts include all observed individuals on the survey transects.

| **Species Group** | **Spring** | **Summer** | **Fall** | **Winter** | **Total Count (Ind.)** |
|---|---|---|---|---|---|
| *Tursiops truncatus* (Bottlenose) | 239 | 400 | 227 | 8 | 874 |
| *Delphinus delphis* (Common) | 65 | 0 | 0 | 144 | 209 |
| *Stenella frontalis* (Spotted) | 0 | 4 | 0 | 0 | 4 |
| Unidentified Delphinid | 11 | 35 | 54 | 13 | 113 |
| **Dolphins Total** | **315** | **439** | **281** | **165** | **1200** |
| *Balaenoptera physalus* (Fin) | 2 | 0 | 0 | 1 | 3 |
| *Balaenoptera borealis* (Sei) | 0 | 0 | 0 | 1 | 1 |
| *Balaenoptera acutorostrata* (Minke) | 0 | 0 | 1 | 2 | 3 |
| *Eubalaena glacialis* (Right) | 1 | 0 | 0 | 0 | 1 |
| *Megaptera novaeangliae* (Humpback) | 0 | 1 | 4 | 7 | 12 |
| Unidentified Whale | 2 | 0 | 3 | 10 | 15 |
| **Whales Total** | **5** | **1** | **8** | **21** | **35** |
| *Caretta caretta* (Loggerhead) | 11 | 52 | 26 | 0 | 89 |
| *Dermochelys coriacea* (Leatherback) | 0 | 9 | 6 | 0 | 15 |
| Unidentified Sea Turtle | 2 | 4 | 4 | 0 | 11 |
| **Sea Turtles Total** | **13** | **65** | **36** | **0** | **114** |
| Percent of Total by Season: | 24.68 | 37.44 | 24.09 | 13.79 | 100 |
| ***Grand Total*** | ***333*** | ***505*** | ***325*** | ***186*** | ***1,349*** |

**Table 15-2. Summary data for aerial surveys by season.** (Spring: March - May; Summer: June - August; Fall: September - November; Winter: December - February). Counts include all observed individuals on the survey transects.

| Species Group | Spring | Summer | Fall | Winter | Total Count (Ind.) |
|---|---|---|---|---|---|
| *Tursiops truncatus* (Bottlenose) | 226 | 265 | 176 | 10 | 677 |
| *Delphinus delphis* (Common) | 11 | 7 | 4 | 39 | 61 |
| *Grampus griseus* (Risso's) | 0 | 0 | 1 | 0 | 1 |
| Unidentified Toothed Whale | 282 | 420 | 454 | 141 | 1297 |
| **Dolphins Total** | **519** | **692** | **635** | **190** | **2036** |
| *Phocoena phocoena* (Harbor Porpoise) | 2 | 0 | 0 | 1 | 3 |
| **Porpoises Total** | **2** | **0** | **0** | **1** | **3** |
| *Balaenoptera physalus* (Fin) | 0 | 0 | 0 | 1 | 1 |
| *Balaenoptera acutorostrata* (Minke) | 1 | 0 | 1 | 1 | 3 |
| *Eubalaena glacialis* (Right) | 3 | 0 | 0 | 5 | 8 |
| *Megaptera novaeangliae* (Humpback) | 0 | 0 | 0 | 2 | 2 |
| Unidentified Whale | 1 | 0 | 0 | 1 | 2 |
| **Whales Total** | **5** | **0** | **1** | **10** | **16** |
| Unidentified Cetacean (Whale or Dolphin) | 2 | 1 | 2 | 0 | 5 |
| **Unidentified Cetaceans Total** | **2** | **1** | **2** | **0** | **5** |
| *Caretta caretta* (Loggerhead) | 60 | 50 | 78 | 0 | 188 |
| *Dermochelys coriacea* (Leatherback) | 2 | 78 | 42 | 0 | 122 |
| *Chelonia mydas* (Green) | 3 | 1 | 7 | 0 | 11 |
| *Eretmochelys imbricate* (Hawksbill) | 0 | 0 | 2 | 0 | 2 |
| *Lepidochelys kempii* (Kemp's) | 13 | 10 | 14 | 1 | 38 |
| Unidentified Sea Turtle | 523 | 438 | 425 | 1 | 1387 |
| **Sea Turtles Total** | **601** | **577** | **568** | **2** | **1748** |
| Percent of Overall Total by Season: | 29.65 | 33.35 | 31.67 | 5.33 | 100 |
| ***Grand Total*** | ***1129*** | ***1270*** | ***1206*** | ***203*** | ***3808*** |

**Table 15-3. Individual marine mammal and sea turtle aerial survey sightings by month in relation to survey effort**. Sightings are summarized by linear transect km, as well as per hour of survey time based on a constant flight speed of 250 km/hr.

| Month | Jan | Feb | Mar | Apr | May | Jun | July | Aug | Sept | Oct | Nov | Dec | Avg |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Number of Sightings/km | 0.00 | 0.06 | 0.07 | 0.00 | 0.61 | 0.25 | 0.77 | 0.31 | 0.38 | 0.18 | 0.00 | 0.10 | 0.23 |
| Number of Sightings/hr | 0.00 | 15.00 | 17.50 | 0.00 | 152.50 | 62.50 | 192.50 | 77.50 | 95.00 | 45.00 | 0.00 | 25.00 | 56.88 |

**Table 15-4. Model selection criterion for bottlenose dolphins in the spring**. The chosen model is highlighted in green. (AIC-Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 92.43 | 43.00 | 0.09 | 0.67 |
| 2 | s(SST) + s(Chl a) | 92.45 | 42.10 | 0.10 | 0.56 |
| 3 | s(SST) + s(DFS) | 92.89 | 37.90 | 0.10 | 0.65 |
| 4 | s(Chl a) + s(DFS) | 90.49 | 42.70 | 0.10 | 0.67 |
| 5 | s(SST) | 103.40 | 16.90 | 0.10 | 0.33 |
| 6 | s(Chl a) | 93.24 | 31.90 | 0.10 | 0.51 |
| 7 | s(DFS) | 92.72 | 33.70 | 0.10 | 0.66 |
| **#** | **GLM Model Covariates** | **AIC** | **% Dev** | **Mean** | **SE** |
| 8 | s(SST) + s(Chl a) + s(DFS) | 100.60 | 20.52 | 0.10 | 2.29 |
| 9 | s(SST) + s(Chl a) | 100.73 | 17.57 | 0.11 | 2.26 |
| 10 | s(SST) + s(DFS) | 99.74 | 18.95 | 0.10 | 2.15 |
| 11 | s(Chl a) + s(DFS) | 98.90 | 20.12 | 0.10 | 0.94 |
| 12 | s(SST) | 107.91 | 4.81 | 0.10 | 2.23 |
| 13 | s(Chl a) | 99.59 | 16.38 | 0.11 | 0.50 |
| 14 | s(DFS) | 98.12 | 18.42 | 0.10 | 0.37 |

**Table 15-5. Model selection criterion for bottlenose dolphins in the summer.** The chosen model is highlighted in green. (AIC-Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 153.67 | 23.90 | 0.20 | 0.32 |
| 2 | s(SST) + s(Chl a) | 152.63 | 24.10 | 0.20 | 0.30 |
| 3 | s(SST) + s(DFS) | 159.96 | 17.20 | 0.20 | 0.25 |
| 4 | s(Chl a) + s(DFS) | 155.81 | 23.20 | 0.20 | 0.31 |
| 5 | s(SST) | 157.24 | 19.70 | 0.20 | 0.29 |
| 6 | s(Chl a) | 158.36 | 17.30 | 0.20 | 0.24 |
| 7 | s(DFS) | 164.66 | 12.50 | 0.20 | 0.25 |
| **#** | **GLM Model Covariates** | **AIC** | **% Dev** | **Mean** | **SE** |
| 8 | s(SST) + s(Chl a) + s(DFS) | 157.75 | 23.16 | 0.20 | 5.75 |
| 9 | s(SST) + s(Chl a) | 156.63 | 22.29 | 0.20 | 0.59 |
| 10 | s(SST) + s(DFS) | 165.98 | 13.18 | 0.20 | 0.60 |
| 11 | s(Chl a) + s(DFS) | 155.82 | 23.09 | 0.20 | 5.62 |
| 12 | s(SST) | 171.80 | 5.56 | 0.20 | 0.52 |
| 13 | s(Chl a) | 157.24 | 19.75 | 0.20 | 5.30 |
| 14 | s(DFS) | 164.66 | 12.52 | 0.20 | 0.29 |

**Table 15-6. Model selection criterion for bottlenose dolphins in the fall.** The chosen model is highlighted in green. (AIC-Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates: | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 126.54 | 23.50 | 0.14 | 0.31 |
| 2 | s(SST) + s(Chl a) | 125.45 | 22.10 | 0.14 | 0.31 |
| 3 | s(SST) + s(DFS) | 125.09 | 22.30 | 0.14 | 0.31 |
| 4 | s(Chl a) + s(DFS) | 127.43 | 19.9 | 0.14 | 0.3 |
| 5 | s(SST) | 138.00 | 10.10 | 0.14 | 0.25 |
| 6 | s(Chl a) | 126.35 | 18.20 | 0.14 | 0.30 |
| 7 | s(DFS) | 125.71 | 19.10 | 0.14 | 0.30 |
| **#** | **GLM Model Covariates** | **AIC** | **% Dev** | **Mean** | **SE** |
| 8 | s(SST) + s(Chl a) + s(DFS) | 128.57 | 18.31 | 0.15 | 4.04 |
| 9 | s(SST) + s(Chl a) | 127.67 | 17.02 | 0.16 | 3.63 |
| 10 | s(SST) + s(DFS) | 127.99 | 16.65 | 0.14 | 5.07 |
| 11 | s(Chl a) + s(DFS) | 128.69 | 15.82 | 0.14 | 1.27 |
| 12 | s(SST) | 139.48 | 0.88 | 0.14 | 4.17 |
| 13 | s(Chl a) | 127.22 | 15.20 | 0.15 | 0.46 |
| 14 | s(DFS) | 127.89 | 14.42 | 0.14 | 0.41 |

**Table 15-7. Model selection criterion for sea turtles in the spring.** The chosen model is highlighted in green. (AIC- Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 1572.07 | 31.40 | 0.72 | 0.08 |
| 2 | s(SST) + s(Chl a) | 1572.56 | 30.90 | 0.73 | 0.08 |
| 3 | s(SST) + s(DFS) | 1570.71 | 31.00 | 0.72 | 0.08 |
| 4 | s(Chl a) + s(DFS) | 1625.67 | 24.30 | 0.72 | 0.07 |
| 5 | s(SST) | 1589.60 | 27.40 | 0.73 | 0.07 |
| 6 | s(Chl a) | 1633.23 | 21.30 | 0.73 | 0.06 |
| 7 | s(DFS) | 1649.61 | 18.40 | 0.72 | 0.06 |
| **#** | **GLM Model Covariates** | **AIC** | **% Dev** | **Mean** | **SE** |
| 8 | s(SST) + s(Chl a) + s(DFS) | 1604.50 | 24.47 | 0.73 | 0.80 |
| 9 | s(SST) + s(Chl a) | 1602.80 | 24.43 | 0.73 | 0.70 |
| 10 | s(SST) + s(DFS) | 1609.10 | 23.50 | 0.74 | 0.70 |
| 11 | s(Chl a) + s(DFS) | 1663.00 | 15.52 | 0.72 | 0.40 |
| 12 | s(SST) | 1627.60 | 20.47 | 0.74 | 0.65 |
| 13 | s(Chl a) | 1667.60 | 14.54 | 0.72 | 0.12 |
| 14 | s(DFS) | 1669.20 | 14.31 | 0.74 | 0.10 |

**Table 15-8. Model selection criterion for sea turtles in the summer.** The chosen model is highlighted in green. (AIC- Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 1725.07 | 34.60 | 0.89 | 0.07 |
| 2 | s(SST) + s(Chl a) | 1723.65 | 34.20 | 0.89 | 0.07 |
| 3 | s(SST) + s(DFS) | 1724.79 | 34.70 | 0.89 | 0.07 |
| 4 | s(Chl a) + s(DFS) | 1780.89 | 27.80 | 0.89 | 0.06 |
| 5 | s(SST) | 1744.02 | 30.90 | 0.89 | 0.06 |
| 6 | s(Chl a) | 1781.20 | 26.80 | 0.89 | 0.07 |
| 7 | s(DFS) | 1813.78 | 21.90 | 0.09 | 0.06 |
| # | GLM Model Covariates | AIC | % Dev | Mean | SE |
| 8 | s(SST) + s(Chl a) + s(DFS) | 1725.80 | 33.49 | 0.90 | 2.20 |
| 9 | s(SST) + s(Chl a) | 1729.30 | 32.72 | 0.89 | 2.19 |
| 10 | s(SST) + s(DFS) | 1725.90 | 33.21 | 0.90 | 2.14 |
| 11 | s(Chl a) + s(DFS) | 1821.50 | 19.88 | 0.89 | 0.31 |
| 12 | s(SST) | 1749.90 | 25.59 | 0.90 | 1.95 |
| 13 | s(Chl a) | 1840.20 | 17.00 | 0.87 | 0.11 |
| 14 | s(DFS) | 1827.00 | 18.84 | 0.91 | 0.13 |

**Table 15-9. Model selection criterion for sea turtles in the fall.** The chosen model is highlighted in green. (AIC- Akaike's Information Criterion, % Dev-Percent of deviance explained from the null model, Mean-average predicted density, SE-Standard error, SST-Sea Surface Temperature, Chl a-Chlorophyll a, DFS- Distance from shore)

| # | GAM Model Covariates | AIC | % Dev | Mean | SE |
|---|---|---|---|---|---|
| 1 | s(SST) + s(Chl a) + s(DFS) | 1494.74 | 8.31 | 0.56 | 0.07 |
| 2 | s(SST) + s(Chl a) | 1501.47 | 5.42 | 0.56 | 0.06 |
| 3 | s(SST) + s(DFS) | 1492.79 | 8.40 | 0.56 | 0.07 |
| 4 | s(Chl a) + s(DFS) | 1493.04 | 8.02 | 0.56 | 0.07 |
| 5 | s(SST) | 1512.83 | 4.74 | 0.56 | 0.06 |
| 6 | s(Chl a) | 1500.29 | 5.29 | 0.56 | 0.06 |
| 7 | s(DFS) | 1491.42 | 7.98 | 0.56 | 0.07 |
| # | GLM Model Covariates | AIC | % Dev | Mean | SE |
| 8 | s(SST) + s(Chl a) + s(DFS) | 1501.20 | 5.80 | 0.56 | 3.11 |
| 9 | s(SST) + s(Chl a) | 1501.50 | 5.42 | 0.56 | 2.67 |
| 10 | s(SST) + s(DFS) | 1509.50 | 4.11 | 0.56 | 2.90 |
| 11 | s(Chl a) + s(DFS) | 1501.80 | 5.36 | 0.56 | 0.34 |
| 12 | s(SST) | 1517.10 | 2.52 | 0.56 | 1.57 |
| 13 | s(Chl a) | 1500.30 | 5.29 | 0.56 | 0.11 |
| 14 | s(DFS) | 1507.90 | 4.04 | 0.56 | 0.13 |

# Chapter 16: Modeling species assignment in strip transect surveys with uncertain species identification

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** *www.briloon.org/mabs*

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# Chapter 16 Highlights

*Modeling loon species assignment in digital video aerial surveys with uncertain species identifications*

### *Context*[1]

High resolution digital video aerial surveys provide numerous advantages to monitoring marine bird abundance and distribution across large spatial areas. In some situations, however, low species identification rates in aerial surveys pose a major challenge when species-specific metrics are of management and conservation interest. While several chapters in Part IV of this report focus on estimating abundance and habitat relationships, this chapter focuses on integrating information from boat-based surveys, with high species identification rates, into a species assignment model to predict species identity in high resolution digital video aerial data.

Common and Red-throated Loons were chosen as the focal species for Chapter 16, due in part to differences in the conservation status of the two species, as well as evidence from Europe of the sensitivity of Red-throated Loons to offshore wind energy development. Loons had low species identification rates in aerial surveys, but high species identification rates in boat surveys. Chapter 19, which focused on combining data from the two survey types, models loons as a genus rather than each species independently, because of the low species identification rate in the aerial data. However, this chapter's results were used to develop species-specific maps of persistent hotspots (see Chapter 17), and it could be used to inform other joint modeling approaches.

### *Study goal/objectives*

Investigate three approaches to modeling species assignment in digital video aerial surveys where species identity was not always completely observed (i.e., some individuals were identified to genus, but not species).

### *Highlights*

- Species identity predictions were qualitatively similar across all three prediction approaches.
- The majority of unidentified loons were assigned as Common Loons, but species assignment varied by survey and with habitat covariates.
- Distance to shore was the strongest habitat predictor of species identity. The probability of assigning an unidentified loon as a Red-throated Loon noticeably decreased with distance from shore.

### *Implications*

Species identification models can assist in prioritizing areas important to specific species of conservation concern, even if current technology cannot identify all observations to the species level. Species-specific conclusions may be of particular interest for Red-throated Loons, which are designated as a species of conservation concern by the U.S. Fish and Wildlife Service.

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[1] For more detailed context for this chapter, please see the introduction to Part III of this report.

## Abstract

Passive survey techniques (e.g., digital aerial surveys) are increasingly used by ecological monitoring programs. High resolution digital video aerial surveys have tremendous potential to aid in monitoring marine bird abundance and distribution across large spatial areas. Low species identification rates in video aerial surveys, however, pose a major challenge when species-specific metrics are of management and conservation interest. For instance, Red-throated Loons are identified by the US Fish and Wildlife Service as the highest priority open-water species for conservation in the mid-Atlantic US, while global populations of Common Loons are generally considered healthy (see Chapters 2 and 21). Species-specific identification rates of loons in high resolution digital video aerial surveys, however, were low in this study, prohibiting estimates of loon species-specific abundance via this survey method (see Chapter 5). In this chapter, we combine data collected during boat and high resolution digital video aerial surveys to evaluate loon species proportions, important spatial covariate predictors of species proportions, and predicted species assignment of unidentified loons. Boat surveys identified ≥95% of individual loons to species (Common Loon or Red-throated Loon) during each survey, and high resolution digital video aerial surveys identified 3% - 63% of individual loons to species during each survey (see Chapters 5 and 8). For each unidentified loon observation in the aerial data, species identity was predicted using three approaches: 1. species proportions from boat data; 2. species proportions from aerial data; and 3. species proportions as a function of spatially varying covariates. Predicted species assignments of unidentified loons in aerial surveys were qualitatively similar across all three prediction approaches, and generally indicated the majority of unidentified loons were Common Loons. However, results provided strong evidence that spatially varying covariates were important predictors of loon species proportions. Distance to shore was correlated with increased Common Loon proportions across all surveys. Sea surface temperature, grain size, and salinity were also important predictors of species proportions, but the nature of these relationships varied by season. Overall, results suggest that joint modeling of boat and aerial data may provide a useful approach to estimate species-specific metrics such as abundance (see Chapters 5, 8, and 12) and the locations of persistent hotspots (see Chapter 17) when species identification rates are low.

## Introduction

Passive monitoring techniques (e.g., digital aerial surveys) are rapidly being incorporated in to numerous ecological monitoring and research programs (Buckland et al. 2012, Conn et al. 2014, Johnston et al. 2015). Surveys of marine birds using high resolution digital video aerial surveys (hereafter “digital video aerial surveys”) are increasingly used to monitor species-specific abundance and distribution (Buckland et al. 2012, Johnston et al. 2015). Digital video aerial surveys provide several important advantages compared to boat-based surveys, including increased spatial coverage, reduced survey time, increased safety and reduced disturbance (Buckland et al. 2012). Digital video aerial surveys also have notable weaknesses, however, such as low species identification rates relative to boat surveys (Johnston et al. 2015).

Species-specific marine bird abundances are increasingly important due to interest in renewable energy development in nearshore and offshore waters of the United States (Winiarski et al. 2014). Common

Loons (*Gavia immer*) and Red-throated Loons (*Gavia stellata*) are two species of particular interest due to their known or suspected sensitivity to displacement from offshore wind energy development (Halley and Hopshaug 2007, Petersen et al. 2006, Furness et al. 2013, Langston 2013). Currently, Red-throated Loons are known to be sensitive to displacement from areas around offshore wind energy developments, but there is limited species-specific information on Common Loons (Petersen et al. 2006, Furness et al. 2013). Additionally, conservation status varies between Common Loons and Red-throated Loons. The U.S. Fish and Wildlife Service designated Red-throated Loons as a "species of conservation concern" on their wintering grounds along the New England and Mid-Atlantic Coast (USFWS 2008). Common Loon populations, however, are seen as stable, and the species is no longer on the national list of Birds of Conservation Concern (Chapters 2 and 21; Evers 2004, USFWS 2008, Evers 2010). Common Loons and Red-throated Loons wintering in the mid-Atlantic United States have wide overlap in body size (Barr et al. 2000, Gray et al. 2014), which may reduce species identification rates in digital video aerial surveys. Aerial survey observations of loons identified to genus (*Gavia* spp.) but not to species provide valuable information on loon abundance and distribution, but pose a major challenge when estimating species-level hotspots (see Chapter 17) or abundance (Johnston et al. 2015).

In this study, we investigate three approaches to model species identity assignment in surveys with uncertain species identification. Our case study focuses on Common Loons and Red-throated Loons during the winter and spring of 2012 and 2013. Loons provided a unique opportunity to investigate uncertain species identification in digital video aerial surveys due to low species identification rates in aerial surveys (see Chapter 5), but high species identification rates in corresponding boat surveys (see Chapter 8). Boat and digital video aerial surveys were conducted during each season and provided information on species- and genus-level observations within the surveyed area. We hypothesized that boat surveys would provide important information to predict species identification in aerial surveys, which often suffered from low species identification rates. Investigation of methods that use both datasets are important first steps in developing broader approaches to estimate species-specific abundance from surveys with uncertain species identification.

## Methods

Boat and aerial surveys were conducted off the coast of Delaware, Maryland, and Virginia (Figure 16-1) from March 2012 to May 2014. During each boat survey, observers recorded data on species identification, number of individuals observed, and locations of observations (see Chapter 7 for details). Digital video aerial surveys recorded similar metrics, but were completed using four high resolution digital video cameras, each surveying a 50 m strip width (total strip width = 200 m). Video data were manually reviewed to record species identification, number of individuals observed, and locations (see Chapter 3 and Chapter 4 for details). For analysis, each transect was divided into 4-km segments and the number of observed loons in a segment was summed by species identification (Common Loon, Red-throated Loon, or unidentified loon). Boat and aerial data collected during similar times of year were compared to reduce possible differences across seasons. Specifically, we paired boat and aerial surveys that occurred during April-May 2012, December 2012, March 2013, and December 2013 (Table 16-1). April-May 2012 surveys included two months, but boat and aerial surveys were separated by <2 weeks (boat and aerial surveys occurred on 25-29 April and 6-7 May, respectively).

Analysis was conducted in two principal stages: estimation of observed species proportions, and prediction of species identity. Spatially referenced boat and aerial data allowed investigation of three approaches to predict species identity, including: 1. species proportions from boat data; 2. species proportions from aerial data; and 3. species proportions as a function of spatially varying covariates.

### *Species proportions*

During boat surveys, ≥95% of all observed loons were identified to species and the remaining ≤5% were excluded from analysis (following the protocol of Johnston et al. 2015). Digital video aerial surveys, however, had lower species identification rates generally and were thus comprised of a large number of unidentified loons, which were retained for analysis (Table 16-1, Figure 16-2).

Survey and segment specific proportions of Common Loons and Red-throated Loons were modeled using generalized linear models with a binomial distribution. For Models 1 and 2, which estimated the species proportions based on boat and aerial data, the counts of Common Loons at segment $i$, $\mathrm{y_i}$, was defined such that:

$$\mathrm{y_i} \sim \mathrm{Binomial}(p, N_i)$$

where $N_i$ is the total number of identified Loons (Common Loons and Red-throated Loons) in segment $i$, and $p$ is the probability that the observation in segment $i$ was a Common Loon. Analyses were conducted separately for each survey date and type (boat and aerial). For each analysis, the estimates of species proportions were assumed to be constant within a survey.

Next, we fit a series of models that allowed species proportions to vary as a function of spatial covariates. We used six covariates in our analyses: three static (distance to shore [DTS], slope, and grain size), and three dynamic (sea surface temperature [SST], salinity [Sal], and chlorophyll-a [Chlor]). The full model for spatially varying species proportions included all six covariates:

$$\mathrm{y_i} \sim \mathrm{Binomial}(p_i, N_i)$$

$$\mathrm{logit}(p_i) = \beta_0 + \beta_1 \mathrm{DTS_i} + \beta_2 \mathrm{SST_i} + \beta_3 \mathrm{Chlor_i} + \beta_4 \mathrm{Sal_i} + \beta_5 \mathrm{Slope_i} + \beta_6 \mathrm{Grain_i}$$

For the static covariates, we calculated distance to shore (m) within ArcGIS (ESRI, Redlands, CA) and extracted slope (% rise, 370-m resolution) and grain size (ϕ = -log2[mean grain diameter in mm], 370-m resolution) from the data layer derived by NOAA/NOS National Centers for Coastal Ocean Science (Kinlan et al. 2013). For the dynamic covariates, we used Marine Geospatial Ecology Tools in ArcGIS (Roberts et al. 2010) to download remotely-sensed data at the highest resolution available for all segments. We compiled daily values for sea surface temperature (°C, 1-km GHRSST L4) and salinity (Practical Salinity Units, 9-km HYCOM GLBa0.08 Equatorial 4D). Due to missing data along the shoreline at higher resolutions, we used monthly composites of chlorophyll concentration (mg/cubic m, 4-km NASA Ocean Color L3 SMI Aqua). Observations without complete covariate information were excluded from analysis. For each of the four paired surveys, we compared all possible subsets of the full model using $\mathrm{AIC_c}$ (Burnham and Anderson 2002). Results are presented as model-averaged parameter estimates as each survey included multiple competing models (Burnham and Anderson 2002).

### *Predicting species identity*

For digital video aerial surveys, where it was not possible to always identify observations to species, unidentified loons were assigned a species (Common Loon or Red-throated Loon) based on the previously estimated species proportions. This resulted in three estimates of predicted species assignment based on: 1. species proportions from boat data (approach 1); 2. species proportions from aerial data (approach 2); and 3. species proportions using spatial covariates (approach 3). Spatial covariate predictions were derived from model-averaged parameter estimates and predicted to covariate values located at the midpoint of each aerial transect segment. The observational nature of this study did not allow for validation of prediction approaches. Results for all three prediction approaches are therefore presented and compared. All analyses were conducted in R version 3.0 (R Development Core Team 2013).

## Results

### *Species proportions*

The total number of observed loons varied by survey, ranging from 320 individuals to >1,300 individuals (Table 16-1, Figure 16-2). Total loon counts were higher in winter surveys (December) compared to spring surveys (March and April; Figure 16-2). Species identification rates were always ≥95% in boat surveys, but varied from 3% – 63% in aerial surveys (Table 16-1, Figure 16-2). Species proportions also varied by survey (Figure 16-3). For instance, Common Loons comprised >90% of identified loon species in both boat and aerial surveys during December 2013 (Figure 16-3). In March 2013, however, Common Loon proportions were much lower in boat surveys (~40%) as compared to aerial surveys (>70%; Figure 16-3).

Species proportions also varied across the surveyed area (Figure 16-4). Model-averaged parameter estimates provided strong evidence that species proportions of Common Loons and Red-throated Loons varied as a function of spatial covariates (Table 16-2). Distance to shore was the strongest spatial predictor of species proportions, and indicated proportions of Common Loons increased as distance to shore increased (Table 16-2, Figure 16-4). Sea surface temperature was also a significant predictor of species proportions, but results were season-specific (Table 16-2). In spring surveys, proportions of Common Loons increased as sea surface temperature increased, but during winter surveys Common Loon proportions decreased as sea surface temperature increased (Table 16-2). Grain size and salinity were also significantly correlated with species proportions during at least two surveys (Table 16-2). Similar to sea surface temperature, however, relationships were opposite during spring (negative) and winter (positive) surveys (Table 16-2). Effects of chlorophyll a and slope varied across surveys, but model-averaged parameter estimates for these covariates often overlapped zero (Table 16-2).

### *Predicting species identity*

Overall predictions of species identity by survey were qualitatively similar across all three prediction approaches (Figure 16-5). All three prediction approaches (aerial proportions, boat proportions, and spatial covariate relationships) assigned the majority of unidentified loons as Common Loons (Table 16-3, Figure 16-5). March 2013 was the lone exception, where boat proportions and spatial covariate approaches predicted more Red-throated Loons than Common Loons, while aerial proportions predicted more Common Loons (Table 16-3, Figure 16-5). Differences in predicted species identity during March 2013 surveys reflected differences in observed proportions during boat and aerial surveys (Figure 16-3) and low numbers of identified loons during aerial surveys (*n* = 14 loons identified to species during aerial surveys; Table 16-1).

Predicted numbers of individual Common Loons and Red-throated Loons within each transect were generally much greater than the observed numbers, due to assignment of unidentified loons to a species (Table 16-3). Increased species-specific numbers were most dramatic in December 2012, March 2013, and December 2013, due to low rates of species identification and high numbers of observed loons in aerial surveys (Table 16-3). For instance, 84 Common Loons were identified in December 2012 (Table 16-3), while 556 were classified as unidentified loons during that same survey. Species predictions for those unidentified loons resulted in predictions ranging from 402 Common Loons (95% CI 357-445) using aerial proportion predictions to 519 (95% CI 461-562). Common Loons using spatial covariate predictions (Table 16-3). Conversely, in April 2012, there were 45 individuals identified to Red-throated Loon, and the mean predicted number of Red-throated Loons was 46 – 71. Thus the predicted value was similar to the observed, i.e., only 1 – 26 individuals more than the number actually identified during aerial surveys (Table 16-3).

## Discussion

Uncertain species identification, currently common in digital aerial surveys, presents a major challenge when estimating species-specific abundance (Conn et al. 2013, Johnston et al. 2015). In some situations, technological solutions to improve species identification (such as higher camera resolution or other equipment modifications) may be available to improve species identification rates. In other these situations, model-based approaches that account for species uncertainty may be more efficient and feasible approaches to estimate species-specific abundances (Johnston et al. 2015).

The approaches we used to assign species identity included relative species proportions from boat and aerial surveys and spatial covariates. These approaches were rather simplistic given the numerous sources of uncertainty that may arise from these types of surveys. For instance, false identification rates were assumed to be zero. False identification rates in either survey, however, would inflate identification rates and bias results to an unknown degree. Further, detection and availability probabilities were assumed to be similar across species. High altitude digital aerial surveys may reduce disturbances that may alter species behavior, but species-specific detection remains confounded with actual species proportions. For boat surveys, distance sampling is often used to account for imperfect detection probability (Buckland et al. 2001). Recently, integrated modeling of line and strip transect surveys have begun to address uncertain species identification (Conn et al. 2013; Johnston et al. 2015). A particular advantage of these recent approaches is their ability to propagate uncertainty at each level, resulting in final estimates of density that account for each source of uncertainty (Johnston et al. 2015). Joint modeling approaches that directly account for uncertain species identification improve our ability to utilize multiple sources of information, and ultimately identify important drivers of species abundance and distribution.

Incorporating spatial covariates allowed species assignments to be informed by underlying habitat characteristics. In this study, distance to shore, sea surface temperature, grain size, and salinity were strong predictors of loon species proportions. These relationships suggested that assignment of unidentified loons may be improved if observation locations (and associated covariate values) are known. Relationships between species proportions and spatial covariates often changed in strength and direction between seasons. The lack of consistent relationships implies that joint models should be cautious about integrating data collected during different temporal periods. Most boat and aerial surveys in this study were separated by <2 weeks, which based on temporal variability of environmental covariates used in our models was likely adequate to meet these assumptions . Coordinated efforts to overlap surveys in both space and time may provide several benefits, including reduced differences in dynamic covariates between surveys (e.g., sea surface temperature) and direct comparisons of survey-specific results.

Uncertain species identification in digital video aerial surveys was evident across several species in this study (e.g., gulls, terns, scoters; see Chapter 5). Loon species identification rates in digital video aerial surveys, however, were some of the lowest for any avian taxon (see Chapter 14). As a result, loons provided a unique opportunity to investigate uncertain species identification in digital video aerial surveys due to the low species identification rates in digital video aerial surveys, but high species identification rates in corresponding boat surveys (Figure 16-2).

Ecological studies are increasingly using passive monitoring techniques, which provide numerous advantages, but often suffer from low species identification rates (Conn et al. 2013; Johnston et al. 2015). Joint models that integrate multiple sources of data may provide useful model-based methods to estimate species-specific abundance and distribution. Future studies should consider expanding these methods to address >2 species (e.g., gulls and terns), uncertain species identification in both boat and aerial surveys (e.g., scoters), and methods to validate model results.

## Ongoing applications

We have integrated approaches developed in this chapter with lease block hotspot mapping (see Chapter 17), allowing us to identify species-specific hotspots even when aerial surveys cannot perfectly assign species. Species assignment approaches were investigated for other species (e.g., scoters), but will require more complex approaches due to imperfect species identification in both video aerial and boat surveys. Exploration of methods to incorporate uncertain species identification in integrated boat and aerial models (Chapter 19) is a possible topic for future research, especially as digital video aerial surveys increase in application.

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## Figures and tables

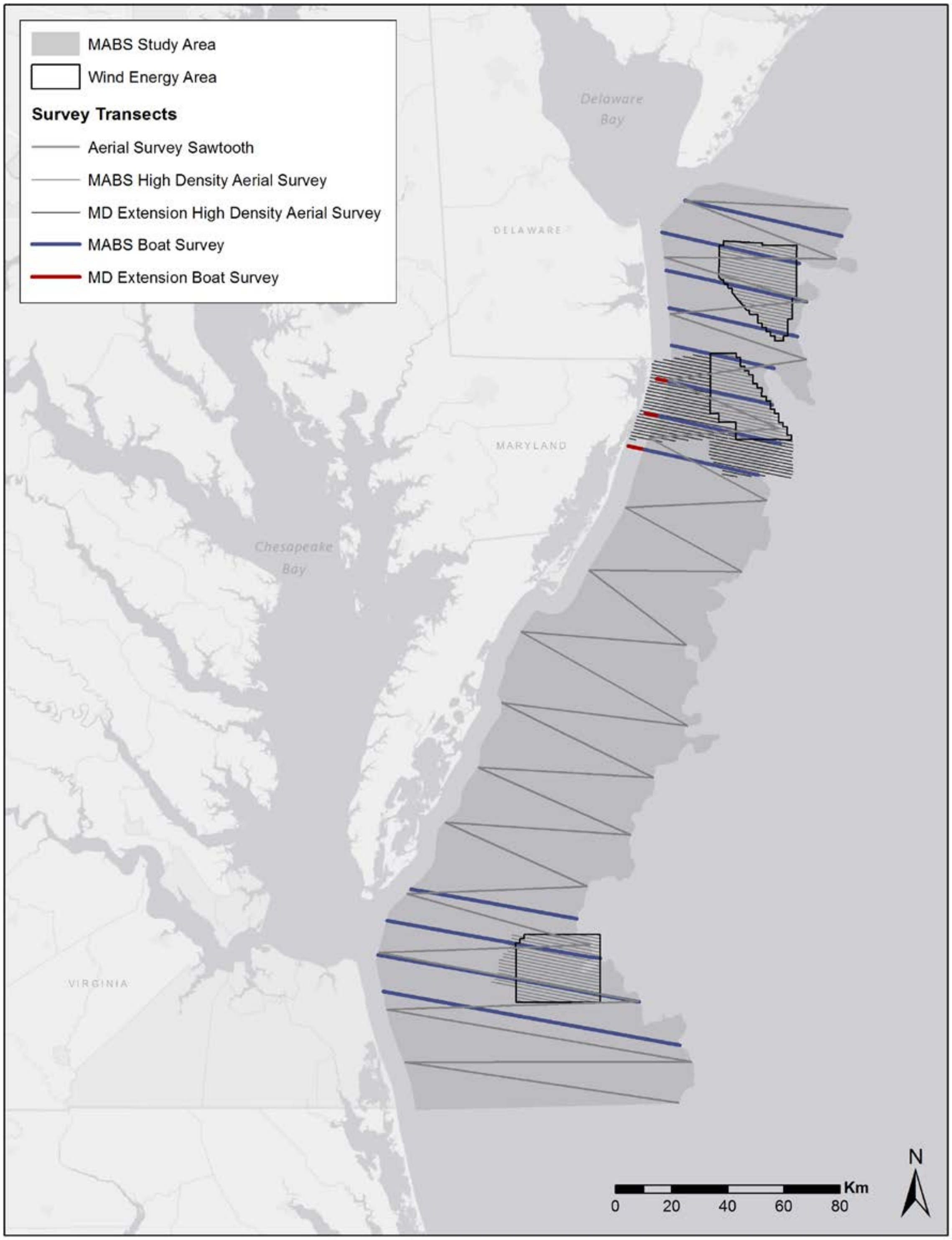

**Figure 16-1. Study area.** Boat transects are shown in blue and red and aerial transects in light and dark grey; Maryland extension transects (funded by the state of Maryland and conducted only in the second year of surveys) are shown in red and dark grey. Department of Energy (DOE)-funded high-density aerial surveys were located within federally designated wind energy areas (WEAs).

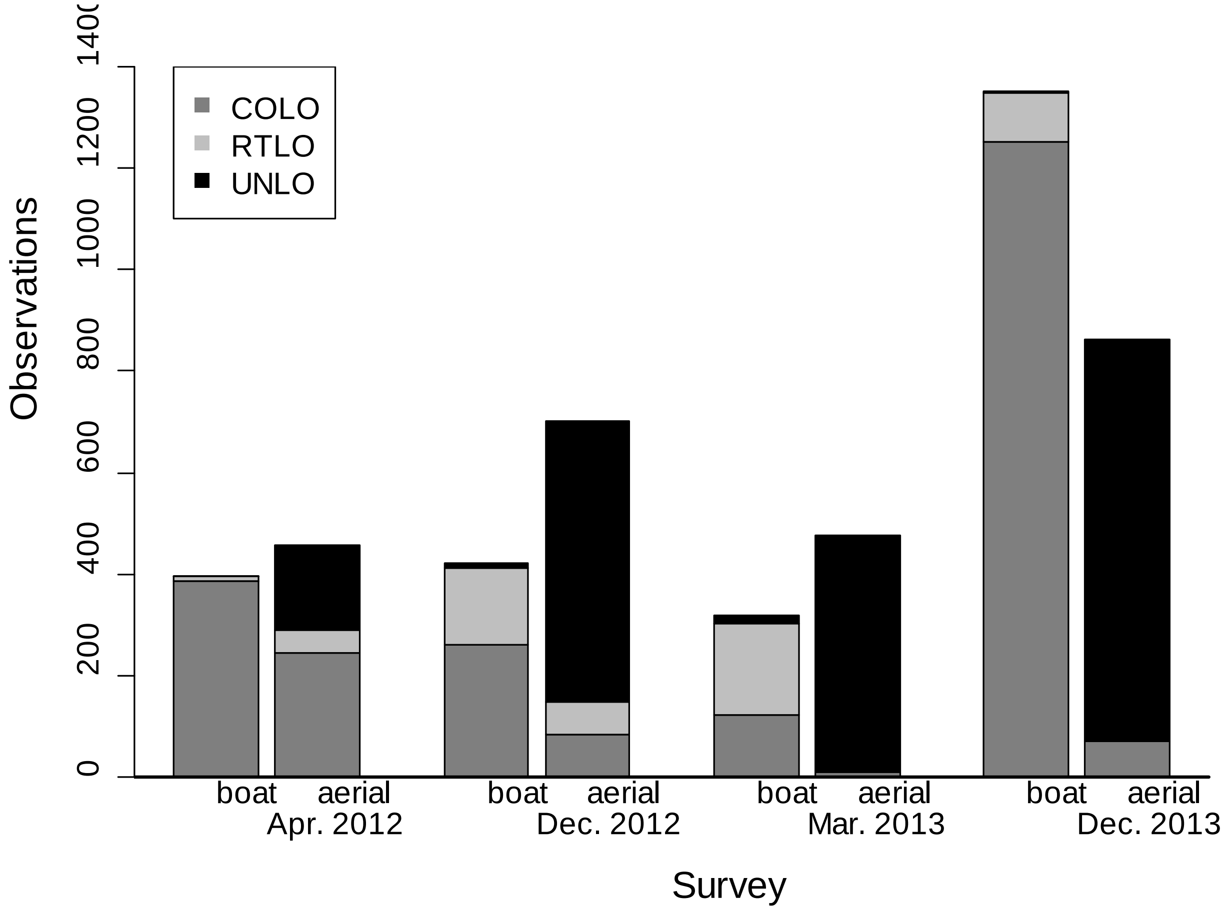

**Figure 16-2. Numbers of individual Common Loons (COLO), Red-throated Loons (RTLO), and unidentified loons (UNLO) observed during corresponding boat and aerial surveys.**

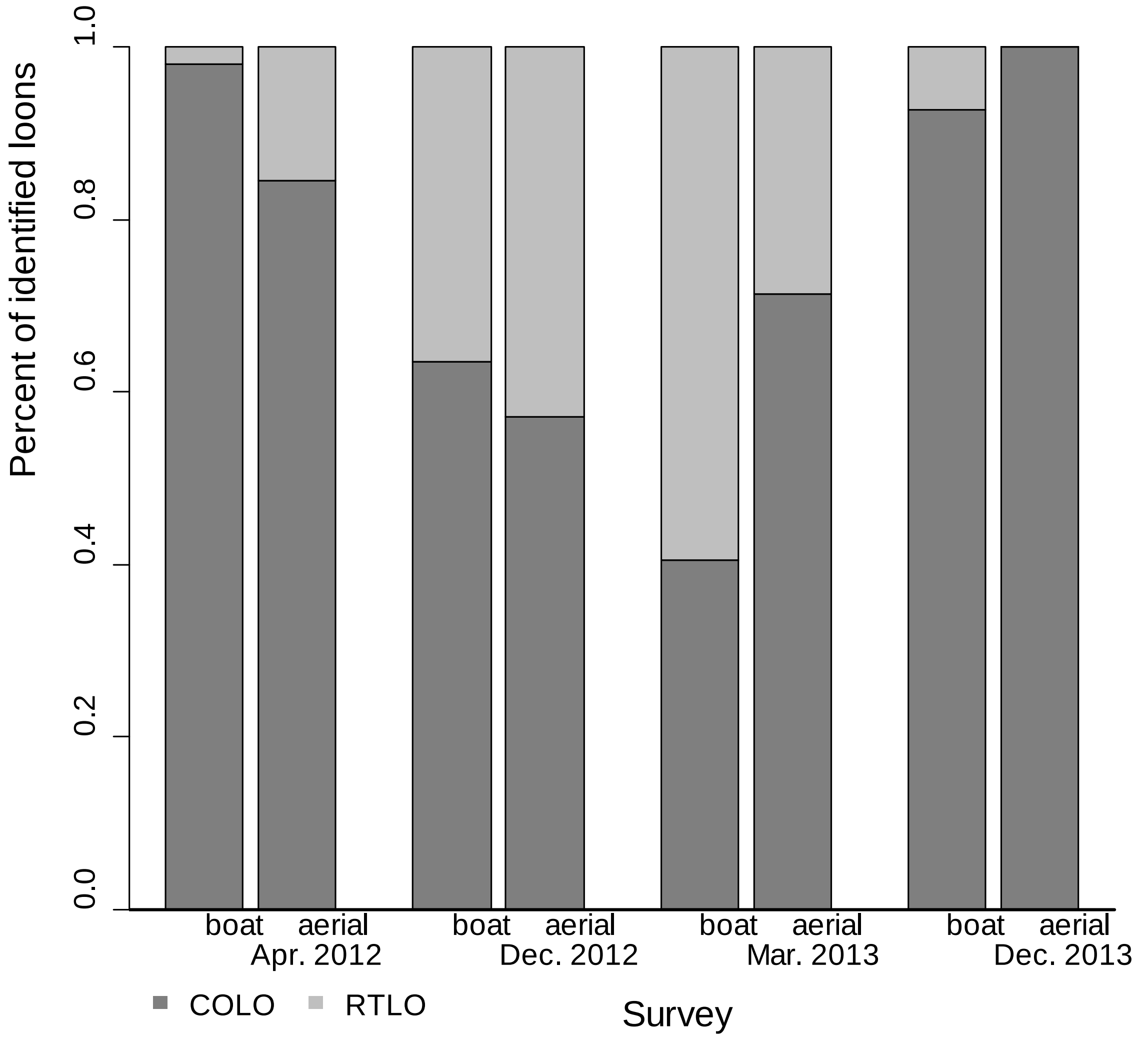

**Figure 16-3. Percent of loons identified to species that were Common Loons (COLO) or Red-throated Loons (RTLO) during corresponding boat and aerial surveys.**

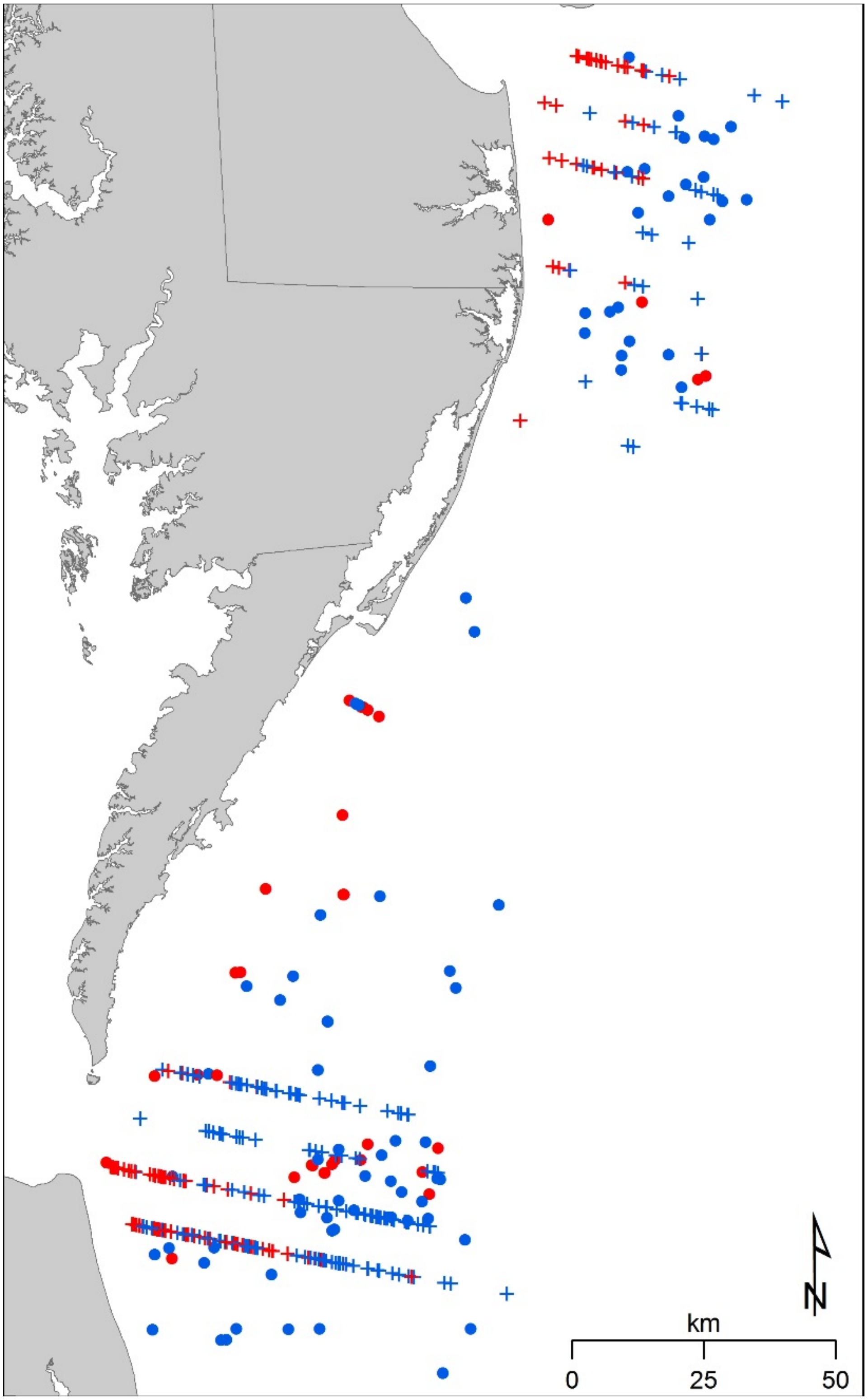

**Figure 16-4. Locations of identified Common Loons (blue) and Red-throated Loons (red) during boat surveys (crosses) and aerial surveys (dots) in December 2012.**

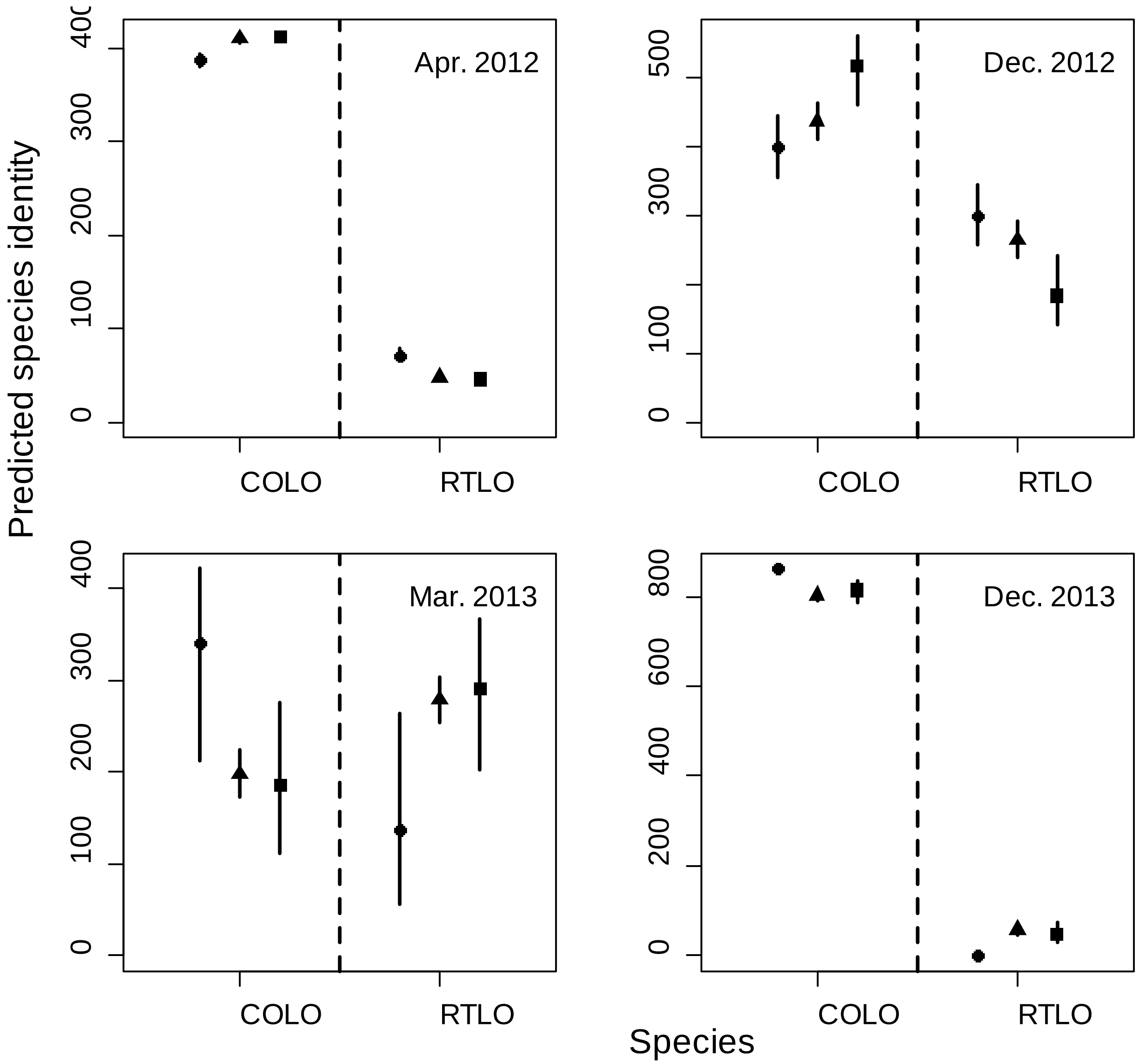

**Figure 16-5. Predicted numbers of Common Loons (COLO) and Red-throated Loons (RTLO) in digital video aerial surveys.** Unidentified loons (UNLO) observed during aerial surveys were assigned to a species using one of three methods: (i) the proportions of identified loons in the aerial data (circles), (ii) the proportions of identified loons in the boat data (triangles), or (iii) model averaged spatial covariate relationships (squares). Error bars denote 95% confidence intervals. Note survey-specific y-axis.

**Table 16-1. Numbers of individual Common Loons (COLO), Red-throated Loons (RTLO), and unidentified loons (UNLO) observed during corresponding boat and aerial surveys.**

| Survey | Boat | | | Aerial | | |
|---|---|---|---|---|---|---|
| | COLO | RTLO | UNLO | COLO | RTLO | UNLO |
| April/May 2012 | 387 | 8 | 0 | 245 | 45 | 168 |
| December 2012 | 262 | 150 | 9 | 84 | 63 | 556 |
| March 2013 | 123 | 180 | 17 | 10 | 4 | 463 |
| December 2013 | 1251 | 98 | 3 | 70 | 0 | 792 |

**Table 16-2. Model-averaged logit-scale parameter estimates (95% CI) predicting loon species proportions across 4 boat surveys.** Model-averaged parameter estimates that did not overlap zero are bolded.

| Parameter | Survey | | | |
|---|---|---|---|---|
| | April/May 2012 | December 2012 | March 2013 | December 2013 |
| Int. | **7.13 (4.32, 9.93)** | **1.15 (0.74, 1.57)** | **-0.37 (-0.67, -0.07)** | **6.77 (5.49, 8.06)** |
| DTS | **4.84 (1.81, 7.87)** | **1.48 (0.25, 2.71)** | 0.26 (-0.32, 0.85) | **5.47 (3.98, 6.95)** |
| SST | **1.44 (0.46, 2.42)** | -0.49 (-1.00, 0.02) | **1.83 (1.10, 2.56)** | **-2.96 (-3.91, -2.01)** |
| Grain size | -0.14 (-1.05, 0.77) | **0.85 (0.42, 1.27)** | **-0.39 (-0.76, -0.03)** | **0.85 (0.36, 1.35)** |
| Salinity | -0.37 (-1.94, 1.20) | 0.57 (-0.23, 1.36) | **-0.77 (-1.45, -0.08)** | **1.32 (0.31, 2.32)** |
| Chlor | 0.47 (-1.14, 2.07) | **-1.70 (-2.66, -0.73)** | -0.45 (-0.93, 0.03) | -1.06 (-2.57, 0.45) |
| Slope | -0.66 (-1.44, 0.13) | -0.34 (-0.75, 0.06) | -0.15 (-0.49, 0.19) | 0.50 (-0.25, 1.25) |

**Table 16-3. Predicted numbers (95% CI) of Common Loons (COLO) and Red-throated Loons (RTLO) in aerial survey transects.** Unidentified loons (UNLO) observed during aerial surveys were assigned to a species using one of three methods: (i) proportions of identified loons in the aerial data (aerial), (ii) proportions of identified loons in the boat data (boat), or (iii) model-averaged spatial covariate relationships (spatial).

| Survey | Method | COLO | RTLO | UNLO |
|---|---|---|---|---|
| April/May 2012 | Observed | 245 | 45 | 168 |
| | Aerial | 387 (379, 393) | 71 (65, 79) | |
| | Boat | 410 (406, 411) | 48 (47, 52) | |
| | Spatial | 412 (408, 413) | 46 (45, 50) | |
| December 2012 | Observed | 84 | 63 | 556 |
| | Aerial | 402 (357, 445) | 301 (258, 346) | |
| | Boat | 438 (411, 463) | 265 (240, 292) | |
| | Spatial | 519 (461, 562) | 184 (141, 242) | |
| March 2013 | Observed | 10 | 4 | 463 |
| | Aerial | 341 (213, 421) | 136 (56, 264) | |
| | Boat | 198 (173, 224) | 279 (253, 304) | |
| | Spatial | 186 (111, 275) | 291 (202, 366) | |
| December 2013 | Observed | 70 | 0 | 792 |
| | Aerial[a] | 862 (NA) | 0 (NA) | |
| | Boat | 804 (792, 815) | 58 (47, 70) | |
| | Spatial | 815 (787, 834) | 47 (28, 75) | |

[a] Confidence interval could not be estimated due to zero observations of Red-throated Loons in the aerial data

# Chapter 17: Integrating data across survey methods to identify spatial and temporal patterns in wildlife distributions

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 17 Highlights

*Identifying spatial and temporal patterns of species abundance and species richness by combining data from boat-based and digital video aerial surveys*

### *Context*[1]

Part IV of this report focuses on the comparison and integration of data from boat surveys and digital video aerial surveys to examine wildlife distributions in the mid-Atlantic. This chapter uses both datasets to identify temporal and spatial patterns of species presence and relative abundance in the study area, including the identification of "persistent hotspots," or geographic areas with consistently high numbers of animals or species through time, which may indicate important habitat use areas. Predicted species identification of Red-throated Loons (*Gavia stellata*) and Common Loons (*G. immer*) presented in Chapter 16 were used in calculating persistent hotspots for these species. Temporal patterns of observations of different species and groups within the study area are also presented in this chapter, and can be used to determine potential exposure to offshore development activities at different times of year.

While this chapter examines patterns in areas that were directly surveyed, several other chapters in Part IV incorporate environmental covariates into modeling efforts, in order to identify environment drivers of these distributions and predict relative densities of wildlife across the study area (Chapters 15-16 and 18-19). In some instances, one survey method was used to predict abundance of specific taxa (e.g., Chapter 15), while in other cases, the two datasets could be combined using an integrated modeling framework (Chapter 19). Additional chapters in Part IV compare boat and aerial survey methodologies (Chapters 13-14), highlighting the strengths and weaknesses of the two methods, and provide context for results presented in this chapter.

### *Study goal/objectives*

Identify persistent hotspots of relative abundance and species richness, as well as temporal patterns of species abundance within both boat survey and digital video aerial survey datasets.

### *Highlights*

- To identify persistent hotspots, boat and digital video aerial survey data were combined for locations with sufficient sample sizes where both datasets were available.
- For most taxa, hotspots were most consistently observed in areas within approximately 30-40 km from shore, particularly offshore of the mouths of Chesapeake Bay and Delaware Bay, and in northern Maryland.
- The presence and relative abundance of different species varied widely by time of year.

### *Implications*

Combining data from two different survey approaches can provide a better view of wildlife populations and distribution patterns than either survey method could provide alone. These results may be helpful for informing the siting and permitting processes for future development projects, and for informing mitigation efforts and construction and operations plans.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

Data on the abundance of marine birds, mammals, and turtles were collected over a two-year period (2012-2014) as part of a baseline study to inform siting and permitting processes for offshore wind energy development. We employed two methods: (1) traditional boat-based surveys, and (2) high resolution digital video aerial surveys. We combined data from both survey methods to examine spatial and temporal patterns of wildlife abundance by calculating persistent hotspots of abundance across all surveys. "Hotspots," or areas with atypically high effort-corrected counts of a taxon from a given survey, were summed across all surveys to calculate relative persistence. Boat and aerial survey data were combined for locations with sufficient sample sizes where both datasets were available. We also used boat and aerial survey data to summarize temporal patterns in species observations throughout the annual cycle and compare results between survey methods.

For most taxa, hotspots were most consistently observed in areas within approximately 30-40 km of shore, particularly offshore of the mouths of Chesapeake Bay and Delaware Bay and in northern Maryland. Exceptions to this general rule included sea turtles (Testudines spp.), Common Dolphins (*Delphinus delphis*), Common Loons (*Gavia immer*), and alcids (Alcidae spp.), all of which tended to have persistent hotspots located farther offshore. The presence and relative abundance of species varied widely by time of year, however, with different species and groups using the study area during non-breeding (summer or winter), breeding, and migratory periods.

## Introduction

The mid-Atlantic region is an extremely important area for a broad range of marine wildlife species throughout the year. This is due to relatively high levels of productivity, fed in part by nutrient inputs from Chesapeake Bay and Delaware Bay, as well as the region's central location on the eastern edge of the continent and in the middle of an important migratory flyway (Chapter 1; Smith and Kemp 1995; Schofield et al. 2008). During this study, our main goal was to gather the baseline information on abundance and movements of marine birds, mammals, and turtles required to inform siting and permitting processes for offshore wind energy development in the mid-Atlantic. We collected this data over a two-year time period (2012-2014) using a variety of technologies and methods to examine spatial patterns and trends, while simultaneously testing a new technology for the first time in the United States (high resolution digital video aerial surveys; hereafter video aerial surveys). Video aerial surveys are a relatively new method for collecting distribution and abundance data on animals in the marine environment (Thaxter and Burton, 2009). Although they have become a common method of collecting baseline data on marine bird mammal distribution and abundance for offshore wind energy planning and monitoring in Europe, the U.S. still relies almost exclusively on boat-based and standard (visual observer) aerial survey methods. This Mid-Atlantic Baseline Studies Project is the largest application of video aerial surveys in the U.S. to date. We also conducted boat-based surveys within the study area to accompany the video aerial surveys; standardized boat-based surveys are a well-established and widely used method of obtaining density data for birds, sea turtles, and marine mammals (Gjerdrum et al., 2012; Tasker et al., 1984).

Boat and video aerial surveys produced markedly different results for some taxa (Chapters 13-14), which can present a challenge when interpreting and/or integrating data obtained from the two survey methodologies. These differences also present an opportunity, however, in that the two surveys can provide complementary data that, in tandem, may be used to provide a better overall view of wildlife distributions and relative abundance in the mid-Atlantic study area. The challenge was to integrate these data in a meaningful way that adds to our understanding of wildlife distributions. Two such integrative efforts are discussed below: the identification of geographic hotspots of persistent abundance, as well as the identification of temporal patterns of persistence and relative abundance of species within the study area.

Both abiotic components (e.g., climatic conditions) and biotic components (e.g., prey and predator distributions) of marine ecosystems can be highly variable in space and time (Kappes et al., 2010; Lehodey et al., 2006; Murphy et al., 1998). Thus, identifying key habitat use areas or locations of high wildlife abundance in the marine environment can be difficult, as such locations may be ephemeral. Several previous studies have attempted to identify hotspots of wildlife abundance in the marine ecosystem; while each study defined hotspots slightly differently, all of these definitions contain an element of temporal as well as spatial persistence. For example, Piatt et al. (2006) define a hotspot for the Short-tailed Albatross (*Phoebastria albatrus*) as a ‘relatively small area in which we expect to find animal aggregations repeatedly’, while Davoren (2007) defines hotspots of seabirds as ‘areas where high abundance of species overlap in space and time’. Suryan et al. (2012) define hotspots of marine predators as ‘regions of consistently high abundance of predators relative to the surrounding area in the open ocean’. Other studies have used varying mathematical definitions to identify hotspots. Zipkin et al. (2015) identifies hotspots as locations with 3x the mean abundance for the study region. Santora and Veit (2013) define hotspots as ‘locations with anomalies that exceed the mean for the entire study region by 1 standard deviation in a given survey’. Likewise, in this study, we apply a quantitative definition to identify hotspots in a consistent, repeatable way across species and surveys. While most similar to the approach taken by Santora and Veit (2013), our exact definition of hotspots also varies slightly from those above, in order to account for highly non-normal distributions of animal counts across our study area, and with a goal of identifying those hotspots that are most consistent throughout the two years of surveys.

Persistent hotspots thus highlight locations where individuals within a species or species group have been consistently observed in greater than average numbers over time, and may indicate the locations of important habitat (Gende and Sigler, 2006; Santora et al., 2010; Sydeman et al., 2006). In this study, we examined spatial patterns of persistent abundance for a wide range of taxa, including seabirds, marine mammals, sea turtles, rays, and bait fish, which were present within the study area for varying amounts of time or in variable numbers depending on each group’s life history traits. Temporal bar charts summarize the temporal patterns of species and species groups within the study area, and allow for comparison of effort-corrected count data for species and species groups through time and between survey methods. Identification of persistent hotspots, paired with temporal bar charts for taxa of interest, can be used in: (1) marine spatial planning efforts; (2) understanding when and where animals may be affected by anthropogenic activities; and (3) identifying species or taxa in particular need of additional study. These data can be used during permitting processes for future development, as well as for siting projects and designing development plans to minimize wildlife impacts on the mid-Atlantic Outer Continental Shelf (OCS). By pairing persistent hotspot maps with temporal bar charts for species and taxonomic groups of interest, we hope to develop a comprehensive picture of geographic and temporal patterns of wildlife within the study area.

## Methods

Between March 2012 and May 2014, we conducted 16 large-scale boat-based visual surveys and 15 large-scale high-resolution digital video aerial surveys across the mid-Atlantic study area. Details on the study area and data collection methods can be found in Chapter 3 and Chapter 7. Several taxa observed during boat and video aerial surveys had insufficient data to calculate persistent hotspots or conduct other spatial analyses, and simple point maps of raw data for several of these taxa are briefly discussed. For taxa with more robust datasets from the boat survey, video aerial survey, or both, additional analyses were conducted.

### *Persistent hotspot analysis*

We adapted the methods of Santora and Veit (2013) to quantify the variance and anomaly persistence of counts for a single species or species group within grid cells across the study area. Aerial and boat survey transects and species observations were binned by Bureau of Ocean Energy Management (BOEM) OCS lease blocks (a 23.0 $km^2$ grid, where each cell is 4.8 x 4.8 km) using ArcGIS version 10.2.2 (ESRI, Redlands, California). The BOEM lease grid was extended west of the Submerged Lands Act boundary (generally 5.6 km from shore) to include the entire Mid-Atlantic Baseline Studies and Maryland Project study areas. Binning survey data by grid cell allowed us to standardize for spatial variation in survey effort within each survey, and combine the resulting hotspot determinations from all 31 surveys (including both survey methods) in a unified hotspot persistence map.

We limited our analysis to the most commonly observed species or species groups, defined by having a minimum of 700 total observations from a survey method over the entire study period. This cut-off point was found to be high enough to show ecologically relevant patterns for most species or groups examined, while eliminating most complications caused by low sample size. The exception to this criterion was cormorants (Phalacrocoracidae spp.) observed by boat; although over 700 individuals were observed, over half of these individuals were recorded in three single observations, which prevented the identification of persistent hotspots according to the criteria used in this chapter. Data were grouped and analyzed by family, instead of by individual species, provided that either (1) most observations within a family likely represented a single species (e.g., unidentified storm-petrels and Wilson's Storm-Petrels [*Oceanites oceanicus*] were mapped together as storm-petrels, Hydrobatidae), or (2) sample sizes for single species were too small to analyze separately, but large enough to be analyzed when aggregated by family (e.g., alcids, Alcidae).

#### *Defining a hotspot*

Boat and video aerial survey data were analyzed independently for hotspot analysis. First, an effort-corrected count was calculated for the species or group of interest per grid cell for each survey. For boat surveys, this was done by dividing the number of individuals observed by the total transect length (km) within each grid cell for each survey. For video aerial surveys, the number of individuals observed was divided by the total surveyed area ($km^2$) within each grid cell per survey. As resulting data were highly non-normal, a gamma distribution was fitted to non-zero effort-corrected counts for each grid cell in a given survey using 'fitdistrplus' package (Delignette-Muller and Dutang, 2015) in the R Statistical Environment (version 1.1-7, R Core Team 2014), and used to assign a probability to each grid cell's value depending on where it fell within the distribution curve for that survey. Fitting a gamma distribution to non-zero effort-corrected counts allowed us to identify cells with high abundance relative to other cells where the taxa was present, on a survey by survey basis. We considered grid cells within the top quartile (>75th percentile in the survey's gamma distribution) of effort-corrected values for a given survey as hotspots for that survey.

### *Determining persistence*

After identifying hotspots for each survey, data were combined in order to index hotspot persistence, or the percentage of time each grid cell was a hotspot across all surveys (within a given survey method). Grid cells that had been surveyed fewer than eight times (i.e., in fewer than half of the surveys) within a survey method were excluded from further analysis. Using these criteria, 168 grid cells were included in further analyses when only boat survey data were analyzed, 410 grid cells were included when only aerial data were analyzed, and a total of 450 cells were included when both boat and aerial data were analyzed and combined in a unified hotspot persistence map for a given species or taxon. In these combined maps, 128 grid cells were surveyed by both methods, 40 cells were surveyed only by boat, and 282 cells were surveyed only by aerial methods (Figure 17-1).

Where only boat or aerial data were analyzed for a given species, the number of times each cell represented a hotspot (hotspot sum) was divided by the number of times the cell was surveyed (survey sum), to calculate persistence as the percentage of surveys in which a cell represented a hotspot for the species or group of interest. For grid cells surveyed by both boat and video aerial survey methods, data were combined. Due to presumed differences in detection and/or identification rates between these two survey methods, we often observed notably different counts of species between the two datasets (Chapters 1 and 14). To account for these differences, we weighted the data by effort-corrected total abundance for each dataset before calculating persistence as described above. Effort-corrected total abundance was calculated by dividing total abundance across all surveys by total area surveyed ($km^2$) across all surveys. Effective strip width for boat survey transects was approximated by multiplying the total transect length by the median distance at which the species/group was observed from the boat, then multiplying by two (to account for the fact that observers surveyed both sides of the boat simultaneously). The resulting ratio of boat to aerial effort-corrected abundance was used to weight data using the following equation:

$$Weighted\ Persistence = \frac{(Hsum_a \times R_a) + (Hsum_b \times R_b)}{(Ssum_a \times R_a) + (Ssum_b \times R_b)}$$

where $R_a$:$R_b$ is the ratio of aerial to boat effort-corrected abundance, *Hsum* is number of times a cell was identified as a hotspot by survey method (*a,* aerial; *b*, boat), and *Ssum* is the number of times a cell was surveyed by each method.

#### *Mapping hotspot persistence*

Persistence values were broken into four distinct classes for mapping purposes, based on breaks at the 75th, 85th, and 95th percentiles of persistence values for cells that were a hotspot in at least one survey. We presented percentiles of persistence values (rather than the persistence values themselves) in order to facilitate comparison between species with different life histories, which may be present in the study area for varying amounts of time throughout the year.

#### *Special case: Common Loons and Red-throated Loons*

Loon abundance data collected by video aerial surveys presented a unique challenge, as only 14% of aerial loon observations were identified to species; the remaining 86% were categorized as unidentified loons, which contained an unknown proportion of either Common Loons (*Gavia immer*) or Red-throated Loons (*G. stellata*). We used the species identification model with environmental covariates developed in this study (Chapter 16) to predict the proportions of Red-throated and Common Loons in each grid cell for four aerial surveys (May 2012, December 2012, March 2013, and December 2013). These surveys had high loon abundance, and also had a boat survey conducted within two weeks of the aerial survey (Chapter 16). For these four aerial surveys, we summed the predicted counts of Red-throated and Common Loons with the identified counts for each species to calculate hotspot persistence in video aerial data. In remaining aerial surveys, only the identified counts (e.g., birds identified as either Common Loons or Red-throated Loons, but not unidentified loons) were used in determining hotspots for the two species.

#### *Special case: Species Richness*

Species richness hotspots were identified using the same analysis methods described above, with two modifications. First, for each grid cell and survey, the datum of interest was considered to be the total count of species observed within the grid cell, rather than the effort-corrected count of an individual taxon observed within the grid cell. The relationship between survey effort and the number of species observed is not linear, so we did not effort-correct species counts within each grid cell, in order to avoid over-estimating counts in cells with very low effort (Gotelli and Colwell, 2010). However, in order to identify hotspots within datasets with similar effort per grid cell, we separated the sawtooth aerial survey transects from the high density aerial survey transects, which were located in the Wind Energy Areas (WEAs) and offshore of Maryland (see study area map in Executive Summary and other chapters throughout this report). We independently identified hotspots within the sawtooth aerial data, high density aerial data, and boat data, and species richness hotspots identified from each dataset were weighted equally when combined to map hotspot persistence.

### ***Temporal bar charts***

We generated temporal bar charts of effort-corrected count data for boat and video aerial survey data independently, because detection and geographic coverage varied between survey methods (Chapters 14-15 and 18). The total count of individuals was summed for each species and species group by two-month time period. Thus, each time period included data from two to four surveys over the two-year study. The two-month length of these time periods was found to best serve for data visualization purposes, as it allowed for variation in the data presented while also controlling somewhat for variation in effort between periods (Chapters 5 and 8).

Bar charts were created in Microsoft Excel (Redmond, WA) for all individual species and species groups that were observed more than 10 times within a survey method over the course of the study. Species and group counts were standardized for survey effort for each survey method (boat-based and digital video aerial surveys), using linear kilometers surveyed within a two-month time period. Effective transect strip width varied greatly by taxon for boat survey data, and using linear kilometers rather than total area surveyed allowed for direct comparisons between the two study methods. Percentiles were calculated for all effort-corrected survey data from both survey types for species groups (Table 17-1) and individual species (Table 17-3). Boat and aerial percentile values, represented by bars of increasing height and greater color intensity, are presented adjacent to one another to allow for comparison between the two study methods.

## Results

Persistent hotspots were identified primarily for groups of species, rather than for individual species, due to sample size limitations and/or difficulties with species identification. Whenever possible, boat and video aerial survey data were combined to develop joint maps of persistent hotspots of abundance for taxa of interest. Insufficient data from one of the two survey methods, however, led to hotspots being estimated with data from a single survey method for some species groups. Thus, some hotspot maps below include data only from boat surveys (for example, for storm-petrels); some hotspots were calculated solely from video aerial surveys (such as sea turtles and rays); and many others used data from both survey methodologies, weighted by the ratio of effort-corrected counts between the two survey types. Temporal bar charts provide context for the maps of hotspots, illustrating the changes in relative abundance of counts and in species composition for both survey types over time.

### *Scoters*

Scoters, a genus of sea ducks that (in the mid-Atlantic) includes Black Scoter (*Melanitta americana*), White-winged Scoter (*M. fusca*), and Surf Scoter (*M. perspicillata*), were observed in 61% of surveys, primarily between September and May (Table 17-2). Though scoters were observed at all longitudes within the study area, observations in the east tended to be sporadic and to involve small numbers of individuals. Scoter flocks, or rafts, were most consistently located in areas within about 30 km of shore, particularly near the mouths of Chesapeake Bay and Delaware Bay (Figure 17-2). The persistent hotspots of scoter abundance identified in Figure 17-2 were some of the largest and most consistent of any species group examined. Surf Scoters (Figure 17-3) and Black Scoters (Figure 17-4) showed strikingly similar patterns of hotspot persistence to each other and to the family as a whole. Additional information on Surf Scoter movements and habitat use in the mid-Atlantic is presented in Chapter 20.

### *Loons*

Loons, including Common Loons and Red-throated Loons, were present in 90% of surveys, with greatest numbers present in the study area between November and May (Table 17-2, Table 17-4). Loons do not form large rafts like many sea duck species, and were more likely to be observed individually or in small groups. Hotspots of loon abundance were less persistent between surveys than for scoters, and showed distinctly different patterns between species (particularly when the species identification model using environmental covariates was used to incorporate unidentified loons into hotspot datasets; Chapter 16). Red-throated Loons showed highest hotspot persistence close to shore along the length of the study area (Figure 17-5). Common Loon hotspots were scattered across the width of the OCS, though many of the most persistent Common Loon hotspots were located offshore of the mouth of Chesapeake Bay (Figure 17-6). Additional information on Red-throated Loon movements and habitat use in the mid-Atlantic is presented in Chapter 21.

### *Storm-petrels*

Storm-petrels were not identified frequently enough from the aerial data to justify mapping persistent hotspots of abundance using those data, but hotspots estimated from boat data for this taxon (primarily Wilson's Storm-Petrels), are presented in Figure 17-7. Storm-petrels were observed in 50% of boat surveys, and almost exclusively in summer (Table 17-2). Identified hotspots of relative abundance included both nearshore and offshore locations (Figure 17-7). Storm-petrels were generally observed individually, rather than in groups, and were abundant for only a few months each year; this led to lease blocks only being considered a hotspot in one out of 16 (6.25%) or two out of 16 (12.5%) boat surveys (that is, 12.5% or 25% of surveys in which the taxon was present in the study area), so persistence classes were consolidated into two categories to display the data for this species group.

### *Northern Gannets*

Northern Gannets (*Morus bassanus*) were observed in 81% of all surveys (13 out of 16 boat surveys and 12 out of 15 video aerial surveys). The species was widely distributed across the study area. The most persistent abundance hotspots for gannets contained large aggregations between 36% and 54% of the time that the species was present in the study area, but the majority of grid cells (70%) were identified as an abundance hotspot during at least one survey (Figure 17-8), indicating that Northern Gannet distribution and abundance patterns varied widely between surveys. The most persistent hotspots tended to be located within about 30-40 km of the shoreline, although abundance hotspots were also consistently observed in several offshore locations (Figure 17-8). Northern Gannets were consistently observed in high numbers from September to April for both boat and video aerial surveys, with low numbers in July and August (Table 17-4). The two survey methods showed very similar temporal variance for Northern Gannets, indicating that detection rates for this species may have been relatively similar between survey methods. Additional information on Northern Gannet movements and habitat use in the mid-Atlantic is presented in Chapters 22 and 24.

### *Alcids*

Family Alcidae, which in the mid-Atlantic generally includes Dovekies (*Alle alle*), Razorbills (*Alca torda*), Atlantic Puffins (*Fratercula arctica*), and both murres (*Uria* spp.), were not identified frequently enough from the aerial data to justify mapping persistent hotspots of abundance. Hotspots estimated from boat data for this taxon are presented in Figure 17-9. Alcids were present almost exclusively in winter (Table 17-2). Identified hotspots of relative abundance included both nearshore and offshore locations, but the largest and most persistent aggregations seemed to occur in the part of the study area located farthest from the shoreline (between about 60-85 km from the coast of southern Virginia; Figure 17-9). Alcids were seldom observed in groups, and the most persistent hotspots for this species were identified in about 30% of the seven boat surveys for which the taxon was present in the study area (2 out of 16 surveys in total).

### *Gulls and terns*

Gulls and terns (Laridae) were observed in all surveys. This is a fairly disparate group in terms of behaviors across species, with some species breeding near the study area, others using this region purely in the non-breeding season, and still others present year-round. Bonaparte's Gull (*Chroicocephalus philadelphia*) and Ring-billed Gull (*Larus delawarensis*), for example, are present primarily in winter, while other gulls are present during fall, winter, and spring (e.g., Laughing Gull, *Leucophaeus atricilla*), and several tern species are present in spring, summer, and fall (Common Tern, *Sterna hirundo*, and Royal Tern, *Thalasseus maximus*). Several gull species use the study area year-round (Herring Gull, *Larus argentatus*, Great Black-backed Gull, *L. marinus*, and Lesser Black-backed Gull, *L. fuscus*; Table 17-4). Likewise, species distributions across the study area vary based on when each species is present. We calculated persistent hotspots for the entire family, as 23% of these aerial observations were not differentiated to subfamily (Chapter 5). But due to the life history and distributional differences between species, we also analyzed data separately for the two main subfamilies (terns, Sterninae; and gulls, Larinae), as well as for the most abundant individual species in our datasets (Bonaparte's Gull, Laughing Gull, Herring Gull, Great Black-backed Gull, and Common Tern).

For the entire family, abundance hotspots were widely distributed throughout the study area, though the most persistent of these were located in the western half of the study area, and particularly in three locations: the mouth of Chesapeake Bay, the mouth of Delaware Bay, and the northern shore of Maryland (Figure 17-10). The same patterns for the most persistent hotspots are present in both the gull-specific (Figure 17-11) and tern-specific (Figure 17-12) maps. A comparison of Figure 17-10 to Figure 17-12, however, indicates that the less persistent hotspots located in many offshore areas in the eastern part of the study area were largely driven by gull distributions, with many fewer tern hotspots in areas >20 km from shore.

Examining hotspot persistence of individual species allowed us to further parse patterns shown in the subfamily maps. Hotspot persistence for both Herring Gulls and Great Black-backed Gulls was most similar to hotspot persistence for the gull subfamily; hotspots occurred across the study area, and were most persistent along the north shore of Maryland and at the mouth of the Chesapeake Bay (Figure 17-15; Figure 17-16). Laughing Gull hotspots were also most persistent in nearshore areas, primarily in the northern parts of the study area (Figure 17-14). In contrast, Bonaparte's Gulls were notably more persistently observed in large numbers in the southern half of the study area, and at a range of distances from shore (Figure 17-13).

Common Terns were the only tern species abundant enough to include in analysis, and were only observed in large numbers by boat (only one Common Tern was identified over the course of the study in digital video aerial surveys). Common Tern hotspots occurred across the OCS, but were most persistent near the mouth of Delaware Bay (Figure 17-17). This differed from the pattern of hotspot persistence observed for terns as a group, which additionally showed high hotspot persistence in Maryland state waters and at the mouth of Chesapeake Bay (Figure 17-12). These differences were likely driven in part by Royal Terns observed from the boat, as well as unidentified terns in both boat and video aerial data (which represented a diverse group of at least six species; Table 17-4) that are mapped in aggregate in Figure 17-12.

### *Rays*

Rays (Batoidea), primarily Cownose Rays (*Rhinoptera bonasus*), were mostly observed in summer and early fall, and were much more frequently observed in video aerial surveys than from the boat (Table 17-2). They were not identified frequently enough from the boat data to justify mapping persistent hotspots of abundance, so only the aerial data are presented in Figure 17-18. Cownose Rays occur in the coastal waters of the western Atlantic Ocean from the northeastern US to Brazil, and migrate seasonally along the Atlantic coast of the US (Goodman et al., 2011). Large and persistent aggregations of rays were commonly observed at the mouth of Chesapeake Bay, the mouth of Delaware Bay, and within about 20-40 km from the coast of Maryland and the north shore of Virginia (Figure 17-18). Further discussion regarding observed Cownose Ray distributions is presented in Chapter 5.

### *Sea turtles*

Sea turtles, including Green (*Chelonia mydas*), Kemp's Ridley (Lepidochelys kempii), Loggerhead (*Caretta caretta*), Hawksbill (*Eretmochelys imbricata*), and Leatherback (*Dermochelys coriacea*) turtles, were mostly observed in warmer months (Table 17-2). They were also much more frequently observed in video aerial surveys than from the boat (Chapter 14), a phenomenon that has been seen for digital aerial surveys elsewhere (Normandeau Associates Inc., 2013). They were not observed frequently enough from boat-based surveys to justify mapping persistent hotspots of abundance, so only the video aerial data are presented in Figure 17-19. Sea turtle species were observed in 80% of video aerial surveys, and were more consistently located in large numbers in the southern half of the study area and farther from shore (Figure 17-19). Further examination of seasonal distribution patterns and possible environmental drivers for this taxon is presented in Chapter 15.

### *Dolphins and porpoises*

Odontoceti, or toothed whales, were observed throughout the study period, with some summertime increases in observations (Table 17-2). The two survey methods showed similar temporal patterns of relative abundance for toothed whales, indicating that detection rates may have been similar between survey methods (Table 17-2; Chapter 14). Almost all identified observations were either Bottlenose Dolphins (*Tursiops truncatus*) or Common Dolphins (*Delphinus delphis*). Bottlenose Dolphins were observed primarily in warmer months, were observed across the study area, and made up a higher proportion of the aerial data than the boat data for most time periods (Table 17-4). Bottlenose Dolphins are distributed into coastal and offshore populations in this area of the Atlantic (Kenney, 1990), so we likely saw individuals from both populations represented in these counts. Common Dolphin counts peaked in the winter for both survey types (Table 17-4), and were mostly observed in the eastern part of the study area. The persistent hotspots identified in Figure 17-20 reflect a combination of these species' distributions. Nearshore hotspots were likely driven by coastal Bottlenose Dolphin populations (Chapter 15), while hotspots located father offshore may represent a combination of offshore Bottlenose Dolphin and Common Dolphin populations. When mapped independently (using boat data only as there were too few aerial observations), Bottlenose Dolphin hotspots occurred primarily on the western half of the study area, and were most persistent at the mouth of Delaware Bay (Figure 17-21). Further examination of seasonal distribution patterns and possible environmental drivers for Bottlenose Dolphin distributions are presented in Chapter 15.

### *Bait balls*

Shoals of small fishes that were not individually distinguishable or identifiable during boat and video aerial surveys were recorded as 'bait balls', and each shoal was counted as a single observation regardless of group size. These large groups of forage fish were much more frequently observed in video aerial surveys than from the boat, and only the aerial data were used in calculating areas of persistent abundance. Bait balls were most often observed in the western half of the study area, and were most persistently observed in highest densities in the nearshore regions of the study area, particularly off the coast of Delaware and Maryland (Figure 17-22). In cells with the highest hotspot persistence, the area was identified as an abundance hotspot in roughly half of the surveys in which bait balls were observed.

### *Persistent hotspots of overall abundance and species richness*

When calculated in aggregate for all taxa observed in this study, abundance hotspots were most consistently observed in nearshore areas (within about 40 km from shore), particularly in northern Maryland, near the mouth of Chesapeake Bay and the North Shore of Virginia, and near the mouth of Delaware Bay (Figure 17-23). Hotspots of species richness were consistently located in similar areas (Figure 17-23). While the aggregate abundance hotspot patterns may be largely driven by a few common species groups (such as scoters and gannets), species richness hotspots display habitat use areas that are valuable to large numbers of species through time. In grid cells that were identified as species richness hotspots, up to 10 species were observed in a single survey; the most persistent species richness hotspots were identified in 88% of surveys.

### *Temporal trends in abundance*

Overall, late fall to early spring was identified by both boat and digital video aerial surveys as a time of year with high effort-corrected counts of animals in the study area, though many aquatic animals peaked in abundance in the summer (Table 17-2). Scoters, gannets, and gulls all contributed greatly to overall abundance, regardless of survey method; loons made up a large proportion of the boat data, in the early winter surveys in particular, while rays were highly abundant in the video aerial surveys in the summer and early fall (Table 17-2). Some differences in temporal patterns between the two survey types are likely reflective of differences in detection for the two methods; for example, both Common Loons and Red-throated Loons make up higher proportions of the boat data compared to the video aerial data, as most loons in aerial surveys were not identified to species. There were peaks in abundance of some alcid and tern species that went almost entirely undetected in the video aerial data, while the video aerial surveys were able to detect temporal trends in abundance of several aquatic species that weren't detected or abundant in the boat surveys (Table 17-4).

### *Distributions of uncommonly observed species*

Several other taxa observed during boat and video aerial surveys had insufficient data to calculate persistent hotspots or conduct other spatial analyses. Cormorants made up a relatively large proportion of the boat data (3.2%), with a total of 2,035 individuals observed. Despite high abundance, there were relatively few sightings; over half of the total individuals observed were reported in three sightings in May and October 2013 at the mouth of Delaware Bay, and only 38 total sightings were reported. Nearly all were identified as Double-Crested Cormorants (*Phalacrocorax auritus*). Only 42 Double-Crested Cormorants were observed by video aerial surveys. Cormorants were observed across the study area, but the largest groups were mostly observed close to the mouths of Delaware Bay and Chesapeake Bay (Figure 17-24). Cormorants were most commonly observed by boat in the spring and fall (Table 17-2).

Passerines made up a small proportion of the aerial data, compared to boat data; 180 passerines were observed by boat and 17 were observed by video aerial survey methods (representing 22 unique identified species; Chapters 5 and 8), with peak numbers of passerines spring through the fall (Table 17-2). Songbirds were observed throughout the study area (Figure 17-25). Swallows (Hirundinidae) were the most frequently observed passerine, particularly in the coastal waters off of Virginia. Warblers (Parulidae) were most commonly observed in the Delaware and Maryland offshore areas. Only Purple Martins (*Progne subis*) and Barn Swallows (*Hirundo rustica*) were abundant enough for temporal persistence charts, with peaks of observed counts in July-August and March-April, respectively (Table 17-4). Additional discussion of passerine migration (which largely occurs nocturnally) may be found in Chapters 26-27.

Shorebirds (Charadriiformes) were also observed primarily on boat-based surveys, with 587 observations of at least 15 species reported, as compared to 74 observations in the aerial data (Chapters 5 and 8). Shorebird observations were distributed broadly across the study area (Figure 17-26). Dunlin (*Calidris alpina*) and Red Phalarope (*Phalaropus fulicarius*) observations peaked in March-April, while Red-necked Phalaropes (*P. lobatus*) were observed primarily in September-October (Table 17-4). Only eight plovers were observed over the course of the study (five Wilson's Plover [*Charadrius wilsonia*], and three Semipalmated Plover [*C. semipalmatus*]), all observed during boat surveys. No identified Red Knots (*Calidris canutus*) were observed, though individuals could have been included among the unidentified shorebirds or unidentified scolopacids (Chapters 5 and 8).

Observations of shearwaters and fulmars (Procellariidae) were more consistent across survey platforms, with 325 individuals of six species of observed by boat surveys, and 112 individuals from at least five species observed during video aerial surveys (Chapters 5 and 8). Great Shearwaters (*Puffinus gravis*) and Cory's Shearwaters (*Calonectris diomedea*) were most commonly observed, typically in the eastern part of the study area (Figure 17-27). Manx Shearwaters (*Puffinus puffinus*) were mostly observed on boat surveys, and a single Audubon's Shearwater (*P. lherminieri*) was also observed on boat surveys. Shearwaters were observed primarily in the spring and fall, while Northern Fulmars (*Fulmaris glacialis*) were observed primarily in winter (Table 17-4).

Fifty-one large whales were observed throughout the study area during surveys, of which 35 were identified to species or family (Figure 17-28). North Atlantic Right Whales (*Eubalaena glacialis*) were detected primarily by video aerial survey methods, with several sightings within the Virginia WEA and on the sawtooth transects between Maryland and Virginia WEAs; an additional Right Whale was observed east of the Virginia WEA during a boat survey. All sightings were reported to NOAA and the New England Aquarium. Other whales, including Humpback Whales (*Megaptera novaeangliae*), Fin Whales (*Balaenoptera physalus*), and Minke Whales (*Balaenoptera acutorostrata*), were also observed throughout the study area (Figure 17-28). Thirty-one of the 51 large whale observations occurred during winter months (Chapters 5, 8, and 15).

Aerial and boat surveys both detected migratory movements of Eastern Red Bats (*Lasiurus borealis*) in the offshore environment in September of 2012 (Hatch et al. 2013) and September of 2013. Seventeen bats were observed altogether, including two during boat surveys and 15 in video aerial surveys. Bats were observed between approximately 16 and 70 km from shore (Figure 17-29), during morning daylight hours (Hatch et al., 2013). Video aerial survey methods allowed for altitude estimation for several of these bats at >200 m above sea level.

## Discussion

The presence and relative abundance of species within the project study area varied widely by time of year, with different species and groups using the study area during non-breeding (summer or winter), breeding, and migratory periods. We obtained insufficient observations for some taxa to develop useful distribution patterns; however, other useful information can be drawn from the raw data on its own. For example, our nine observations of North Atlantic Right Whales, the most critically endangered large whale along the Atlantic coast of North America, provide an important contribution to our collective knowledge of this species given their small population size and our general lack of detailed data on their movements and habitat use in the mid-Atlantic. Additionally, raw observation data for bats provides insight into their offshore migration patterns. In 2012, bats were observed during a period with relatively strong tailwinds and average barometric pressure, suggesting that their presence offshore may have been facultative (e.g., taking advantage of favorable migratory conditions), rather than because storms or other factors pushed them offshore. Direction of movement was noted to be southwest in 10 out of the 15 video aerial observations, further suggesting migratory movements. Little is known about the migration and movements of tree bat species in North America, but anecdotal observations of migrating bats over the Atlantic Ocean (particularly during fall migration periods) have been reported since at least the 1890s (Hatch et al., 2013). The observations from this study provide new evidence of bat movements offshore, and offer insight into their flight heights above sea level and the times of day at which such migrations may occur.

For species or groups with sufficient data, we developed products to visualize both temporal and spatial variation in distribution and relative abundance. Calculating persistent abundance hotspots provides a means for identifying locations where individuals of a species or species group are most often found in large aggregations relative to their typical distribution patterns. These areas likely provide important habitat for foraging, roosting, and/or other activities (Gende and Sigler, 2006; Santora and Veit, 2013; Santora et al., 2010; Sydeman et al., 2006). Calculating persistent abundance hotspots can be particularly useful for highly mobile marine wildlife, because this analysis identifies patterns of high abundance that persist over time. For example, while hotspots of Northern Gannet abundance occurred across the study area and throughout the year, the majority of the most consistent hotspots during our surveys occurred at the mouth of Chesapeake Bay. This pattern only emerged when data were aggregated across repeated surveys. Similarly, summarizing aggregated data across survey methods and across years allowed us to examine temporal patterns of abundance for many species present within the study area. Identifying such patterns may provide useful insight to future siting and permitting processes within the region.

Species of interest were widely distributed across the study area, but for many taxa, larger aggregations were more consistently observed in the western part of the study area, and particularly offshore of the mouths of Chesapeake Bay and Delaware Bay, and in northern Maryland (Figure 17-23). Some exceptions to this general rule included sea turtles, Common Dolphins, Common Loons, and alcids, which were more evenly distributed across the OCS or were more commonly observed in areas farther from shore. The area offshore of northern Maryland, while likely a real hotspot for many species such as gulls and terns, may have emerged as an important habitat use area in part because this was the only region in which boat and video aerial surveys were conducted in inshore state waters (e.g., within three miles of the shoreline), as well as the only area with high density aerial survey transects in nearshore federal waters (e.g., between state waters and the WEA). While high numbers of some species may be consistently present in other nearshore areas as well, similar surveys were not conducted in nearshore or state waters elsewhere during this study.

In some instances, our analyses revealed unexpected patterns of hotspot persistence that may contribute new information about the distribution and relative abundance of a taxon. For example, large and persistent aggregations of rays (primarily Cownose Rays) were observed at the mouth of the Chesapeake Bay, and it is likely that many of the rays observed in our study area moved into the Chesapeake Bay and its tributaries in the summer months, as found in previous studies (Blaylock, 1993; Fisher, 2010). However, our analyses also reveal persistent hotspots at the mouth of Delaware Bay, and within about 20-40 km from the coast of Maryland and the north shore of Virginia, suggesting that this population may also use Delaware Bay and possibly other locations during the summer. Considering that Cownose Rays are thought to summer exclusively in bays and estuaries (Grusha, 2005), and have been particularly well studied in the Chesapeake Bay (e.g., Smith and Merriner 1985; Smith and Merriner 1987), the hotspots calculated from video aerial survey data in this study include areas much farther north and father offshore than might have been expected.

### *Caveats for persistent hotspot analyses*

Several characteristics or limitations of persistent abundance hotspot maps should be noted, and carefully considered when using these maps for management or planning purposes. These maps do not indicate a species' full range of habitat use within the study area; rather, grid cells that were never identified as a hotspot simply never had abundance levels 'above the norm' for a particular species and survey. Quite often, blocks that were never identified as a hotspot still consistently hosted individuals of the species of interest. It is also important to note that persistent hotspot maps are intended to identify persistent geographic patterns at a regional scale; while values are presented by lease block, these individual grid cell 'persistence' values should be interpreted with caution. Minor changes to the display of these data (for example, slightly changing how a hotspot is defined within a survey, or using different persistence categories for mapping) may change individual grid cell values, though overall patterns of animal distributions remained quite robust to such adjustments.

It should also be noted that taxa of interest aggregate to varying degrees. A hotspot for alcids, for example, includes many fewer individuals than does a hotspot for scoters, because alcid species simply do not flock to the same degree within our study area. Likewise, scoter hotspots were much more consistent through time than were hotspots identified for some other species groups. Hotspot persistence calculations in this study were designed to be reasonably comparable between species, but all values presented in maps represent relative, rather than absolute, abundance.

Finally, it is important to consider the number of surveys conducted and the length of our study period when evaluating hotspot persistence. In this study we analyzed data from 31 surveys conducted over a two year period; although our timeframe was relative brief, we conducted a comparable number of surveys to previous studies examining hotspot persistence. Zipkin et al. (2015), for example, used compiled data from 32 data sets collected over a span of 32 years, while Santora and Veit (2013) used data from 14 surveys conducted over 9 years, and Gende and Sigler (2006) used data from 34 surveys conducted over 3 years. Although our study's timeframe is on the lower end of this spectrum, a recent analysis of interannual variation in wildlife distributions suggests that 2-3 years of surveys may be sufficient to capture longer-term (e.g., decadal) levels of variation (Kinlan et al., 2012). There are also several benefits to expending high survey effort over a relatively short time frame. First, this study design provides extensive data within a relatively small study area. Additionally, our study design provides a complete picture of what is happening year round during the course of the study, compared to studies that only survey within a single season (e.g., Santora and Veit, 2013). Combining data from boat-based (16 surveys) and digital video aerial (15 surveys) methodologies also provides a more complete picture of wildlife distributions than a single survey method, by providing complementary data collected during the same time frame in the same location. One drawback to this study design, however, is that trends of persistence may not be accurately captured for species that are present in the study area for short periods of time throughout the year, and thus have fewer opportunities to be sampled. For example, alcids were present almost exclusively in the winter, and the most persistent hotspots for alcids were only hotspots in two out of 16 boat surveys (alcids were not abundant enough in the aerial dataset to conduct persistent hotspot analysis). For these less commonly observed species, we may simply lack the number of sampling events required to adequately characterize lease blocks as hotspots (Zipkin et al., 2015). Data collected from surveys conducted over a greater number of years would provide greater opportunity for sampling of less commonly observed species, and would perhaps capture finer scale patterns of persistence with greater statistical rigor.

### *Ecology of persistent hotspots*

This study focused on identifying the locations and persistence of hotspots within the mid-Atlantic OCS, and did not examine drivers of hotspot occurrence. In some instances, however, we can infer that distinct populations within a taxon may be partial drivers of observed patterns of hotspot persistence. For example, persistent hotspots of Bottlenose Dolphins were generally located in nearshore regions within the study area. As Bottlenose Dolphins are distributed into coastal and offshore populations in this area of the Atlantic, this pattern was likely partially driven by the consistency of locations and numbers for the coastal ecotype of this species, as compared to the more variable and transient populations offshore (Gannon and Waples, 2004; Kenney, 1990).

Patterns of persistent hotspots for various taxa are also likely driven by environmental factors, as previous studies have shown that persistent hotspots likely indicate locations of important habitat for the taxa examined. Piatt et al. (2006) showed that Short-tailed Albatross hotspots in the Aleutian Islands were closely associated with shelf-edge habitats where upwelling and strong vertical mixing occurred, supporting high primary and secondary productivity. Similarly, Suryan et al. (2012) found that persistently high levels of primary productivity (chlorophyll *a*) are a significant predictor of seabird hotspots. Other studies have shown the relationship between hotspots of prey species and hotspots of marine predators (Gende and Sigler, 2006; Santora et al., 2010).

In our study, the most common persistent hotspots tended to occur in nearshore areas, particularly in northern Maryland and areas near and directly south of the mouths of Chesapeake Bay and Delaware Bay. These nearshore regions, particularly those adjacent to the regional bays, contained the most persistent hotspots of overall abundance and species richness, in addition to persistent hotspots for many individual taxa examined. These areas are likely attractive to a wide variety of high trophic level species, such as seabirds and marine mammals, due to their consistently higher primary productivity relative to the broader study area (Chapter 1; Smith and Kemp 1995; Schofield et al. 2008). These areas typically have the highest levels of chlorophyll *a* in the study area due to their close proximity to highly productive estuarine ecosystems, where strong tidal currents and year-round mixing of saline and fresh waters boost productivity. More generally, in shallow coastal waters sunlight is able to penetrate a high proportion of the water column, fueling photosynthetic activity and phytoplankton growth where nutrients are available (Schofield et al., 2008; Xu et al., 2011). This primary productivity forms the base of the pelagic food chain on which nearly all species observed during this study rely; thus, these areas likely serve as key wildlife habitats within the study area, and the locations of these areas should be considered carefully in relation to any future offshore development activities in the region.

Our results present an opportunity for future studies to explicitly examine the relationship between the location and persistence of hotspots (as determined in this study) and the potential environmental predictors of such hotspots. Of particular note, future studies could explore the relationship between persistent hotspots of bait balls and those of marine predators; as populations of forage fishes that form bait balls likely serve as a prey base for many upper trophic level predators, the distribution of persistent bait ball hotspots has the potential to help explain the similar nearshore distribution observed for many other taxa. There may also be more direct relationships between hotspots of higher trophic level taxa, as the location and persistence of hotspots are likely influenced by competitive and/or facilitative species interactions (Chapter 18 Appendix A; Ainley et al., 2009; Camphuysen and Webb, 1999).

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# Figures and tables

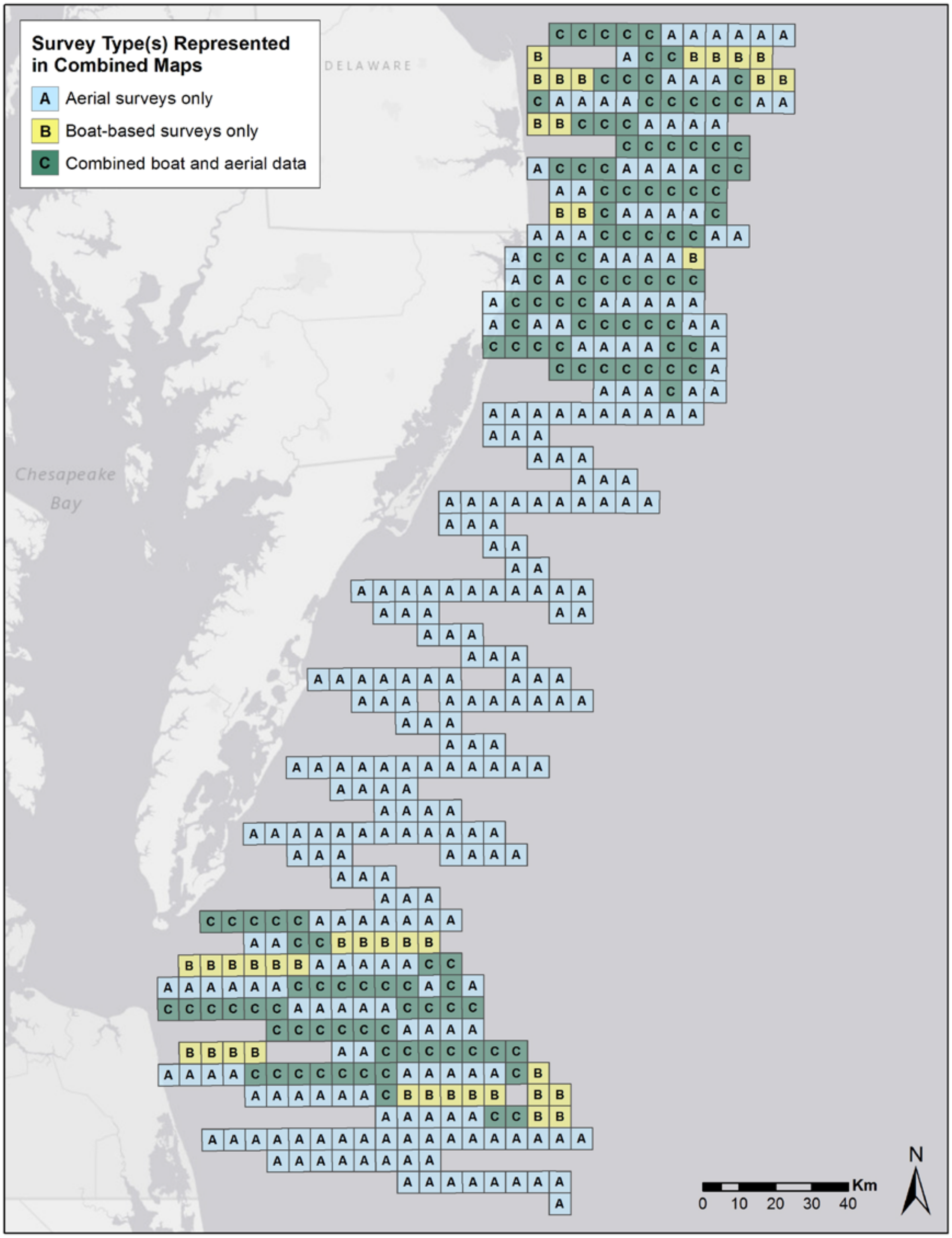

**Figure 17-1. Survey method by grid cell.** Sixteen boat-based and 15 video aerial surveys were conducted across the study area, resulting in a total of 450 surveyed grid cells: (A) 262 grid cells surveyed by video aerial surveys only, (B) 40 grid cells surveyed by boat-based surveys only, and (C) 128 grid cells surveyed by both methods.

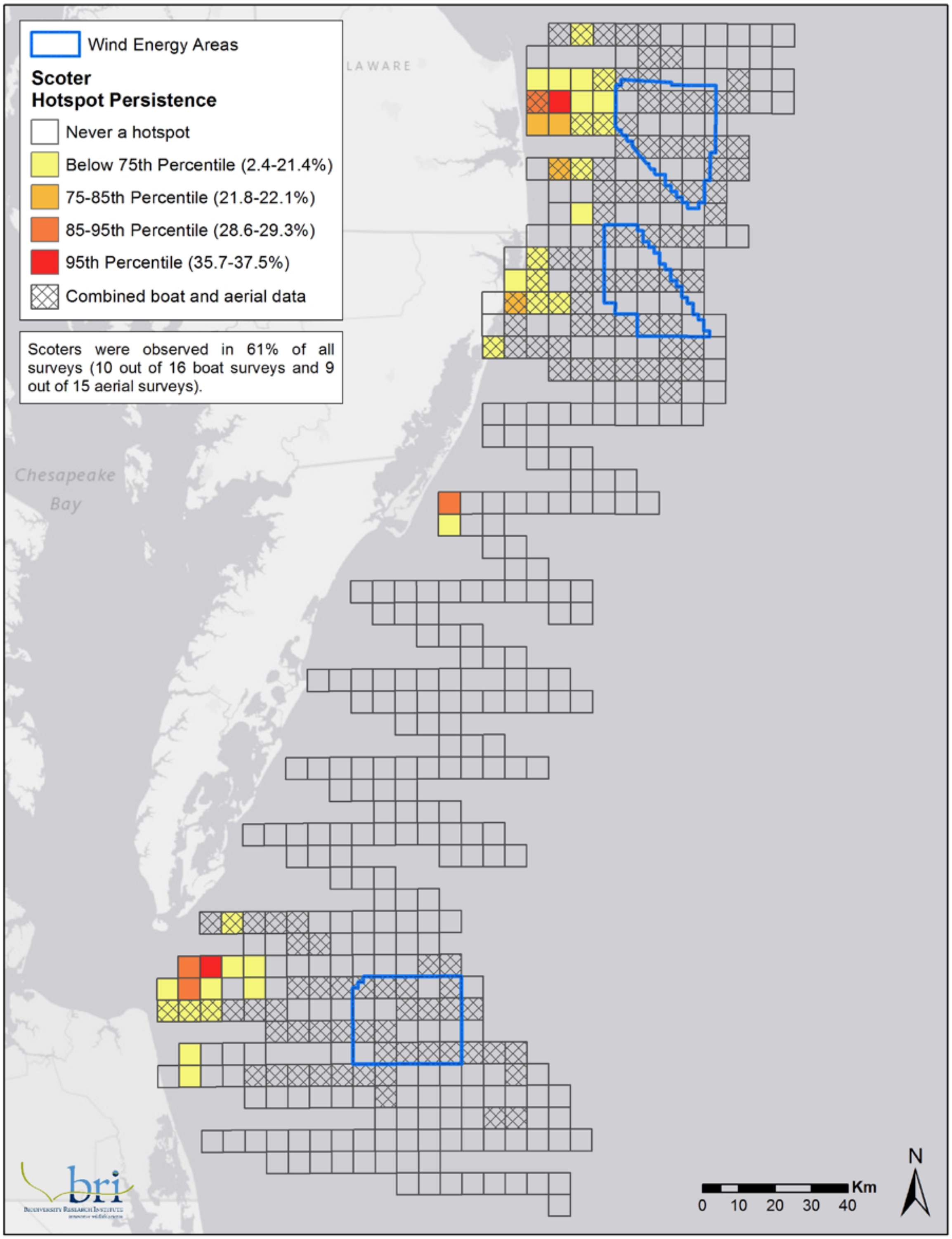

**Figure 17-2. Classified persistent abundance hotspots for scoters (*Melanitta spp.*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

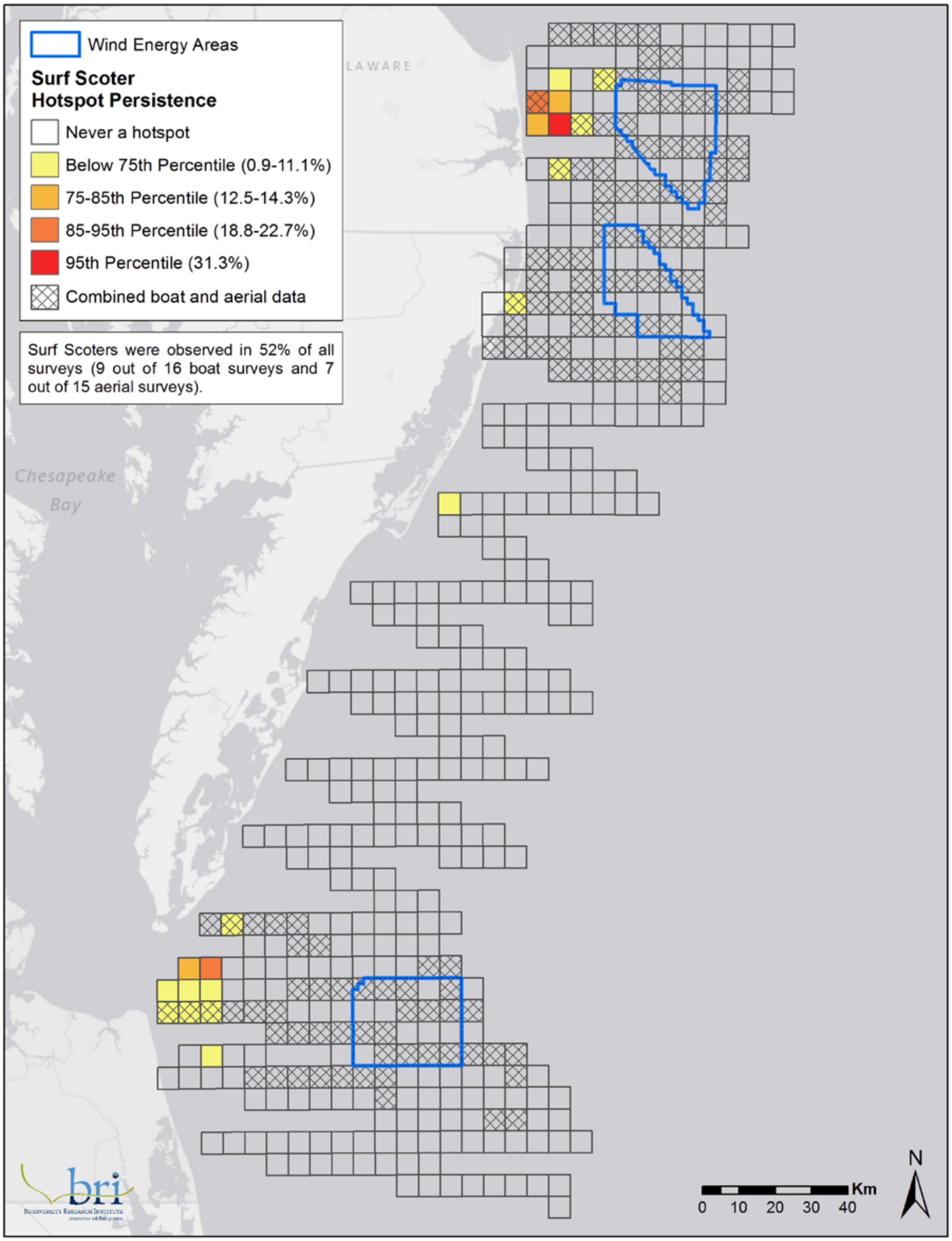

**Figure 17-3. Classified persistent abundance hotspots for Surf Scoters (*Melanitta perspicillata*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

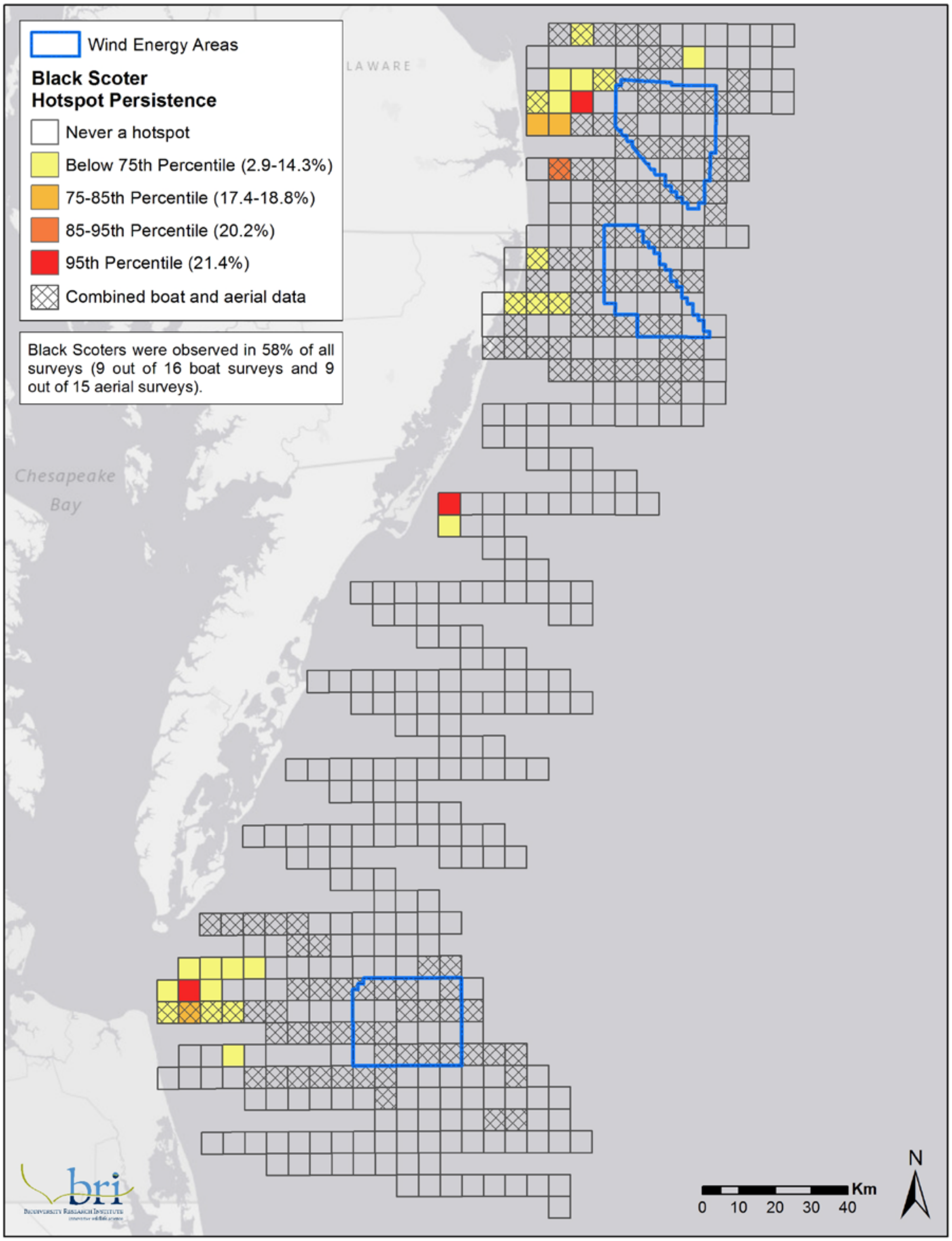

**Figure 17-4. Classified persistent abundance hotspots for Black Scoters (*Melanitta americana*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

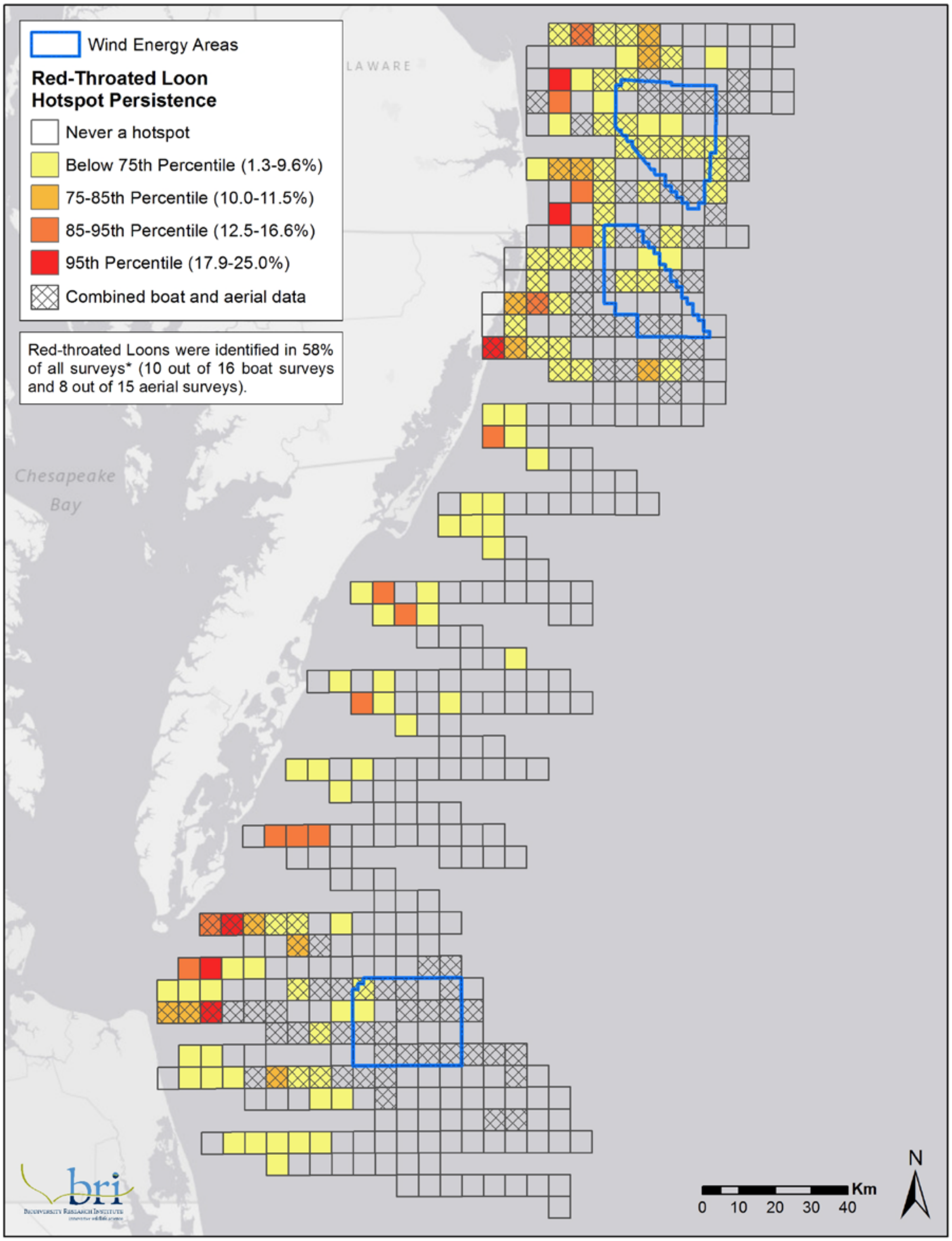

**Figure 17-5. Classified persistent abundance hotspots for Red-throated Loons (*Gavia stellata*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods. *Red-throated Loons were identified to species in 7 aerial surveys, and were predicted to be present in one additional survey using the species identification model (Chapter 17).

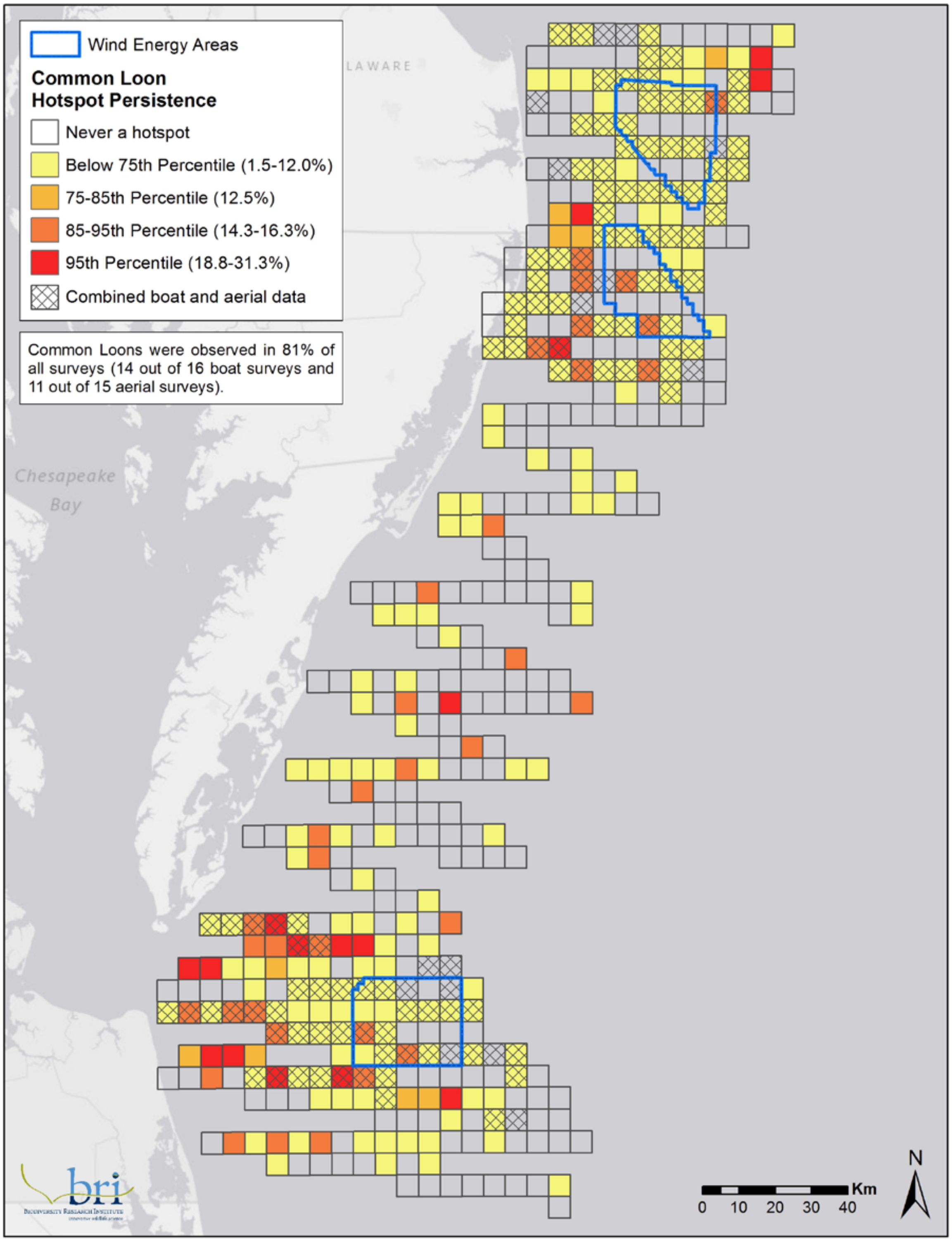

**Figure 17-6. Classified persistent abundance hotspots for Common Loons (*Gavia immer*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

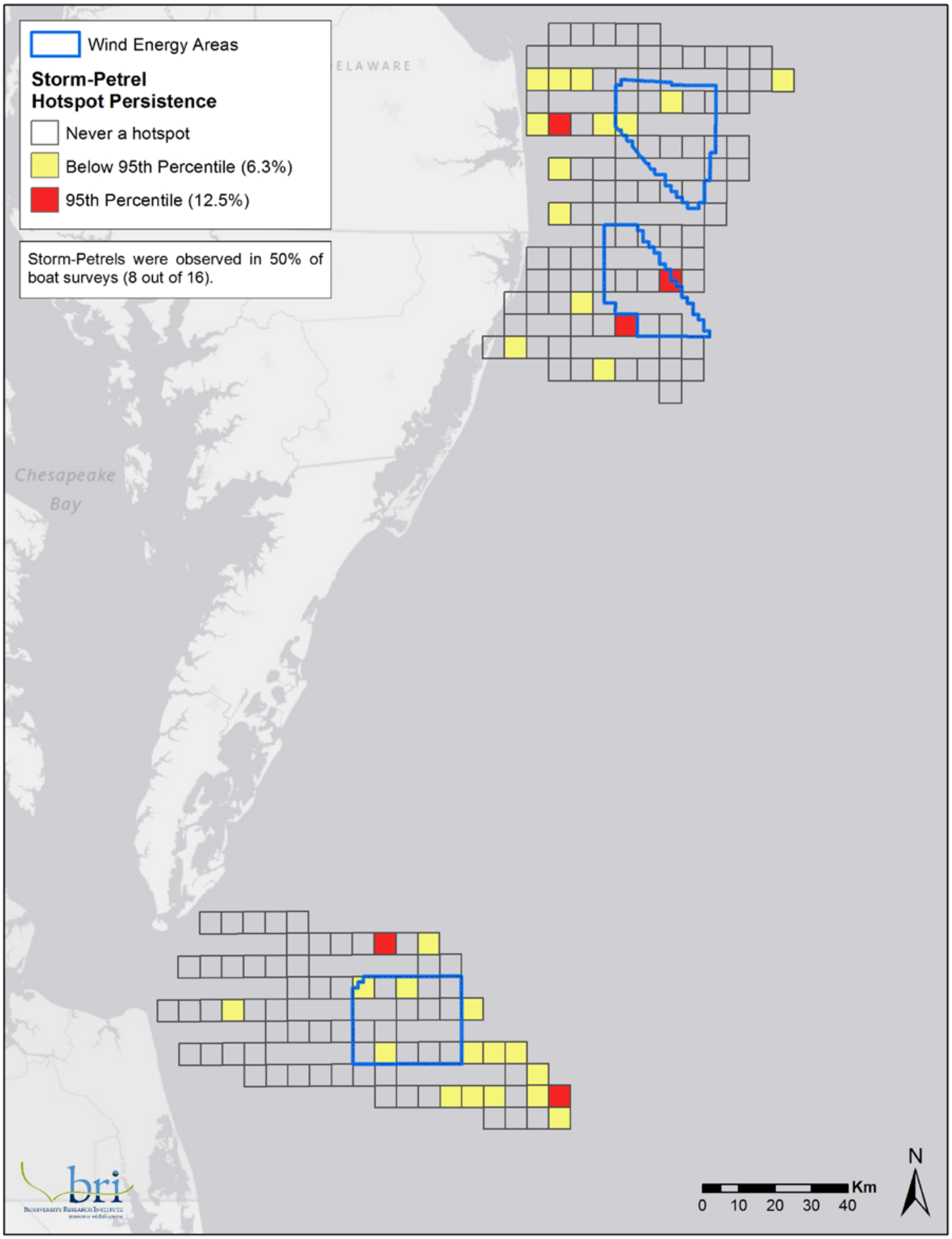

**Figure 17-7. Classified persistent abundance hotspots for storm-petrels (Hydrobatidae spp.) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Data are split into only two persistence classes as only two distinct persistence values were calculated.

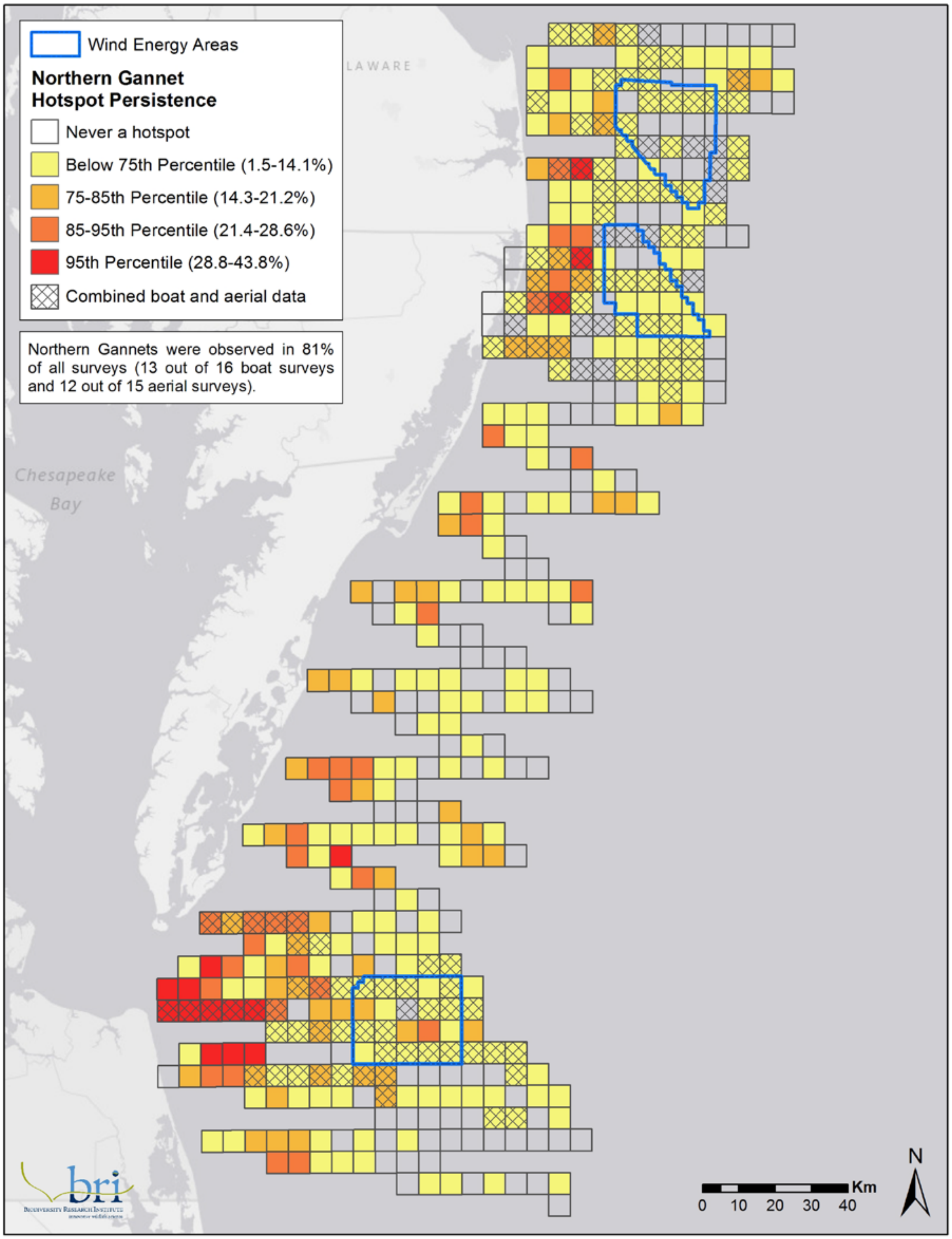

**Figure 17-8. Classified persistent abundance hotspots for Northern Gannets (*Morus bassanus*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

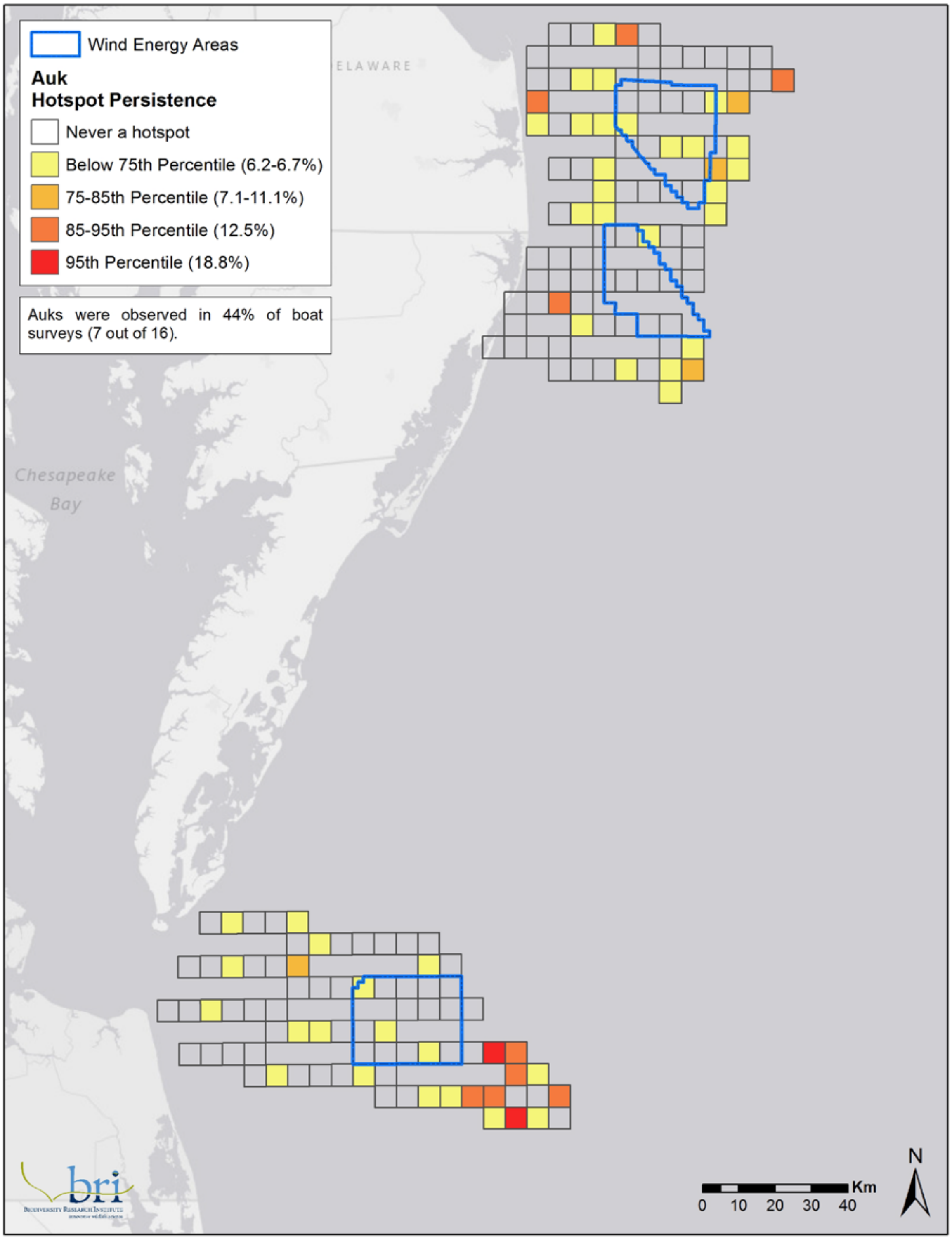

**Figure 17-9. Classified persistent abundance hotspots for alcids (Alcidae spp.) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot.

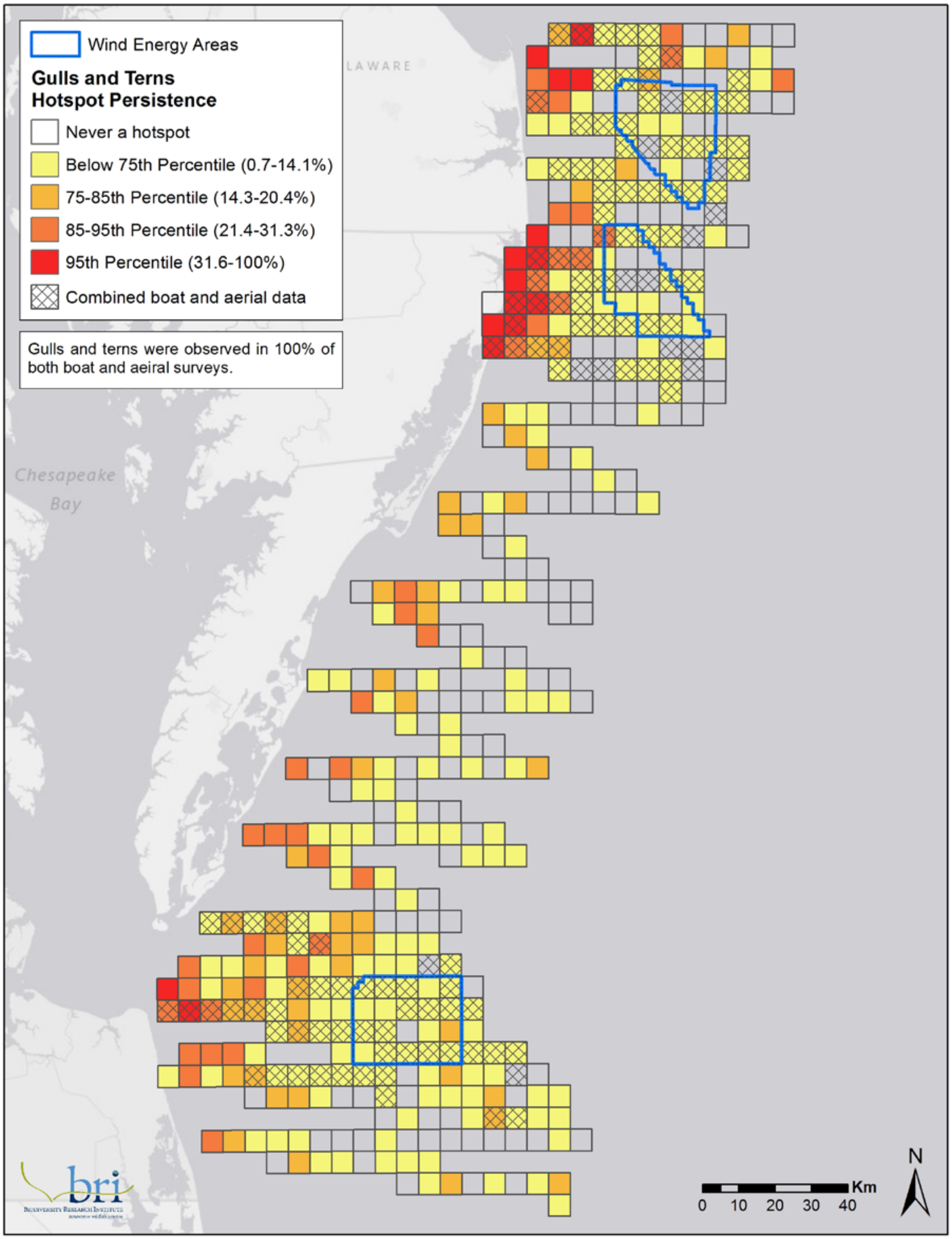

**Figure 17-10. Classified persistent abundance hotspots for gulls and terns (Laridae spp.) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

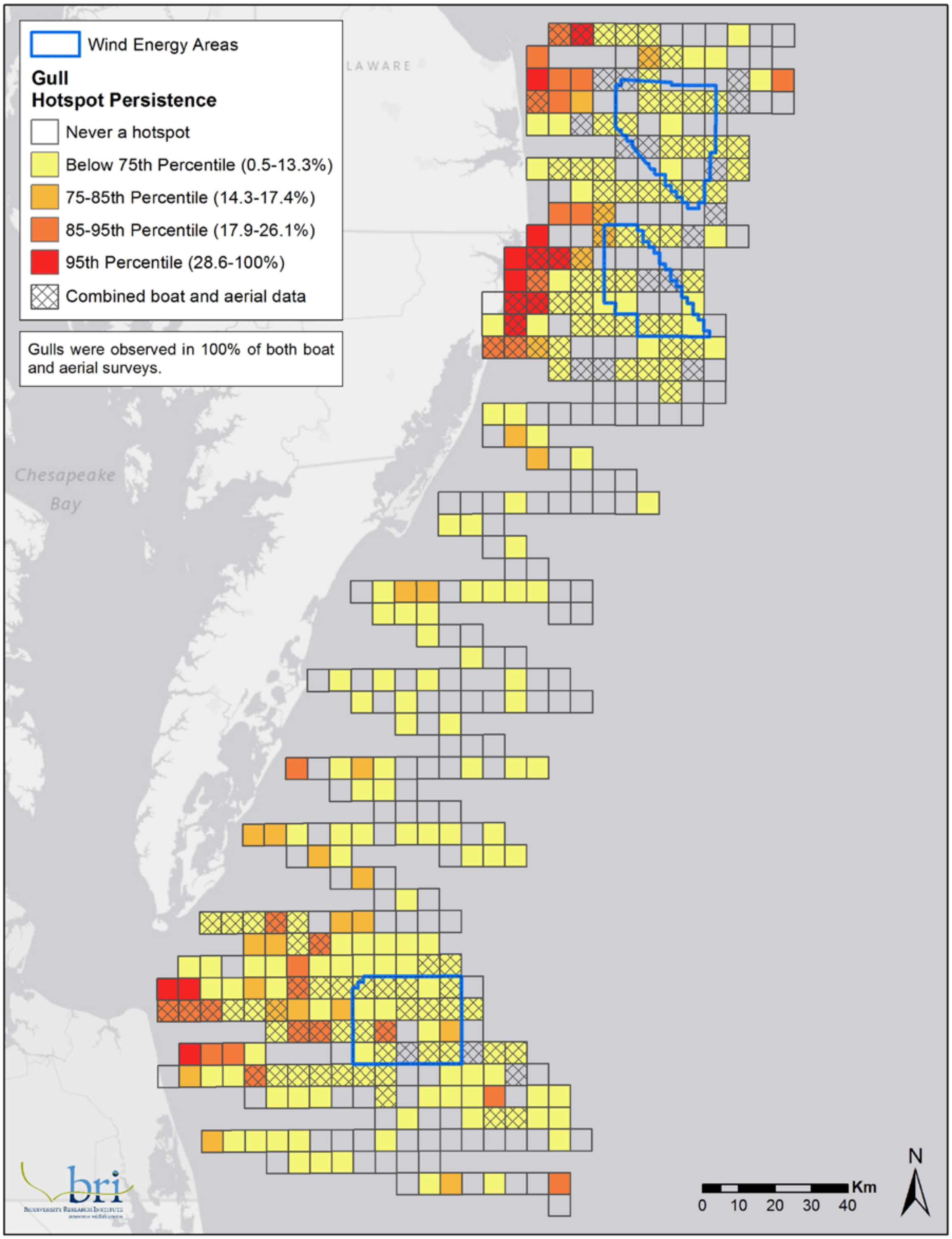

**Figure 17-11. Classified persistent abundance hotspots for gulls (Larinae spp.) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

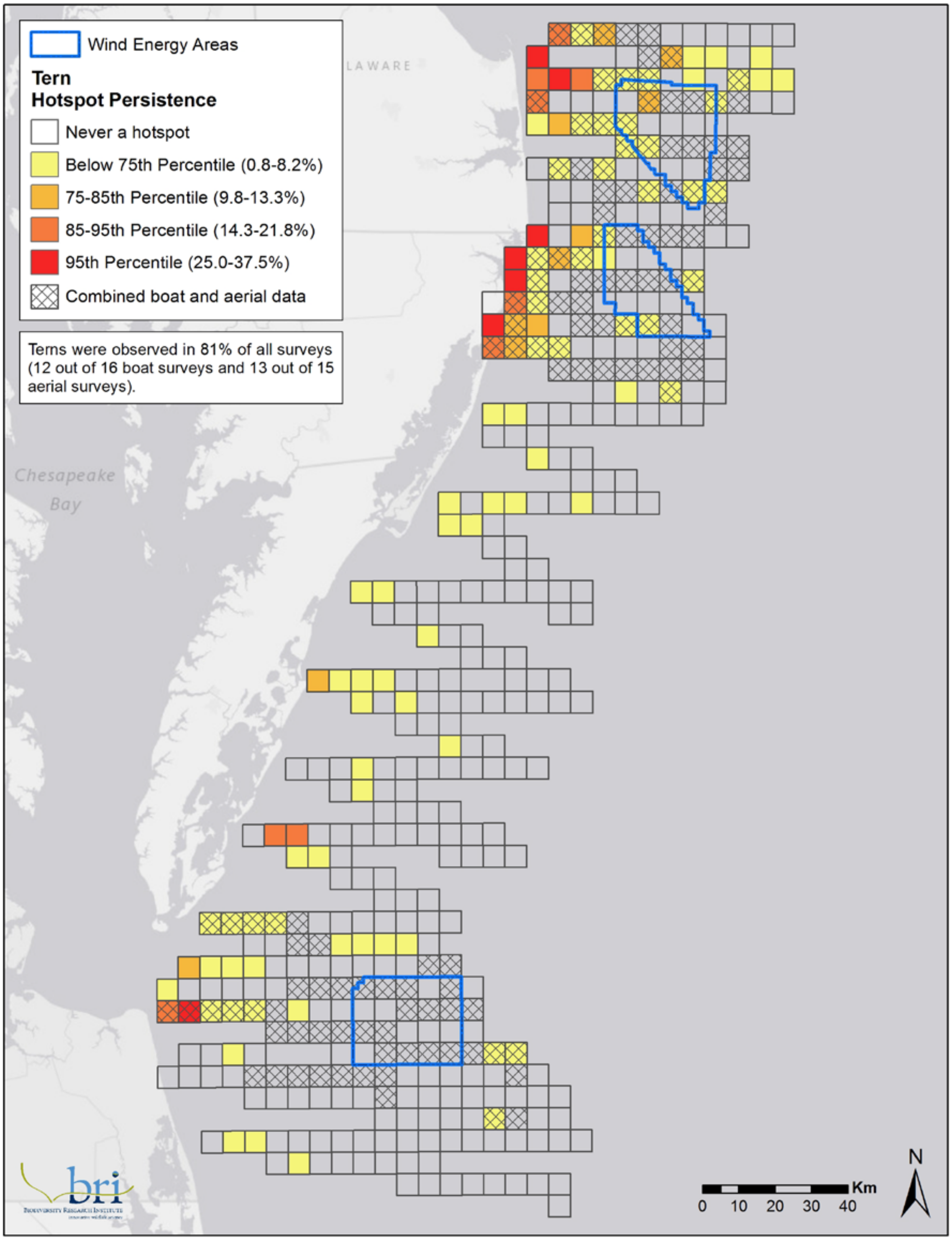

**Figure 17-12. Classified persistent abundance hotspots for terns (Sterninae spp.) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

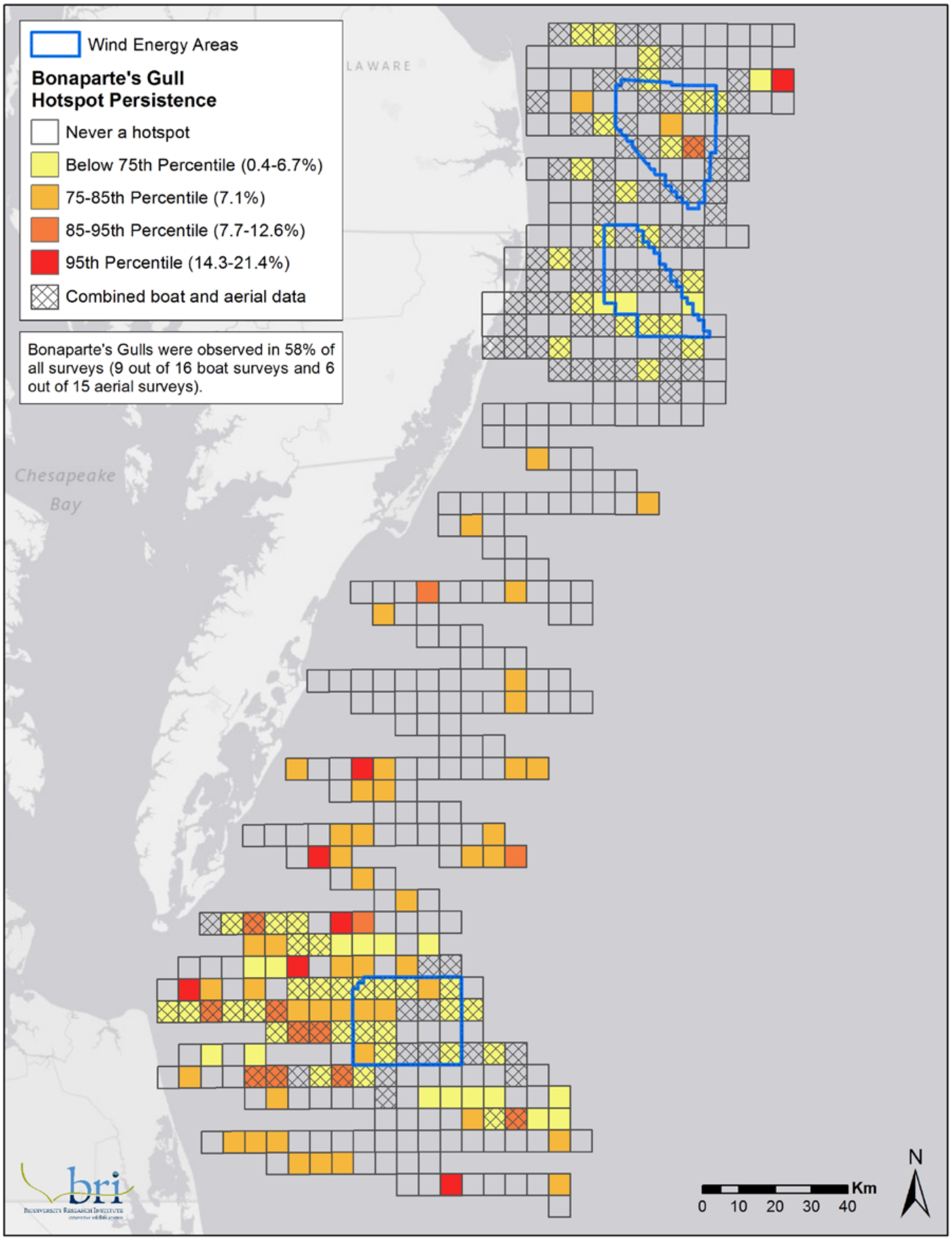

**Figure 17-13. Classified persistent abundance hotspots for Bonaparte's Gulls (*Chroicocephalus philadelphia*) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

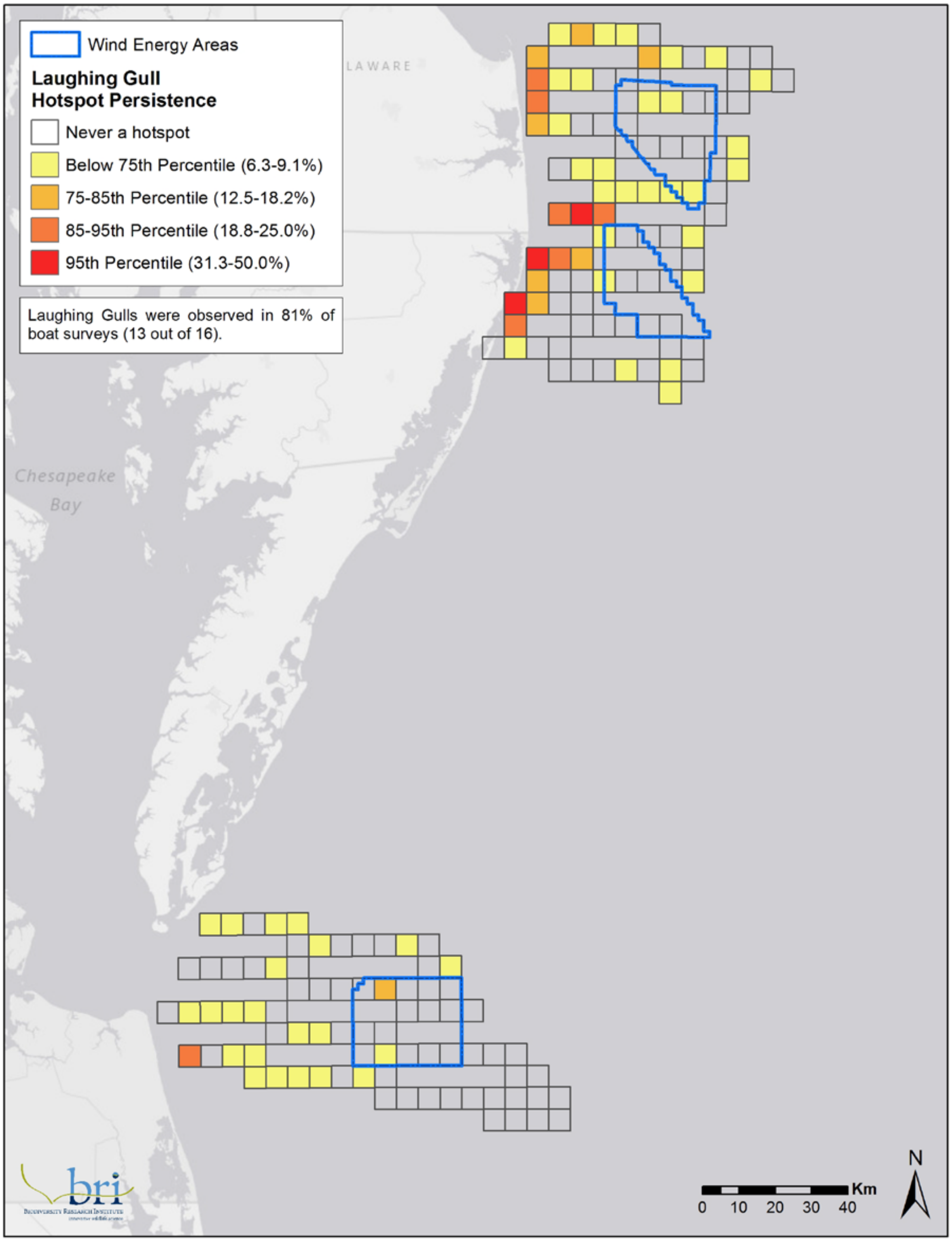

**Figure 17-14. Classified persistent abundance hotspots for Laughing Gulls (*Leucophaeus atricilla*) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot.

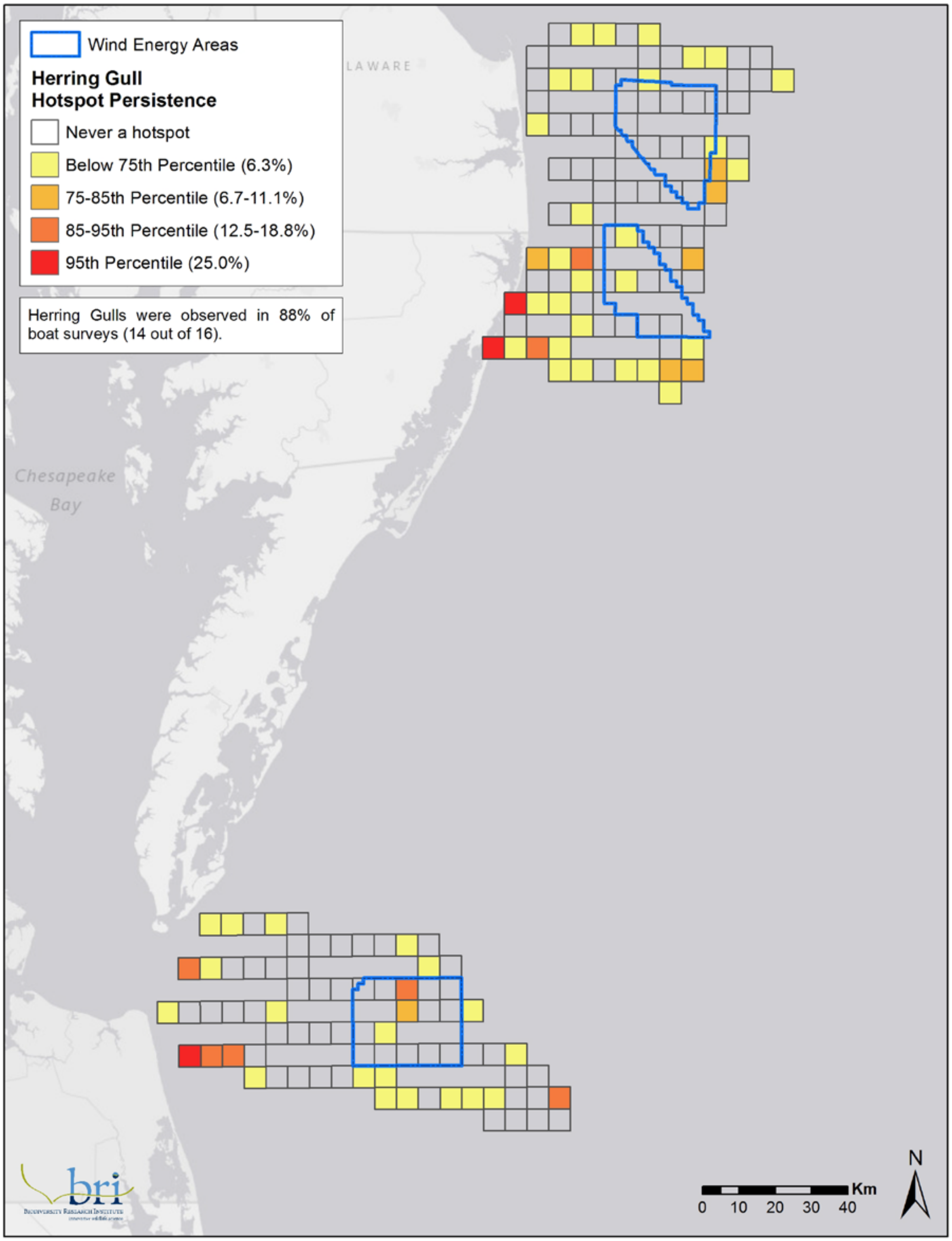

**Figure 17-15. Classified persistent abundance hotspots for Herring Gulls (*Larus argentatus*) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot.

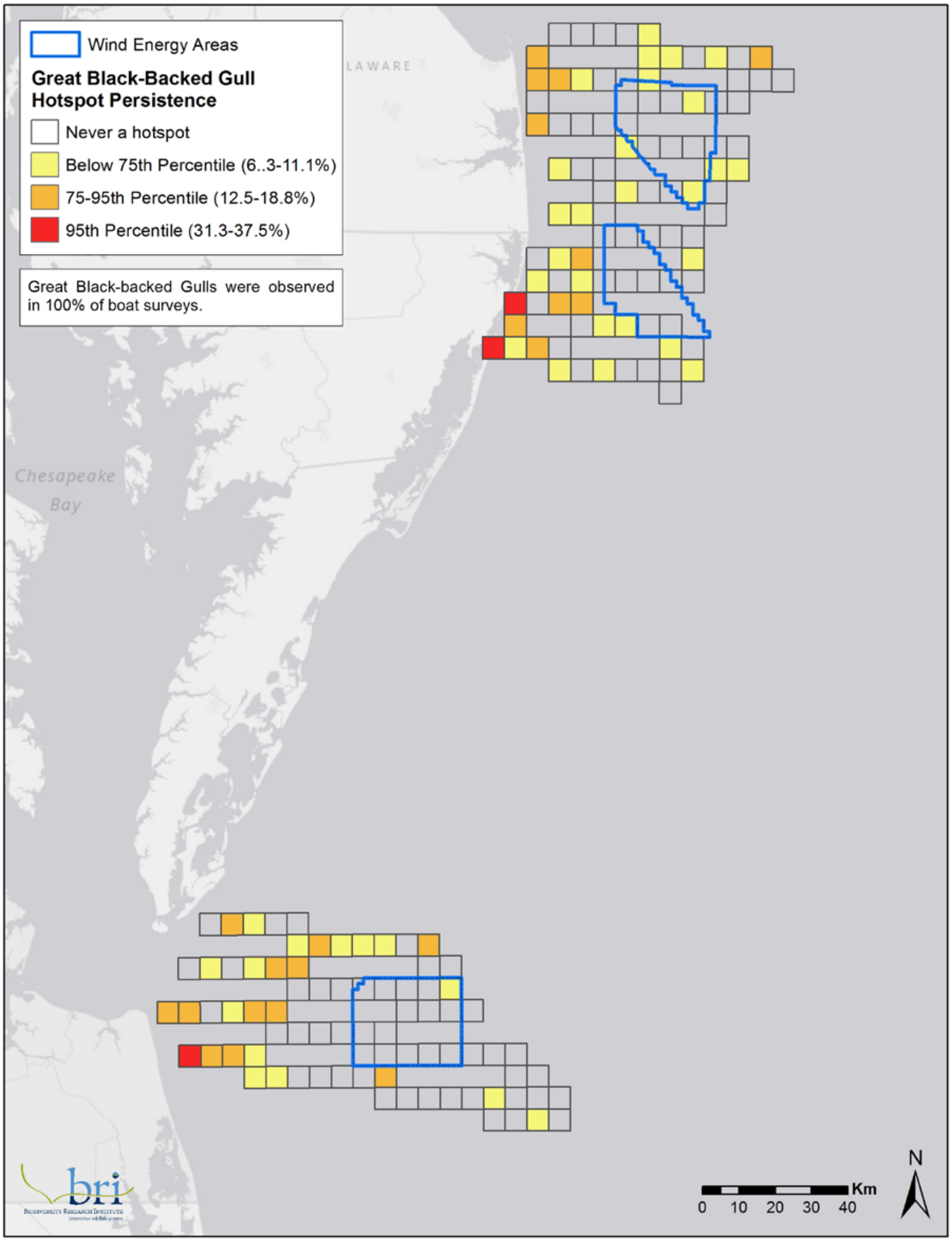

**Figure 17-16. Classified persistent abundance hotspots for Great Black-backed Gulls (*Larus marinus*) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Data are split into only three persistence classes as the 75th and 85th percentile of persistence fell at the same value (12.5%).

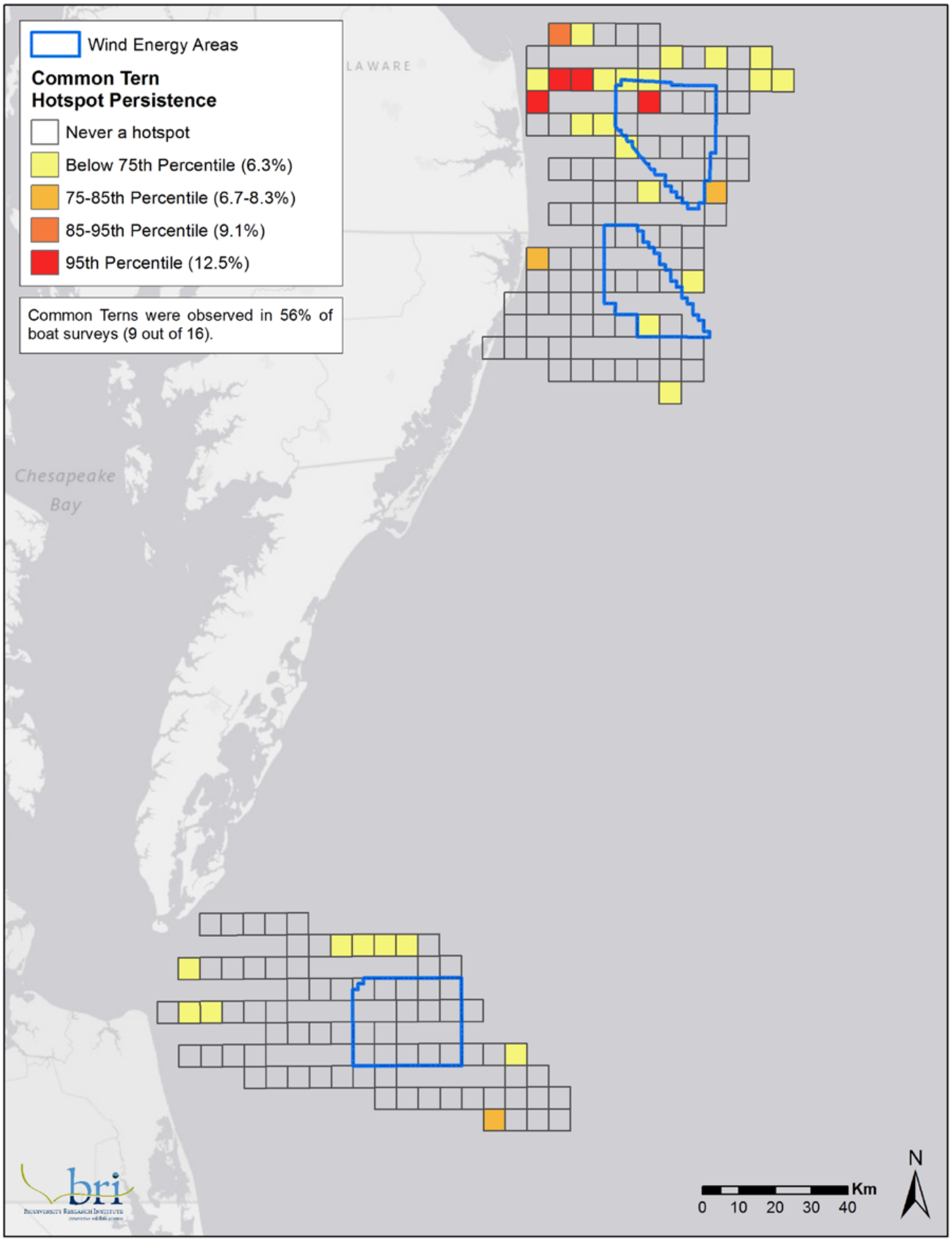

**Figure 17-17. Classified persistent abundance hotspots for Common Terns (*Sterna hirundo*) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot.

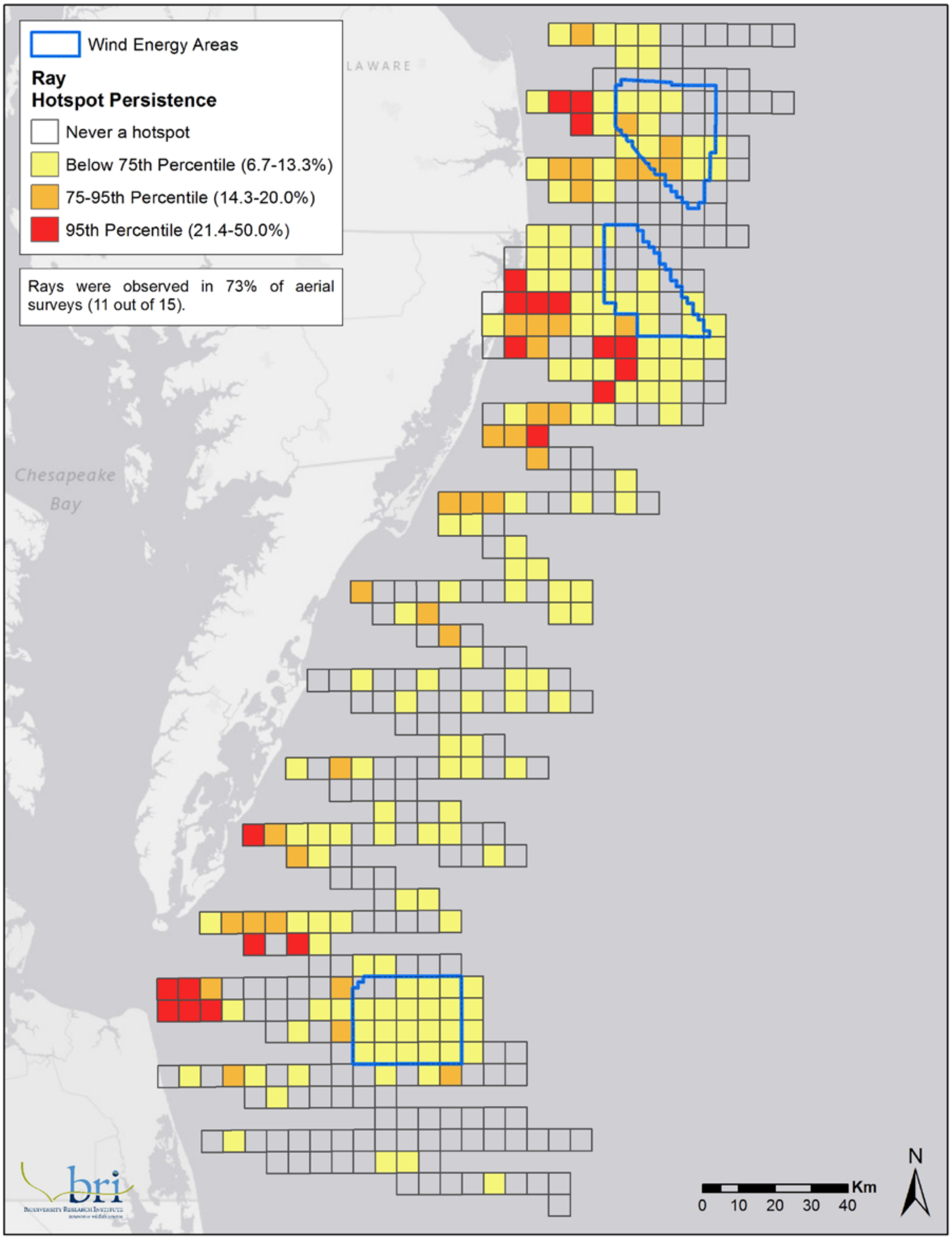

**Figure 17-18. Classified persistent abundance hotspots for rays (Batoidea spp.) observed in video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Data are split into only three persistence classes as the 75th and 85th percentile of persistence fell at the same value (14.3%).

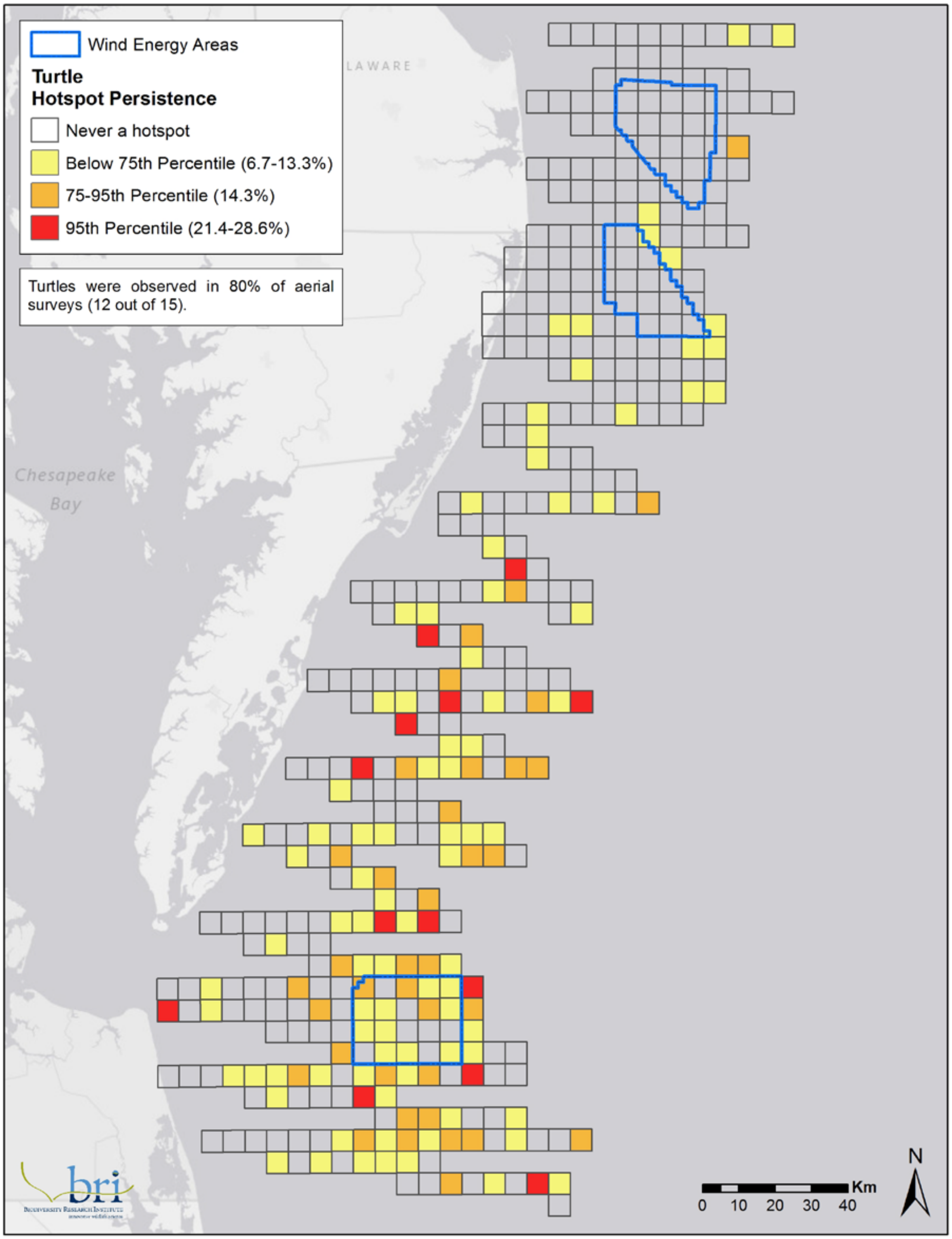

**Figure 17-19. Classified persistent abundance hotspots for turtles (Testudines spp.) observed in video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Data are split into only three persistence classes as the $75^{th}$ and $85^{th}$ percentile of persistence fell at the same value (14.3%).

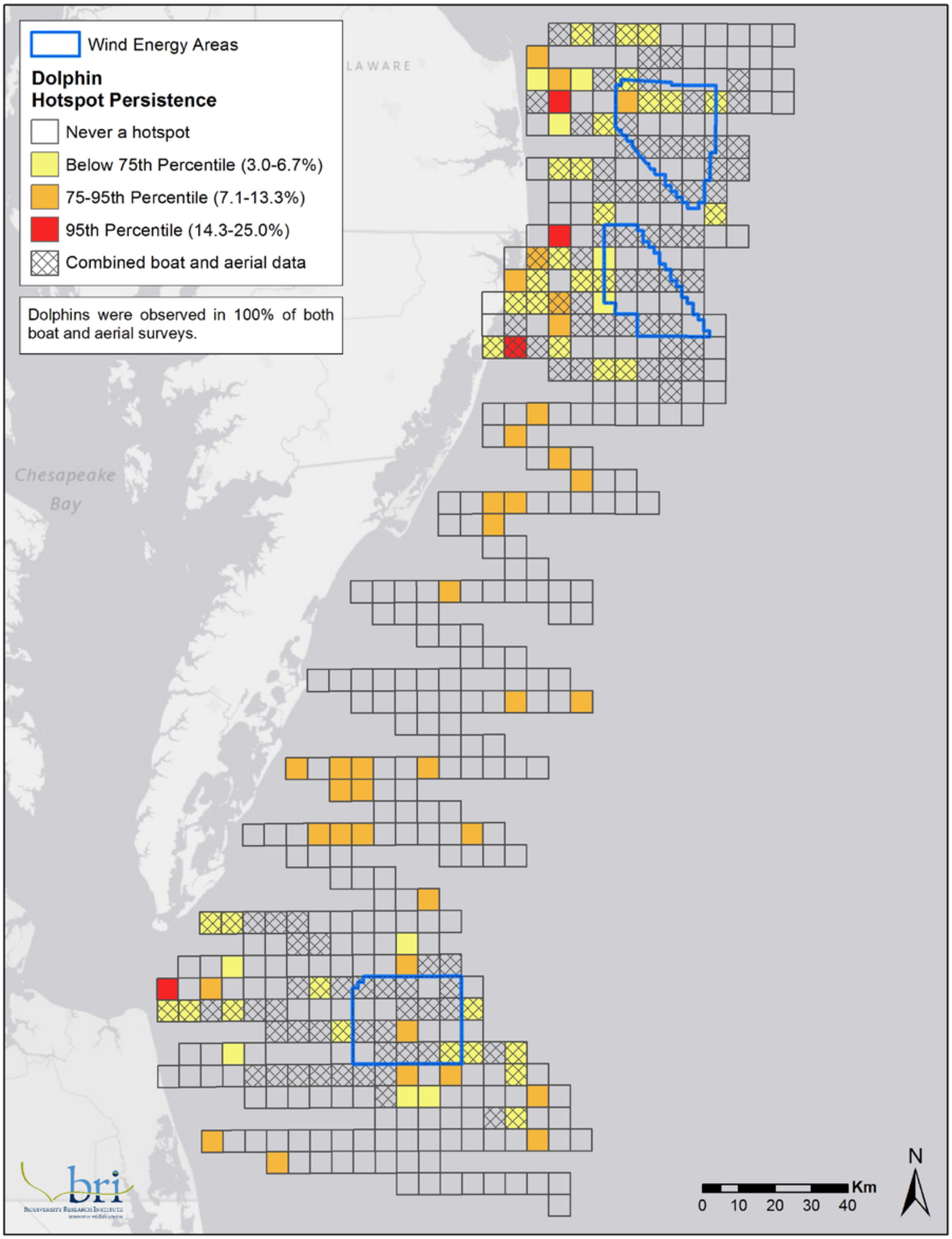

**Figure 17-20. Classified persistent abundance hotspots for dolphin (Odontoceti spp.) observed in boat and video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods. Data is split into only three persistence classes as the 75th and 85th percentile of persistence values fell at the same value (7.1%).

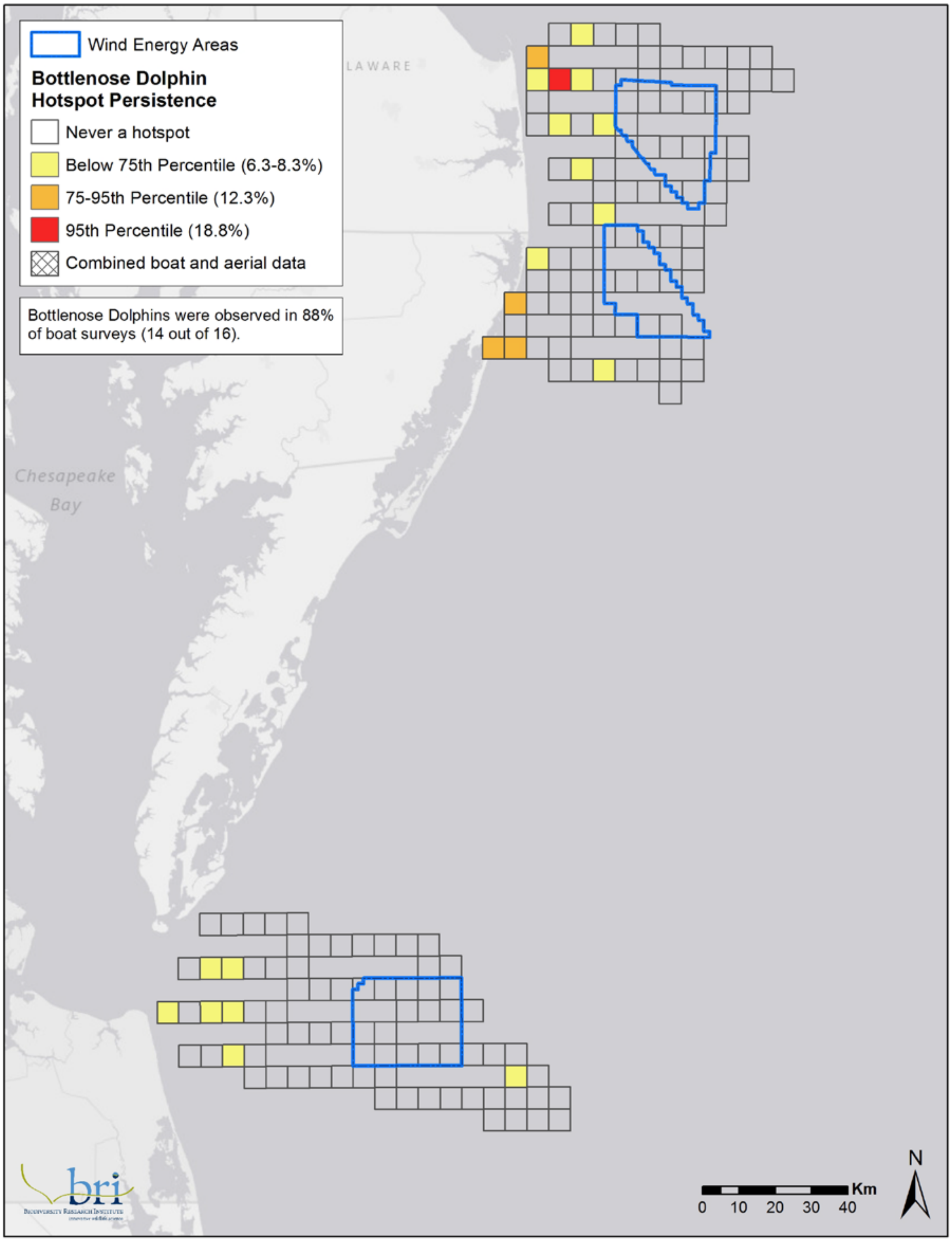

**Figure 17-21. Classified persistent abundance hotspots for Bottlenose Dolphins (*Tursiops truncates*) observed in boat surveys, April 2012 – April 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot. Data is split into only three persistence classes as the 75th and 85th percentile of persistence values fell at the same value (12.3%).

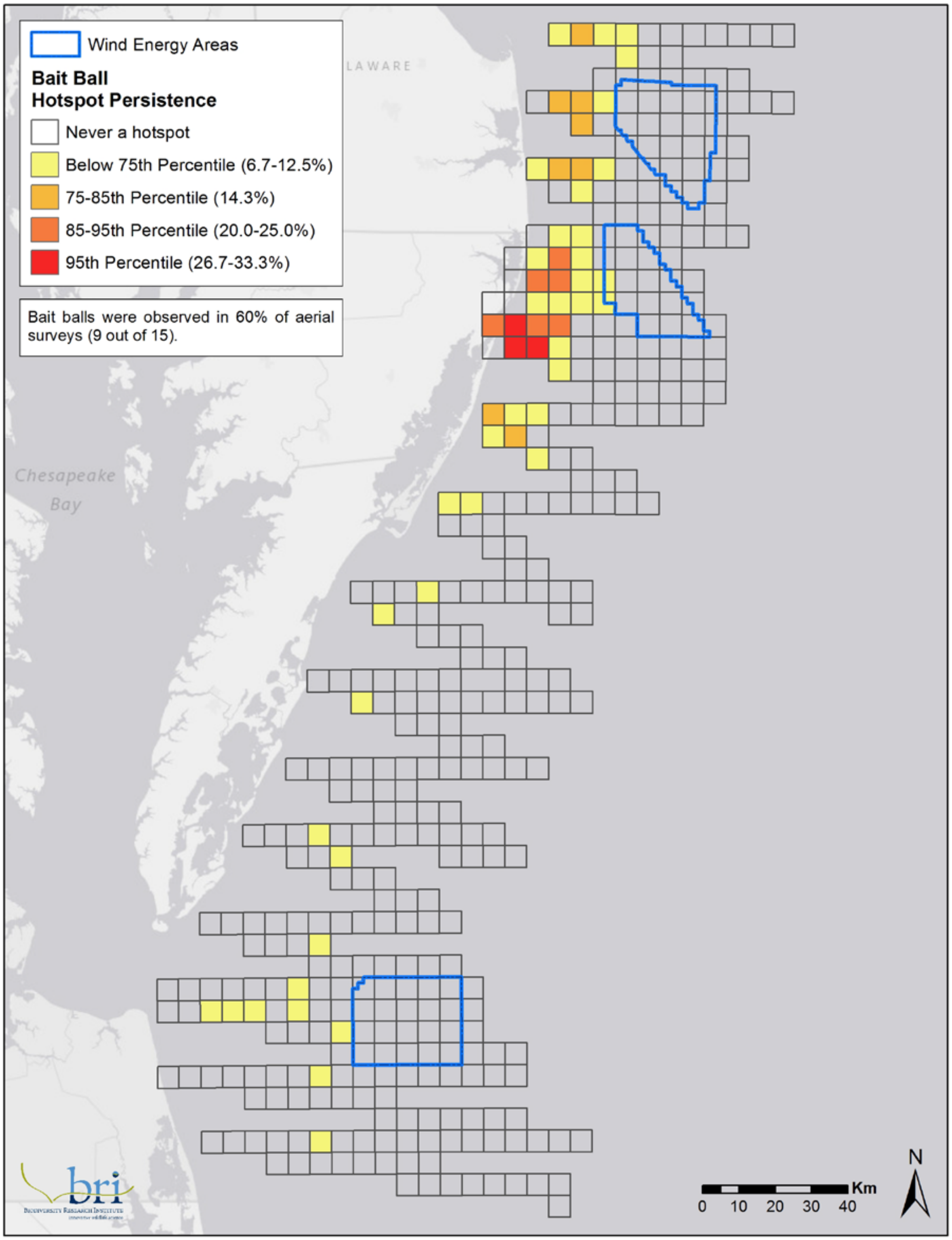

**Figure 17-22. Classified persistent abundance hotspots for bait balls observed in video aerial surveys, March 2012 – May 2014.** For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance to be considered a hotspot.

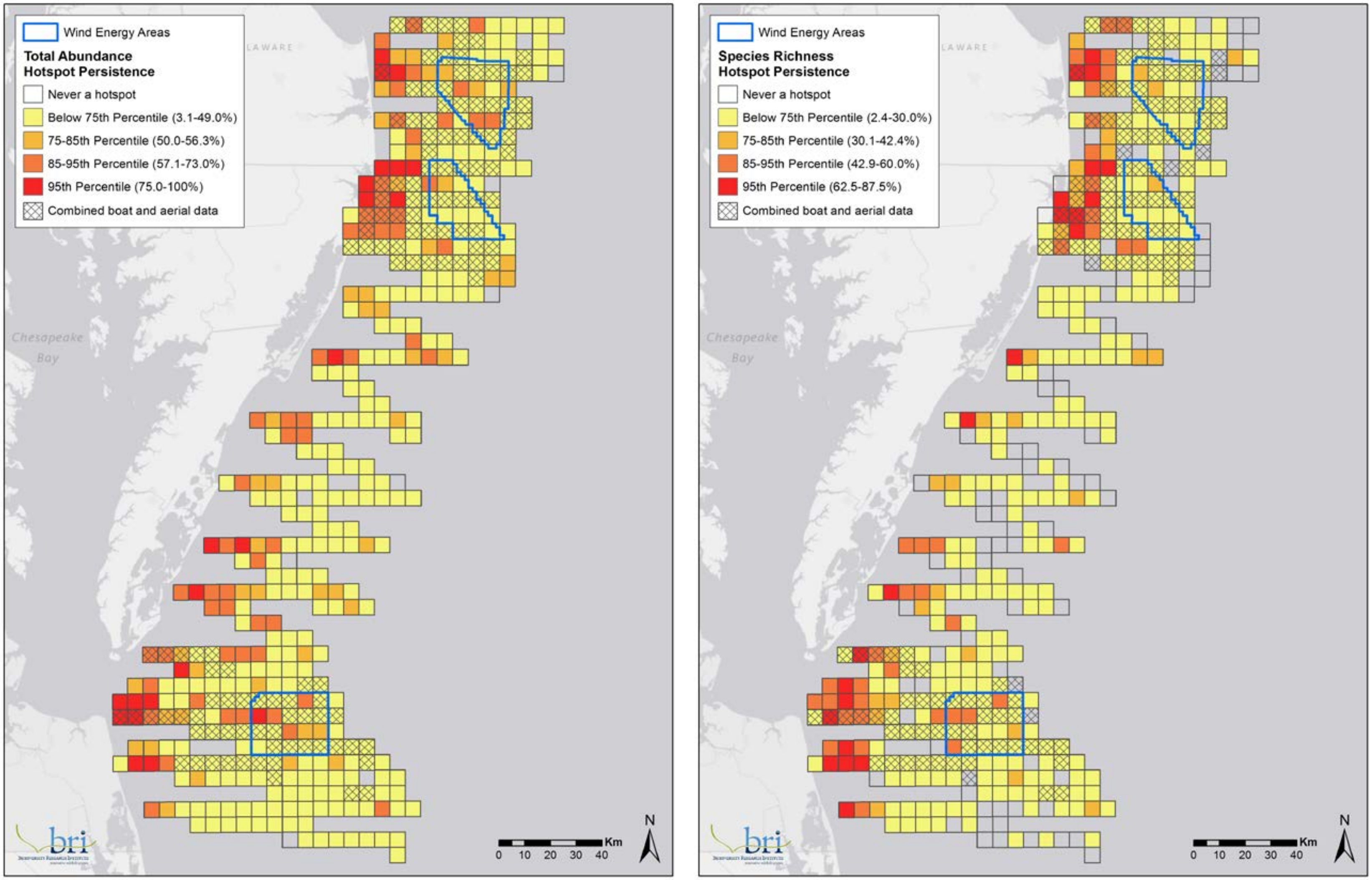

**Figure 17-23. Persistent abundance hotspots identified across all taxa (left) and persistent species richness hotspots (right).** These maps highlight areas where the greatest numbers of individuals across all taxa (left) and the greatest numbers of species (right) were consistently observed over the course of the study. For each percentile category shown in the legend, the corresponding percentage of time a cell was a hotspot (including all surveys) is shown parenthetically. Blank cells never had high enough abundance or high enough species counts to be considered a hotspot. Crosshatched cells integrate data from both boat and video aerial survey methods.

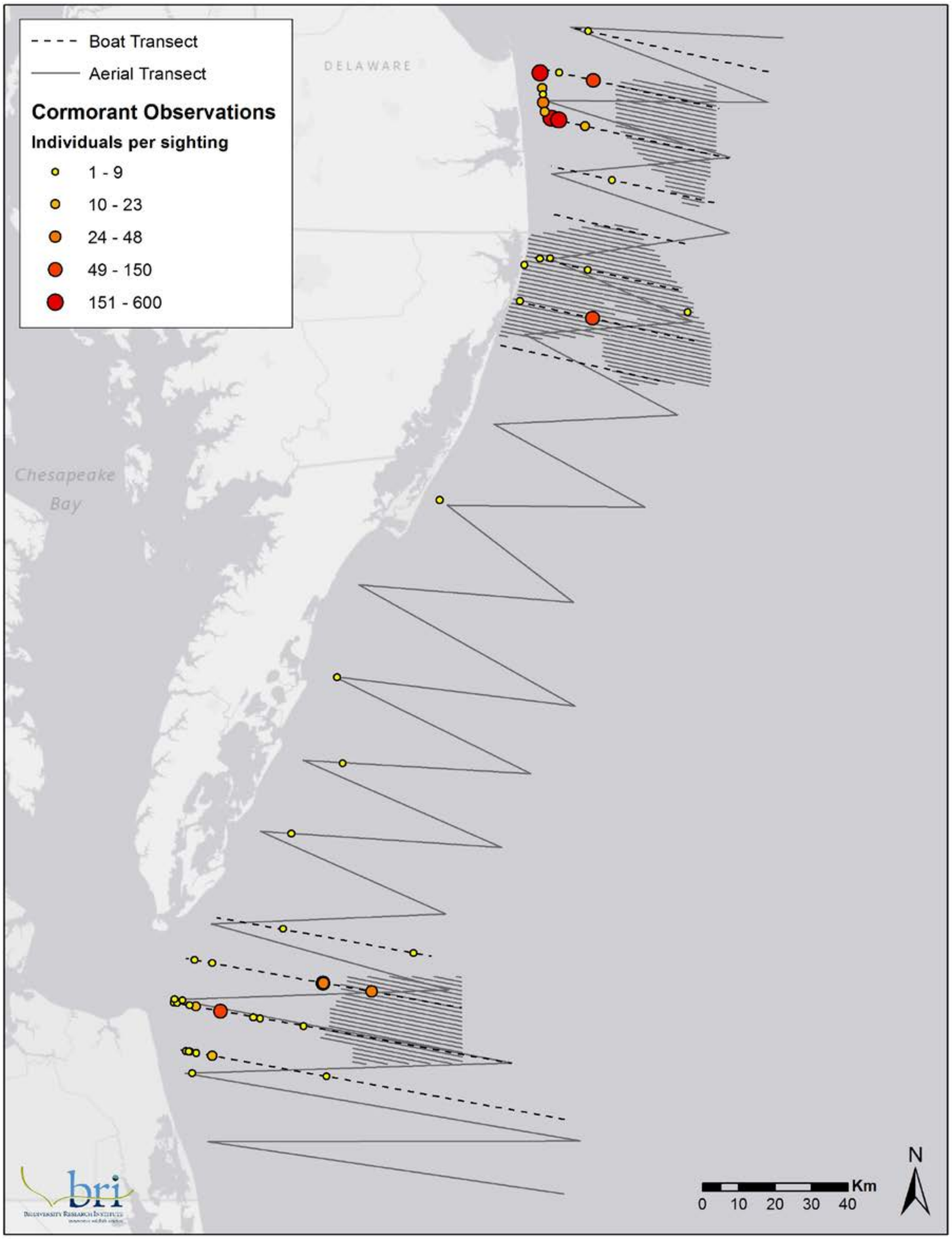

**Figure 17-24. Cormorants (Phalacrocoracidae) observed by boat and video aerial surveys.** A total of 2,077 cormorants were observed (2,035 by boat; 42 by aerial) over the course of the study, March 2012 – May 2014. Over half of these individuals were observed in three sightings by boat surveys at the northern end of the study area.

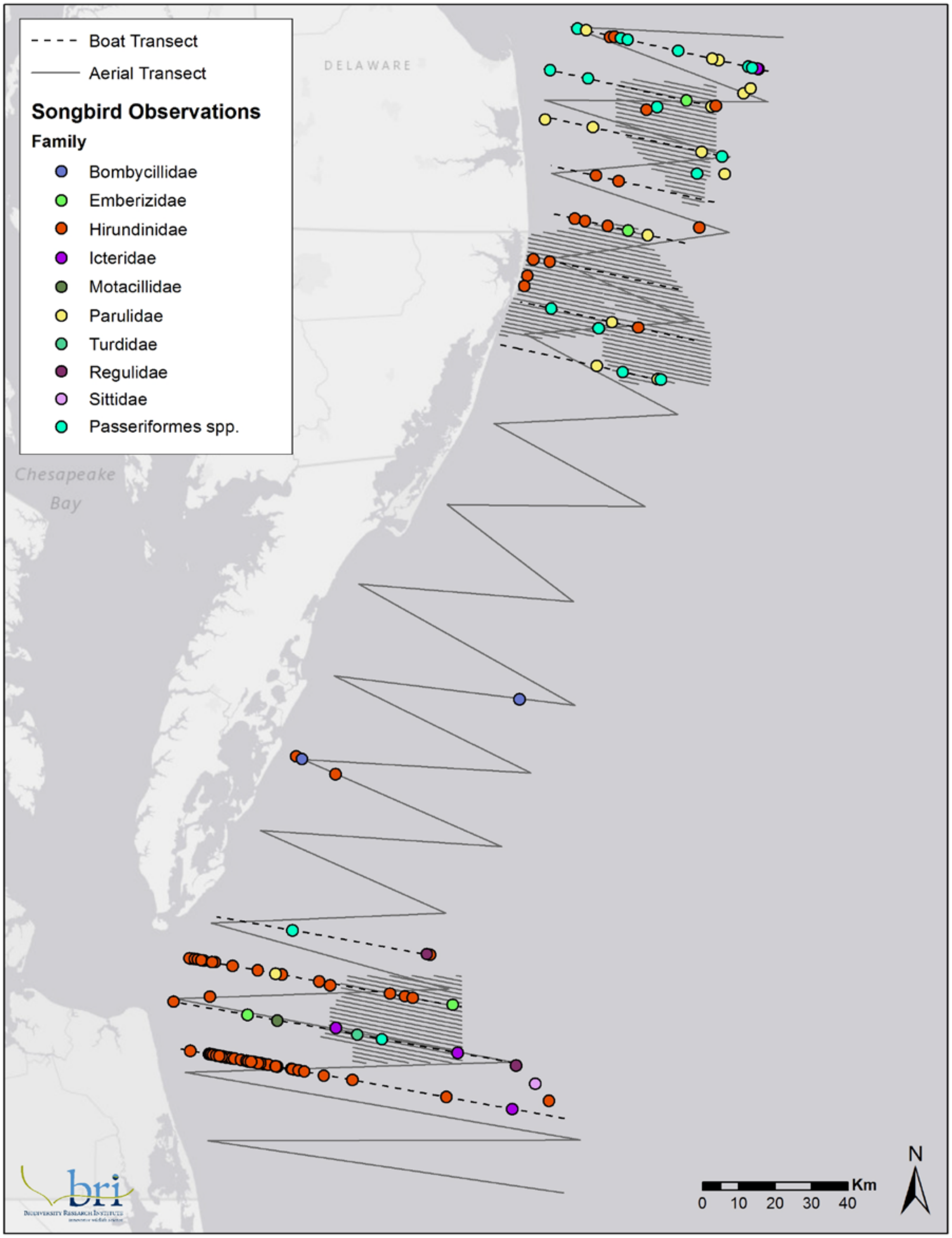

**Figure 17-25. Songbirds observed in the boat and video aerial surveys displayed by family.** Families include waxwings (Bombycilldae), sparrows (Emberizidae), swallows (Hirundinidae), blackbirds and cowbirds (Icteridae), pipits (Motacillidae), warblers (Parulidae), kinglets (Regulidae), nuthatches (Sittidae), robins (Turdidae), and unidentified passerines (Passeriformes).

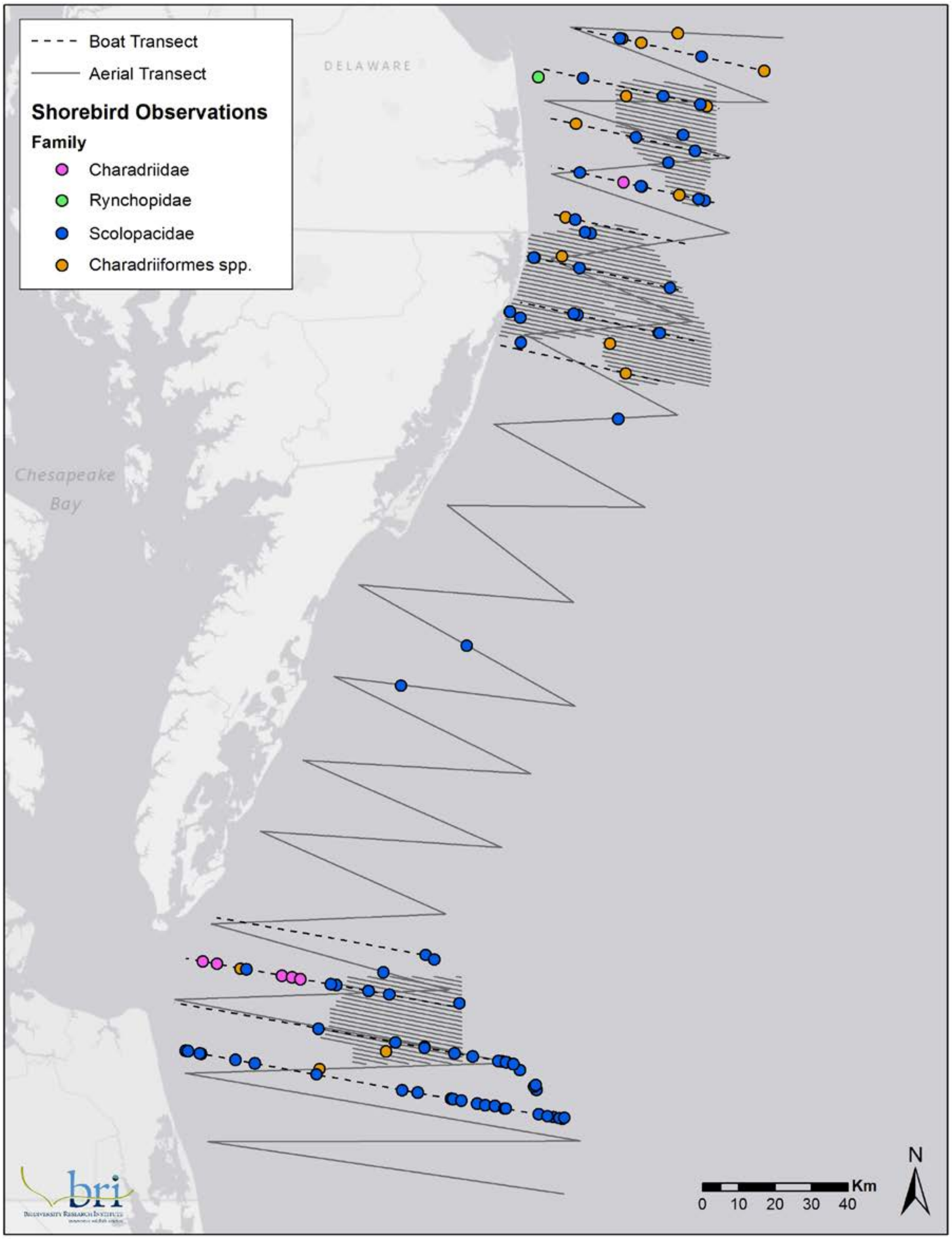

**Figure 17-26. Shorebirds observed in the boat and video aerial surveys, displayed by family.** Families include plovers (Charadriidae); skimmers (Rynochopidae); sandpipers, phalaropes, and other shorebirds (Scolopacidae); and unidentified shorebirds (Charadriiformes).

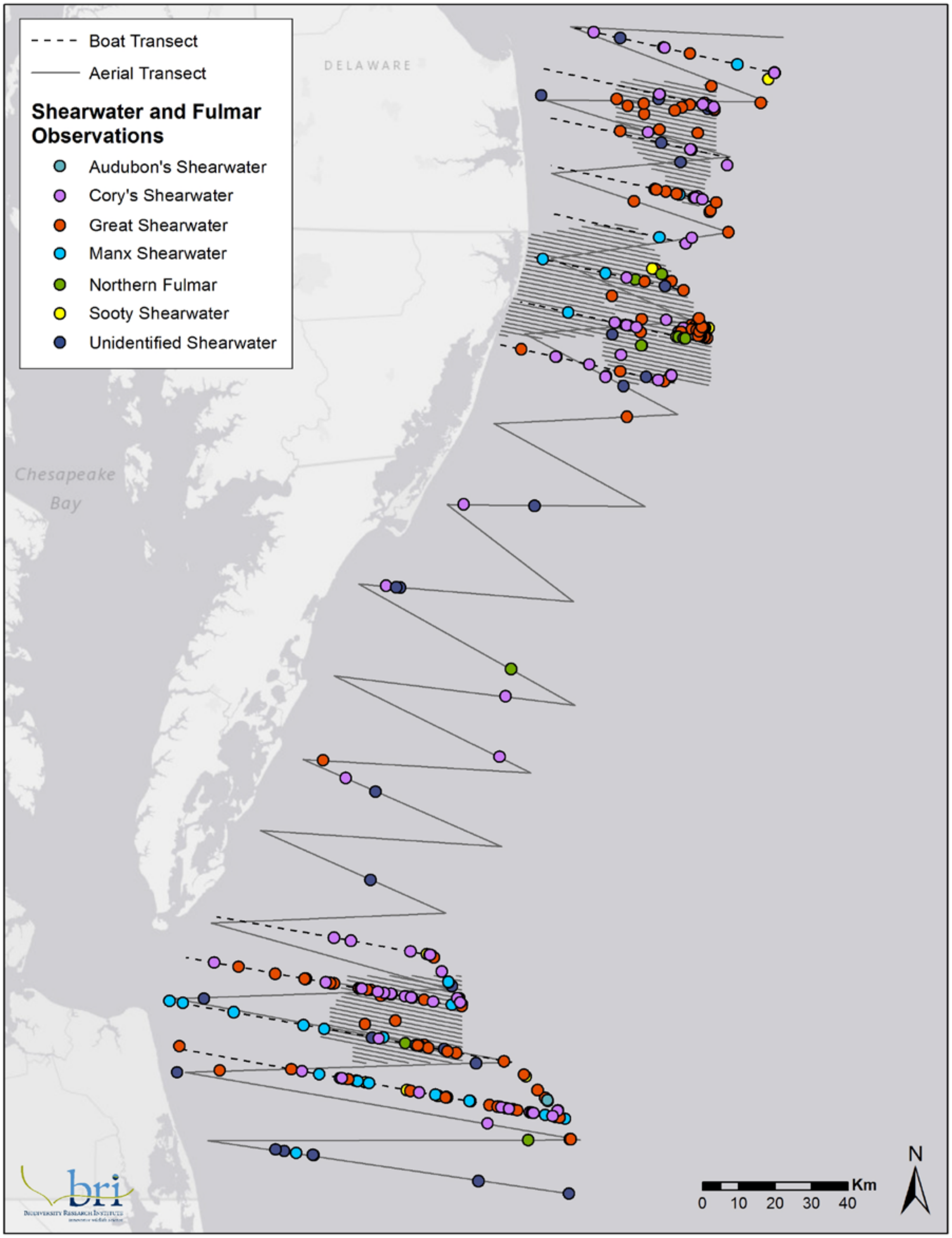

**Figure 17-27. Shearwaters and fulmars (Procellariidae) observed on boat and video aerial surveys (March 2012-May 2014).**

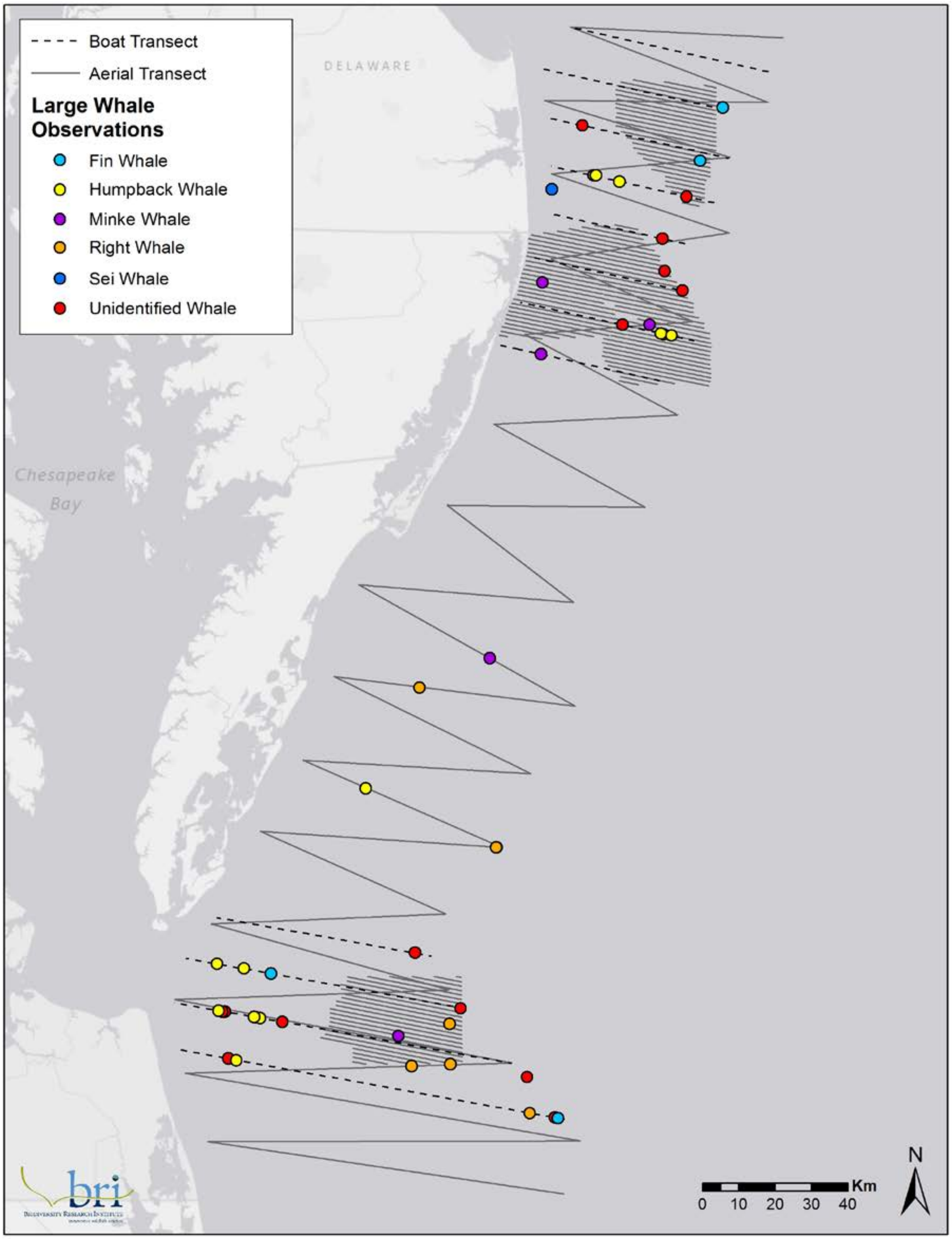

**Figure 17-28. Large whale observations (Mysticeti) from boat and video aerial surveys (March 2012-May 2014).**

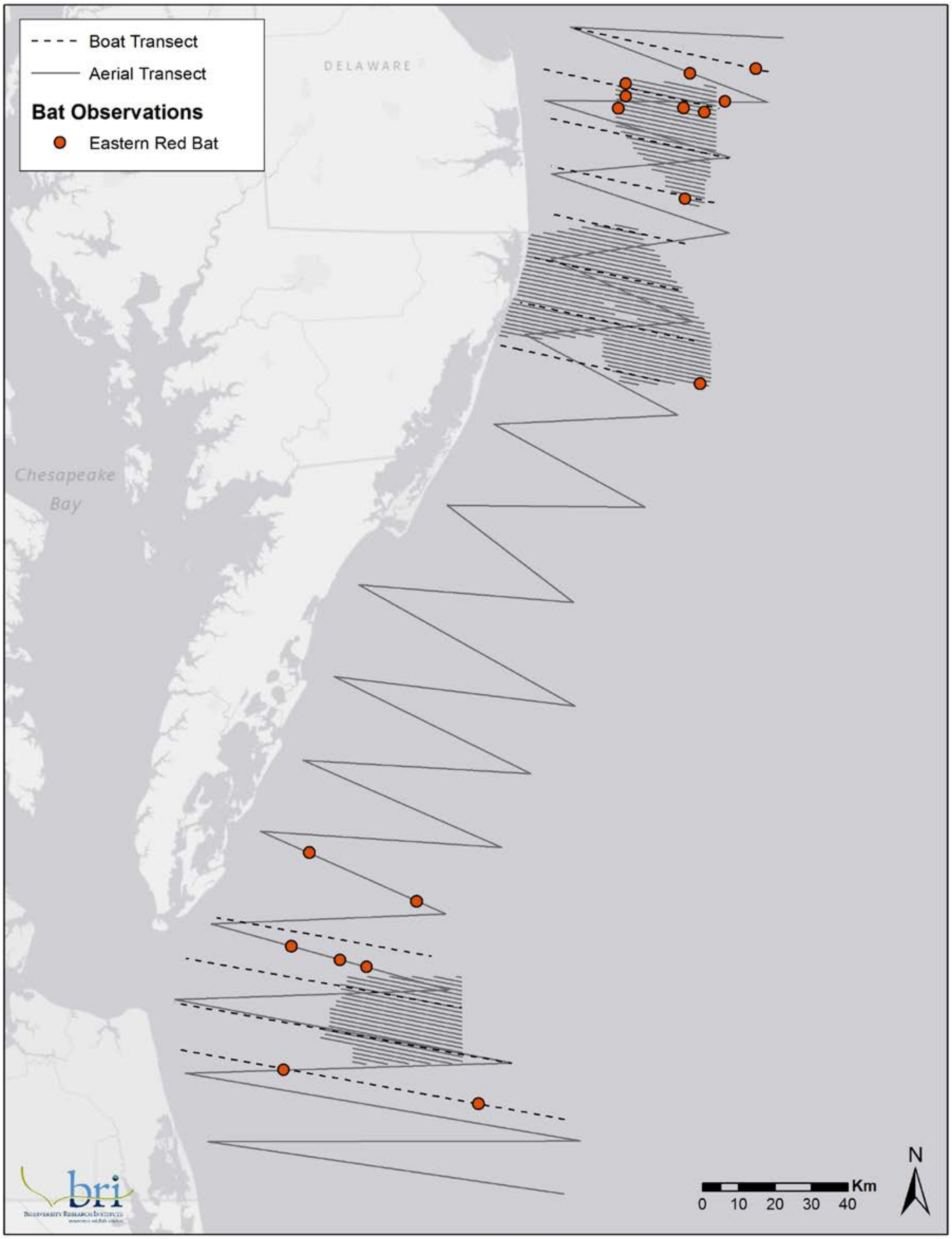

**Figure 17-29. Red bat observations from boat and video aerial surveys (September 2012 and September 2013).**

**Table 17-1. Figure legend for grouped species temporal charts.** Darker and larger bars show time periods when a species or group was more commonly observed in surveys. Effort-corrected counts that correspond with percentile values are shown in kilometers.

| Percentile: | 0 | ≤50% | 50-60% | 60-70% | 70-80% | 80-90% | 90-95% | 95-100% |
|---|---|---|---|---|---|---|---|---|
| Effort-corrected count: | 0 | 0.00319 | 0.0101 | 0.0354 | 0.0997 | 0.371 | 0.892 | 6.661 |
| Method: | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial |

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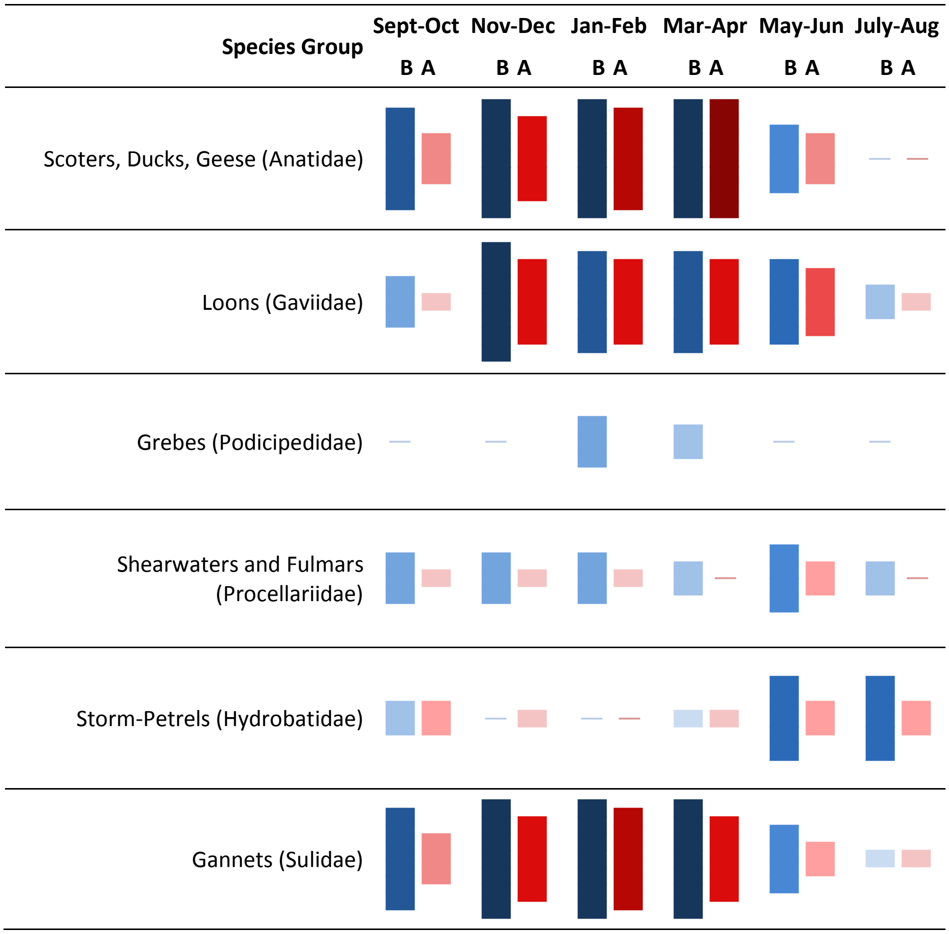
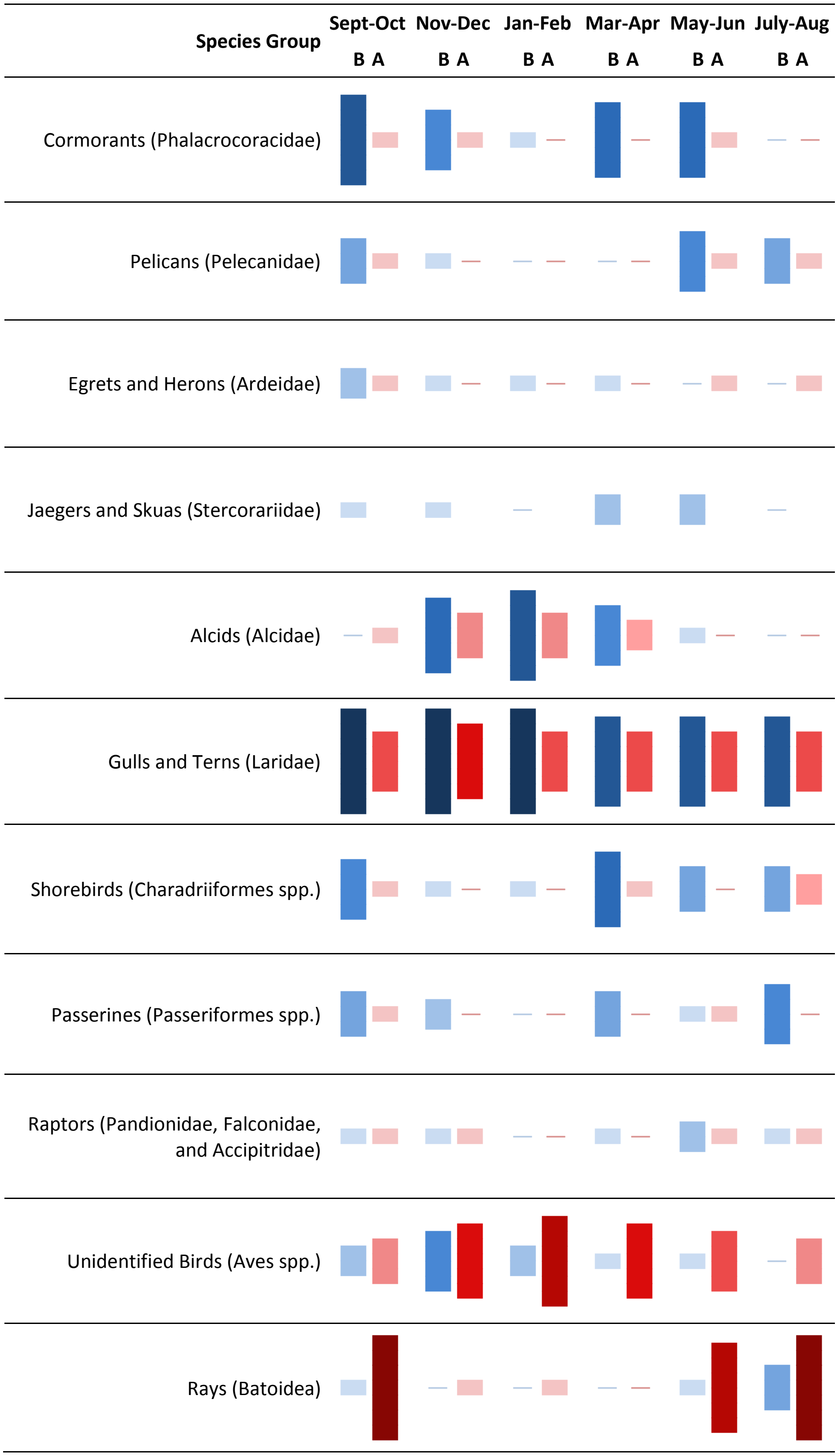
**Table 17-2. Temporal bar charts for all taxonomic groups with more than 10 observations in the boat (B) and video aerial (A) surveys.** When fewer than ten animals were observed in one survey type they were left blank for that survey type (e.g. bats in the boat survey). Avian and non-avian animals are presented in taxonomic order.




Species Group
Sept-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
July-Aug
B A
B A
B A
B A
B A
B A
Cormorants (Phalacrocoracidae)
Pelicans (Pelecanidae)
Egrets and Herons (Ardeidae)
Jaegers and Skuas (Stercorariidae)
Alcids (Alcidae)
Gulls and Terns (Laridae)
Shorebirds (Charadriiformes spp.)
Passerines (Passeriformes spp.)
Raptors (Pandionidae, Falconidae, and Accipitridae)
Unidentified Birds (Aves spp.)
Rays (Batoidea)

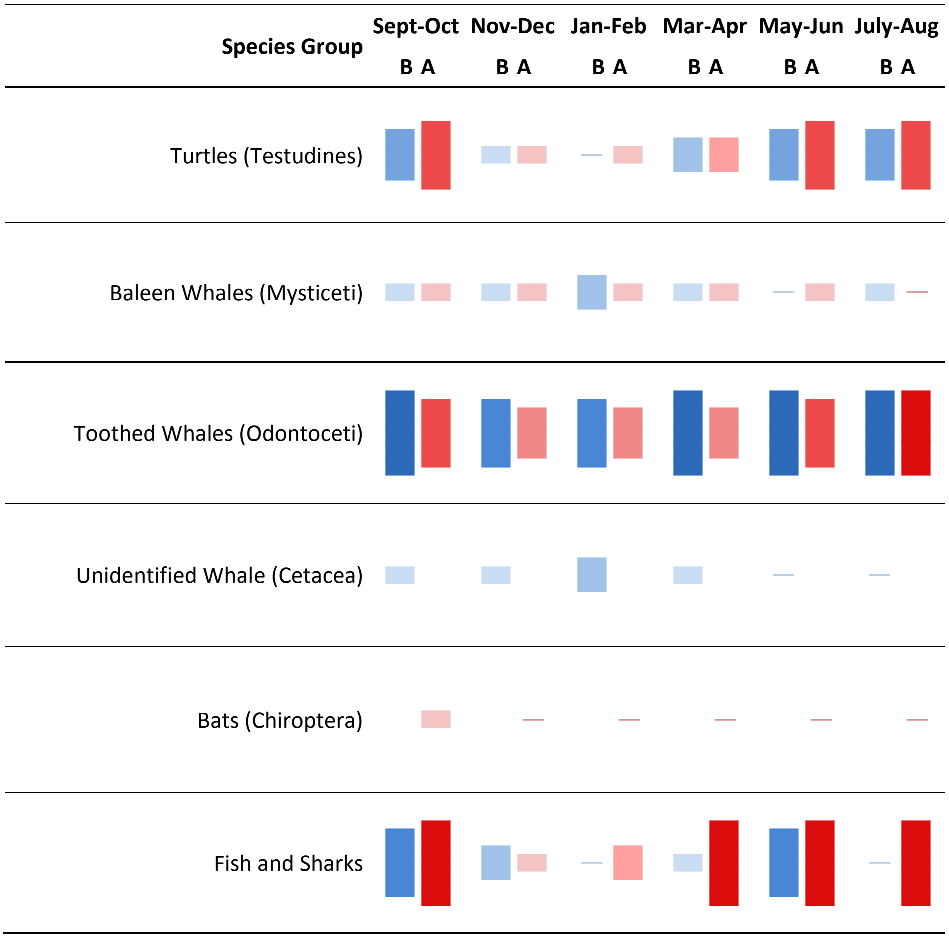
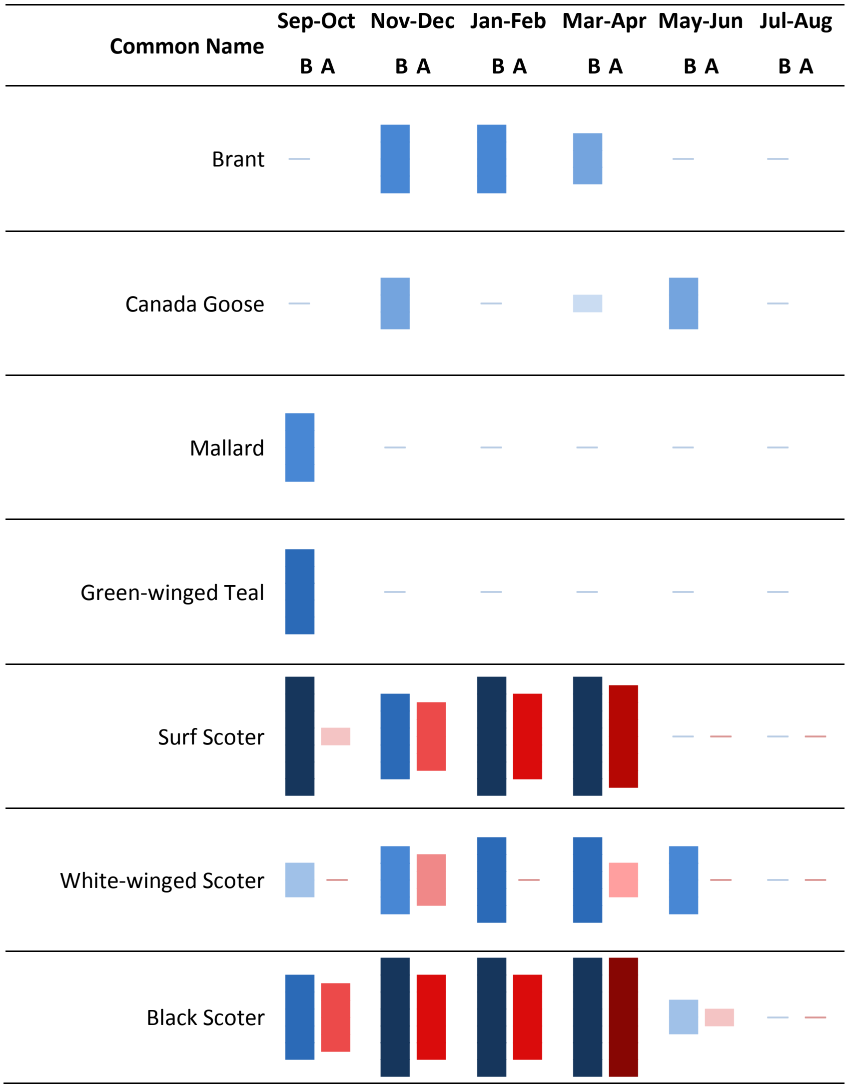
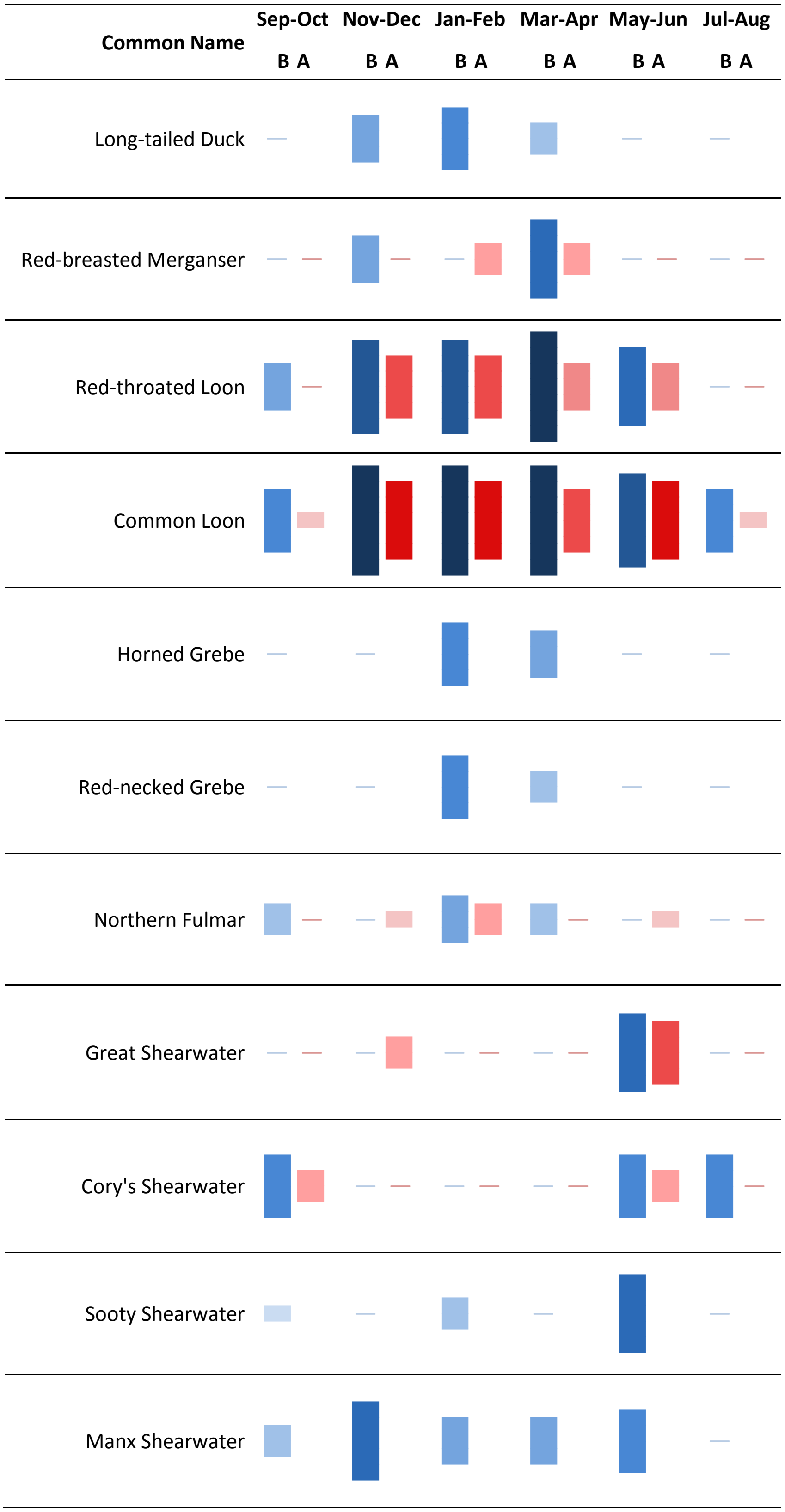

Species Group
Sept-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
July-Aug
B A
B A
B A
B A
B A
B A
Turtles (Testudines)
Baleen Whales (Mysticeti)
Toothed Whales (Odontoceti)
Unidentified Whale (Cetacea)
Bats (Chiroptera)
Fish and Sharks

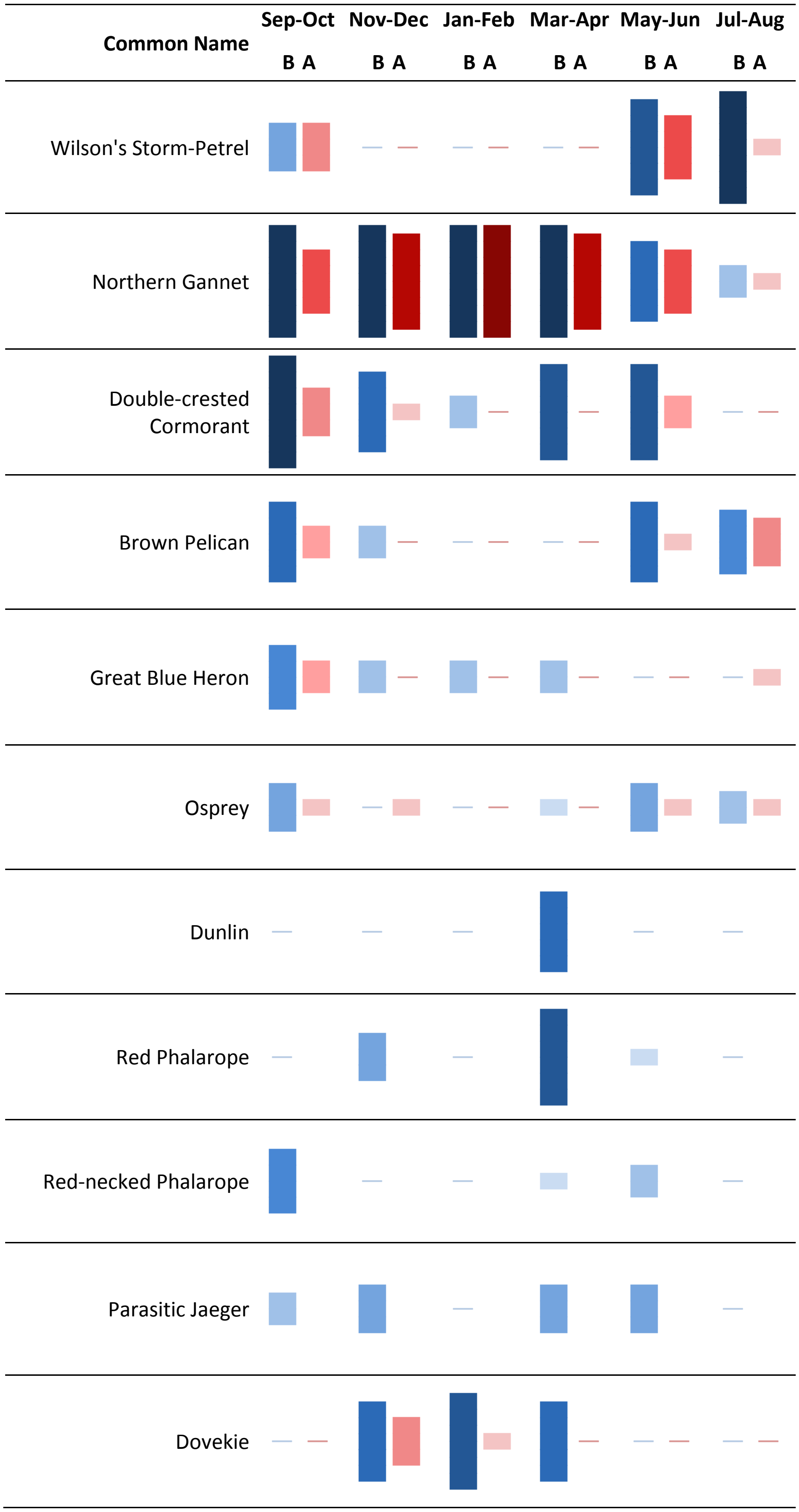
**Table 17-3. Figure legend for individual species temporal charts.** Darker and larger bars show time periods when a species or group was more commonly observed in surveys. Effort-corrected counts that correspond with percentile values are shown in kilometers.

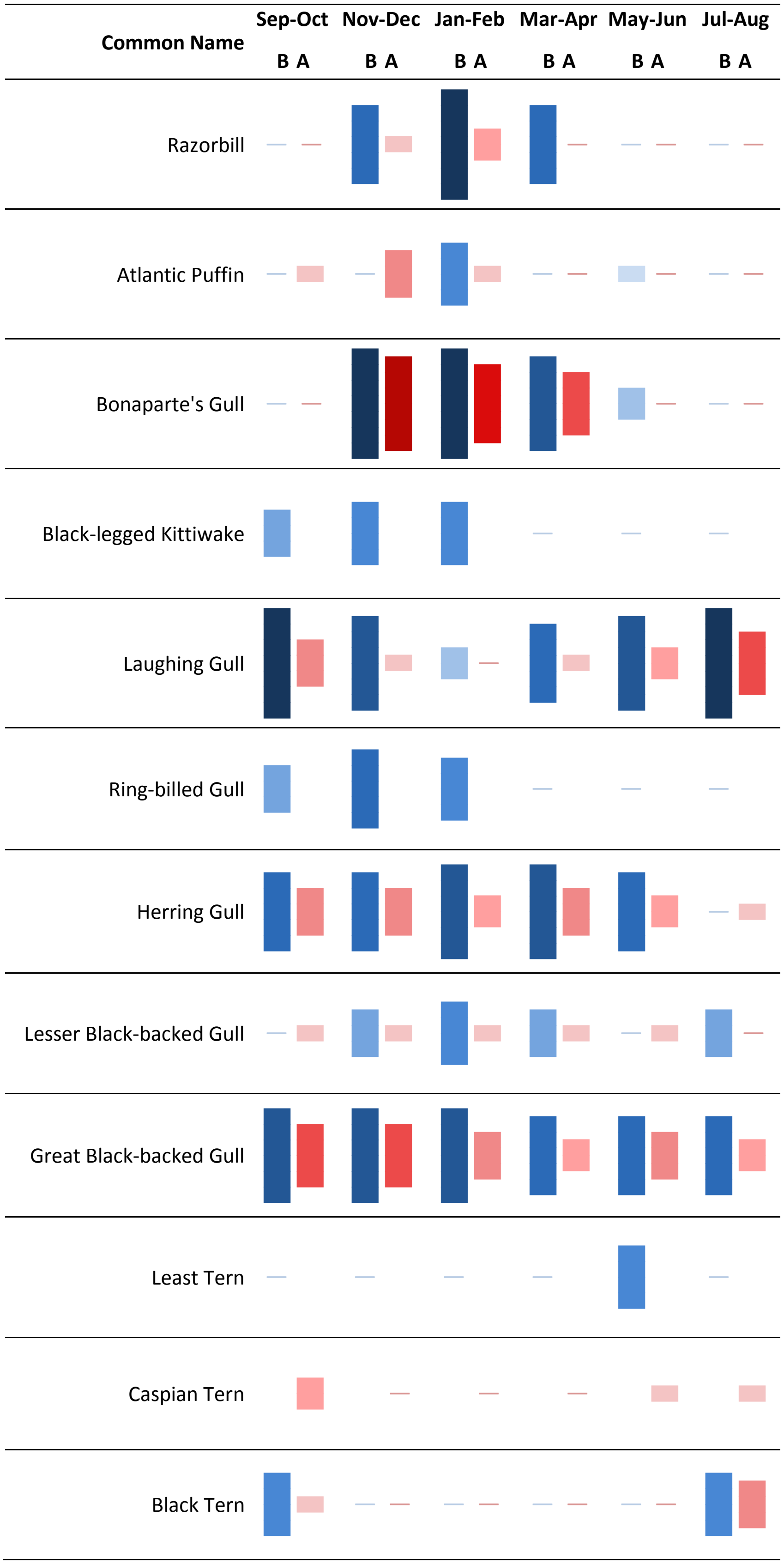
| Percentile: | 0 | ≤50% | 50-60% | 60-70% | 70-80% | 80-90% | 90-95% | 95-100% |
|---|---|---|---|---|---|---|---|---|
| Effort-corrected count: | 0 | 0.000543 | 0.00153 | 0.00490 | 0.0178 | 0.118 | 0.309 | 3.902 |
| Method: | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial | Boat Aerial |

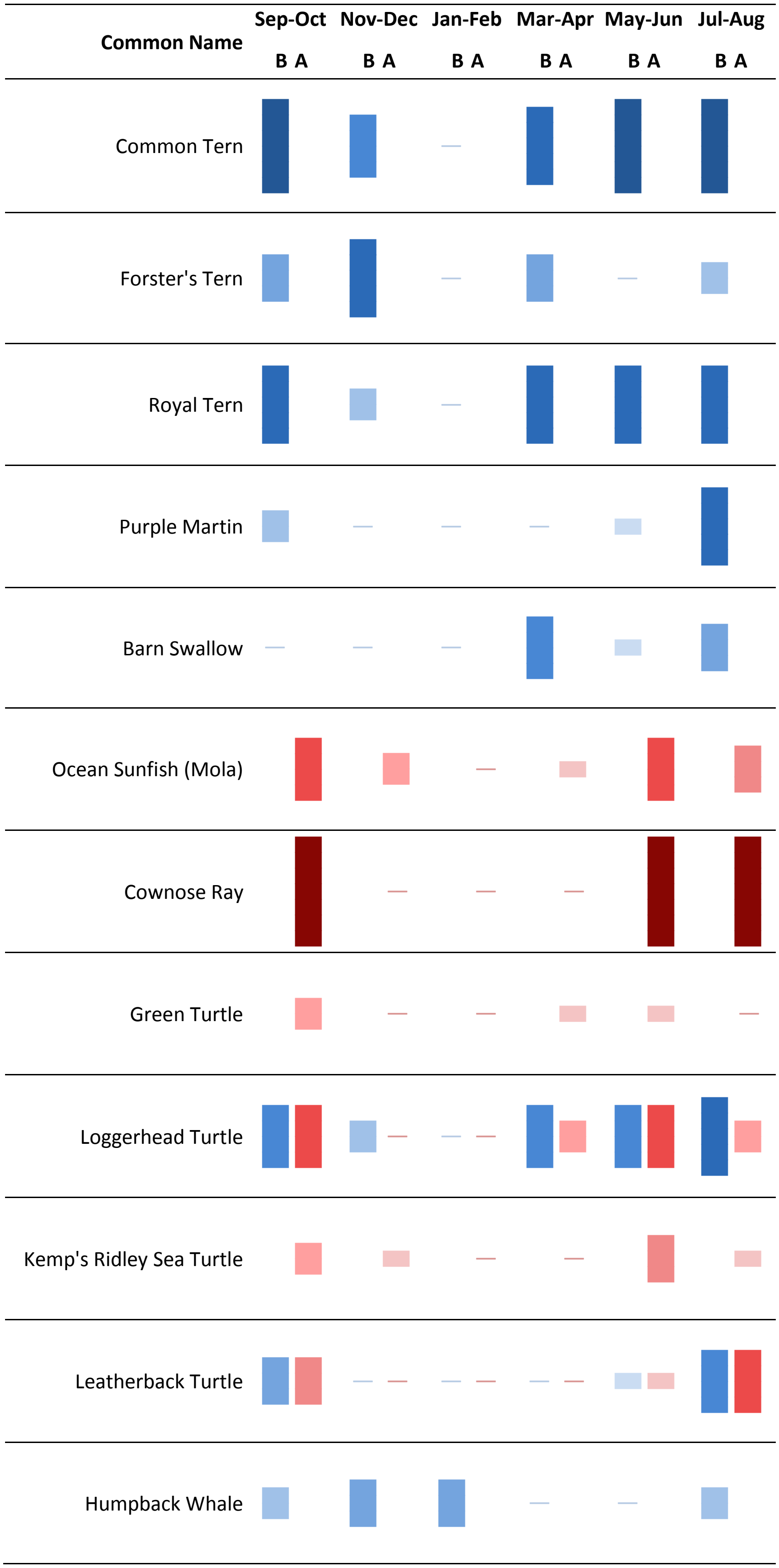
**Table 17-4. Temporal bar charts for all individual species with more than 10 observations in the boat (B) and video aerial (A) surveys.** When fewer than ten animals were observed in one survey type they were not calculated for that survey type (e.g. Brants in the video aerial survey). Avian and non-avian animals are presented in taxonomic order.

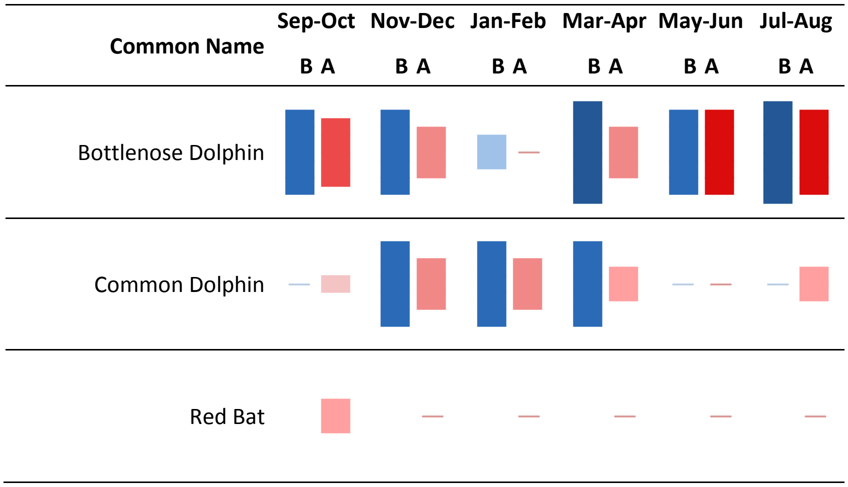



Common Name
Sep-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
Jul-Aug
B A
B A
B A
B A
B A
B A
Long-tailed Duck
Red-breasted Merganser
Red-throated Loon
Common Loon
Horned Grebe
Red-necked Grebe
Northern Fulmar
Great Shearwater
Cory's Shearwater
Sooty Shearwater
Manx Shearwater


Common Name
Sep-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
Jul-Aug
B A
B A
B A
B A
B A
B A
Wilson's Storm-Petrel
Northern Gannet
Double-crested Cormorant
Brown Pelican
Great Blue Heron
Osprey
Dunlin
Red Phalarope
Red-necked Phalarope
Parasitic Jaeger
Dovekie


Common Name
Sep-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
Jul-Aug
B A
B A
B A
B A
B A
B A
Razorbill
Atlantic Puffin
Bonaparte's Gull
Black-legged Kittiwake
Laughing Gull
Ring-billed Gull
Herring Gull
Lesser Black-backed Gull
Great Black-backed Gull
Least Tern
Caspian Tern
Black Tern


Common Name
Sep-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
Jul-Aug
B A
B A
B A
B A
B A
B A
Common Tern
Forster's Tern
Royal Tern
Purple Martin
Barn Swallow
Ocean Sunfish (Mola)
Cownose Ray
Green Turtle
Loggerhead Turtle
Kemp's Ridley Sea Turtle
Leatherback Turtle
Humpback Whale


Common Name
Sep-Oct
Nov-Dec
Jan-Feb
Mar-Apr
May-Jun
Jul-Aug
B A
B A
B A
B A
B A
B A
Bottlenose Dolphin
Common Dolphin
Red Bat

# Chapter 18: Comparison of boat and aerial models of seabird abundance with environmental covariates

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** www.briloon.org/mabs

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## Chapter 18 Highlights

*Comparison of habitat relationships and abundance estimates from boat and digital aerial surveys across the study area*

### *Context*[1]

Identifying the exposure of seabird species to proposed development projects often requires an understanding of how their abundance relates to environmental covariates. When multiple survey approaches are used, we must additionally determine how such sampling methods differ in estimating species' abundance in relation to these covariates. In this chapter, we focus on comparing data between survey methods for the purpose of determining how best to combine boat and digital aerial survey data for analysis. We tried to make the models as similar between survey types and species as possible, to facilitate comparison, which meant sometimes using slightly different formulations of models from other chapters. We analyzed the boat data similarly to Chapters 11-12, but with single species instead of a community. The digital aerial data are modeled similarly to Chapter 15 where generalized linear models were used.

This chapter presents a preliminary analysis of data from four seabird groups (terns, gannets, loons, and alcids) across the seasons when they were present in the study region. Remotely-collected environmental data were incorporated into separate boat and digital aerial models, to compare and contrast the estimated effects of habitat on seabird abundance using data from each sampling method. Chapter 19 builds upon these results and examines an integrated modeling approach for these taxa.

### *Study goal/objectives*

Compare the estimated effect of habitat on the predicted abundance of marine bird species by season for models based on boat and aerial digital videography data.

### *Highlights*

- Distance to shore was generally the most common predictor of abundance across species and surveys.
- Similar habitat relationships were estimated between the two survey types for gannets, terns, and loons; alcids were less consistent between the survey types and years.
- Accounting for imperfect detection in the boat data resulted in higher abundance for the boat-based than the aerial models.

### *Implications*

Boat-based and digital aerial survey data provide comparable estimates of habitat relationships. This suggests that a model that can combine both data types may be the most powerful for understanding seabird distributions, although there are many ways to jointly model the data. Based on these results, caution should be taken for species like alcids, where different patterns were observed between survey types. Such differences may be due to differences in the sampling domain, detectability, or temporal variation.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

This chapter is a preliminary analysis that explores the patterns of seabird abundances observed in the shipboard and digital aerial surveys. Other chapters in this report (Chapters 11-12) focus on analysis of only the boat survey data, but the goal in this chapter is to compare the boat data with the aerial data to determine how best to combine the data types into one joint analysis. These results are not meant to be compared with other chapters that focused on abundance estimates, but instead just to evaluate the patterns and differences between the survey types. As such, this chapter uses slightly different approaches than other chapters, in order make the models as similar between survey types and species as possible and facilitate direct comparisons. The surveys have some spatial and temporal mismatch, which may cause variation in the observations. Additionally, there has been little previous work that jointly models boat surveys with distance-sampling and aerial digital videography surveys, thus demonstrating the need to conduct a preliminary exploration of the two datasets.

Our results indicate that for the species and groups included in this analysis (terns, alcids, loons, and gannets), we generally find that the habitat relationships are consistent between survey types, with distance to shore being the most common significant predictor of abundance. For alcids, we did see a lack of consistency in the patterns, both between years and survey types. We also found that the estimated abundance was generally higher for the boat surveys, likely due to the ability of our models to address imperfect detection in the boat sampling. The findings in this chapter were used to inform the development of a joint model, presented in Chapter 19.

## Introduction

Shipboard and traditional aerial survey methodologies have been compared extensively in their performance at estimating species richness and abundance (for overview, see Camphuysen et al. 2004). Comparisons between shipboard and digital aerial surveys, however, remain sparse given the novelty of high resolution digital videography (Buckland et al. 2012). Digital videography covers a larger geographic area in a faster time frame, but the technology used in this study was limited by a few components: 1) only 200 meters of width was sampled, which is a small snapshot of the marine realm, 2) the angle and resolution of the video restricted most objects to being identified to family or group, as opposed to individual or species, and 3) there is no method to address issues of detection and availability, which likely vary by species, season, weather, or other factors. We evaluate the variation that may arise in digital videography data and identify issues related to inherent detection and identification constraints. We postulate that, were the digital aerial and boat surveys to provide similar parameter and abundance estimates, then both surveys would not need to be conducted simultaneously; however, if there are differences in the datasets, then finding ways that make use of the information in both datasets (a ‘joint model’) will be very informative. Before taking the next step in creating a joint model, we first aim to compare the two methods of sampling by using a suite of species (terns, alcids, loons, and gannets) and examining their habitat relationships across different seasons.

Our objectives include:

1. Compare the habitat parameter estimates from boat and aerial habitat models for various species across different seasons.

2. Based on the results of the habitat modeling, compare the predicted abundance from boat-based and digital videography estimates.

3. Evaluate the strength of association between marine mammals and seabirds, based on survey type (Appendix 18A).

It is important to note that there are methodological differences in sampling from the boat versus digital videography. Some differences are inherent to the two survey methods, such as transect width; the boat surveys sample wider transect widths for most species, and use distance sampling to account for variation in detection. Other differences are specific to the survey design utilized in this study (e.g., boat and aerial transects were located in slightly different geographic areas and occurred at different days and times). To minimize the study-specific sources of variation, we used an offset for area sampled, and compiled data from multiple surveys within each survey year. We expected that boat-based models would estimate higher abundance as a result of accounting for imperfect detection in the sampling.

## Methods

Three wind energy areas (WEAs) are designated in our study area, off the coasts of Delaware (DE), Maryland (MD), and Virginia (VA; Figure 18-1). Field methods for the aerial and boat surveys are explained elsewhere in this report (Chapters 3 and 7, respectively). Aerial identification protocols for video analysis are discussed in Chapter 4. For this comparison, we used boat survey observations that were sampled from the forward quadrant on one side of the vessel, extending up to 1 km from the trackline, and digital aerial observations that were collected from 4 cameras, which each recorded a 50 m band (totaling 0.2 km strip width). For both the boat and aerial surveys, we divided survey transects into 4 km segments ('sites'); this resulted in some shorter segments at the transect ends which is included in our analysis as an offset for 'site' area, the segment length by strip width (1 km for boat, 0.2 km for aerial). The number of individuals for each species was summed by 4 km segment per survey (defined as the time period over which the entire study area was sampled). Many species, including terns, gannets, and loons, are seasonally present in the study area or observed in low numbers, so surveys were combined within each year for analysis but varied depending on the species. We compared gannets, terns, loons, and alcids between the two survey methods; due to slight differences between each taxonomic group, we created group-specific models, described below.

### *Covariates*

We used five covariates in our analyses: three static (distance to shore, slope, and grain size), and two dynamic (sea surface temperature, and salinity). We excluded chlorophyll-*a* in these analyses because it was co-linear with distance to shore in some of the surveys and we wanted to keep the covariates consistent across surveys for the purpose of comparison within each species; due to missing data at higher resolutions it also varied monthly, which is a lower temporal resolution than the other dynamic covariates. Remotely sensed covariate data corresponded to the values located at the midpoint of each transect segment. For the static covariates, we calculated distance to shore (m) within ArcGIS 10.2 (ESRI, Redlands, CA) and extracted slope (% rise, 370-m resolution) and grain size (ϕ = -og2[mean grain diameter in mm], 370-m resolution) from the data layer derived by NOAA/NOS National Centers for Coastal Ocean Science (Kinlan et al. 2013). For the dynamic covariates, we used Marine Geospatial Ecology Tools in ArcGIS (Roberts et al. 2010) to download remotely-sensed data at the highest resolution available for all segments. We compiled daily values for sea surface temperature (SST, °C, 1-km GHRSST L4) and salinity (Practical Salinity Units, 9-km HYCOM GLBa0.08 Equatorial 4D). In the boat survey analysis, we additionally included one covariate on detection: Beaufort sea state on the binary scale, which varied by segment (0 = calm seas, Beaufort state 0-2; 1 = rough seas, Beaufort state 3-6).

### *Models*

To facilitate comparisons, we ran the same model across both the boat and aerial data for each species, except that the boat-based model included an additional component for estimating detection using distance sampling (see Chapter 11). For each species or group, we conducted preliminary diagnostics to evaluate the data and select the best model for abundance, considering the Poisson, Negative Binomial, and zero-inflated versions of both distributions. For the boat-based models, we considered a detection as a single individual, thus breaking down each flock into separate detections of individuals (as opposed to modeling the flock, which we did previously in Chapters 11 and 12), so that we could compare parameters directly with the aerial surveys.

#### *Terns*

Terns included Least Terns (*Sternula antillarum*), Caspian Terns (*Hydroprogne caspia*), Black Terns (*Chlidonias niger*), Common Terns (*Sterna hirundo*), Roseate Terns (*Sterna dougallii*), Royal Terns (*Thalasseus maximus*), and Sandwich Terns (*Thalasseus sandvicensis*), as well as those individuals classified as “unidentified terns.” Vague identifications that could have included other species such as gulls (e.g., “large tern or small gull,”) were excluded. Terns were primarily present in the study area during spring, summer and fall (Chapters 5, 8, 12), so we compared three boat and two aerial surveys from Jun 2012 – September 2012 (first year), and Jun 2013 – September 2013 (second year; this did not include the August aerial surveys, as the August 2013 aerial survey covered only the MD WEA and surrounding areas). For the tern models we used a Negative Binomial distribution on abundance and a Negative Exponential distribution on detection (only in the boat survey models). The abundance component of the model for both boat and aerial surveys was constructed such that each count of terns at segment $i$, $\mathrm{y_i}$, was modeled as:

$$\mathrm{y_i} \sim \mathrm{NegBin}(\lambda_i,\ r)$$

$$\log(\lambda_i) = \beta_0 + offset\ (site\ area) + \beta_1 \mathrm{Dst_i} + \beta_2 \mathrm{Slp_i} + \beta_3 \mathrm{Grn_i} + \beta_4 \mathrm{Sst_i} + \beta_5 \mathrm{Sal_i}$$

where Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and $r$ is the overdispersion parameter.

#### *Gannets*

The only gannets in the area are Northern Gannet (*Morus bassanus*), thus all unidentified gannets were considered Northern Gannets. Gannets were primarily present in the study area in late fall to early spring (Chapters 5, 8, 12), so we compared three boat and three aerial surveys from October 2012 –

February 2013 (first year), and October 2013 – February 2014 (second year). For these gannet models we used a Negative Binomial distribution on abundance, and a Half Normal distribution on detection (only in the boat survey models). The abundance component of the model for both boat and aerial surveys was constructed such that each count of gannets at segment $i$, $\mathrm{y_i}$, was defined:

$$\mathrm{y_i} \sim \mathrm{NegBin}(\lambda_i,\, r)$$

$$\log(\lambda_i) = \beta_0 \; + offset\ (site\ area) + \beta_1 \mathrm{Dst_i} + \beta_2 \mathrm{Slp_i} + \beta_3 \mathrm{Grn_i} + \beta_4 \mathrm{Sst_i}$$

where Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, and Sst = sea surface temperature. We removed salinity in these models because it was highly co-linear with SST and distance to shore.

*Loons*

We considered loons by species (Common Loons, *Gavia immer*, and Red-throated Loons, *G. stellata*) and as a group (all loons, which included both species and all unidentified loon observations), to examine whether habitat relationships varied by species (Chapter 16). Loons were primarily present in the study area from late fall to early spring (Chapters 5, 8, 12), so we included three boat and three aerial surveys from December 2012 – March 2013 (first year), and December 2013 – May 2014 (second year). For all of the loon models, we used a Negative Binomial distribution on abundance, and a Half Normal distribution on detection (only in the boat survey models). The abundance component of the model for both boat and aerial surveys was constructed such that each count of loons at segment $i$, $\mathrm{y_i}$, was defined:

$$\mathrm{y_i} \sim \mathrm{NegBin}(\lambda_i,\, r)$$

$$\log(\lambda_i) = \beta_{0,survey} + offset\ (site\ area) + \beta_1 \mathrm{Dst_i} + \beta_2 \mathrm{Slp_i} + \beta_3 \mathrm{Grn_i} + \beta_4 \mathrm{Sst_i} + \beta_5 \mathrm{Sal_i}$$

where Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and a survey specific intercept to address interannual variation in the survey counts.

*Alcids*

The alcid group included Razorbills (*Alca torda*), Dovekies (*Alle alle*), Atlantic Puffins (*Fratercula arctica*), Common Murres (*Uria aalge*), Thick-billed Murres (*U. lomvia*), and Black Guillemots (*Cepphus grille*), as well as those individuals classified as “unidentified alcids.” Alcids were primarily present in the study area during winter (Chapters 5, 8, 12), therefore we compared two boat and two aerial surveys from December 2012 – February 2013 (first year), and December 2013 – February 2014 (second year). For all alcid models we used a Negative Binomial distribution on abundance, and a Half Normal distribution on detection (only in the boat survey models). To model abundance for both boat and aerial surveys we defined the counts of alcids at segment $i$, $\mathrm{y_i}$, such that:

$$\mathrm{y_i} \sim \mathrm{NegBin}(\lambda_i,\, r)$$

$$\log(\lambda_i) = \beta_0 \; + offset\ (site\ area) + \beta_1 \mathrm{Dst_i} + \beta_2 \mathrm{Slp_i} + \beta_3 \mathrm{Grn_i} + \beta_4 \mathrm{Sst_i} + \beta_5 \mathrm{Sal_i}$$

where Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, and Sal = salinity. In the second year of the aerial surveys, we had to set the overdispersion parameter $r = 0.02$ in order to achieve convergence in the model. There were only 45 transects with observed alcids during this year, which may indicate that our model is overparameterized with five covariates and an intercept. This requires further attention and investigation.

### *Implementation*

We implemented all models in a Bayesian framework using the package “rjags” to run the software JAGS (Plummer 2003) in program R version 2.15.3 (R Development Core Team 2014). We standardized the covariates for analysis to center them on a mean = 0, with a variance close to 1. We initialized three parallel Markov chains at different values and ran them for 30,000 iterations (boat models) or 10,000 (aerial models) following a burn-in of 1,000 iterations. We checked for chain convergence visually (posterior density and trace plots), and quantitatively using the Gelman-Rubin statistic (Gelman et al. 2014). This statistic (termed R-hat) is a measure of among-chain versus between-chain variance and values < 1.1 indicate convergence (Gelman et al. 2014). We also assessed goodness of fit by computing Bayesian p-values. We used Freeman-Tukey fit statistics to evaluate the model for abundance, and to select the Negative Exponential or Half Normal detection function (Gelman et al. 2014). Fitting the models resulted in estimated abundance to the sampled transects, summed across segments and surveys. Using the posterior means of each model parameter, we additionally predicted the abundance of each wintering species to habitat covariates from a representative day (25 Dec 2012), which covered unsampled locations in the three WEAs proposed for our study area.

## Results

Overall, we found that fewer individuals were observed on the aerial surveys than the boat surveys for smaller species (e.g., terns, alcids), and the observations varied by survey date, year, and species (Table 18-1). Accounting for detection resulted in higher abundance in the boat than the aerial surveys, which carried through to the predicted number of birds in each of the WEAs. In Year 1, the estimated number of gannets was very similar for the boat and aerial surveys, with the Virginia WEA having a lower predicted number of birds from the boat survey than the aerial (Table 18-1). Similarly, in Year 2, the number of predicted alcids in all the WEAs from the aerial survey was near 40 birds, while it was near zero birds from the boat survey, though 127 alcids were predicted to the entire study region (Table 18-1). These two cases are the only situations where the boat surveys did not predict higher abundance of birds than the aerial survey, and are likely due to the strong effect of proximity to shore reducing the numbers predicted to the VA WEA (see below for more details on parameter effects).

Across both the boat and aerial surveys, proximity to shore was the most important predictor of abundance. The abundance of terns, gannets, and loons increased with proximity to shore (Table 18-2 through Table 18-6). Alcids associated more closely with the shoreline in Year 2, but they were farther from shore in Year 1 boat surveys (Table 18-7). The detectability of terns, loons, and alcids decreased as seas became rougher, whereas gannets showed no change in detectability in Year 1, and an opposite effect in Year 2 (Table 18-2 to Table 18-7).

The general patterns in habitat relationships between the aerial and boat surveys were consistent. Terns showed similar parameter estimates for the habitat covariates between survey methods, though this was not true for all parameters (Table 18-2). For example, terns were associated with warm water in the first year aerial surveys and with fine sand in the second year aerial surveys. We found that gannets had a positive relationship with cold water in all surveys except the first year of the boat surveys, when we found no significant relationship (Table 18-3). Gannets associated strongly with all four habitat covariates in the second year of surveys, with similar parameter estimates to all except that they associated with coarse sand in the aerial surveys and fine sand in the second year boat surveys.

For loons, we found similar patterns as with gannets and terns; however, it is useful to note that there were some differences when using species-level data (Table 18-4 to Table 18-6). For example, in Year 2 of the aerial survey, only one Red-throated Loon was identified, while there were 2062 total loon observations (Table 18-1). This meant that we were unable to model the distribution of Red-throated Loons that year for the aerial data; the boat data that same year had 754 observed Red-throated loons. In comparison, the Year 2 aerial survey for Common Loons had the most number of significant covariates of any of the loon analyses (Table 18-5). Looking at the boat survey results for year 1, Red-throated Loons had significant negative effects of distance to shore, slope, and salinity (Table 18-6); Common Loons also had a significant negative effect of salinity, but additionally a positive effect of sea surface temperature (Table 18-5). The model for all loons had a significant negative effect of salinity and distance to shore and a significant positive effect of sea surface temperature (Table 18-4). Thus the combined model smooths out the individual species effects, losing the importance of slope on Red-throated Loons and suggesting a relationship with distance to shore that was not detected in Common Loons. Similar results were observed in the first year aerial survey: Common Loons had no significant effects (Table 18-5), Red-throated Loons had a significant negative effect of distance to shore (Table 18-6), and all loons had a significant negative effect of distance to shore and a positive effect of sea surface temperature (Table 18-4). Here, we may be seeing some differences due to increases in sample size; as we add observations from unidentified loons, more patterns can be detected.

Bayesian p-values suggest that model fit was generally adequate for all of the abundance model components (Table 18-8); the aerial data for the gannets and the combined loons did not fit very well, and thus other distributions may be explored in these cases. Futher investigation into the detection component may be necessary, but in general the estimates of abundance have been rather insensitive to the detection model (Half Normal vs. Negative Exponential; unpublished results), so the results are not likely to change significantly even under a different detection model in these cases.

## Discussion

As expected, proximity to shore was the main consistent driver of abundance patterns for all species/groups in this study. Chlorophyll concentration also increased with proximity to shore, and while it was not included in models due to this co-linearity, this suggests that distance from shore may be a proxy for primary productivity in this region. A large effect of primary productivity on predator distributions may indicate strong bottom-up forcing in this region. This is consistent with studies suggesting that, in waters off the east coast of the US where productivity and species richness are relatively high, bottom-up control dominates and resource limitation induces positive predator-prey relationships (Ainley and Hyrenbach 2010; Frank et al. 2007; Hunt and McKinnell 2006).

The boat surveys generally resulted in higher estimates of abundance compared to the aerial surveys, taking effort into account. The total length of an aerial survey's transects (3,613 km including the Maryland extensions, as stated in Chapter 3) is much greater than in a boat survey (572 km, Chapter 7). The strip width is 1/5 of the 1 km truncation distance we used for the detection function in the boat survey models. Therefore, the aerial survey effort (total area sampled) is 1.3 fold greater, so we would expect to estimate more individuals in the aerial surveys. However, our results show the opposite: that the boat survey models consistently estimated and predicted higher abundance, which is primarily due to accounting for imperfect detection. The differences are particularly noticeable with the smaller species (e.g., terns and alcids), indicating that they were more easily identified in the boat surveys.

Detection decreased with increased sea state for all species except gannets in the Year 2 boat surveys. The observer team moved into the pilot house during rough seas, following safety protocol, which likely contributed to reduced visibility. We suspect that the increased detection of gannets in rough seas was a result of differences in behavior, as gannets are less likely to sit on the water during rough seas, and flying gannets are generally considered to be more visible.

In general, habitat relationships were similar within a season and between survey types (boat and aerial), with a few exceptions. These exceptions could be due to (1) more habitat sampled between the MD and VA WEAs in the aerial surveys, or to (2) the extreme habitat values that occurred in the shipping channel to Delaware Bay, which were sampled on Transect 2 of the boat surveys (e.g., steep slopes and a strong salinity front). Gulf Stream waters on the outer edge of the continental shelf tend to be warmer than coastal waters, and salinity also tends to decrease with distance from the freshwater outlets inshore of the Delaware and Chesapeake Bays. Thus, the significant influence of warm water and fine sand on abundance of terns in the aerial surveys (unlike the boat surveys) may be due to aerial observations of them close to shore between the MD and VA WEAs. Opposite effects of sediment grain size on gannets occurred in the Year 2 boat surveys and aerial surveys, which may have been due to differences in sampling effort by survey type, where aerial surveys covered more area between the MD and VA WEA footprints. In Year 2, the boat surveys also showed that Common Loons associated with steep slope, and with more gradual slope in the aerial surveys, which again could be due to occurrences between the MD and VA WEAs, where the bottom is relatively flat. Low salinity had a strong effect on Red-throated Loons in the boat surveys, but not in the first year aerial survey. However, Red-throated Loon data from the aerial surveys should be taken with caution, since many Red-throated Loons were not identified to species (Chapter 16), which may cause biased results. Alcids were likely to be far from shore, associating with cold water in the first year boat surveys (similarly to Chapter 12), but the first year aerial surveys showed an association only with warmer water, which may be a result of co-linearity between SST and distance to shore in gulf stream waters on the outer edge of the continental shelf.

Similarities between survey types were most pronounced with proximity to shore, which had consistently significant effects on (1) terns and loons across both seasons and survey types (Chapter 12), (2) gannets across three of the four models (Ch. 11), and (3) alcids in the second year across both survey types.

Significant effects were consistent across both survey types in year 2, with respect to cold water and gannets, as well as grain size and Common Loons. Significant effects of warm water on loons occurred across both seasons and survey types. Common and Red-throated Loons also associated with low salinity in different survey types and seasons. Our results suggest that using both boat and aerial surveys can provide more complete ecological context compared to either survey type alone.

### *Future work*

The results of this chapter suggest that combining the two survey types into one comprehensive model would be fruitful. The results between the boat and aerial surveys were generally consistent for the species we examined, and variations between the methods may be due to differences in the sampled area (larger coverage with aerial) and in detection (accounted for by distance sampling in boats). Further data exploration of yearly differences (as opposed to survey-specific) in covariate values and patterns would be useful (e.g., to address issues of co-linearity). Additionally, testing the impacts of localized habitat on the results for the entire study area would be informative, and could be achieved by removing parts of the dataset to evaluate changes in the results (for example transect 2 of the boat survey, which sampled some extreme covariate values). Teasing apart differences due to variation in survey type, inter-annual differences, and sampling space will help to better understand the differences observed in the relationships between seabird abundance and habitat covariates.

Combining the data into a single model would likely play to the strengths of both survey methodologies and provide more reliable inferences about the underlying ecological drivers of seabird distributions and abundance. In a first attempt at this, we have implemented an integrated model described in Chapter 19. There are a number of approaches that can be taken when developing a joint model, and we are continuing to pursue those options in an addendum to this final report. One issue to be addressed is how to deal with availability in the digital aerial surveys (see Winiarski et al. 2014); we currently have no measure of availability, and this would be difficult to acquire for all species. Thus, in addition to a joint modeling approach to combine the survey types, we suggest also conducting an analysis of model sensitivity to availability and detection, to better understand the impact of these processes on abundance estimates for the digital aerial surveys.

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## Figures and tables

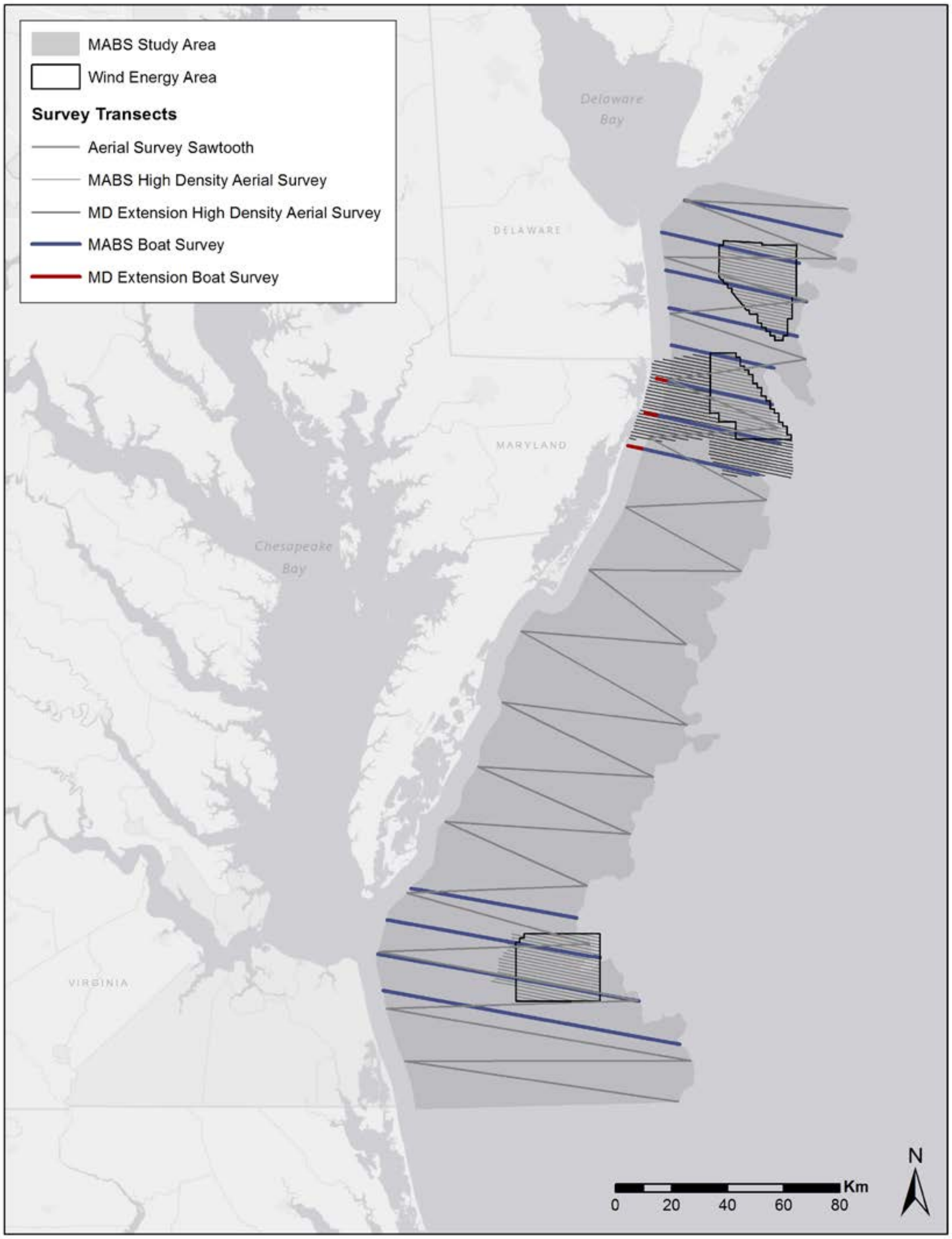

**Figure 18-1**. **Study area.** Boat transects are shown in blue and red, aerial transects in light and dark gray, and wind energy areas in black.

**Table 18-1. Surveys used in the analysis for each species/group and the abundance of each species/group.** Observed (Obs.) refers to raw counts and estimated abundance (Estim.) is fit to the sampled transects (summed across the listed surveys). We predicted (Predicted) the abundance of each species to a representative summer or winter day (25 Jul 2012 for terns; 25 Dec 2012 for gannets, loons, and alcids) in each wind energy area (WEA) by state: Delaware (DE), Maryland (MD), and Virginia (VA). [a]Predictions used first survey intercept.

| **Year** | **Boat Surveys** | | | | **Aerial Surveys** | | | | **Group** | **Boat abundance** | | | | | **Aerial abundance** | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | **Obs.** | **Estim.** | **Predicted** | | | **Obs.** | **Predicted** | | |
| | | | | | | | | | | **trans.** | **trans.** | **DE WEA** | **MD WEA** | **VA WEA** | **trans.** | **DE WEA** | **MD WEA** | **VA WEA** |
| **First** | **Jun-12** | **Aug-12** | **Sep-12** | | | **Jun-12** | **Sep-12** | | Terns | 534 | 3,378.4 | 1,151.5 | 1,382.4 | 152.4 | 108 | 93.4 | 109.6 | 11.2 |
| | **Nov-12** | **Dec-12** | **Jan-13** | | **Oct-12** | **Dec-12** | **Feb-13** | | Gannets | 3,998 | 8,960.5 | 1,215.5 | 1,313.6 | 408.0 | 4,190 | 1,158.3 | 932.6 | 1,022.6 |
| | | **Dec-12** | **Jan-13** | **Mar-13** | | **Dec-12** | **Feb-13** | **Mar-13** | Loons[a] | 996 | 3,811.2 | 1,139.6 | 804.8 | 1,356.0 | 1,661 | 368.7 | 329.8 | 307.5 |
| | | | | | | | | | Common L.[a] | 517 | 2,094.7 | 647.3 | 462.5 | 1,140.0 | 173 | 74.9 | 59.7 | 67.5 |
| | | | | | | | | | Red-thr. L.[a] | 441 | 1,805.9 | 360.7 | 260.3 | 185.4 | 117 | 64.2 | 62.1 | 30.7 |
| | | **Dec-12** | **Jan-13** | | | **Dec-12** | **Feb-13** | | Alcids | 598 | 3,495.1 | 1,409.3 | 889.5 | 2839.9 | 339 | 122.9 | 191.9 | 201.8 |
| **Second** | **Jun-13** | **Aug-13** | **Sep-13** | | | **Jul-13** | **Sep-13** | | Terns | 243 | 1,877.9 | 269.4 | 309.0 | 74.4 | 154 | 19.8 | 29.0 | 1.5 |
| | **Oct-13** | **Dec-13** | **Jan-14** | | **Oct-13** | **Dec-13** | **Feb-14** | | Gannets | 4,723 | 5,693.9 | 5,340.3 | 1,578.3 | 2,272.4 | 1,612 | 420.3 | 413.5 | 152.8 |
| | | **Dec-13** | **Jan-14** | **Apr-14** | | **Dec-13** | **Feb-14** | **May-14** | Loons[a] | 2,626 | 10,884.9 | 1,476.7 | 1,512.9 | 941.5 | 2,062 | 666.7 | 521.9 | 697.6 |
| | | | | | | | | | Common L.[a] | 1,851 | 8,453.8 | 407.2 | 510.5 | 250.6 | 122 | 55.8 | 42.1 | 105.5 |
| | | | | | | | | | Red-thr. L.[a] | 754 | 2,586.0 | 216.0 | 187.1 | 97.9 | 1 | NA | NA | NA |
| | | **Dec-13** | **Jan-14** | | | **Dec-13** | **Feb-14** | | Alcids | 578 | 1,769.4 | 0.3 | 0.9 | 0.1 | 102 | 10.8 | 12.6 | 19.6 |

**Table 18-2. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of terns.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, and all abundance parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Mam = marine mammal density, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **Terns** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept** | -0.33 | 0.21 | -0.74 | 0.08 | ***-0.94*** | 0.21 | -1.33 | -0.53 | ***-3.81*** | 0.24 | -4.31 | -3.36 | ***-4.18*** | 0.26 | -4.70 | -3.70 |
| | **Dst** | ***-1.96*** | 0.33 | -2.64 | -1.35 | ***-1.42*** | 0.20 | -1.83 | -1.05 | ***-1.61*** | 0.22 | -2.05 | -1.19 | ***-2.14*** | 0.20 | -2.55 | -1.77 |
| | **Slp** | -0.32 | 0.19 | -0.71 | 0.07 | -0.02 | 0.17 | -0.36 | 0.32 | 0.01 | 0.13 | -0.24 | 0.26 | 0.08 | 0.10 | -0.12 | 0.28 |
| | **Grn** | -0.12 | 0.19 | -0.48 | 0.25 | 0.24 | 0.19 | -0.13 | 0.62 | 0.18 | 0.14 | -0.11 | 0.45 | ***0.37*** | 0.13 | 0.13 | 0.63 |
| | **Sst** | 0.62 | 0.37 | -0.10 | 1.33 | 0.17 | 0.16 | -0.15 | 0.49 | ***0.64*** | 0.22 | 0.20 | 1.07 | 0.02 | 0.12 | -0.21 | 0.25 |
| | **Sal** | -0.31 | 0.32 | -0.90 | 0.32 | -0.08 | 0.18 | -0.44 | 0.26 | -0.39 | 0.25 | -0.87 | 0.09 | -0.30 | 0.16 | -0.63 | 0.02 |
| | **Overdisp;** $r$ | 0.12 | 0.02 | 0.09 | 0.15 | 0.21 | 0.04 | 0.14 | 0.31 | 0.22 | 0.07 | 0.12 | 0.38 | 0.31 | 0.08 | 0.18 | 0.49 |
| **Detection** | **Beaufort 0-2** | ***5.25*** | 0.07 | 5.12 | 5.38 | ***5.16*** | 0.09 | 5.00 | 5.33 | | | | | | | | |
| | **Beaufort 3-6** | ***4.84*** | 0.07 | 4.70 | 4.98 | ***4.46*** | 0.11 | 4.24 | 4.68 | | | | | | | | |

**Table 18-3. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of gannets.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, and all abundance parameters are on the log scale . Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **Gannets** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept** | ***0.92*** | 0.10 | 0.72 | 1.13 | ***1.07*** | 0.09 | 0.89 | 1.26 | ***-0.14*** | 0.07 | -0.28 | 0.01 | ***-0.80*** | 0.06 | -0.92 | -0.67 |
| | **Dst** | ***-1.21*** | 0.12 | -1.43 | -0.97 | ***-1.08*** | 0.13 | -1.34 | -0.83 | 0.11 | 0.08 | -0.05 | 0.27 | ***-0.78*** | 0.07 | -0.92 | -0.65 |
| | **Slp** | -0.11 | 0.11 | -0.30 | 0.11 | ***0.30*** | 0.09 | 0.12 | 0.48 | -0.09 | 0.07 | -0.23 | 0.05 | ***0.26*** | 0.05 | 0.15 | 0.37 |
| | **Grn** | 0.13 | 0.10 | -0.07 | 0.31 | ***-0.29*** | 0.12 | -0.54 | -0.07 | ***0.20*** | 0.07 | 0.05 | 0.34 | ***0.15*** | 0.06 | 0.03 | 0.26 |
| | **Sst** | -0.02 | 0.12 | -0.25 | 0.21 | ***-0.76*** | 0.10 | -0.96 | -0.56 | ***-1.87*** | 0.10 | -2.06 | -1.68 | ***-0.65*** | 0.08 | -0.80 | -0.50 |
| | **Overdisp**; $r$ | 0.28 | 0.02 | 0.23 | 0.32 | 0.25 | 0.02 | 0.22 | 0.30 | 0.14 | 0.01 | 0.13 | 0.16 | 0.14 | 0.01 | 0.13 | 0.16 |
| **Detection** | **Beaufort 0-2** | ***5.86*** | 0.02 | 5.82 | 5.89 | ***5.67*** | 0.02 | 5.63 | 5.72 | | | | | | | | |
| | **Beaufort 3-6** | ***5.91*** | 0.02 | 5.87 | 5.95 | ***5.82*** | 0.01 | 5.80 | 5.85 | | | | | | | | |

**Table 18-4. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of all loons (Common, Red-throated, and unknowns combined).** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, and all abundance parameters are on the log scale (from the count process). Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| Loons | | Boat | | | | | | | | Aerial | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | First year | | | | Second year | | | | First year | | | | Second year | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept (survey 1)** | 0.04 | 0.23 | -0.40 | 0.50 | ***1.20*** | 0.24 | 0.72 | 1.70 | **-0.95** | 0.21 | -1.36 | -0.54 | **0.13** | 0.06 | 0.01 | 0.25 |
| | **Intercept (survey 2)** | ***0.56*** | 0.15 | 0.28 | 0.85 | ***1.32*** | 0.21 | 0.92 | 1.74 | **0.67** | 0.10 | 0.48 | 0.87 | **0.72** | 0.08 | 0.55 | 0.88 |
| | **Intercept (survey 3)** | ***1.37*** | 0.19 | 1.01 | 1.74 | ***1.31*** | 0.15 | 1.01 | 1.61 | 0.01 | 0.18 | -0.35 | 0.35 | **-2.06** | 0.13 | -2.31 | -1.82 |
| | **Dst** | ***-0.25*** | 0.10 | -0.45 | -0.05 | ***-0.76*** | 0.17 | -1.10 | -0.43 | **-0.65** | 0.06 | -0.76 | -0.54 | **-0.28** | 0.04 | -0.35 | -0.20 |
| | **Slp** | -0.11 | 0.08 | -0.26 | 0.04 | ***0.21*** | 0.08 | 0.06 | 0.37 | -0.02 | 0.04 | -0.09 | 0.05 | -0.07 | 0.04 | -0.14 | 0.00 |
| | **Grn** | -0.08 | 0.08 | -0.23 | 0.07 | ***0.37*** | 0.09 | 0.20 | 0.57 | -0.06 | 0.04 | -0.13 | 0.02 | 0.00 | 0.03 | -0.06 | 0.07 |
| | **Sst** | ***0.35*** | 0.12 | 0.12 | 0.57 | ***0.53*** | 0.12 | 0.30 | 0.76 | **0.75** | 0.11 | 0.53 | 0.96 | **0.45** | 0.07 | 0.32 | 0.58 |
| | **Sal** | ***-0.50*** | 0.12 | -0.74 | -0.26 | -0.12 | 0.21 | -0.53 | 0.30 | -0.09 | 0.12 | -0.33 | 0.14 | **-0.23** | 0.05 | -0.32 | -0.13 |
| | **Overdisp;** $r$ | 0.67 | 0.07 | 0.54 | 0.82 | 0.44 | 0.04 | 0.37 | 0.52 | 0.51 | 0.03 | 0.46 | 0.58 | 0.66 | 0.04 | 0.58 | 0.75 |
| **Detection** | **Beaufort 0-2** | ***5.38*** | 0.03 | 5.32 | 5.43 | ***5.61*** | 0.03 | 5.56 | 5.66 | | | | | | | | |
| | **Beaufort 3-6** | ***5.28*** | 0.04 | 5.21 | 5.36 | ***5.14*** | 0.02 | 5.10 | 5.17 | | | | | | | | |

**Table 18-5. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of Common Loons.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, and all abundance parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **COLO** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept (survey 1)** | ***-0.94*** | 0.31 | -1.54 | -0.35 | ***1.23*** | 0.29 | 0.66 | 1.80 | **-1.98** | 0.48 | -2.93 | -1.03 | **-2.37** | 0.16 | -2.70 | -2.06 |
| | **Intercept (survey 2)** | 0.26 | 0.17 | -0.06 | 0.60 | ***0.78*** | 0.26 | 0.27 | 1.28 | **-2.00** | 0.24 | -2.47 | -1.54 | **-2.24** | 0.25 | -2.74 | -1.76 |
| | **Intercept (survey 3)** | ***0.54*** | 0.24 | 0.08 | 1.02 | ***0.52*** | 0.19 | 0.15 | 0.90 | **-4.37** | 0.51 | -5.41 | -3.40 | **-5.73** | 0.46 | -6.67 | -4.86 |
| | **Dst** | -0.15 | 0.13 | -0.41 | 0.11 | ***-0.79*** | 0.22 | -1.22 | -0.34 | -0.21 | 0.15 | -0.52 | 0.08 | 0.07 | 0.11 | -0.14 | 0.29 |
| | **Slp** | -0.04 | 0.09 | -0.21 | 0.14 | ***0.28*** | 0.10 | 0.09 | 0.48 | 0.05 | 0.10 | -0.14 | 0.24 | **-0.30** | 0.14 | -0.57 | -0.04 |
| | **Grn** | -0.07 | 0.09 | -0.25 | 0.11 | ***0.42*** | 0.12 | 0.18 | 0.67 | -0.06 | 0.10 | -0.26 | 0.13 | **0.22** | 0.11 | 0.01 | 0.46 |
| | **Sst** | ***0.88*** | 0.16 | 0.57 | 1.21 | ***0.71*** | 0.15 | 0.41 | 1.00 | -0.04 | 0.26 | -0.55 | 0.46 | **0.81** | 0.20 | 0.41 | 1.21 |
| | **Sal** | ***-0.45*** | 0.15 | -0.76 | -0.15 | 0.23 | 0.26 | -0.28 | 0.71 | -0.07 | 0.28 | -0.62 | 0.49 | **-0.39** | 0.13 | -0.65 | -0.14 |
| | **Overdisp**; $r$ | 0.51 | 0.07 | 0.39 | 0.66 | 0.29 | 0.03 | 0.24 | 0.35 | 0.26 | 0.07 | 0.17 | 0.42 | 0.40 | 0.21 | 0.17 | 0.97 |
| **Detection** | **Beaufort 0-2** | ***5.32*** | 0.04 | 5.25 | 5.40 | ***5.47*** | 0.04 | 5.41 | 5.54 | | | | | | | | |
| | **Beaufort 3-6** | ***5.20*** | 0.06 | 5.09 | 5.31 | ***5.09*** | 0.02 | 5.05 | 5.13 | | | | | | | | |

**Table 18-6. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of Red-throated Loons.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, ***r*** is the overdispersion parameter, and all abundance parameters are on the log. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics. There was only 1 observed Red-throated Loon in the second year aerial surveys, so no model was fit to these data.

| **RTLO** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept (survey 1)** | ***-0.99*** | 0.37 | -1.72 | -0.30 | ***-2.29*** | 0.33 | -2.94 | -1.66 | **-1.88** | 0.90 | -3.66 | -0.15 | NA | NA | NA | NA |
| | **Intercept (survey 2)** | ***-1.43*** | 0.27 | -1.98 | -0.91 | 0.01 | 0.22 | -0.42 | 0.44 | **-2.60** | 0.45 | -3.48 | -1.72 | NA | NA | NA | NA |
| | **Intercept (survey 3)** | ***0.85*** | 0.32 | 0.24 | 1.50 | 0.33 | 0.16 | 0.03 | 0.64 | **-6.12** | 0.87 | -7.88 | -4.48 | NA | NA | NA | NA |
| | **Dst** | ***-0.78*** | 0.17 | -1.12 | -0.46 | ***-1.13*** | 0.18 | -1.50 | -0.80 | **-0.62** | 0.31 | -1.25 | -0.03 | NA | NA | NA | NA |
| | **Slp** | ***-0.30*** | 0.12 | -0.53 | -0.07 | -0.12 | 0.09 | -0.30 | 0.06 | -0.26 | 0.17 | -0.59 | 0.06 | NA | NA | NA | NA |
| | **Grn** | -0.08 | 0.13 | -0.33 | 0.17 | ***0.23*** | 0.09 | 0.05 | 0.41 | -0.05 | 0.17 | -0.38 | 0.26 | NA | NA | NA | NA |
| | **Sst** | -0.22 | 0.17 | -0.56 | 0.11 | ***0.33*** | 0.12 | 0.09 | 0.58 | -0.34 | 0.46 | -1.22 | 0.58 | NA | NA | NA | NA |
| | **Sal** | ***-0.92*** | 0.25 | -1.43 | -0.45 | ***-0.54*** | 0.22 | -0.98 | -0.08 | 0.21 | 0.54 | -0.85 | 1.28 | NA | NA | NA | NA |
| | **Overdisp;** $r$ | 0.35 | 0.05 | 0.26 | 0.46 | 0.65 | 0.09 | 0.50 | 0.84 | 0.06 | 0.01 | 0.04 | 0.08 | NA | NA | NA | NA |
| **Detection** | **Beaufort 0-2** | ***5.34*** | 0.05 | 5.25 | 5.44 | ***5.70*** | 0.04 | 5.63 | 5.77 | | | | | | | | |
| | **Beaufort 3-6** | ***5.20*** | 0.05 | 5.10 | 5.30 | ***5.26*** | 0.04 | 5.19 | 5.33 | | | | | | | | |

**Table 18-7. Parameter estimates by year from the boat and high resolution digital video aerial surveys, using a Negative Binomial distribution to model counts of alcids.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, and all abundance parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics. There was only 1 observed Red-throated Loon in the second year aerial surveys, so no model was fit to these data.

| **Chapter 19 Alcids** | | **Chapter 20 Boat** | | | | | | | | **Chapter 21 Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept** | ***1.00*** | 0.12 | 0.78 | 1.24 | ***-0.51*** | 0.16 | -0.83 | -0.19 | ***-1.71*** | 0.12 | -1.95 | -1.47 | ***-2.95*** | 0.21 | -3.35 | -2.52 |
| | **Dst** | ***0.54*** | 0.15 | 0.24 | 0.84 | ***-1.45*** | 0.24 | -1.93 | -0.94 | 0.04 | 0.13 | -0.20 | 0.29 | **-0.61** | 0.28 | -1.18 | -0.07 |
| | **Slp** | 0.13 | 0.10 | -0.06 | 0.34 | -0.03 | 0.14 | -0.30 | 0.27 | -0.19 | 0.12 | -0.43 | 0.05 | -0.43 | 0.23 | -0.88 | 0.02 |
| | **Grn** | -0.20 | 0.12 | -0.44 | 0.04 | 0.13 | 0.19 | -0.23 | 0.50 | 0.26 | 0.14 | -0.02 | 0.53 | 0.02 | 0.25 | -0.47 | 0.50 |
| | **Sst** | ***-0.28*** | 0.13 | -0.54 | -0.02 | 0.00 | 0.18 | -0.37 | 0.34 | ***0.65*** | 0.15 | 0.36 | 0.95 | ***-0.58*** | 0.23 | -1.03 | -0.13 |
| | **Sal** | -0.09 | 0.11 | -0.30 | 0.13 | ***2.06*** | 0.26 | 1.53 | 2.54 | ***0.79*** | 0.12 | 0.55 | 1.03 | 0.38 | 0.29 | -0.18 | 0.95 |
| | **Overdisp;** $r$ | 0.38 | 0.05 | 0.29 | 0.49 | 0.25 | 0.04 | 0.18 | 0.33 | 0.08 | 0.01 | 0.06 | 0.10 | 0.02 | fixed | | |
| **Detection** | **Beaufort 0-2** | ***5.19*** | 0.04 | 5.12 | 5.27 | ***5.61*** | 0.05 | 5.52 | 5.72 | | | | | | | | |
| | **Beaufort 3-6** | ***4.56*** | 0.06 | 4.45 | 4.67 | ***5.54*** | 0.04 | 5.46 | 5.61 | | | | | | | | |

**Table 18-8. Bayesian p-values for the abundance and detection components of the models.** Values close to 0.5 indicate good model fit.

| | | Boat | | | | Aerial | |
|---|---|---|---|---|---|---|---|
| | | First year | | Second year | | First year | Second year |
| Group | Sub-group | Abundance | Detection | Abundance | Detection | Abundance | Abundance |
| Terns | | 0.58 | 0.50 | 0.51 | 0.39 | 0.39 | 0.58 |
| Gannets | | 0.66 | 0.45 | 0.71 | 0.75 | 0.99 | 0.72 |
| Loons | All | 0.5 | 0.81 | 0.6 | 0.99 | 0.85 | 0.65 |
| | COLO | 0.53 | 0.55 | 0.63 | 0.88 | 0.42 | 0.42 |
| | RTLO | 0.55 | 0.62 | 0.55 | 0.97 | 0.51 | NA |
| Alcids | | 0.52 | 0.54 | 0.51 | 0.58 | 0.48 | 0.49 |

## Supplementary Material

### Appendix 18A. Evaluating species and habitat associations in a marine bird community, using shipboard and digital video aerial surveys

#### *Abstract*

Marine spatial planning efforts have recently brought attention to the need for baseline studies of marine wildlife in relation to proposed offshore wind energy development on the US Atlantic Outer Continental Shelf. From March 2012 to May 2014, we collected line transect data from 16 shipboard surveys, and novel high resolution digital videography data from 15 aerial surveys, off the coast of Delaware, Maryland, and Virginia. We implemented hierarchical models to evaluate the relationships between marine birds and mammals, while accounting for variation in habitat covariates and observed seabird behaviors. We selected six covariates on avian abundance: three static (distance to shore, slope, and grain size), and three dynamic (sea surface temperature, salinity, and marine mammals). We hypothesized that aerial foragers would show positive associations with marine mammals, and found support for this in a subset of surveys and species. Video aerial surveys detected more submerged animals compared to boat surveys, which detected more smaller seabird species and accommodated variability in detection using distance sampling. This resulted in higher abundance estimates than those derived from video aerial surveys. Our results are consistent with a growing body of research suggesting that facilitative interactions occur among pelagic communities, where subsurface predators drive shared prey upwards for increased detectability and accessibility to surface-feeding seabirds. Our study highlights the importance of quantifying behavioral and ecological influences on avian abundance, particularly in predicting the potential exposure of protected marine birds and mammals to offshore development.

#### *Introduction*

Over the past few decades, the field of community ecology has evolved from a focus on competition to the function of facilitation in species interactions (Bruno et al. 2003). While negative (competitive) and positive (facilitative) species interactions can co-occur, competition is more likely to dominate when resources are limited, and facilitation may eclipse competition when resources are abundant (Stachowicz 2001). The spatial and temporal patterns of marine habitat and organisms tend to be highly dynamic and aggregated, which can result in areas of limited resources and high interference competition, or ephemeral patches of superabundant resources that promote positive, facilitative species interactions (Camphuysen and Webb 1999; Fauchald et al. 2011; Poysa 1992). Marine birds face the challenging task of searching for inconspicuous prey located below the sea surface. A growing body of research has documented positive associations of high density marine fish, mammals, and birds near the sea surface, where the subsurface predators drive shared prey upwards from deeper strata (Evans 1982; Hebshi et al. 2008; Skov et al. 1995). This has shaped the hypothesis that subsurface predators increase the accessibility and visibility of prey to marine birds, via commensal relationships and local enhancement (Ashmole and Ashmole 1967; Au and Pitman 1986). Local enhancement is a social mechanism by which individuals cue into the act of other foraging for food for improved detectability in an otherwise relatively “featureless” ocean environment (Grünbaum and Veit 2003; Silverman et al. 2004).

We hypothesized that aerial-foraging seabird species would show positive associations with marine mammals at the sea surface in our study area. By aerial foragers, we refer primarily to plunge-divers that usually initiate feeding bouts from the air, thereby excluding species that sit on the water prior to dives (e.g., benthivores such as scoters that dive to the bottom, and pursuit-divers such as alcids and loons that swim across the water column at substantial depths). Aerial foragers such as terns and gannets are highly visual predators and can rely on social foraging cues seen from high flight heights at substantial distances (Haney et al. 1992; Tremblay et al. 2014). Therefore we expected them to associate positively with marine mammals. Several studies have compared shipboard and traditional aerial survey methodologies and shown that boat surveys are better at detecting smaller seabird species (for overview, see Camphuysen et al. 2004). However, aerial observers often perform better at detecting fully submerged animals located at or just below the sea surface (Chapter 5, Chapter 14), so we expected the aerial digital video surveys to show stronger positive seabird-mammal associations.

### *Methods*

Models of tern and gannet abundance used a similar model formulation to that described in the main text of Chapter 18, but also included in the abundance model a covariate of 'observed marine mammals' to explore the potential relationship between seabird abundance and marine mammal counts. We used raw counts of marine mammals along each transect segment. For the boat surveys, this included only those observations in the sampled quadrant (forward, on one side of the vessel, extending up to 1 km from the trackline); for the aerial video surveys, observations occurred within a strip width of 200 m. For the boat survey data, we used seabird foraging behavior as a covariate on detection, rather than Beaufort sea state. Foraging behaviors included feeding, diving, pattering, kleptoparasitizing, and milling (i.e., flying along a circular path), while non-foraging behaviors included those classified as traveling (i.e., flying along a straight path) and resting.

As described in Chapter 18, for the boat surveys, we used a Negative Exponential distribution on detection and a Negative Binomial distribution on abundance. The intercept on abundance, however, was behavior-specific, as we observed some differences in detection between foraging and non-foraging individuals. Counts at segment $i$, $\mathrm{y_i}$, were modeled such that:

$$\mathrm{y_i} \sim \mathrm{NegBin}(\lambda_i,\ r)$$

$$\log(\lambda_i) = \beta_{0,behavior} + offset\ (site\ area) + \ \beta_1 \mathrm{Dst_i} + \beta_2 \mathrm{Slp_i} + \beta_3 \mathrm{Grn_i} + \beta_4 \mathrm{Sst_i} + \beta_5 \mathrm{Sal_i} + \beta_6 \mathrm{Mam_i}$$

where $site\ area$ is the transect segment length multiplied by its width (one km for the boat surveys, 0.2 km for the digital aerial surveys), Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, and Mam = observed marine mammal counts.

For gannets, we used the same formulation as the tern model above, except with a Half Normal detection function and without salinity as a covariate on abundance, due to its co-linearity with SST (especially in the aerial and second year winter boat surveys).

### *Results*

Gannets were more abundant than terns, and accounting for detection resulted in higher estimates in the boat than the aerial video surveys (Table 18A-1). The Negative Binomial distribution fit well to tern and gannet abundance (except for some aerial surveys), the Negative Exponential foraging behavior model fit well to tern detection, and the Half Normal foraging behavior model component fit well to gannet detection (Table 18A-2). Across all boat surveys, foraging individuals had significantly higher detection probabilities, except in the second year boat surveys when non-foraging gannets were more detectable.

Between survey methods, terns and gannets showed comparable patterns in response to the habitat covariates, though this was not true for all parameters. For example, across all summer boat and aerial video surveys, terns associated in close proximity with the shoreline (Table 18A-3). However, SST was found to be significant in the first year of the aerial survey whereas slope and salinity were significant in the second year of the aerial survey. We found that gannets also responded to colder water in all winter surveys except the first year of the boat surveys, when we found no significant relationship (Table 18A-4). We again found some differences between the results for the two survey methodologies, where gannets associated in close proximity with the shoreline across all except the first year aerial video survey, when they showed no significant association with the coast. Gannets responded similarly to all four habitat covariates in the second year aerial and boat surveys, except for the second year boat survey where they associated with coarse sand, rather than with fine sand as in both aerial surveys. During the first year, terns and gannets associated positively with marine mammals (terns in the summer aerial surveys and gannets in the winter boat surveys; Table 18A-3, Table 18A-4).

### *Discussion*

We expected that the aerial video surveys would show stronger positive associations between surface-feeding seabirds and marine mammals than the boat surveys, as a result of better detectability of predators located below the sea surface. Results were not consistent with this hypothesis. Instead, our results suggest that boat and aerial video surveys are both useful in identifying community associations (Camphuysen et al. 2004; Camphuysen and Garthe 2004), and that the information derived from one survey complements that derived from the other, depending on species or survey. For example, in the first year, we found that high abundance of seabirds associated positively with high marine mammal counts, for gannets in the boat surveys and terns in the aerial video surveys. This lends considerable support to our hypothesis for facilitative interactions between these taxa (i.e., via local enhancement and commensalism; Ashmole and Ashmole 1967; Au and Pitman 1986). In the pelagic realm, facilitative and competitive interactions can co-occur, yet vary in degree, depending on scale and resource availability (Fauchald et al. 2011; Goyert et al. 2014; Safina 1990). Presumably, higher densities would stimulate interference competition, especially with limited resources in small prey patches; however, facilitative interactions are feasible when resources are abundant, even at small scales.

Our results suggest that the distributions of resources and marine mammals can drive high abundance of seabirds, which likely associate with suitable habitat until other predators become available to exploit for improved foraging success. Across both seasons and surveys, terns adhered tightly to the shoreline (Chapter 12). Additionally, terns associated with cold water, which could be due to their distribution along the coast, away from the warm Gulf Stream waters on the Outer Continental Shelf. Gannets associated in close proximity with the shoreline across all except the first year aerial video survey; they also responded to cold water in all except the first year boat surveys. Primary productivity increased with proximity to shore in our study region, which suggests that terns and gannets may have associated with marine mammals in productive areas.

Northern Gannets and Common Terns are examples of feeding generalists that rely heavily on social or opportunistic strategies in search of food (Goyert 2015; Montevecchi et al. 2009). As aerial foragers, their use of visual cues and behavioral mechanisms such as local enhancement may be especially advantageous (Goyert 2014; Thiebault et al. 2014; Tremblay et al. 2014; as in Cape Gannets, *Morus capensis*). Spatial or temporal variability (i.e., seasonality) may play a role in whether associations result as positive or negative between seabirds and marine fish or mammals (Goyert et al. 2014). Predictability or persistence of resource hotspots can also drive the formation of facilitative feeding assemblages, as shown in gannets (Davoren et al. 2010). Our results suggest that aerial video surveys complement boat surveys in documenting competitive or facilitative species interactions between marine mammals and birds, with facilitative interactions contributing to increased detectability and accessibility of shared prey. Such dynamic associations between seabirds and marine mammals suggest that their exposure to offshore wind energy development depends on either the persistence or volatility of shared resource hotspots.

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## *Tables*

**Table 18A-1. Surveys used in the analysis for each species/group and the abundance of each species/group.** Observed (Obs.) refers to raw counts and estimated abundance (Estim.) is the predicted abundance for the sampled transects (summed across the listed surveys).

| Year | Boat Surveys | | | Aerial Surveys | | | Group | Boat abundance | | Aerial abundance | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | Obs. | Estim. | Obs. | Estim. |
| First | Jun-12 | Aug-12 | Sep-12 | May-12 | Jun-12 | Sep-12 | Terns | 534 | 3,447.33 | 238 | 238.50 |
| | Nov-12 | Dec-12 | Jan-13 | Oct-12 | Dec-12 | Feb-13 | Gannets | 3,998 | 8,962.84 | 4,190 | 4189.45 |
| Second | Jun-13 | Aug-13 | Sep-13 | Jul-13 | Aug-13 | Sep-13 | Terns | 243 | 1,751.09 | 223 | 223.27 |
| | Oct-13 | Dec-13 | Jan-14 | Oct-13 | Dec-13 | Feb-14 | Gannets | 4,723 | 11,910.73 | 1,612 | 1612.17 |

**Table 18A-2. Bayesian p-values for the abundance and detection components of the models with marine mammals as a covariate.** Values close to 0.5 indicate good model fit.

| Group | Boat | | Boat | | Aerial | Aerial |
|---|---|---|---|---|---|---|
| | First year | | Second year | | First year | Second year |
| | Abundance | Detection | Abundance | Detection | Abundance | Abundance |
| Terns | 0.59 | 0.43 | 0.52 | 0.42 | 0.46 | 0.68 |
| Gannets | 0.70 | 0.42 | 0.67 | 0.48 | 0.99 | 0.71 |

**Table 18A-3. Parameter estimates by year from the boat and high resolution digital aerial videography for terns.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, ***r*** is the overdispersion parameter and all abundance parameters are on the log scale.. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Mam = marine mammal density, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **Terns** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept (Foraging)** | ***-3.2*** | 0.3 | -3.8 | -2.5 | ***-2.5*** | 0.3 | -3.1 | -2.0 | ***-3.1*** | 0.2 | -3.5 | -2.8 | ***-3.9*** | 0.2 | -4.3 | -3.5 |
| | **Intercept (Not Foraging)** | -0.3 | 0.2 | -0.8 | 0.2 | ***-1.2*** | 0.2 | -1.7 | -0.8 | | | | | | | | |
| | **Dst** | ***-1.9*** | 0.3 | -2.5 | -1.4 | ***-1.4*** | 0.2 | -1.8 | -1.0 | ***-1.5*** | 0.2 | -1.9 | -1.2 | ***-2.1*** | 0.2 | -2.5 | -1.8 |
| | **Slp** | -0.2 | 0.2 | -0.6 | 0.2 | -0.1 | 0.2 | -0.4 | 0.2 | 0.0 | 0.1 | -0.3 | 0.2 | ***0.2*** | 0.1 | 0.0 | 0.3 |
| | **Grn** | -0.1 | 0.2 | -0.4 | 0.3 | 0.3 | 0.2 | -0.1 | 0.7 | 0.0 | 0.1 | -0.2 | 0.3 | 0.1 | 0.1 | 0.0 | 0.3 |
| | **Sst** | 0.6 | 0.4 | -0.2 | 1.3 | 0.1 | 0.2 | -0.2 | 0.4 | ***-0.5*** | 0.1 | -0.7 | -0.2 | 0.0 | 0.1 | -0.2 | 0.2 |
| | **Sal** | -0.5 | 0.3 | -1.2 | 0.1 | 0.0 | 0.2 | -0.4 | 0.3 | -0.2 | 0.2 | -0.6 | 0.2 | ***-0.4*** | 0.1 | -0.7 | -0.1 |
| | **Mam** | 0.1 | 0.2 | -0.3 | 0.4 | -0.1 | 0.1 | -0.3 | 0.2 | ***0.3*** | 0.2 | 0.0 | 0.7 | 0.1 | 0.1 | 0.0 | 0.2 |
| | **Overdisp**; *r* | 0.1 | 0.0 | 0.1 | 0.1 | 0.1 | 0.0 | 0.1 | 0.2 | 0.1 | 0.0 | 0.1 | 0.1 | 0.4 | 0.1 | 0.3 | 0.7 |
| **Detection** | **Foraging** | ***5.7*** | 0.2 | 5.4 | 6.0 | ***5.2*** | 0.1 | 5.0 | 5.5 | | | | | | | | |
| | **Not Foraging** | ***5.0*** | 0.1 | 4.9 | 5.1 | ***4.8*** | 0.1 | 4.7 | 5.0 | | | | | | | | |

**Table 18A-4. Parameter estimates by year from the boat and high resolution digital aerial videography for gannets.** SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, ***r*** is the overdispersion parameter, and all abundance parameters are on the log scale (from the count process). Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Mam = marine mammal density, and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% Bayesian credible interval (BCI) does not overlap zero are in bold italics.

| **Gannets** | | **Boat** | | | | | | | | **Aerial** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **First year** | | | | **Second year** | | | | **First year** | | | | **Second year** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept (Foraging)** | ***-0.6*** | 0.1 | -0.9 | -0.4 | ***0.4*** | 0.1 | 0.2 | 0.7 | -0.1 | 0.1 | -0.3 | 0.0 | ***-0.8*** | 0.1 | -0.9 | -0.7 |
| | **Intercept (Not Foraging)** | ***0.7*** | 0.1 | 0.4 | 0.9 | ***0.6*** | 0.1 | 0.3 | 0.8 | | | | | | | | |
| | **Dst** | ***-1.4*** | 0.1 | -1.6 | -1.2 | ***-1.0*** | 0.1 | -1.2 | -0.8 | 0.1 | 0.1 | 0.0 | 0.3 | ***-0.8*** | 0.1 | -0.9 | -0.6 |
| | **Slp** | -0.1 | 0.1 | -0.3 | 0.2 | ***0.3*** | 0.1 | 0.1 | 0.5 | -0.1 | 0.1 | -0.2 | 0.0 | ***0.3*** | 0.1 | 0.2 | 0.4 |
| | **Grn** | 0.1 | 0.1 | -0.1 | 0.2 | ***-0.3*** | 0.1 | -0.6 | -0.1 | ***0.2*** | 0.1 | 0.1 | 0.3 | ***0.1*** | 0.1 | 0.0 | 0.3 |
| | **Sst** | 0.0 | 0.1 | -0.2 | 0.2 | ***-0.7*** | 0.1 | -0.9 | -0.6 | ***-1.9*** | 0.1 | -2.0 | -1.7 | ***-0.6*** | 0.1 | -0.8 | -0.5 |
| | **Mam** | ***0.4*** | 0.1 | 0.2 | 0.7 | 0.2 | 0.1 | 0.0 | 0.5 | -0.2 | 0.2 | -0.6 | 0.1 | 0.0 | 0.2 | -0.4 | 0.5 |
| | **Overdisp**; *r* | 0.2 | 0.0 | 0.1 | 0.2 | 0.1 | 0.0 | 0.1 | 0.1 | 0.1 | 0.0 | 0.1 | 0.2 | 0.1 | 0.0 | 0.1 | 0.2 |
| **Detection** | **Foraging** | 5.9 | 0.0 | 5.9 | 6.0 | 5.4 | 0.0 | 5.4 | 5.5 | | | | | | | | |
| | **Not Foraging** | 5.9 | 0.0 | 5.8 | 5.9 | 5.9 | 0.0 | 5.8 | 5.9 | | | | | | | | |

# Chapter 19: Developing an integrated model of marine bird distributions with environmental covariates using boat and digital video aerial survey data *This chapter is in draft form

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 19 Highlights

*Developing an integrated model of marine bird distributions with environmental covariates using boat and digital video aerial survey data*

### *Context*[1]

A broad geographic and temporal scale of analysis is required to assess exposure to wildlife from proposed development projects. In this study, data were collected via traditional methods (boat-based distance sampling) and with newer technologies (high definition videography) with the intention of addressing similar questions related to marine wildlife abundance and distribution. Chapter 18 explored the two datasets to determine if similar patterns were detected by each sampling method. Based on those results, Chapter 19 aims to develop a method of integrating these two datasets into a combined approach that makes use of the strengths of each survey type, to produce a single prediction of marine bird abundance and distribution.

In this approach, predictions of marine bird abundance and distribution are jointly informed by aerial surveys, which encompassed a large geographic area, and boat surveys, which allowed for estimation of detection probability. We analyze data from the same four species groups as Chapter 18 (terns, alcids, loons, and Northern Gannets), and incorporate remotely collected environmental covariate data into the hierarchical modeling structure. This approach accounts for imperfect detection to estimate “true” abundance, and predicts marine bird distributions to help identify important habitat use areas and patterns.

### *Study goal/objectives*

Evaluate potential exposure of the marine bird community to offshore development by: (1) developing a model to integrate data from the two survey platforms; and (2) producing a single prediction of abundance and distribution to identify ecological drivers of distribution, abundance, and local hotspots.

### *Highlights*

- Distance to shore was generally the most common predictor of abundance, as was found in Chapters 12 and 18.
- Integrated models predicted species-specific hotspots that generally concurred with Chapters 12 and Chapter 17, with terns largely distributed along near shore habitats, alcids distributed across large areas of the study area, loon hotspots near the mouth of the Chesapeake Bay, and Northern Gannet hotspots consisting of large, localized aggregations.
- Integrated models improved the identification of abundance hotspots (and areas of lower than expected abundance) within a survey period, but further work is required to explore their predictive ability across surveys.

### *Implications*

Developing new approaches to jointly model disparate datasets can improve identification of important habitat use areas, while also providing a framework to combine historical and new sources of data.

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[1] For more detailed context for this chapter, please see the introduction to Part IV of this report.

## Abstract

We investigated an approach to combine shipboard and digital video aerial survey data for marine birds to produce a single prediction of marine bird abundance and distribution. Modeling frameworks were similar to those in other chapters (for example, the boat survey data are modeled similarly to Chapters 11, 12, and 18, but with single species instead of a community), but we aimed to create a method for integrating the boat and aerial datasets. Our approach in this chapter creates a covariate based on the data from the digital video aerial surveys to be included as a predictor variable in the boat surveys. We compare models with and without the aerial covariate and evaluate the model performance. As in Chapter 18, we focused on terns (summer 2013), Northern Gannets (winter 2012), loons (winter 2012), and alcids (winter 2012). The preliminary results showed that integrated and boat-only models predicted similar total abundance across the study area, but distributions and hotspot locations often varied between approaches. Northern Gannets and loons showed strong associations between aerial and boat data, which led to concentrated hotspots for both species. The influence of integrated models were less evident for terns and alcids, but differences in predicted spatial patterns of abundance were still evident. Model evaluation indicated that integrated models outperformed models that only used boat data when predicting back to the same boat and aerial data used for the analysis, but boat-only models were better at predicting distributions from separate surveys (i.e., boat and aerial surveys conducted in the same season but during a different month than the data used in the analysis). The results of this chapter generally support conclusions of Chapters 12, 17, and 18, which found that the distribution of marine birds was often patchy, species- and survey-specific, and correlated with habitat covariates. Developing new joint modeling approaches can improve identification of important habitat use areas (particularly local dynamic hotspots) and provides a framework to compare historical and new sources of data. Further exploration of seasonal, annual, and species-specific results will be useful to evaluate the performance of integrated models to predict important habitat use areas.

## Introduction

Shipboard and traditional aerial survey methodologies have been compared extensively in their performance at estimating marine bird species richness and abundance (see Camphuysen et al. 2004 for an overview). Advantages of shipboard and aerial survey methods vary by species of interest, logistical constraints, size of area surveyed, and specific research questions (Camphuysen et al. 2004). High resolution digital video aerial surveys (hereafter “digital video aerial surveys”) have become common practice for monitoring of marine wildlife in Europe in relation to offshore wind energy development, but the technology is still relatively new, and thus has seldom been compared to more traditional survey approaches. Digital video aerial surveys provide several benefits, including coverage of a larger geographic area in a faster timeframe than is possible with boat surveys (Buckland et al. 2012). Likewise, boat-based surveys provide advantages such as the ability to estimate detection probability based on distance sampling, and thereby adjust raw counts to develop estimates of abundance (Buckland et al. 1993, see Chapters 11-12 for details). Application of concurrent shipboard and digital video aerial surveys in this study provided a unique opportunity to integrate data collected from both these survey types, thus utilizing the strengths of each survey platform.

Herein, we investigate an approach to combine shipboard and digital aerial surveys for marine birds into a single model that uses information from both datasets (hereafter “integrated model”). We aim to compare integrated models and models that use strictly boat data (hereafter “boat-based”). This chapter includes the same suite of species as Chapter 18 (terns, Sternidae; alcids, Alcidae; loons, *Gavia* spp.; and Northern Gannets, *Morus bassanus*) to allow for improved comparisons and insights across multiple species groups.

Our objectives include:

1. Develop an integrated modeling approach that combines digital video aerial and boat-based data to produce a single prediction of marine bird abundance and distribution.
2. Compare the performance of integrated and boat-based models by evaluating their predictive ability to (1) the original boat and aerial datasets (Fitted surveys) and (2) independent boat and aerial surveys (i.e., independent surveys conducted in the same season but during a different month).

It is important to note that there are methodological differences in sampling from the boat versus digital videography. Some differences are inherent to the two survey methods, such as transect width; the boat surveys sample wider transect widths for most species, and use distance sampling to account for variation in detection, while aerial surveys sample a defined strip width for all species. Other differences are specific to the survey design utilized in this study (e.g., boat and aerial transects were located in slightly different geographic areas and occurred at different dates and times). To minimize the study-specific sources of variation, we accounted for the differences in area sampled, and only used data from boat and aerial surveys that occurred within a similar temporal period (i.e., within the same or consecutive months of each other).

In general, our approach first utilized the large geographic coverage of digital video aerial surveys to predict areas of higher or lower than average expected marine bird abundance across the study area, essentially mapping raw hot and cold spots for each species. Smoothed counts from the digital video aerial surveys (i.e., the degree to which grid cells were above or below expected values) were then integrated as a covariate in analysis of boat surveys. As in Chapters 11-12, 15, and 18, habitat covariates were also included in the analysis. This approach allowed predicted marine bird abundance from boat data to not only vary by habitat covariates (similar to Chapters 11-12, and 18), but also by information on expected abundance derived from digital video aerial surveys. This additional covariate should inform model predictions if similar trends in distribution and abundance were observed in both boat and digital video aerial surveys (Chapter 18). Specific details on our modeling approach are provided in the Methods section.

We caution the reader that the approach we use in this chapter is not a fully integrated model, as we use the aerial survey data as a covariate for analyzing the boat survey data. However, we implemented this approach to see if the aerial data would provide useful information for estimating abundance and local hotspots of abundance. After building the models, we compare and contrast those with and without the aerial data covariate to see how well the models do at (1) estimating overall abundance and local hotspots and (2) predicting future seabird patterns. This chapter is an important step towards simultaneously modeling the two data types, which is the ultimate goal and will continue to be pursued in an addendum to the final report, to be completed in 2016.

## Methods

Field methods for the aerial and boat surveys were explained elsewhere in this report (Chapters 3 and 7, respectively). Aerial identification protocols for video analysis were discussed in Chapter 4. For this comparison, we used boat survey observations that were sampled from the forward quadrant on one side of the vessel, extending up to 1 km from the trackline, and digital aerial observations that were collected from four cameras, which each recorded a 50 m band (totaling 0.2 km strip width). For both the boat and aerial surveys, we divided survey transects into 4 km segments; this resulted in some shorter segments at the transect ends, and segment area (the segment length by the abovementioned strip widths for each survey method) was included in our analysis as an offset. The number of individuals for each species was summed by segment and survey. We compared two modeling methods to estimate abundance and covariate relationships for the same species groups examined in Chapter 18, using data from boat and aerial surveys that were closely coincident in time.

### *Species*

We investigated the same suite of species as Chapter 18 (terns, alcids, loons, and Northern Gannets).

#### *Terns*

Terns included Least Terns (*Sternula antillarum*), Caspian Terns (*Hydroprogne caspia*), Black Terns (*Chlidonias niger*), Common Terns (*Sterna hirundo*), Roseate Terns (*Sterna dougallii*), Royal Terns (*Thalasseus maximus*), and Sandwich Terns (*Thalasseus sandvicensis*), as well as those individuals classified as "unidentified terns." Vague identifications that could have included other species such as gulls (e.g., "large tern or small gull,") were excluded. Terns were primarily present in the study area during spring, summer and fall (Chapters 5, 8, 12), so we focused on boat and aerial surveys during the summer (Chapter 18), specifically August (boat) and September (aerial) 2013 (Table 19-1).

#### *Northern Gannets*

Northern Gannets are the only gannet species found in the study area. Because Northern Gannets were primarily present in the study area in late fall to early spring (Chapters 5, 8, 12), it made sense to focus on boat and aerial surveys during the winter season (Chapter 18), and for this analysis, we selected the survey from December 2012.

#### *Loons*

We considered loons as a group (all loons, which included Common Loons, *Gavia immer*, Red-throated Loons, *G. stellata,* and all unidentified loon observations). Similar to Northern Gannets, loons were primarily present in the study area from late fall to early spring (Chapters 5, 8, 12). We used the same survey, December 2012, for the analysis of loons.

#### *Alcids*

The alcid group included Razorbills (*Alca torda*), Dovekies (*Alle alle*), Atlantic Puffins (*Fratercula arctica*), Common Murres (*Uria aalge*), Thick-billed Murres (*U. lomvia*), and Black Guillemots (*Cepphus grille*), as well as those individuals classified as “unidentified alcids.” Alcids were primarily present in the study area during winter (Chapters 5, 8, 12), so again we focused on boat and aerial surveys during the winter season (Chapter 18), specifically December 2012.

### Covariates

As in Chapter 18, we used five covariates in our analyses: three static (distance to shore, slope, and grain size), and two dynamic (sea surface temperature and salinity). We excluded chlorophyll-*a* in these analyses because it was co-linear with distance to shore in some of the surveys and we wanted to keep the covariates consistent across species. As in Chapter 18, remotely sensed covariate data corresponded to the values located at the midpoint of each transect segment. For the static covariates, we calculated distance to shore (m) within ArcGIS 10.2 (ESRI, Redlands, CA) and extracted slope (% rise, 370-m resolution) and grain size (ϕ = -log2[mean grain diameter in mm], 370-m resolution) from the data layer derived by NOAA/NOS National Centers for Coastal Ocean Science (Kinlan et al. 2013). For the dynamic covariates, we used Marine Geospatial Ecology Tools in ArcGIS (Roberts et al. 2010) to download remotely-sensed data at the highest resolution available for all segments. We compiled daily values for sea surface temperature (SST, °C, 1-km GHRSST L4) and salinity (Practical Salinity Units, 9-km HYCOM GLBa0.08 Equatorial 4D). In the boat survey analysis, we additionally included Beaufort sea state on the binary scale as a covariate to detection, which varied by segment (0 = calm seas, Beaufort state 0-2; 1 = rough seas, Beaufort state 3-6; see Chapters 12 and 18).

We overlaid a predictive grid (approximately 4x4 km grid cells) that encompassed the entire sampled area, including the three proposed wind energy areas (Figure 19-1 and Figure 19-2; note that this predictive grid is a restricted version of the one used in Chapter 12, to represent the area covered by digital aerial surveys). As in Chapter 12, we used data from the midpoint of each cell and the central date for each season to predict overall flock abundance to a representative day (summer [terns]: 25 July 2013, winter [Northern Gannets, loons, and alcids]: 25 December 2012).

### Models

To facilitate comparisons, we used the same modeling approach across all species. We summarized the aerial data such that $y_i$ is the count at segment $i$. For each species or group, we then modeled the aerial data using an overdispersed Poisson conditional autoregressive (CAR) model. This approach allowed us to capture excess heterogeneity in counts at the segment level (overdispersion), while also allowing spatial clustering of counts at a broader level with the CAR portion. To implement the model, we assigned each segment to the predictive grid cell that it fell within, thus the notation $i[j]$ indicates that segment $i$ is within grid cell $j$.

The model for aerial surveys was:

$$y_i \sim \text{Poisson}(\lambda_i)$$

$$\log(\lambda_i) = \alpha_0 + offset\ (segment\ area_i) + \varepsilon_i + \theta_{i[j]}$$

where $\alpha_0$ is the intercept, $\varepsilon_i$ is a random effect at the segment level ($i$), and $\theta_{i[j]}$ is the spatially correlated random effect at the predictive grid cell level ($j$). Random segment effects ($\varepsilon_i$) were distributed $\mathrm{Normal}(0, \tau^2)$. Spatial autocorrelation was evaluated at the grid cell level, thus repeated segments ($i$) within grid cell $j$ were assigned the same $\theta_{i[j]}$. Specifically,

$$\theta_j |\, \theta_k \neq \theta_j \sim \mathrm{Normal}\left(\frac{1}{m_j}\sum_{k\in c_j}\theta_k\,, \frac{\sigma^2}{m_j}\right)$$

where $m_j$ is the number of neighbors for predictive grid cell $j$ and $c_j$ is the specific set of neighbors for predictive grid cell $j$. The set of neighbors for each predictive grid cell ($c_j$) was all adjacent grid cells (i.e., Queen's neighborhood). The CAR model also allows inference to unsampled grid cells by utilizing the spatial correlation observed in counts across the sampled area. Grid cell specific random effects ($\theta_j$), which indicate higher or lower expected grid cell abundance, were then used to inform abundance estimates from the boat data.

Next, for each species or group, we conducted preliminary diagnostics to evaluate boat-based data and select the best model for flock abundance, considering the Poisson and Negative Binomial distributions (see Chapters 11-12 and 18 for details). It should be noted that in this chapter, as well as Chapters 11 and 12, abundance of flocks was the sampling unit of analysis; however, Chapter 18 used individuals instead of flocks to make direct comparisons between boat and aerial surveys. Boat-based models for Northern Gannets, loons, and terns used a Negative Binomial distribution on abundance. The abundance component of the boat-based model was constructed such that the flock abundance at segment $i$, $N_\mathrm{i}$, was modeled as:

$$N_\mathrm{i} \sim \mathrm{NegBin}(\lambda_i,\ r)$$

$$\log(\lambda_i) = \beta_0 + offset\ (segment\ area_i) + \beta_1\mathrm{Dst_i} + \beta_2\mathrm{Slp_i} + \beta_3\mathrm{Grn_i} + \beta_4\mathrm{Sst_i} + \beta_5\mathrm{Sal_i} + \ \beta_5\theta_{i[j]}$$

where Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, $\theta_{i[j]}$ = the estimated spatially correlated random effect defined above from the aerial data, and $r$ is the overdispersion parameter. In this approach, a positive parameter estimate for the aerial covariate ($\beta_5$) indicates that aerial and boat surveys were observing similar trends, and data integration is informative for understanding the abundance patterns between the two surveys. Alcid boat-based data were adequately fit with a Poisson distribution, which was identical to the above model except that $N_\mathrm{i} \sim \mathrm{Pois}(\lambda_\mathrm{i})$.

Raw count of flocks at segment $i$, $n_\mathrm{i}$ were linked to true abundance ($N_\mathrm{i}$) through estimation of detection probability using a half-normal distribution (i.e., distance sampling, see Chapter 11-12, 18; Buckland et al. 1993). As in Chapters 12 and 18, we allowed the detection function to vary by a binary indicator of sea state.

### *Model evaluation*

We evaluated integrated and boat-based models by predicting abundance to the original boat and aerial survey data (i.e., fitted surveys) and independent boat and aerial data (i.e., independent surveys in the same season). Specifically, we compared tern predictions to the original survey data, August 2013 (boat) and September 2013 (aerial), as well as to independent surveys conducted during September 2013 (boat) and July 2013 (aerial). For Northern Gannets, loons, and alcids we compared predictions to the original survey data in December 2012 (boat and aerial) and independent surveys conducted during January 2013 (boat and aerial). Using the posterior means of each model parameter, we predicted the abundance of each species using survey-specific habitat covariates. Predictive ability was evaluated using root mean squared error (smaller indicates better model fit) with each segment considered as a replicate.

### *Implementation*

We implemented all models in a Bayesian framework using the package "R2OpenBUGS" (Sturtz et al. 2010) to run the software OpenBUGS (Thomas et al. 2006) in program R version 3.2.0 (R Core Team 2014). We standardized the covariates for analysis to center them on a mean = 0, with a variance close to 1. We ran three parallel Markov chains for 40,000 iterations following a burn-in of 20,000 iterations, thinning by 5. We checked for chain convergence visually (posterior density and trace plots), and quantitatively using the Gelman-Rubin statistic (Gelman et al. 2014). This statistic (termed R-hat) is a measure of among-chain versus between-chain variance and values < 1.1 indicate convergence (Gelman et al. 2014). We also assessed goodness of fit by computing Bayesian p-values. We used Freeman-Tukey fit statistics to evaluate the model for abundance, and the Half Normal detection function (Gelman et al. 2014).

## Results

The number of species/group-specific observations were > 100 for loons, Northern Gannets, and alcids in both boat (flocks) and digital aerial surveys (individuals), and > 60 for terns (Table 19-1). Mean flock size varied by species, but most observed flocks were ≤ 2 individuals (Table 19-1). Distributions of flock size were right-skewed for all species, with larger flocks (≥ 18) only observed on rare occasions (Table 19-1). Bayesian p-values suggested that model fit was adequate for all of the abundance and detection model components (Table 19-2).

Integrated and boat-based models often resulted in similar abundance estimates (Table 19-3). For example, predicted tern abundance from the integrated and boat-based models was 3,367 and 3,727, respectively. Covariate relationships were also similar between integrated and boat-based models (Table 19-4 to Table 19-7). The noticeable exception to this trend was parameter estimates for distance to shore, especially for Northern Gannets. Integrated models for Northern Gannets predicted a significant positive relationship, while boat-based models predicted a negative relationship (Table 19-6). A reversal of the distance to shore parameter estimate was also observed for loons and alcids, but differences were generally smaller or non-significant (Table 19-6, Table 19-7, see next paragraph and Discussion for detailed explanations). Parameter estimates for the aerial covariate were positive and significant for both Northern Gannets and Loons (Table 19-5, Table 19-6). Positive parameter estimates indicated that variation in expected grid cell abundance in the aerial-based models were positively correlated with variation in expected grid cell abundance estimated from the boat data. Mean parameter estimates for terns and alcids were negative, but not significant (Table 19-4, Table 19-7).

Predicted distribution and abundance from both integrated and boat-based models showed similar relationships, with terns, loons, and Northern Gannets predicted to be closer to shore (Figure 19-2 to Figure 19-4). Integrated models, however, often identified hotspots that were not predicted in boat-based models (Figure 19-2 to Figure 19-5). Hotspots were particularly evident for Northern Gannets (Figure 19-3) and loons (Figure 19-4). Predicted distributions for terns and alicds were often more similar across models, but areas of higher and lower abundances were still noticeable (Figure 19-2, Figure 19-5). For instance, boat-based models for Northern Gannets predicted a rather uniform trend in abundance that decreased with distance to shore. Integrating information from aerial surveys, however, predicted a much more clustered distribution, even though total abundance was similar between the models (Figure 19-3, Table 19-1). Similarly, boat-based models for terns predicted a relatively ubiquitous near-shore abundance that quickly decreased with distance to shore (Figure 19-2). Integrated models predicted a similar trend with distance to shore for terns, but indicated several areas of higher and lower abundances in the nearshore environment (Figure 19-2). Over the entire study area, the range of predicted grid cell-specific flock abundances was much smaller for alcids (range = 0 - 40 flocks per grid cell) and terns (0 - 60 flocks per grid cell) relative to Northern Gannets (0 - 400 flocks per grid cell) and loons (0 - 200 flocks per grid cell; note species-specific scales for Figure 19-2 to Figure 19-5).

Root mean squared error for integrated models was generally lower, or at least equal to boat-based models when predicting back to the original boat and aerial survey data (Table 19-8). Boat-based predictions, however, often outperformed integrated models when predicting to independent boat and aerial survey data (i.e., predicting to a different survey in the same season, Table 19-8).

## Discussion

Jointly modeling aerial and boat survey data can improve our understanding of several important ecological phenomena important to proposed wind energy development, especially (1) clustering of marine wildlife within the study area and (2) relationships between marine wildlife abundance and spatially varying habitat covariates. An integrated approach utilizes beneficial aspects of both survey methods, with study area predictions informed by both aerial surveys, which encompassed a large geographic area, and boat surveys that allowed for estimation of detection probability (see Chapters 11-12, and 18; Winiarski et al. 2014). The integrated model presented herein had noticeable improvements in predicting local hotspots and marine bird distribution relative to models that only included boat-based data. The integrated model, however, had relatively low predictive power to independent surveys (data collected from a survey different than the one used to fit the models), which was likely a consequence of interseasonal variation in local hotspots, changes in habitat covariates, and possibly changes in the relationships with those covariates (Winiarski et al. 2013, Winiarski et al. 2014, Chapter 18).

In general, habitat relationships were similar to those presented in Chapter 18. For instance, relationships with distance to shore were consistently negative for terns, loons, and Northern Gannets in boat-based models herein and in Chapter 18. Conversely, parameter estimates for distance to shore were sometimes reversed (positive) in the integrated model. Both boat-based and integrated models, however, still predicted increased nearshore distribution patterns (e.g., areas of higher predicted abundance for Northern Gannet were generally closer to shore). Distance to shore and the aerial covariate used in the integrated model were collinear for all species ($\leq 0.60$), but likely not strong enough to completely explain the differences in the distance to shore parameter estimate. Instead, integrating aerial data likely provided more information on areas of particularly high or low abundances, and was a more informative covariate than the less variable distance to shore covariate. These examples demonstrate the necessity of investigating both parameter estimates and predicted abundance maps to determine proper interpretations from boat-based and integrated models.

In line with our hypotheses, integrated models improved the identification of abundance hotspots and areas of lower than expected abundances. The greater spatial coverage of aerial surveys improved the detection of latitudinal gradients and hotspots, especially those occurring outside of areas surveyed by the boat. Covariate relationships identified during boat surveys, however, remained important predictors of marine bird abundance and distribution. Differences between modeling approaches were most evident for Northern Gannets and loons, where integrated models predicted increased clustering relative to boat-based model predictions. For loons, integrated models predicted both a distance to shore effect and latitudinal gradient, with abundances highest near the mouth of the Chesapeake Bay (southwest region of the study area). Similarly, integrated model predictions for Northern Gannets were clustered near the mouth of the Chesapeake Bay and a few nearshore areas off the coasts of Virginia.

Due to the high proportion of birds in the aerial dataset that were not identified to species, we could not model species-specific abundances (e.g., Common and Red-throated Loons; see Chapter 16 for details). However, one goal for creating an integrated model with the two datasets is to estimate species-specific abundances by accounting for birds that were not identified to species. This is not straightforward, as the work in Chapter 16 uses the information from the boat surveys to inform species identification in the aerial survey, while in the joint model, we want to use both data simultaneously to estimate abundance. After developing the fully integrated models, we hope to incorporate a model for species identification like the one presented in Chapter 16. Until that point, our integrated approach is limited to the best available data for each survey type, and we will continue to use species groups (e.g., loons).

Integrated models were an improvement over boat-only models at identifying hotspots and predicting to the original surveys. Boat-only models, however, were better at predicting patterns observed in independent surveys conducted in the same season, which are likely longer-term patterns of abundance. Short-term changes in local hotspot locations, possibly due to shifting prey distributions, affect the predictive ability of both modeling approaches. Predictions from integrated models may be particularly affected by shifting hotspot locations, as aerial data from one sampling period will not necessarily improve predictions for a different sampling period. Further exploration of seasonal, annual, and species-specific differences beyond those explored here will help evaluate the performance of integrated models. Overall, integrated models improved identification of important habitat use areas, but further work is required to explore their predictive ability across surveys.

Developing new approaches to jointly model disparate datasets can improve identification of important habitat use areas, while also providing a framework to compare and possibly combine historical (i.e., boat-based) and new sources of data (i.e., boat-based and high definition videography). Here, we have used the aerial data as a covariate for estimating abundance from the boat data, but have not formally integrated the two data types. Identification of positive relationships between boat and aerial surveys suggests that these survey types often identify similar trends in abundance and distribution across the study area. Verifying the consistency of these results across multiple surveys, species, and geographic areas will be vital to developing fully integrated modeling approaches.

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## Figures and tables

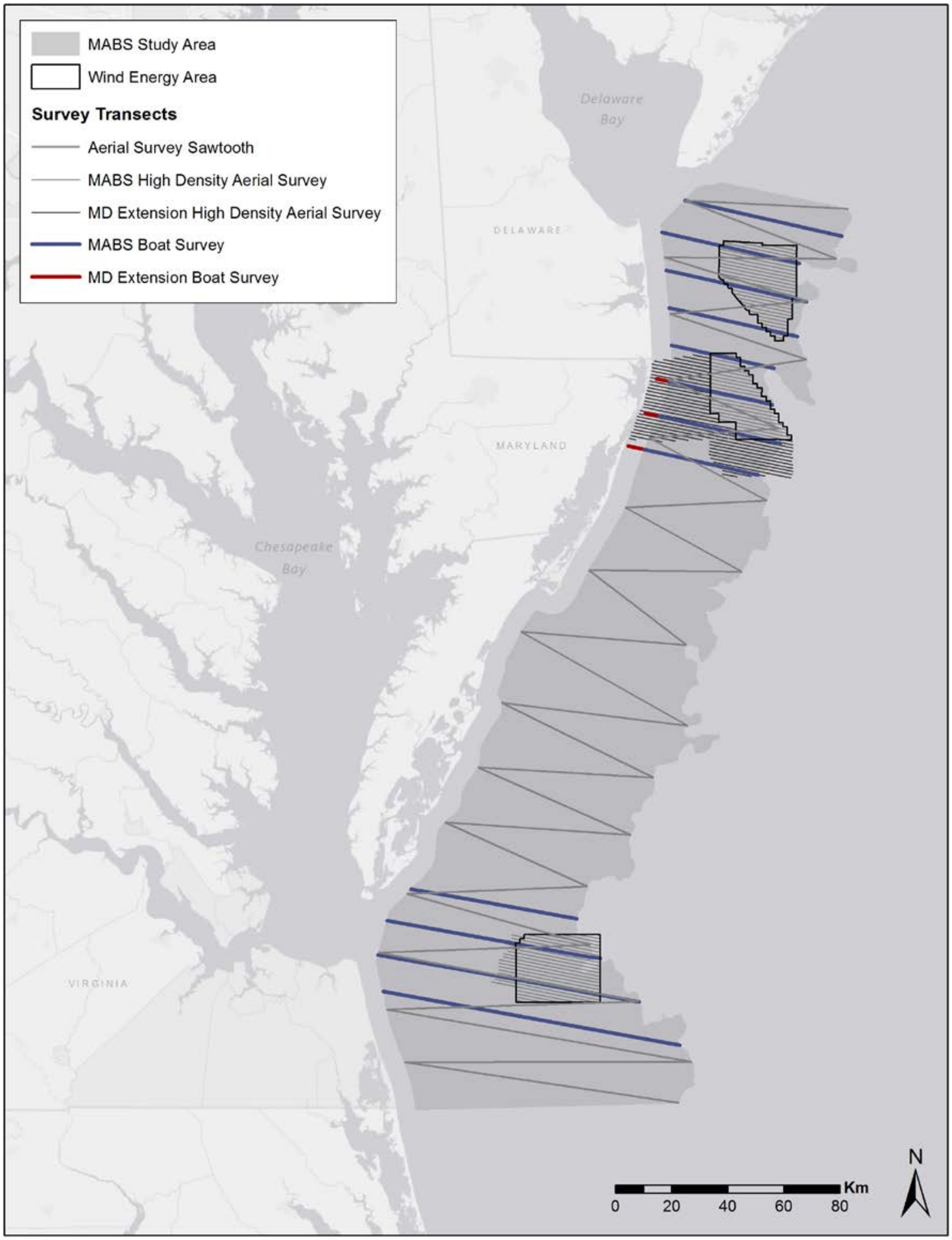

**Figure 19-1. Study area.** Boat transects are shown in blue and red, and aerial transects in gray; Maryland extension transects (funded by the state of Maryland and conducted only in the second year of surveys) are shown in red (boat) and dark gray (aerial). Department of Energy (DOE)-funded high density aerial surveys were located within federally designated wind energy areas (WEAs).

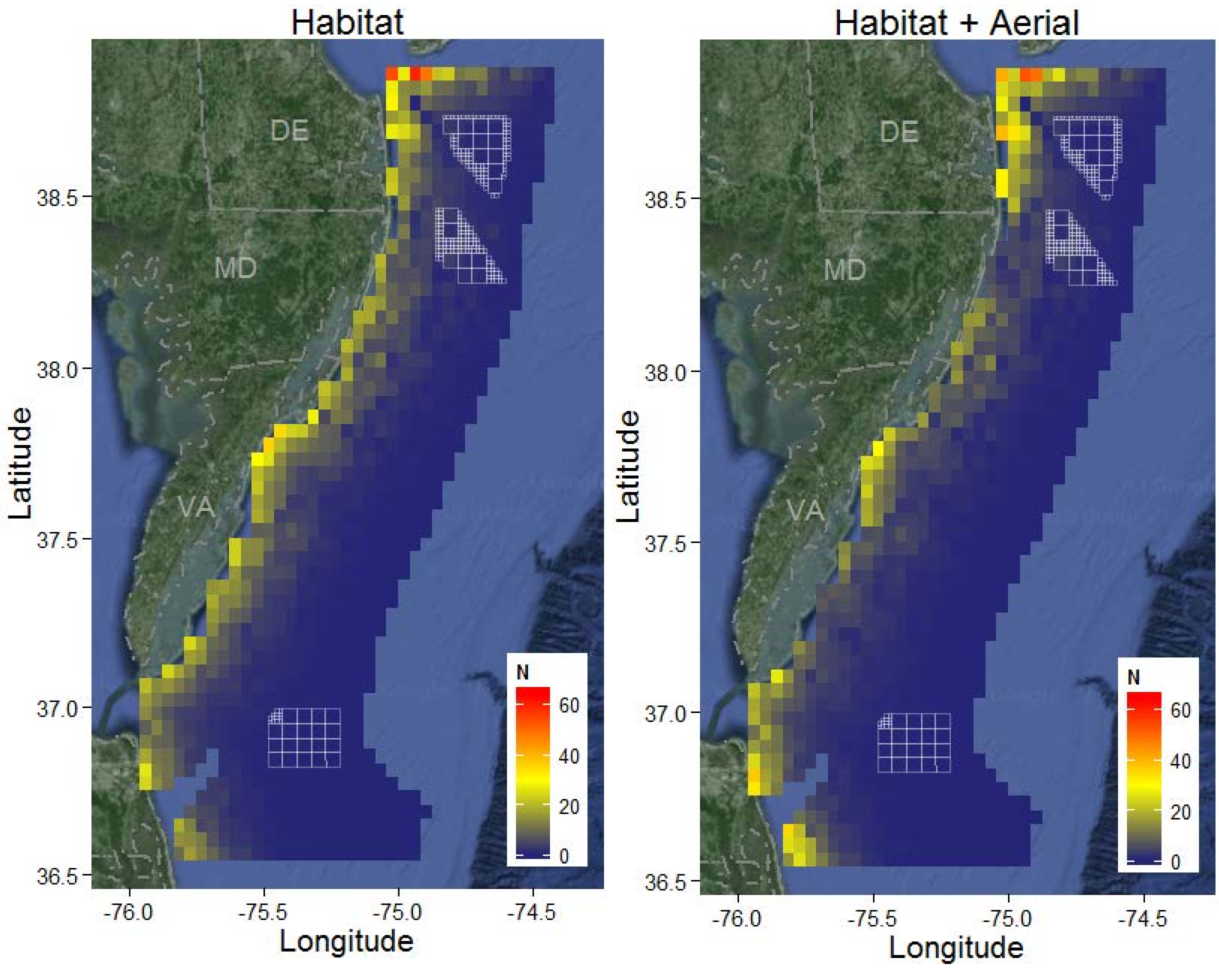

**Figure 19-2. Predicted abundance of tern flocks using the boat-based model (Habitat, left) or integrated model (Habitat + Aerial, right).** Note species-specific flock abundance scale. Median tern flock size was 1.0 individuals (Table 19-1). Covariate values were from the midpoint date for the summer season (25 July 2013). White grids represent proposed wind energy areas. See Table 19-1.

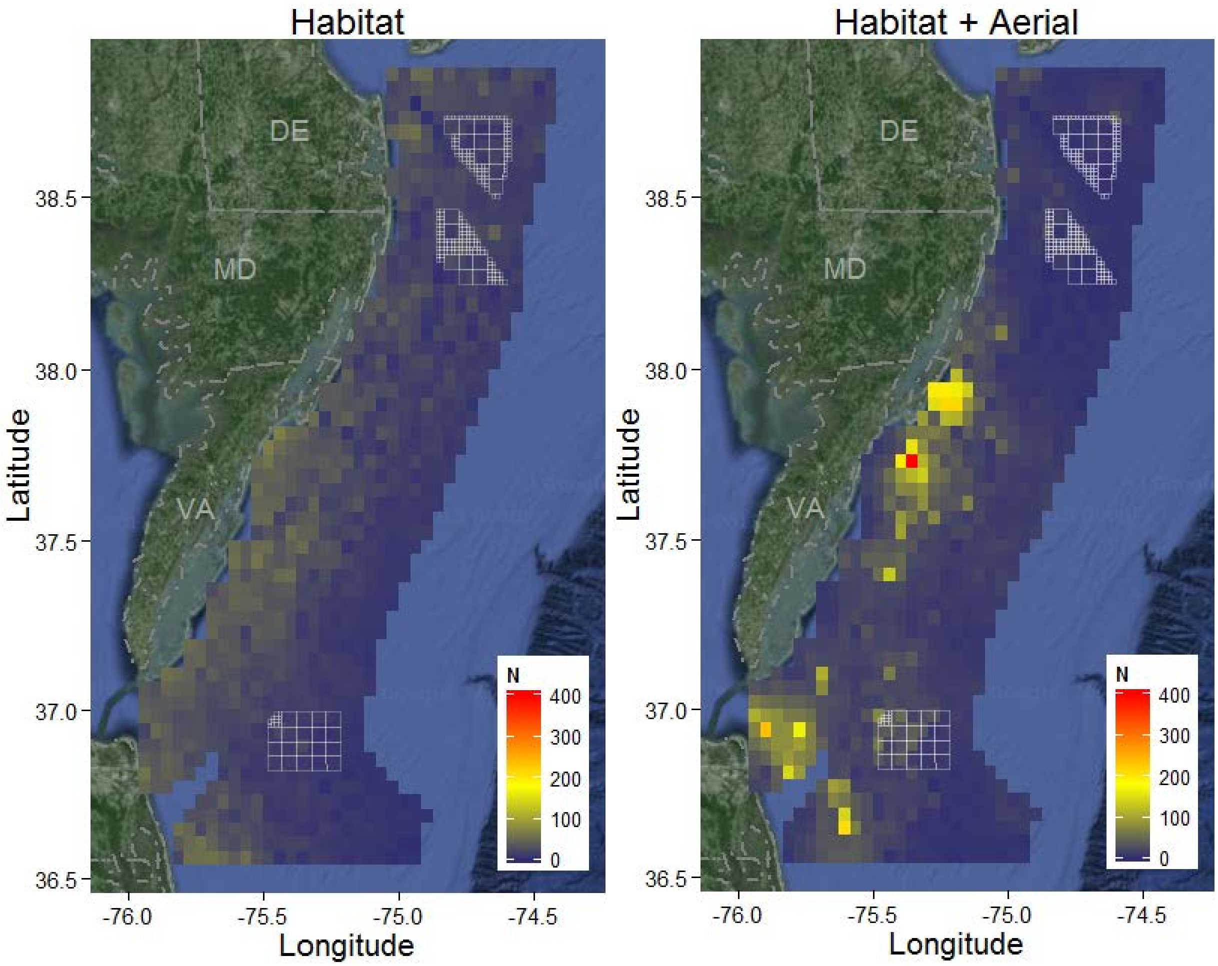

**Figure 19-3. Predicted abundance of Northern Gannet flocks using the boat-based model (Habitat, left) or integrated model (Habitat + Aerial, right).** Note species-specific flock abundance scale. Median Northern Gannet flock size was 1.0 individuals (Table 19-1). Covariate values were from the midpoint date for the winter season (25 December 2012). White grids represent proposed wind energy areas.

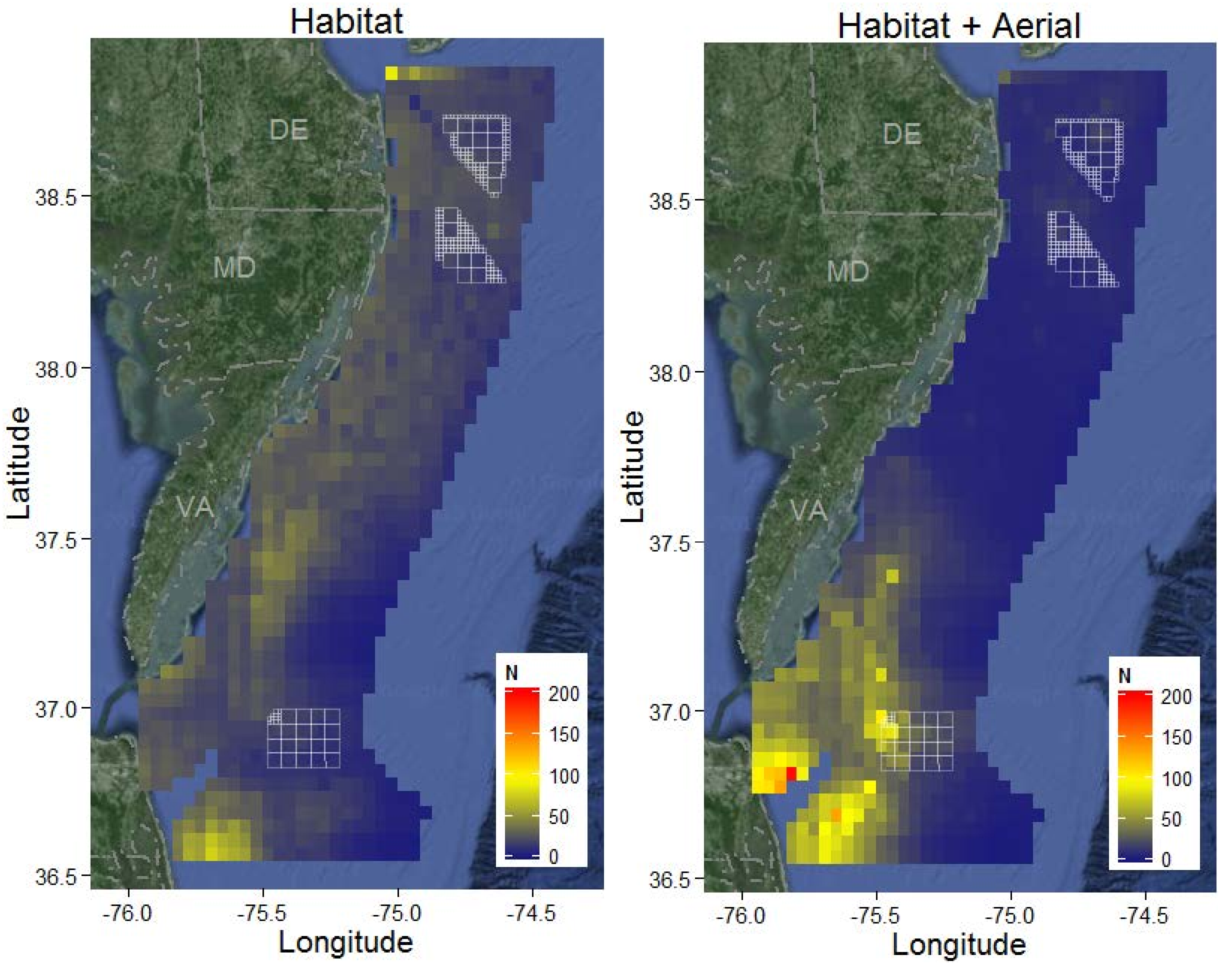

**Figure 19-4. Predicted abundance of loon flocks using the boat-based model (Habitat, left) or integrated model (Habitat + Aerial, right).** Note species-specific flock abundance scale. Median loon flock size was 1.0 individuals (Table 19-1). Covariate values were from the midpoint date for the winter season (25 December 2012). White grids represent proposed wind energy areas.

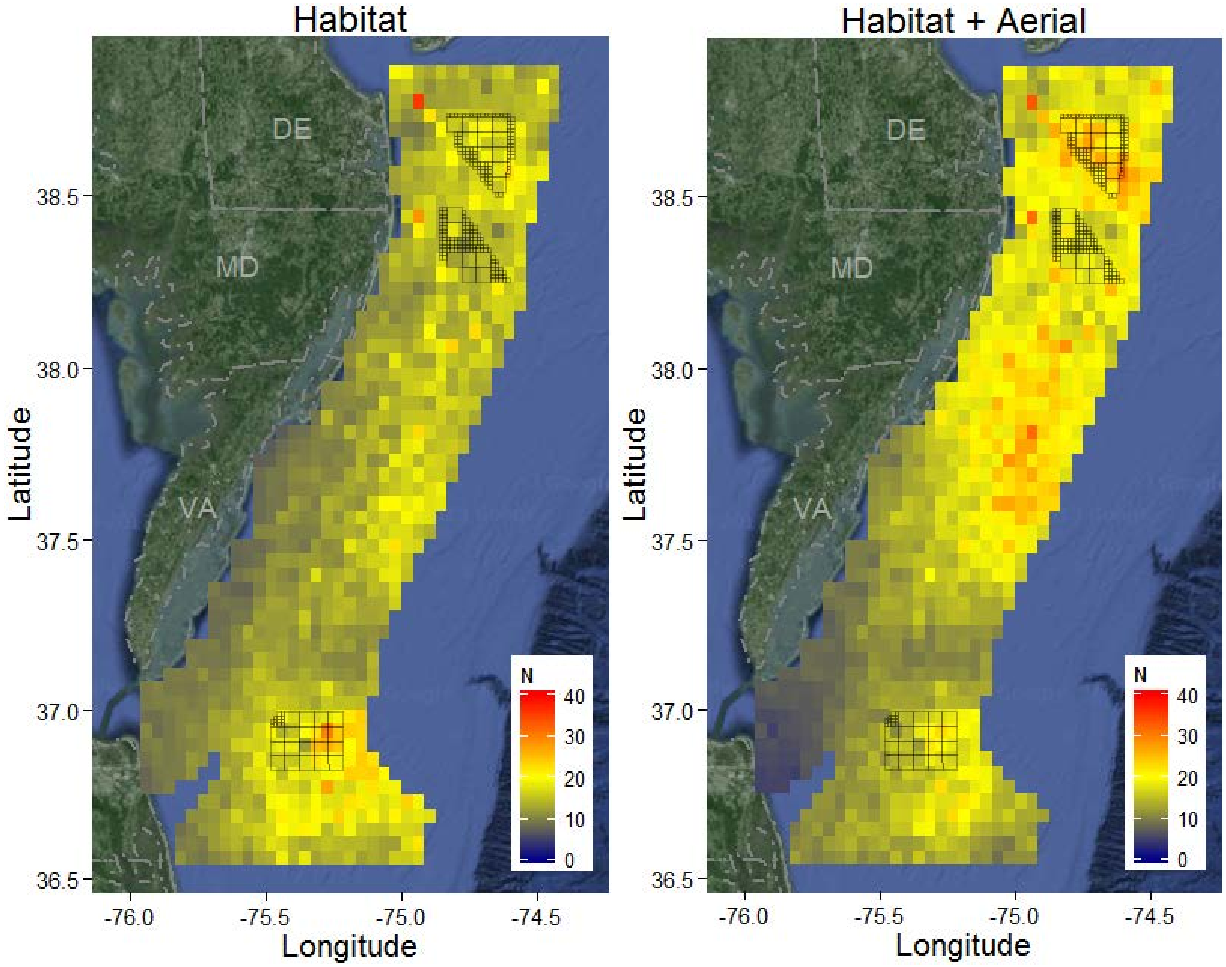

**Figure 19-5. Predicted abundance of alcid flocks using the boat-based model (Habitat, left) or integrated model (Habitat + Aerial, right).** Note species-specific flock abundance scale. Median alcid flock size was 2.0 individuals (Table 19-1). Covariate values were from the midpoint date for the winter season (25 December 2012). Black grids represent proposed wind energy areas.

**Table 19-1. Surveys used in the analysis for each species/group, and the raw counts (observations) for each species/group.**

| Boat Survey | Aerial survey | Group | Boat observations | | | | | Aerial observations |
|---|---|---|---|---|---|---|---|---|
| | | | Flocks | Flock size | | | | |
| | | | | Mean | Median | Min | Max | Individuals |
| Aug 2013 | Sep 2013 | Terns | 67 | 1.7 | 1.0 | 1 | 30 | 69 |
| Dec 2012 | Dec 2012 | Northern Gannets | 306 | 3.4 | 1.0 | 1 | 350 | 407 |
| Dec 2012 | Dec 2012 | Loons | 299 | 1.4 | 1.0 | 1 | 25 | 703 |
| Dec 2012 | Dec 2012 | Alcids | 122 | 2.3 | 2.0 | 1 | 18 | 148 |

**Table 19-2. Bayesian p-values for the abundance and detection components of the models, using either the boat-based model (Habitat) or integrated model (Habitat +Aerial).** Values close to 0.5 indicate good model fit.

| Group | Model | Boat | | Aerial |
|---|---|---|---|---|
| | | Abundance | Detection | Abundance |
| Terns | Habitat | 0.51 | 0.48 | NA |
| | Habitat + Aerial | 0.51 | 0.48 | 0.44 |
| Northern Gannets | Habitat | 0.44 | 0.23 | NA |
| | Habitat + Aerial | 0.48 | 0.23 | 0.57 |
| Loons | Habitat | 0.57 | 0.32 | NA |
| | Habitat + Aerial | 0.58 | 0.31 | 0.26 |
| Alcids | Habitat | 0.58 | 0.31 | NA |
| | Habitat + Aerial | 0.58 | 0.31 | 0.44 |

**Table 19-3. Predicted flock abundance of each species/group to a representative summer day (25 Jul. 2013; terns) or winter day (25 Dec. 2012; Northern Gannets, loons, and alcids) using either the boat-based model (Habitat) or integrated model (Habitat +Aerial).** Prediction area was constant across species and designed to represent the surveyed area (see Figure 19-2, Figure 19-3, Figure 19-4, and Figure 19-5).

| Boat survey | Species/Group | Predicted abundance | |
|---|---|---|---|
| | | Habitat | Habitat + Aerial |
| Aug 2013 | Terns | 3,726.8 | 3,366.6 |
| Dec 2012 | Northern Gannets | 19,576.9 | 20,275.4 |
| Dec 2012 | Loons | 18,503.6 | 17,371.6 |
| Dec 2012 | Alcids | 12,969.0 | 14,653.6 |

**Table 19-4. Parameter estimates for terns from the boat based-model (Habitat) or integrated model (Habitat + Aerial).** Abundance was modeled using a Negative Binomial distribution. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $\boldsymbol{r}$ is the overdispersion parameter, $\boldsymbol{\alpha}$ and $\boldsymbol{\beta}$ parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Aerial = aerial covariate (i.e., smoothed aerial counts), and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% credible interval does not overlap zero are in bold italics.

| **Terns** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Negative Binomial** | | **Habitat** | | | | **Habitat + Aerial** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept**; $\alpha_0$ | ***-2.12*** | 0.35 | -2.85 | -1.48 | ***-2.29*** | 0.39 | -3.10 | -1.58 |
| | **Dst**; $\alpha_1$ | ***-1.91*** | 0.37 | -2.67 | -1.23 | ***-2.12*** | 0.42 | -2.99 | -1.35 |
| | **Slp**; $\alpha_2$ | -0.33 | 0.25 | -0.86 | 0.15 | -0.39 | 0.26 | -0.94 | 0.10 |
| | **Grn**; $\alpha_3$ | 0.22 | 0.23 | -0.22 | 0.68 | 0.18 | 0.23 | -0.27 | 0.64 |
| | **Sst**; $\alpha_4$ | -0.20 | 0.30 | -0.79 | 0.41 | -0.29 | 0.33 | -0.96 | 0.33 |
| | **Sal**; $\alpha_5$ | -0.31 | 0.23 | -0.77 | 0.14 | -0.07 | 0.29 | -0.62 | 0.52 |
| | **Aerial**; $\alpha_6$ | - | - | - | - | -0.26 | 0.18 | -0.63 | 0.08 |
| | **Overdisp.**; $r$ | 1.10 | 0.74 | 0.38 | 2.89 | 1.17 | 0.93 | 0.39 | 3.17 |
| **Detection** | **Beaufort 0-2**;$\beta_0$ | ***5.28*** | 0.11 | 5.08 | 5.50 | ***5.28*** | 0.11 | 5.08 | 5.50 |
| | **Beaufort 3-6**; $\beta_1$ | ***5.02*** | 0.14 | 4.76 | 5.32 | ***5.03*** | 0.14 | 4.77 | 5.32 |

**Table 19-5. Parameter estimates for Northern Gannets from the boat based-model (Habitat) or integrated model (Habitat + Aerial).** Abundance was modeled using a Negative Binomial distribution. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $\boldsymbol{r}$ is the overdispersion parameter, $\boldsymbol{\alpha}$ and $\boldsymbol{\beta}$ parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Aerial = aerial covariate (i.e., smoothed aerial counts), and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% credible interval does not overlap zero are in bold italics.

| **Northern Gannets** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Negative Binomial** | | **Habitat** | | | | **Habitat + Aerial** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept**; $\alpha_0$ | 0.11 | 0.14 | -0.17 | 0.41 | 0.11 | 0.13 | -0.15 | 0.37 |
| | **Dst**; $\alpha_1$ | ***-0.48*** | 0.21 | -0.88 | -0.07 | ***0.80*** | 0.32 | 0.21 | 1.46 |
| | **Slp**; $\alpha_2$ | -0.33 | 0.18 | -0.68 | 0.02 | -0.31 | 0.16 | -0.64 | 0.00 |
| | **Grn**; $\alpha_3$ | 0.20 | 0.16 | -0.13 | 0.52 | 0.00 | 0.15 | -0.31 | 0.29 |
| | **Sst**; $\alpha_4$ | 0.06 | 0.18 | -0.30 | 0.40 | ***-0.86*** | 0.24 | -1.33 | -0.39 |
| | **Sal**; $\alpha_5$ | -0.14 | 0.19 | -0.54 | 0.21 | -0.24 | 0.19 | -0.64 | 0.12 |
| | **Aerial**; $\alpha_6$ | - | - | - | - | ***0.86*** | 0.17 | 0.54 | 1.24 |
| | **Overdisp.**; $r$ | 0.54 | 0.10 | 0.37 | 0.76 | 0.77 | 0.17 | 0.50 | 1.14 |
| **Detection** | **Beaufort 0-2**;$\beta_0$ | ***5.67*** | 0.05 | 5.58 | 5.77 | ***5.66*** | 0.05 | 5.57 | 5.76 |
| | **Beaufort 3-6**; $\beta_1$ | ***5.80*** | 0.08 | 5.65 | 5.97 | ***5.83*** | 0.08 | 5.68 | 6.01 |

**Table 19-6. Parameter estimates for loons from the boat based-model (Habitat) or integrated model (Habitat + Aerial).** Abundance was modeled using a Negative Binomial distribution. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, $\alpha$ and $\beta$ parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Aerial = aerial covariate (i.e., smoothed aerial counts), and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% credible interval does not overlap zero are in bold italics.

| **Loons** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Negative Binomial** | | **Habitat** | | | | **Habitat + Aerial** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept**; $\alpha_0$ | ***0.26*** | 0.12 | 0.03 | 0.49 | 0.02 | 0.12 | -0.22 | 0.25 |
| | **Dst**; $\alpha_1$ | ***-0.86*** | 0.20 | -1.25 | -0.48 | 0.09 | 0.26 | -0.45 | 0.60 |
| | **Slp**; $\alpha_2$ | -0.08 | 0.11 | -0.30 | 0.15 | 0.02 | 0.11 | -0.20 | 0.24 |
| | **Grn**; $\alpha_3$ | 0.06 | 0.12 | -0.17 | 0.30 | 0.00 | 0.11 | -0.21 | 0.22 |
| | **Sst**; $\alpha_4$ | ***0.69*** | 0.15 | 0.40 | 0.98 | -0.39 | 0.26 | -0.89 | 0.15 |
| | **Sal**; $\alpha_5$ | ***-0.51*** | 0.14 | -0.79 | -0.25 | ***-0.44*** | 0.12 | -0.68 | -0.22 |
| | **Aerial**; $\alpha_6$ | - | - | - | - | ***1.26*** | 0.26 | 0.75 | 1.77 |
| | **Overdisp.**; $r$ | 1.42 | 0.42 | 0.81 | 2.42 | 1.94 | 0.65 | 1.04 | 3.55 |
| **Detection** | **Beaufort 0-2**; $\beta_0$ | ***5.42*** | 0.05 | 5.33 | 5.52 | ***5.41*** | 0.05 | 5.33 | ***5.50*** |
| | **Beaufort 3-6**; $\beta_1$ | ***5.48*** | 0.09 | 5.33 | 5.66 | ***5.53*** | 0.09 | 5.36 | ***5.71*** |

**Table 19-7. Parameter estimates for alcids from the boat based-model (Habitat) or integrated model (Habitat + Aerial).** Abundance was modeled using a Poisson distribution. SD is the standard deviation, 2.5% and 97.5% are the respective quantiles, $r$ is the overdispersion parameter, $\alpha$ and $\beta$ parameters are on the log scale. Dst = distance to shore, Slp = slope of the seafloor, Grn = sediment grain size, Sst = sea surface temperature, Sal = salinity, Aerial = aerial covariate (i.e., smoothed aerial counts), and Beaufort sea state 3-6 are rough seas (as opposed to calm, 0-2). The posterior mean for covariates where the 95% credible interval does not overlap zero are in bold italics.

| **Alcids** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Poisson** | | **Habitat** | | | | **Habitat + Aerial** | | | |
| **Component** | **Term** | **Mean** | **SD** | **2.5%** | **97.5%** | **Mean** | **SD** | **2.5%** | **97.5%** |
| **Abundance** | **Intercept**; $\alpha_0$ | -0.02 | 0.12 | -0.26 | 0.21 | 0.03 | 0.12 | -0.21 | 0.27 |
| | **Dst**; $\alpha_1$ | 0.09 | 0.15 | -0.20 | 0.37 | -0.20 | 0.25 | -0.70 | 0.30 |
| | **Slp**; $\alpha_2$ | 0.08 | 0.09 | -0.10 | 0.25 | 0.07 | 0.09 | -0.12 | 0.23 |
| | **Grn**; $\alpha_3$ | -0.12 | 0.10 | -0.33 | 0.08 | -0.10 | 0.10 | -0.31 | 0.10 |
| | **Sst**; $\alpha_4$ | -0.02 | 0.14 | -0.28 | 0.25 | 0.30 | 0.26 | -0.21 | 0.82 |
| | **Sal**; $\alpha_5$ | -0.15 | 0.10 | -0.33 | 0.06 | -0.17 | 0.10 | -0.36 | 0.04 |
| | **Aerial**; $\alpha_6$ | - | - | - | - | -0.27 | 0.19 | -0.65 | 0.11 |
| **Detection** | **Beaufort 0-2**; $\beta_0$ | ***5.07*** | 0.07 | 4.93 | 5.22 | ***5.07*** | 0.07 | 4.94 | 5.22 |
| | **Beaufort 3-6**; $\beta_1$ | ***4.79*** | 0.13 | 4.54 | 5.05 | ***4.77*** | 0.13 | 4.53 | 5.03 |

**Table 19-8. Root mean squared error (RMSE) evaluating the ability of each model to predict abundance in (a) the original boat and aerial datasets (Original surveys) and (b) independent boat and aerial datasets (Independent surveys)**. Predictions were from the boat-based (Habitat) or integrated model (Habitat + Aerial). Dates for the original surveys and independent surveys are provided. Root mean squared error values closer to zero indicate better model fit (lower value for each comparison is in bold italics).

| **(a) Original surveys** | | **Original survey date** | **RMSE** | |
|---|---|---|---|---|
| **Group** | **Dataset** | | **Habitat** | **Habitat + Aerial** |
| Terns | Boat | Aug-13 | 0.9 | 0.9 |
| | Aerial | Sep-13 | 0.7 | ***0.6*** |
| Northern Gannets | Boat | Dec-12 | 3.6 | ***3.1*** |
| | Aerial | Dec-12 | 1.8 | ***1.0*** |
| Loons | Boat | Dec-12 | 2.2 | ***2.0*** |
| | Aerial | Dec-12 | 3.2 | ***2.7*** |
| Alcids | Boat | Dec-12 | 1.0 | 1.0 |
| | Aerial | Dec-12 | ***1.1*** | 1.4 |

| **(b) Independent surveys** | | **Independent survey date** | **RMSE** | |
|---|---|---|---|---|
| **Group** | **Dataset** | | **Habitat** | **Habitat + Aerial** |
| Terns | Boat | Sep-13 | 1.0 | 1.0 |
| | Aerial | Jul-13 | ***0.5*** | 0.6 |
| Northern Gannets | Boat | Jan-13 | ***4.5*** | 14.4 |
| | Aerial | Feb-13 | ***21.5*** | 22.1 |
| Loons | Boat | Jan-13 | ***2.1*** | 3.9 |
| | Aerial | Feb-13 | ***3.2*** | 3.4 |
| Alcids | Boat | Jan-13 | 1.6 | 1.6 |
| | Aerial | Feb-13 | 1.7 | ***1.6*** |

# Introduction to Part V
# Individual movements and habitat use for focal bird species

## Report structure

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure I).

Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure I. In Parts II (Chapters 3-6) and III (Chapters 7-12), we describe methods and results for high resolution digital video aerial surveys and boat surveys, respectively. Part IV of this report (Chapters 13-19) combines data from both survey approaches to develop a comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area. Part V (Chapters 20-25) focuses on individual movements and habitat use of focal avian species, tracked via satellite telemetry; and Part VI (Chapters 26-27) focuses on population-level migratory movements over the oceans, using several approaches for studying nocturnal avian migration. An additional study effort, which further explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

## Part V: Individual movements and habitat use of focal bird species

Satellite telemetry allows us to track the movements of individual animals within their environment, and is an effective tool for understanding movement patterns and habitat use for a large number of marine species. Transmitters were only deployed on birds in this study, but there is analogous technology available for turtles and mammals (being deployed as part of the Atlantic Marine Assessment Program for Protected Species, or AMAPPS,[1] study among other efforts). With this method, temporal coverage is limited only by battery power and tag longevity. Due to power limitations such tracking is seldom continuous, but it is possible to track movements of individual birds over seasons or years regardless of weather or time of day. This level of individual temporal and spatial coverage requires expensive

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[1] www.nefsc.noaa.gov/psb/AMAPPS/

technology, however, which often has the effect of limiting samples sizes, and it can be difficult to extrapolate population-wide distributions from a few individuals (Lindberg and Walker 2014).

The project team investigated the spatial and temporal patterns of offshore bird migration and winter habitat use through a combination of satellite telemetry data, and determined how these data covary with environmental conditions using remotely collected covariate information. We tracked the movements of individuals from three focal avian taxa: seabirds, including the Red-throated Loon (*Gavia stellata)* and Northern Gannet (*Morus bassanus*); sea ducks (the Surf Scoter, *Melanitta perspicillata*); and raptors (the Peregrine Falcon, *Falco peregrinus*). The six chapters in Part V of this report are:

| | |
|---|---|
| Chapter 20. | Wintering movements and habitat use of Surf Scoters in the mid-Atlantic U.S. |
| Chapter 21. | Wintering movements and habitat use of Red throated Loons in the mid-Atlantic U.S. |
| Chapter 22. | Wintering movements and habitat use of Northern Gannets in the mid-Atlantic U.S. |
| Chapter 23. | Incorporating temporal variation in seabird telemetry data: time variant kernel density models. |
| Chapter 24. | Using state-space models to identify areas of persistent winter activity and their associated environmental covariates in Northern Gannets. |
| Chapter 25. | Offshore migration of Peregrine Falcons along the Atlantic flyway. |

Animal movement modeling generally falls into two categories: more “place-based” models that can be applied to groups of individuals, and models based on random walk behavior, which are more likely to be applied to individual animals (Smouse et al. 2010). Both of these approaches were utilized in this report.

### *“Place-based” group models: kernel density estimates and utilization distributions*

Traditional modeling approaches for tracking data are focused on predicting the probability of an animal’s occurrence at a given point in space, and have historically included “home range” analyses (Smouse et al. 2010, Fischer et al. 2013). Modern variations include the development of utilization distributions, which quantify the relative frequency distribution of an animal’s occurrence in space and time, and can be used to examine environmental correlates to high-use areas (Worton 1989; Smouse et al. 2010; e.g., Loring et al. 2014) and how frequency of use changes over time (Keating and Cherry 2009). In these approaches, consecutive observations are not linked (e.g., the sequences of positions are not used to add behavioral changes of individuals into the models). These modeling approaches are appropriate for understanding species’ habitat use within area-confined ranges. These methods were used in this report to examine wintering movements and habitat use of seabirds in the mid-Atlantic study area, as birds were largely expected to remain within the region during the winter period.

Kernel-based utilization distributions and resource selection functions identified important habitat use areas for Surf Scoters (Chapter 20). Scoters are likely to utilize more geographically stable prey resources than are Red-throated Loons or Northern Gannets, however (Chapter 18, Appendix 18A), and modifications to this approach using different resource selection methods and temporally variable environmental covariates were applied to these two species (Chapters 21 and 22, respectively). Surf Scoters in core-use areas utilized shallow (<40 m) areas within 4.5 km from shore. Red-throated Loons also tended to use nearshore areas, and in our sample, the greatest chance for interaction between Red-throated Loons and potential wind energy areas generally occurred during the spring migration period (late March to early May). In contrast, Northern Gannets ranged widely over the Outer Continental Shelf during winter (Chapter 22). Though core habitat within the study area included the protected inshore waters of the major bays and bay mouths, individual birds displayed extensive movements up and down the eastern seaboard between the core use areas, increasing the likelihood that they would encounter offshore wind developments in the region repeatedly throughout the winter.

Chapters 20-22 explored spatial patterns and movement of three target species using fairly traditional methods, which collapse the temporal component of movement data into a single period for analysis. Time-variant kernel density analysis allows for a more explicit presentation of habitat use areas through time (Keating and Cherry 2009; Chapter 23), which may prove useful for understanding the timing of these species' presence in and around development areas. Time variant kernel density maps proved to be an effective tool for showing use of the study area at fine temporal scales. Analysis showed limited overlap of habitat use with wind energy areas, with the most overlap occurring for Northern Gannets.

### *Individual-based models of movement and behavior*

Individual-based models attempt to estimate the detailed movements of an animal through the landscape, rather than aggregating observations into home ranges or similar distributions. These approaches are often based on correlated random walk models, in which stochastic differential equations are used to predict animal movements between observed locations (Smouse et al. 2010). State-space models can infer positions of animals where data gaps are present by modeling the underlying individual behavior that is used to move through space (Patterson et al. 2008, Schick et al. 2008, Smouse et al. 2010). Dynamic Brownian Bridge Movement Models (dBBMM) are data-intensive, but can serve as a useful "hybrid" approach that uses animals' movement paths to develop more refined utilization distributions than traditional place-based approaches (Kranstauber et al. 2012).

#### *State-space models*

Satellite telemetry can provide the opportunity to understand more detailed behaviors, as well as movements and general habitat use. Northern Gannet interactions with offshore wind energy development are hypothesized to largely occur as a result of their foraging behaviors, which include a large proportion of time spent soaring at or near the altitude of the rotor-sweep zone for offshore wind turbines (Garthe et al. 2000, Langston 2010). Being able to differentiate between foraging and other behaviors in telemetry data will allow us to better determine areas of potential conflict between offshore wind energy development and Northern Gannet habitat use, and could inform the siting and permitting of offshore wind energy development.

In Chapter 24, we use positional data in a behavioral state switching state-space model (SSSM) in a Bayesian modeling framework (Jonsen et al. 2007), to analyze telemetry data from the nonbreeding period. This more complex version of a correlated random walk model allows us to identify when these seabirds were employing "area-restricted search" foraging behavior versus transient searching behavior (when animals were likely moving between areas of high foraging availability). This analysis provides information on locations that were consistently used by Northern Gannets for foraging, and what habitat characteristics (e.g., water depth and sea surface temperature [SST] front density) defined these foraging areas. Weekly SST front density was a very strong predictor of foraging activity, which indicates that Northern Gannets dynamically responded to either the change in water temperature itself, or to the increase in prey availability that is likely occurring in areas with high front density. In order to understand what locations are most important for this species, it will be important to develop accurate models for predicting temperature fronts.

### *Dynamic Brownian bridge movement models*

Unlike the above seabird species, Peregrine Falcons migrate through the project study area but seldom winter in or near the mid-Atlantic. Peregrine Falcons are sometimes thought of as a terrestrial species, but they are probably the most commonly encountered non-piscivorous raptor in marine settings. Peregrines are commonly observed foraging or perching far from shore at offshore islands, oil drilling platforms, and large offshore vessels (Voous 1961, Cochran 1975, 1985, Russell 2005, Johnson et al. 2011, Desorbo et al. 2012). Their migratory movements, as well as the higher number of data points afforded by solar transmitters, led to the use of different methodological approaches to understand falcon movements and use of space within the project study area during fall migration (Chapter 25).The dBBMM was well-suited for this scenario as it accounts for the changing probabilities of space-use as the speed of animal also changes. For migratory species like the Peregrine Falcon, this method generates utilization distributions that are more accurate in depicting high and low use areas and migratory corridors compared to traditional methods (Kernohan et al. 2001, Kie et al. 2010, Kranstauber et al. 2012, Fischer et al. 2013).

During this study, falcons regularly used habitat hundreds of kilometers offshore along the Atlantic coast, including within and east of the mid-Atlantic study area. Twelve of the 13 tracked falcons that continued their fall migration beyond the mid-Atlantic coast initiated a significant transoceanic flight from coastal North Carolina to the Caribbean or South America. Birds tracked in this study were all captured on offshore islands, and it remains unclear what proportion of the full Peregrine Falcon population ventures offshore. However, findings from this study are consistent with observations elsewhere (Cochran 1975, Fuller et al. 1998, Desorbo et al. 2012), and suggest that this species commonly uses offshore habitats along the Atlantic flyway.

## *Implications*

The inclusion of satellite telemetry provides information on broad-scale movements of specific species in the environment, including nocturnal movements and habitat use, which is missing from our survey data. Depending upon the chosen analytical approach, satellite telemetry data can provide information at a variety of scales. The SSSM in Chapter 24 focuses on individual foraging decisions, for example, and thus examines Northern Gannet behavior at a different scale than the utilization distributions of Chapters 20-23 (which are "populations" of tagged birds), or boat or aerial surveys of population distributions (Parts II-IV of this report). Our ability to combine broad-scale analyses of population distributions and important habitat areas with finer-scale analyses of behavioral data may have implications for assessing risk. Predictions of the effects of offshore wind energy development on Northern Gannets in the mid-Atlantic, for example, must consider exposure (e.g., whether they are present in an area in large numbers) as well as well as hazards (that is, whether they are foraging, a behavior which has been suggested to be linked to collision risk as well as displacement; Lindeboom et al. 2011, Furness et al. 2013, Vanermen et al. 2014). Within the core use areas and persistent habitat use areas identified elsewhere in this report (shallower waters closer to shore; Chapters 12, 17, 19, and 22), gannets seem to be preferentially foraging in deeper portions of these areas with high numbers of temperature fronts; as a result, these may be the highest risk areas for interactions with offshore wind, even if population-level abundance is lower in those areas than in some shallower locations.

**Acknowledgments:** Part V of this report includes data from several collaborative field studies focused on the movement patterns of Red-throated Loons, Northern Gannets, Surf Scoters, and Peregrine Falcons in the Atlantic flyway, carried out by numerous principal investigators and agencies.

The work reported in Chapters 20-24 constitutes part of two multi-year collaborative projects: 1) Determining Offshore Use of Diving Bird Species in Federal Waters of the Mid-Atlantic United States Using Satellite Tracking, developed by the Bureau of Ocean Energy Management (BOEM) and the U.S. Fish & Wildlife Service (USFWS) and 2) the Atlantic and Great Lakes Sea Duck Migration Study, developed by the USFWS, Canadian Wildlife Service (CWS), and other state and federal partners. The first study was designed in partnership with the U.S. Geological Survey (USGS)-Patuxent Wildlife Research Center, Biodiversity Research Institute (BRI), and Memorial University of Newfoundland (MUN), and funded by BOEM, DOE, BRI, MUN, and The Bailey Wildlife Foundation. The second project was primarily funded by the USFWS through their support of the Sea Duck Joint Venture, as well as by DOE and by BOEM. In addition to these funding entities and the authors of these chapters, other partners that provided logistical or financial support include the Maryland Department of Natural Resources, Virginia Department of Game and Fisheries, University of Rhode Island, Delaware Division of Fish and Wildlife, and North Carolina Wildlife Resource Commission. Chapters 20-23 cover all data through the end of 2013, including the first two years of the BOEM/USFWS study and all prior years of the SDJV study. Chapter 24 includes Northern Gannet data through the end of 2014, including the first three years of the BOEM/USFWS study.

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**Figure I. Organization of chapters within this final report.**

# Chapter 20: Wintering movements and habitat use of Surf Scoter (*Melanitta perspicillata*) in the mid-Atlantic U.S.

### Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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[8]Sea Duck Joint Venture, Anchorage, AK

**Project webpage:** *www.briloon.org/mabs*

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# Chapter 20 Highlights

*Wintering movements and habitat use of Surf Scoters in the mid-Atlantic U.S.*

## *Context*[1]

Wintering habitat use and migratory pathways are key issues facing sea ducks in the mid-Atlantic U.S. and elsewhere due to the potential for offshore wind energy development. A comprehensive understanding of important winter habitats and environmental characteristics determining sea duck abundance and distribution is paramount in advising marine spatial planning efforts in the region and identifying key resource areas for sea ducks to minimize potential threats posed by offshore wind energy development. This chapter explores spatial patterns and movement of Surf Scoters using satellite telemetry data to identify migratory chronology and pathways, as well as determine utilization distributions and core-use areas during the wintering period. Remotely-collected environmental covariate data were also incorporated to develop resource selection models to help identify valuable scoter habitat. Similar studies were conducted on Red-throated Loons and Northern Gannets, and are discussed in Chapters 21 and 22, respectively. Additionally, time variant kernel density models were developed using movements of all three species to examine broad scale movement patterns throughout the year (Chapter 23).

## *Study goal/objectives*

Investigate the spatial patterns, temporal patterns, and environmental variation in Surf Scoter migration and winter habitat use through a combination of satellite telemetry data and remotely collected environmental covariate information.

## *Highlights*

- Surf Scoters in core-use areas utilized shallow (<40 m) areas within 4.5 km from shore.
- Resource selection models suggest that other dynamic variables such as sea surface temperatures, productivity, and salinity (and selected interactions among them) may also be important in determining valuable scoter habitat.
- Migration chronology of birds tracked in this study suggests that Surf Scoters wintering and migrating throughout the mid-Atlantic region could encounter future offshore wind energy facilities between mid-October and early May.

## *Implications*

This analysis indicates Surf Scoters tagged along near-shore areas of the mid-Atlantic have a minimal likelihood of overlapping with current mid-Atlantic Wind Energy Areas (WEAs). Activities associated with construction within WEAs, such as installation of transmission lines or vessel traffic within nearshore areas, would have a higher likelihood of overlapping with wintering Surf Scoters in the mid-Atlantic.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

Wintering habitat use and migratory pathways are key issues facing sea ducks in the mid-Atlantic U.S. and elsewhere due to the potential for offshore wind energy development. A comprehensive understanding of important winter habitats and environmental characteristics determining sea duck abundance and distribution is paramount in advising marine spatial planning efforts in the region and identifying key resource areas for sea ducks. We captured and tracked 101 Surf Scoters to investigate the spatial patterns, temporal patterns, and environmental variation in migration and winter habitat use through a combination of satellite telemetry data and remotely collected environmental covariate information. We found that Surf Scoters in core-use areas utilized shallow (<40 m) areas within 4.5 km from shore. Resource selection models suggest that other dynamic variables such as sea surface temperatures, productivity, and salinity (and selected interactions among them) may also be important in determining valuable scoter habitat. Migration chronology of birds tracked in this study suggests that Surf Scoters wintering and migrating throughout the mid-Atlantic region could encounter future offshore wind energy facilities between mid-October and early May. Our analyses indicate Surf Scoters tagged along near-shore areas of the mid-Atlantic have a minimal likelihood of overlapping with current Wind Energy Areas (WEAs) in the mid-Atlantic, though activities associated with construction within WEAs, such as installation of transmission lines or vessel traffic within nearshore areas, or possible development of wind farms closer to shore and outside currently designated WEAs, may have a higher likelihood of overlapping with wintering Surf Scoters in the mid-Atlantic.

## Introduction

As development of renewable energy sources such as offshore wind power progresses towards large-scale development in the United States, the need for effective pre-construction surveys and site planning is paramount. Several development sites along the mid-Atlantic Outer Continental Shelf of the United States have been proposed for large offshore wind energy facilities. To date, commercial wind energy leases have been issued for offshore areas in Massachusetts, Rhode Island, Delaware, Maryland, and Virginia. These areas coincide with important staging and wintering habitat for several sea duck species (Silverman et al. 2013), for which habitat use, migration pathways, and general biology has only recently been studied (Sea Duck Joint Venture 2014). Population declines in several species of North American sea ducks (Bowman et al. 2015, Sea Duck Joint Venture Management Board 2014) have led to increased research on how environmental and anthropogenic factors in various stages of the annual cycle may affect survival, productivity, habitat use, and site fidelity (Provencher et al. 2014, Perry et al. 2007, Skerratt et al. 2005, Merkel 2004).

No operational offshore wind energy facilities currently exist in U.S. waters, however the Deepwater Wind project located off the coast of Block Island, RI is currently under construction. Although planning for offshore wind energy and assessing its potential effects on wildlife is a recent development in the U.S. (Loring et al. 2014, Goodale and Milman 2014), European nations have been developing offshore wind energy and studying its impacts on wildlife for several decades (Guillemette and Larsen 2002, Desholm and Kalhert 2005, Langston 2013). The main concerns regarding potential interactions between birds and wind farms involve collision risk, disturbance, and indirect effects such as impacts to habitat and prey base (Fox et al. 2006). European studies have demonstrated that collision risk is probably minimal for sea ducks in many locations (Desholm and Kahlert 2005), but avoidance behaviors may have an effect. Common Eiders (*Somateria mollissima*) at a wind farm in Denmark were found to avoid flying close to or amongst turbines, determining that habitat use within and around wind farms may be greatly reduced (Larsen and Guillemette 2007). These avoidance behaviors may be trivial when considering the energetic costs of a long-distance migration, but the cumulative effects of multiple developments along a migration route could be significant (Madsen et al. 2009).

Sea ducks feed primarily on mollusks and benthic invertebrates in shallow, subtidal areas where wind farms are likely to be developed, at least initially. A study at the Horns Rev 1 wind facility in Denmark found a short-term (three year) displacement of Common Scoter (*Melanitta nigra*) followed by a return to the wind farm area (Petersen and Fox 2007). It is unclear whether habituation, changes in prey distribution, or other factors initiated the birds' return. This potential loss of foraging habitat (temporary or otherwise) in areas with large congregations of wintering sea ducks could have detrimental population-level effects. Recent studies have suggested that habitat conditions and prey availability during the wintering period may have strong impacts on reproductive success and productivity during the subsequent breeding season (Camphuysen et al. 2002, Oosterhuis and van Dijk 2002). A recent study tracking Black Scoters (*Melanitta americana)* through the migratory and wintering period in southern New England found considerably larger home range sizes in scoters than other sea duck species (Loring et al. 2014). That study also demonstrated the tendency for scoters to occasionally venture outside near shore core-use areas to locations further offshore, suggesting an increased likelihood of encountering offshore wind energy facilities scattered throughout coastal areas.

In this study, we captured and outfitted Surf Scoters (*Melanitta perspicillata*) with abdominally-implanted satellite transmitters to track daily and seasonal movements during the wintering and migratory periods off the mid-Atlantic coast of the U.S. Data collected from satellite-tagged Surf Scoters from Sea Duck Joint Venture (SDJV) collaborative studies, dating as far back as 2000, are also included in this report. To assess the extent of potential exposure to proposed offshore wind energy development, we determined median arrival dates, departure dates, and length of stay in the mid-Atlantic study area. We also performed habitat utilization analyses and determined prominent biotic and abiotic characteristics of core-use areas. This information is invaluable in delineating offshore areas that have the habitat and environmental characteristics associated with high densities of wintering sea ducks. These data can be used to ensure that currently-proposed and future offshore wind farms are sited in areas where we would predict minimal negative impacts to sea ducks.

## Methods

### *Study area*

Surf Scoters were captured and tagged by several principal investigators and agencies between 2000 and 2014 (Sea Duck Joint Venture 2014). Captures and satellite transmitter deployments occurred at various locations in Canada and the US. These locations include Labrador, Québec (Forestville and Chaleur Bay), New Brunswick (Chaleur Bay), Rhode Island, Delaware Bay, Chesapeake Bay, and North Carolina (Pamlico Sound and Swanquarter Bay; Figure 20-1).

### *Satellite deployment and individual tracking*

Scoters were captured during molting, fall and spring staging, and wintering periods (Table 20-1). Scoters were captured using a variety of techniques based on location, seasonal timing, and effective capture approaches in different habitats. These techniques included floating mist-nets (Brodeur et al. 2008), net-gunning from a boat, night-lighting, and gill-netting of molting birds (Sea Duck Venture 2014). Sea Duck Joint Venture (SDJV) partners deployed most of the transmitters prior to the initiation of the DOE- and BOEM-funded studies, and priority was placed on tagging adult female birds to delineate breeding populations. At some sites, however, males and second-year birds did receive transmitters, depending on the study objectives. This dataset encompasses scoters captured and tagged between 2000 and 2014 by several collaborative study partners including U.S. Department of Energy, Biodiversity Research Institute (BRI), USGS Patuxent Wildlife Research Center, Canadian Wildlife Service, U.S. Fish and Wildlife Service, Sea Duck Joint Venture, Bureau of Ocean Energy Management, Ducks Unlimited, Maryland Department of Natural Resources, Virginia Department of Game and Fisheries, Rhode Island Division of Fish and Wildlife, University of Rhode Island, and North Carolina Wildlife Resource Commission.

Platform transmitting terminal satellite transmitters (PTTs) were implanted into the abdominal cavity by a trained veterinarian using sterile surgery procedures and techniques described by Korschgen et al. (1996). PTTs weighed 26 or 38 g with an estimated 450-1400 hours of data transmission at an average internal bird body temperature ($105^0$ F; Figure 20-2). Transmitters were wrapped in a sterile mesh to provide additional surface area for adhesion to the body wall and anchor points to stabilize the PTT within the abdominal cavity. All efforts were made to closely monitor the recovery of the birds and return them as quickly as possible to the area of capture. During recovery and transport, birds were held separately in small crates. Each crate was equipped with a raised mesh floor platform that allowed the birds to remain clean and dry. Crate doors were also covered with soft padding to avoid bill damage. Each bird was held for a minimum of one hour following surgery and released at the discretion of the veterinarian when it was judged that the bird had fully recovered from anesthesia and was showing no signs of complications. In some cases birds were subcutaneously hydrated with Lactated Ringers Solution during recovery. Birds were released at or near their capture site (Figure 20-3).

Individual location data, body temperature, and PTT operational information were transmitted according to pre-determined duty cycles. Most PTTs deployed on wintering areas were programmed to transmit more frequently during fall and winter months (4 hours ON, 13 hours OFF) to determine spatiotemporal movements and habitat use in areas of proposed large-scale offshore wind power development during periods of peak migration and staging. PTTs deployed in Surf Scoters as part of the BOEM study (2012-2013) used a duty cycle designed to yield two locations per day during the migratory and wintering periods, and a less frequent duty cycle during the rest of the year to preserve battery power. To conserve PTT battery life, transmission periods were greatly reduced during the breeding season (2 hours ON, 120 hours OFF) when birds were primarily sedentary. All PTTs provided through SDJV were programmed with duty cycles that represented a compromise between PTT longevity and frequency of location data. This was intended to allow PTTs to last at least one year, but possibly up to three years, and enable an analysis of annual variation in timing of migration, habitat use, and site fidelity. Exceptions to these duty cycles were made for partners who customized transmitters to meet specific local objectives that required different (usually more frequent) duty cycles (e.g., daily movements during winter to evaluate potential conflicts with proposed offshore wind projects).

Telemetry data from PTTs are available via the Argos system of satellites. Argos records data for known PTTs and stores these data for one year on its servers, but makes available only the last ten days of data to the end-user. BRI's process for handling Argos data management and mapping tasks for the SDJV and BOEM studies included nightly downloads by telnet process of the last five days of data using a customized program written in the Python programming language (Python 2.7[2]). All active programs were cycled through to download data, and any active tags that transmitted during this period provided data for download that were archived for later use. BRI archived ds, diag, and last message data from Argos telnet servers. Argos data from this study were stored or archived at several locations including MoveBank[3] and servers at the USGS Patuxent Wildlife Research Center. Specific subsets of the location data were also uploaded to *wildlifetracking.org* or *seaturtle.org*, free online services that host animal tracking projects and update maps of individual birds' movements with new daily location data (on a fairly coarse scale).

Once data were archived, they were compiled and filtered to remove redundant data and errant points using the Douglas Argos Filter[4] (DAF). The DAF is a threshold filter that has several user-defined parameters to flag improbable locations in satellite tracking data (Douglas et al. 2012). The parameters are adjustable based on species' movement behaviors and the scale of the area under observation. With the DAF, data are retained if they pass (1) a spatial redundancy test and/or (2) a movement rate and turning angle test. Since bird data contain both short-distance, local movements and long-distance migratory events, we employed the hybrid filter of both the distance, angle and rate (DAR) and minimum redundant distance (MRD) filters. This achieved the best of both filters and in practice has produced very clean data with few erroneous points. Using DAF, we also identified the best representative point per duty cycle for each animal to reduce redundant daily positional information.

A database of deployment and life-history data was built for every PTT tag that was deployed or archived by the SDJV project (2000-2014). Data were stored in a shared Google spreadsheet accessible by all project investigators. The database contained information about the PTT tags themselves as well as data on the capture and deployment history of all birds. Furthermore, life-history period start and end dates were noted for every animal, following species-specific life-stage criteria defined by the SDJV (Sea Duck Joint Venture 2014). We identified the following periods: breeding, molting, fall-staging, wintering, and spring-staging, for every year that the animal was alive and transmitted locations.

Final dispositions and the date of disposition were assigned for all non-active tags. Sensor data were assessed for every tag to identify confirmed mortality (by internal temperature sensor or mortality sensor) or battery/tag failure due to low voltage. We listed the day after last transmission as the last date of disposition for all birds with tags that stopped transmitting for either low voltage or unknown

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[2] http://www.python.org/

[3] www.movebank.org

[4] http://alaska.usgs.gov/science/biology/spatial/douglas.html

reasons. Tag duration dates were then calculated from the deployment start date and the final disposition date. The DAF filter works off of deployment dates to correctly parse PTT data. Since PTT Argos ID numbers can be deployed multiple times, it was necessary to re-run DAF filters once final dispositions were determined to allow the DAF filter to correctly parse the data. Maps presented in this report include DAF-filtered data through 21 October 2014.

BRI wrote a custom script in Python 2.7 for ArcGIS to automate map production. This script used the DAF-filtered data and the corresponding life-period data from the deployment database to map locations for each individual. Only one year's worth of data per life period per animal was included in analyses in order to avoid biasing the dataset towards individuals with more years of data due to tag longevity or mortality. As the movement patterns of birds may be adversely affected in the first year following capture and implant of satellite tags (S. G. Gilliland, unpublished data), we chose Year 1 data if only one year of data existed, and Year 2 data preferentially over Year 3 data. Only birds that transmitted >60 days after release were included to reduce bias from birds that could have been negatively affected by transmitter implantation and handling.

In order to determine if, and during what time periods, Surf Scoters may be exposed to offshore wind facilities in the mid-Atlantic, we calculated arrival date, departure date, and average length of stay for all birds wintering within the study area. Arrival date was determined by finding the date the bird arrived and subsequently remained in the wintering area, typically between October and November. Scoters included in this analysis consisted of birds marked outside of the wintering areas or birds marked originally on the wintering area and returning in subsequent years after marking. Departure date was determined by finding the date the bird made significant movements away from the wintering area, typically between March and April (Sea Duck Joint Venture 2014). A one-way analysis of variance (ANOVA) was used to test significance between arrival and departure dates between sexes. If significance was determined, non-parametric Wilcoxon multiple comparison or Tukey-Kramer HSD tests were used to compare fall arrival dates, spring departure dates, and total length of stay in the study area by age, sex, year, and capture location. Results of statistical tests were considered significant at $p<0.05$.

### *Wintering area distribution*

We produced two basic map types: (1) movement maps showing mean location points for each animal per period (i.e., breeding, molt, and winter) connected by approximate migratory path lines and (2) kernel density maps that show broad-scale utilization distribution for all birds. Movement maps were created to show male movements from *winter to molting*, which included spring migration and movements during the breeding and post-breeding periods, and *molting to winter*, which included fall migration. Female movement maps consisted of *winter to breeding,* which included spring migration, and *breeding to winter,* which included post-breeding movements, molt, and fall migration. Different maps were produced for each sex to reliably delineate both breeding and molting locations. Period locational means for winter, breeding, and molting locations were used for these maps, calculated from all best locations per duty cycle available for those periods. Movement lines for each individual were created from all intermediate points between "book-end" periods such as winter and breeding to show the linkage between these periods; increasing line density where tracks were overlaid indicate areas used by multiple birds. Lines are straight-line paths between consecutive points and do not necessarily reflect the true path of each bird; the broadness and semitransparency of the lines used in these maps was intended to illustrate generalized migratory paths.

Kernel density maps were created for various life-history periods. Kernel density estimation involves the use of point data from telemetry to estimate relative spatial use during specified time intervals. For each location, the bird's habitat use was estimated to be greatest directly on the point, and to decrease with distance from the point (reaching zero at a distance specified by the user, after Worton 1989). Seventy DAF-filtered locations were randomly selected from each individual's winter period. The number of locations selected per bird (*n*=70) was based on the lower quartile of the distribution of location data available for all birds, and was chosen to minimize bias related to varying amounts of available location data among individuals. For maps of wintering and staging in this report, separate kernel density estimates were developed for each individual bird, and those results were summed at each location ("location" meaning a degree of latitude/longitude), to form a composite kernel density map for all birds during these periods. This prevented birds that spent greater time in the wintering area from disproportionately influencing estimates of population utilization. Bandwidth (distance) was calculated for all pooled locations using the likelihood cross-validation estimator (Loring et al. 2014).

### *Habitat selection and use*

We examined winter habitat use by satellite-tagged Surf Scoters by comparing relevant environmental covariates at locations within the core use area (0.5 isopleth) to environmental characteristics available throughout the utilization distribution (0.95 isopleth). Following Loring et al. 2014, habitat data were in raster format and resampled to a standardized cell size of 250$m^2$ (hereafter: resource units). We then randomly sampled habitat variables from 25% of the resource units within both the available and core-use areas. We measured water depth (m) and slope (degrees) within each resource unit using the NOAA National Geophysical Data Center 3 arc-second Coastal Relief Model for the United States (NOAA 2014a). To estimate distance to shore, the Euclidean distance (m) between scoter locations and the nearest segment of the NOAA Medium Resolution Digital Vector Shoreline (1:70,000; NOAA 2014b) was calculated using the Near Tool in Arcmap 10.2.2 (ESRI, Redlands, CA). For dynamic variables with a temporal component, we relied on Marine Geospatial Ecology Tools to create seasonal and long-term winter mean climatology rasters for sea surface temperature, chlorophyll-*a*, and sea surface salinity (Roberts et al. 2010). We used smoothed daily Sea Surface Temperature (SST; degrees Celsius) estimates based on optimal interpolation of data derived from high resolution satellite imagery and floating buoys (Stark et al. 2007); these data are produced by the UK Met Office on a global scale at a spatial resolution of 0.054 degrees latitude and longitude. Sea surface salinity (in psu – practical salinity units) was estimated using the Hybrid Coordinate Ocean Model (HYCOM) produced by the National Ocean Partnership Program at a spatial scale of 1.5 degrees latitude and longitude (Chassignet et al. 2009). To estimate ocean productivity, we obtained monthly estimates of Chlorophyll-*a* concentration (mg/$m^3$) produced by the NASA Goddard Space Flight Center's Ocean Data Processing System. These data have a spatial resolution of 4 km and are derived from radiometric measurements of chlorophyll fluorescence made by the Aqua sensor aboard the MODIS satellite system (Mueller et al. 2003). We calculated long-term averages for sea surface temperature (10 years), Chlorophyll-*a* (10 years), and sea surface salinity

(6 years) parameters to best account for the approximately 15 year span of sampling efforts in this study.

Pearson product-moment correlations were performed to test for relationships between pairs of habitat parameters (distance to shore, water depth, slope, long-term chlorophyll-*a* concentrations, sea surface salinity, and sea surface temperature). Variance inflation factors (VIF) were calculated to quantify multicollinearity of variables. Throughout the entire study period, pair-wise correlation among habitat covariates did not exceed 0.31 and VIF values were <1.5. Therefore all variables were included in modeling.

Logistic regression was used to model habitat covariate effects, including interactions between covariates, on used versus available locations throughout the study period. We selected the best of 10 a priori candidate models using Akaike's Information Criterion corrected for small sample size ($AIC_c$). Models were ranked using $AIC_c$ differences ($\Delta AIC_c$) and $AIC_c$ weights ($w_i$) to evaluate the relative likelihood of each candidate model (Loring et al. 2014, Žydelis et al. 2006, Burnham and Anderson 2002).

## Results

### *Arrival/departure dates and length of stay*

A total of 101 Surf Scoters, tagged by collaborating investigators, were tracked to the study area and made available for data analysis in this report (Table 20-2). The median fall arrival date to the mid-Atlantic study area across all capture years was 8 November (range: 14 Oct – 21 Dec, *n*=83). We found no significant difference in fall arrival dates based on age ($F_{[1,81]}$=0.0032, *p*=0.955), sex ($F_{[1,82]}$=1.9211, *p*=0.169), or capture year ($F_{[5,78]}$=0.7531, *p*=0.586). Birds captured in Chesapeake Bay and tracked through the subsequent fall migration arrived on the wintering grounds significantly later in the fall than those captured in Pamlico Sound, Labrador, and Forestville (*p*=0.028, *p*=0.008, *p*<0.0001, respectively).

The median spring departure date across all capture years was 5 April (range: 1 Jan – 12 May, *n*=83). We found no significant difference in spring departure dates based on age ($F_{[1,81]}$=2.4036, *p*=0.125), sex ($F_{[1,82]}$=0.4539, *p*=0.502), or capture location ($F_{[6,77]}$=1.2640, *p*=0.284). Scoters wintering in the study area during the winters of 2002 and 2004 departed on spring migration significantly earlier ($F_{[5,78]}$=3.2618, *p*=0.010) than in all other years.

Mean (±SD, range) length of stay within the study area for all scoters was 133 days (±28, 60-184) and did not significantly differ based on age ($F_{[1,81]}$=2.1382, *p*=.0147) or sex ($F_{[1,81]}$=0.0000, *p*=0.996). Birds wintering in 2002 and 2004 stayed for a significantly shorter period of time ($F_{[5,78]}$=3.7533, *p*=0.004) than those in other years, while birds captured in Forestville stayed in the study area for significantly longer than those captured in Chesapeake Bay (*p*=0.047).

### *Wintering area distribution*

Kernel density estimations for both sexes showed that core-use areas during the wintering period encompassed the majority of both Chesapeake Bay and Delaware Bay, with additional smaller core-use areas occurring south of Cape Cod near Nantucket Shoals, in Long Island Sound, and in Pamlico Sound, NC (Figure 20-4). This distribution was similar when broken out by sex, with females (Figure 20-5) demonstrating slightly heavier use of the areas south of Cape Cod than males (Figure 20-6). Composite core-use areas throughout the study area encompassed an area of just over 4400 $km^2$. Core-use areas for males were much more condensed (~2700 $km^2$) than for females. It should be noted that core-use areas primarily occurred near mid-Atlantic capture locations (Figure 20-1), which could bias analysis.

Spring migration movement data from satellite-tagged Surf Scoters showed that most birds followed a coastal migration route along the eastern seaboard to staging areas near the Gulf of St. Lawrence and its inner estuary. Most birds then migrated over land to breeding and molting areas in northern Québec, east of Hudson Bay. At least one bird appears to have taken a more direct over-land route, briefly stopping over on Lake Ontario and Lake Huron before continuing north. Other breeding locations west of Hudson Bay in northeastern Manitoba and southeastern Nunavut were also documented (Figure 20-7).

Fall migration data showed that birds used migration routes very similar to their spring migration. Birds left breeding and molting areas in northern Canada and flew east to stage briefly in the St. Lawrence Estuary, before following the eastern seaboard south to wintering areas in the mid-Atlantic region (Figure 20-8).

### Winter site fidelity

Surf Scoters appear to exhibit strong site fidelity to wintering areas. However, the vast majority of the satellite transmitters deployed in Surf Scoters contained duty cycles focused on detailed winter movements, and therefore compromised overall battery life longevity. Most marked scoters did not provide satellite locations for multiple (>1) complete winters. A total of 22 marked scoters (9M, 13F) provided satellite locations for two complete winter seasons; 91% (20/22) of the birds occupied the same wintering areas in consecutive winters. Two individuals deviated from their wintering areas slightly, utilizing Chesapeake Bay in the first winter and both Chesapeake Bay and Delaware Bay in the second winter.

There was a high tendency (92% or 36/39) for most marked scoters to return to the same wintering area in which they were originally captured and marked. Most scoters were captured on the wintering areas in March, and transmitters provided locations through the following complete winter, and terminated prior to the third winter. We cannot positively determine if these birds occupied the same wintering area throughout the winter prior to capture and marking.

### Habitat selection and use

Surf Scoter core-use areas were closer to shore and in shallower water depths than the rest of the winter utilization areas. Core-use areas also had higher chlorophyll-*a* concentrations and slightly lower salinity (Table 20-3). In both the core-use and utilization distributions, percentage use was highest in areas with mid-range sand grain size; however this percentage use was proportionally larger in core-use areas (Table 20-4). The top model for this study accounted for 0.97 of Akaike weight and included the parameters water depth, distance to shore, sediment type, 10-year mean chlorophyll concentration, 10-year mean sea surface temperature, 6-year mean sea surface salinity, and seafloor slope. Our best fit model also included interactions between sea surface temperature and sea surface salinity, chlorophyll concentration and sea surface temperature, as well as chlorophyll concentration and sea surface salinity. No other models were considered competitive as all had $\Delta AIC_c$ values >2 (Table 20-5).

We found that the majority of locations within core use areas, for both sexes, occurred in bays (near mid-Atlantic capture locations) rather than open-ocean or tidal rivers. It should be noted, capture location could influence subsequent habitat associations. The majority of marked Surf Scoters were captured in protected bays, which are conducive to scoter capture techniques, rather than turbulent offshore areas. During the study period, scoters utilized areas between 0 and ≥22.2 km from shore. Distances from shore were not influenced by sex. We found a trend of decreasing temperatures when moving from the 0.95 isopleth inward to the core-use areas. We found no significant difference in slope between isopleths or by sex. Both sexes in all locations utilized virtually flat areas with minimal slope.

## Discussion

### *Arrival/departure dates and length of stay*

The preliminary results of this study underline the importance of the mid-Atlantic coastal region for migrating and wintering sea ducks. The highest densities of wintering Surf Scoters on the Atlantic coast are located in Chesapeake Bay, Delaware Bay, and along the Maryland-Delaware coast. Smaller concentrations winter in Pamlico Sound and Nantucket Sound, MA (Silverman et al. 2013). Satellite tagged Surf Scoters arrived to the mid-Atlantic region between mid-October and late December (mean: 08 Nov, range: 14 Oct – 21 Dec). Spring departure for satellite tagged Surf Scoters occurred between January through May (mean: 5 April, range: 1 Jan – 12 May). The migration chronology of birds within this study suggests that Surf Scoters wintering and migrating throughout the mid-Atlantic region could encounter future offshore wind energy facilities between mid-October and early May. These dates concur with past research on sea duck migration on the east coast (Loring et al. 2014, Sea Duck Joint Venture 2014, Veit and Petersen 1993). Satellite-tagged scoters in this study spent nearly 40% (mean: 133 days, range: 60-184 days) of their annual cycle on wintering grounds within the study area. This is similar to a recent study in which Black Scoters spent an average of 147 days on wintering grounds in southern New England (Loring et al. 2014). Another study found that King Eiders (*Somateria spectabilis*) in the Bering Sea spent an average of 160 days on their wintering grounds (Oppel et al. 2008).

We found no difference in migration chronology or length of stay on wintering grounds between males and females. Spring departure dates spanned a nearly 5 month period, with no significant differences between sexes. Fall arrival dates were also not significantly different between sexes. This differs from movement data on Black Scoters that documented males arriving on the wintering grounds a week or more earlier than females (Loring et al. 2014, Bordage and Savard 1995).

### *Wintering area distribution*

Composite kernel density estimation core-use areas throughout the study area encompassed an area of just over 4400 $km^2$. Core-use areas for males were much more condensed (~2700 $km^2$) than for females. Surf Scoters contained a mean winter utilization distribution of 3,008 $km^2$ (n=68) and ranged in size of <1.0 to 30,393 $km^2$. A study on Black Scoters in southern New England found winter utilization distributions ranging from <20 to >10,000 $km^2$ (Loring et al. 2014) while satellite-tagged King Eiders in the Bering Sea had winter ranges between 13 and >66,000 km$^2$ (Oppel et al. 2008). Winter ranges for common eiders in Greenland averaged only 67.8 km$^2$ (Merkel et al. 2006).

We evaluated the potential occurrence of satellite tagged Surf Scoters within the Wind Energy Areas (WEAs) through evaluation of both core use areas (0.5 isopleth) and utilization distributions (0.95 isopleth). Locations of Surf Scoters were not recorded within any WEAs, and 0.2% of all core use and 4.0% utilization distributions were within the mid-Atlantic study area. This analysis indicates Surf Scoters tagged along nearshore areas of the mid-Atlantic have a minimal likelihood of overlapping with current WEAs. Activities associated with construction within WEAs, such as installation of transmission lines or vessel traffic within nearshore areas, or possible construction in nearshore waters outside of currently designated WEAs, would have a higher likelihood of overlapping with wintering Surf Scoters in the mid-Atlantic.

Sea ducks during the wintering period are exposed to several variable factors that can affect movement patterns and distributions. Large winter ranges may allow flexibility in responding to factors such as changing habitat conditions or depleted food sources (Lok et al. 2008, Kirk et al. 2008). The tendency to expand outside the typical utilization distributions, as reported in black scoters (Loring et al. 2014) and king eiders (Oppel et al. 2008), may also increase the probability that wintering scoters will encounter and potentially be affected by wind energy developments scattered throughout the coastal landscape.

While core use areas in this study occurred primarily in bays, this may have been partially influenced by the locations of capture efforts. The majority of marked Surf Scoters were captured in protective bays, which are conducive to scoter capture techniques, rather than turbulent offshore areas.

### *Habitat selection and use*

Surf Scoters in core-use areas utilized shallow (<40 m) areas within 8.5 km from shore. We analyzed distance to shore measurements on an individual level for all scoters within the data set and found a range between 0.16 and 12.8 km. The mean range for individual birds was 8.5 km with a maximum range of 17.5 km. Thirty-six of the scoters sampled had ranges of movement in excess of 10 km, demonstrating the ability and likelihood of scoters to utilize a wide area within the nearshore environment. This concurs with Black Scoter habitat use documented by Loring et al. (2014) where birds generally utilized shallow (<20 m) areas within 5 km from shore. Our resource selection models also suggest that other dynamic variables such as sea surface temperatures, productivity, and salinity (and selected interactions among them) may also be important in determining valuable scoter habitat. Existing literature on scoter species (velvet [*Melanitta fusca*] and common) in Europe document that these species forage mainly on small bivalve mollusks found within the upper layers of sandy substrates less than 20 m deep (Fox 2003). Scoter diet studies along the Atlantic coast have indicated the preferred winter diet is blue mussels and hooked mussels (*Ischadium recurvum*), found among hard substrate (M. Perry unpubl. data). During the wintering period, sea ducks are well known to congregate in areas of high-quality feeding habitat and prey abundance (Lewis et al. 2008, Guillemette et al. 1993). Depending on prey availability, sea ducks are capable of frequent movements between different feeding areas and increased foraging effort (Kirk et al. 2008, Kirk et al. 2007, Larsen and Guillemette 2000). Placement of offshore wind energy facilities in shallow (<40 m) areas in this project's study area could potentially displace wintering sea ducks from high-quality feeding areas. Molluscivorous sea ducks in a similar study were temporarily displaced for at least 3 years after construction of offshore wind energy developments in Europe (Peterson et al. 2007). Flocks of sea ducks began to reappear within the wind farm areas in the fourth year post construction. This potential displacement may necessitate increased flight and foraging efforts, thus increasing energy expenditure during critical periods of the annual cycle.

### *Next steps*

Future analysis of this dataset should include the development of individualized kernel-based home range size estimates, as well as updating current results with data gathered from birds during the 2013-2014 and 2014-2015 winter periods. The inclusion of data from additional consecutive wintering periods will also allow for more thorough and comprehensive analysis of wintering site fidelity. Subsequently, once all movement data from all wintering periods has been obtained, habitat covariate data used in modeling should be updated to include long-term mean values that match the duration of the PTT data. This may improve model fitness or uncover new interactions between covariates not currently recognized in the existing dataset.

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## Figures and tables

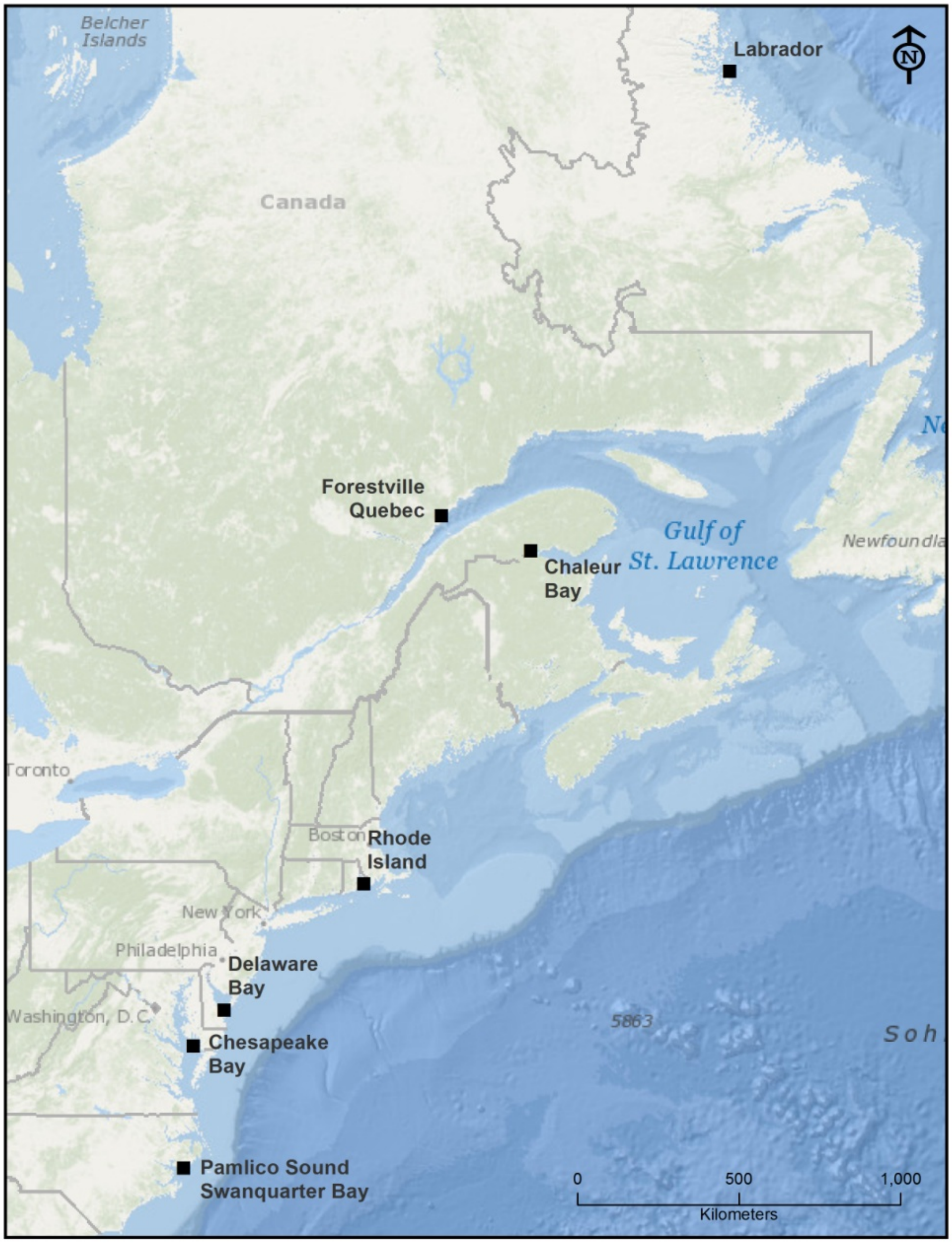


**Figure 20-1. Capture locations of Surf Scoters implanted with PTTs between 2000 and 2014.**

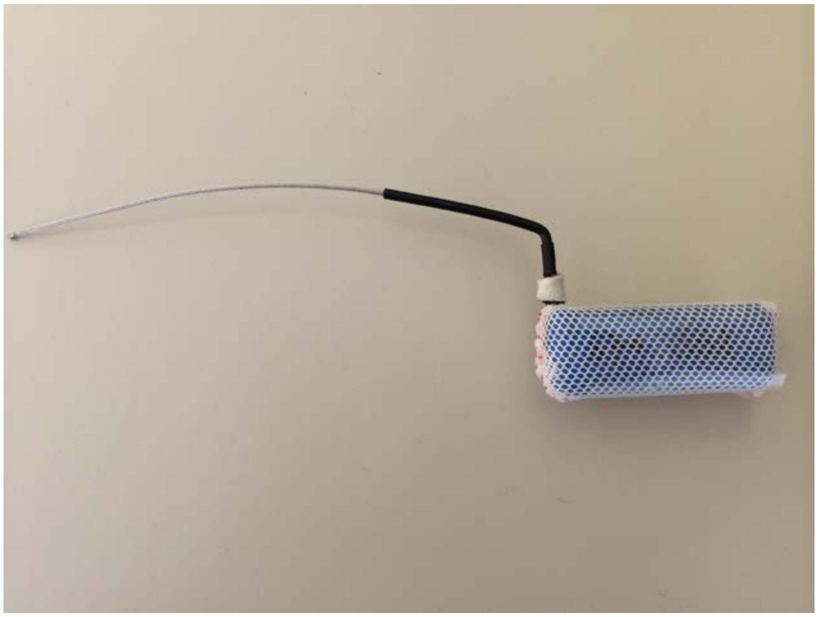


**Figure 20-2. Example of implantable PTT transmitter deployed in Surf Scoters.** (Photograph: L. Savoy/BRI)

**Figure 20-3. Adult male Surf Scoter with implanted PTT prior to release.** (Photograph: L. Savoy/BRI)

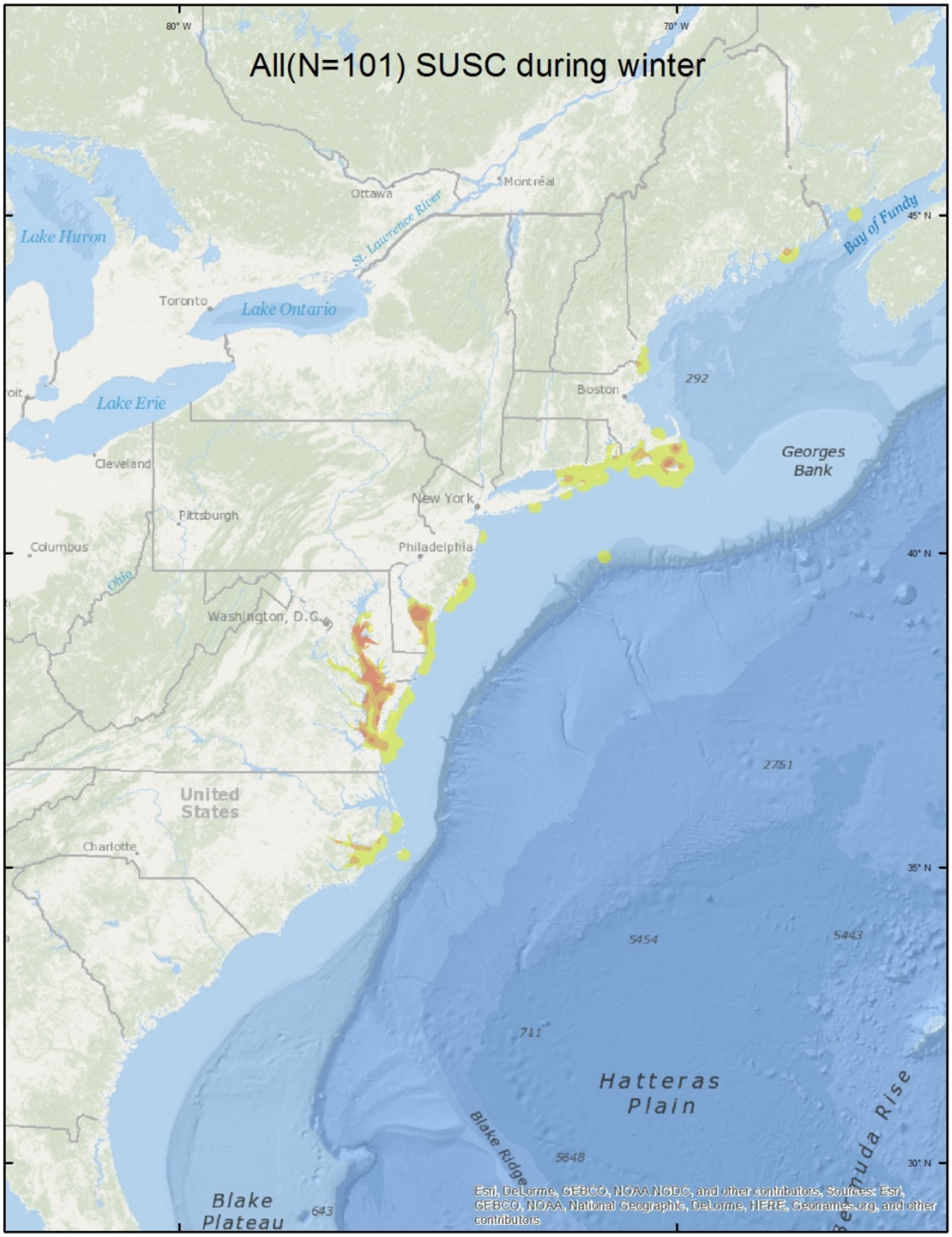
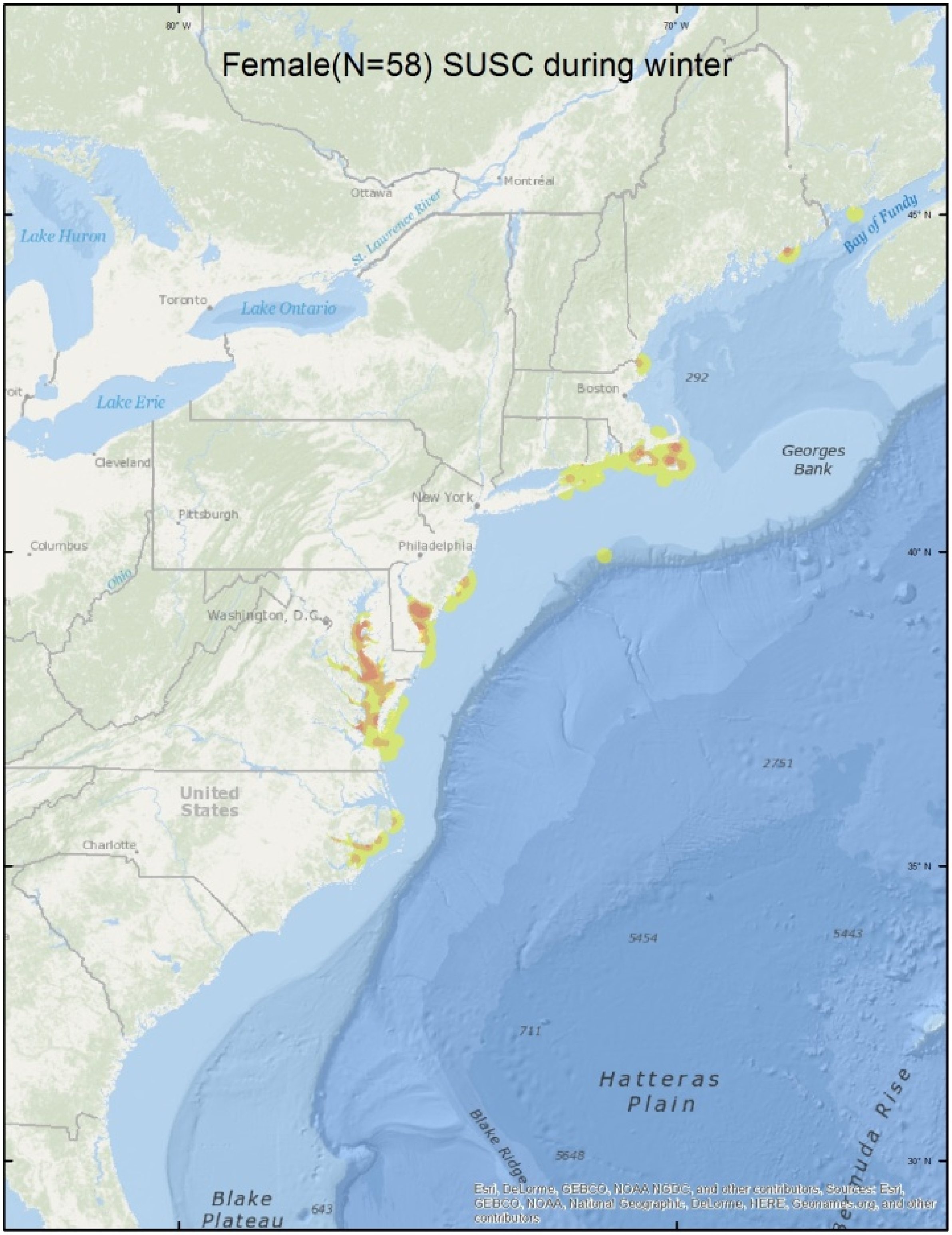


**Figure 20-4. Composite kernel density estimations of wintering ground utilization distributions for satellite-tagged Surf Scoters.** Intensity of use ranges from lowest (yellow) to greatest (red).



Esri, DeLorme, GEBCO, NOAA NGDC, and other contributors, Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

**Figure 20-5. Composite kernel density estimations of wintering ground utilization distributions for satellite-tagged female Surf Scoters.** Intensity of use ranges from lowest (yellow) to greatest (red).

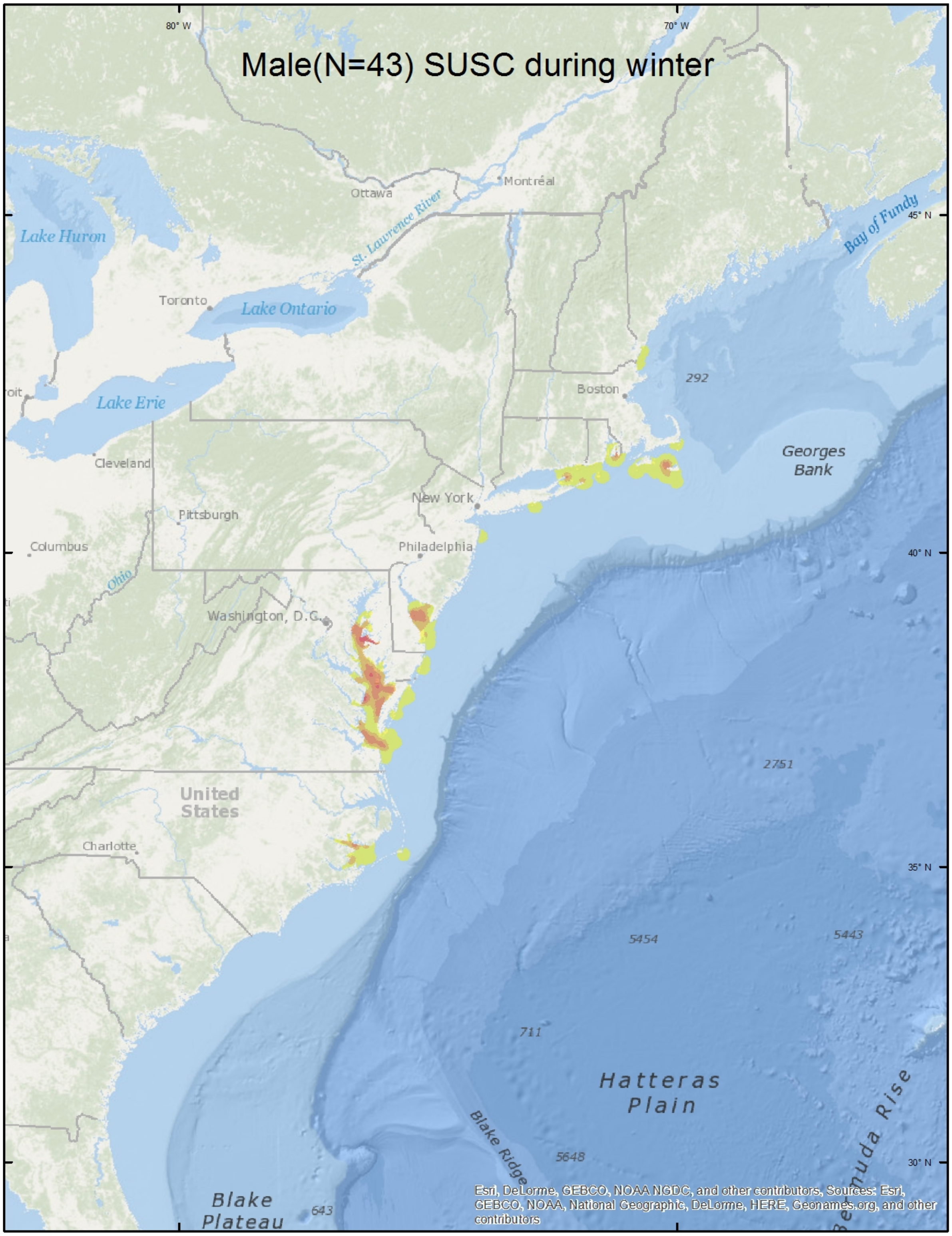


**Figure 20-6. Composite kernel density estimations of wintering ground utilization distributions for satellite-tagged male Surf Scoters.** Intensity of use ranges from lowest (yellow) to greatest (red).

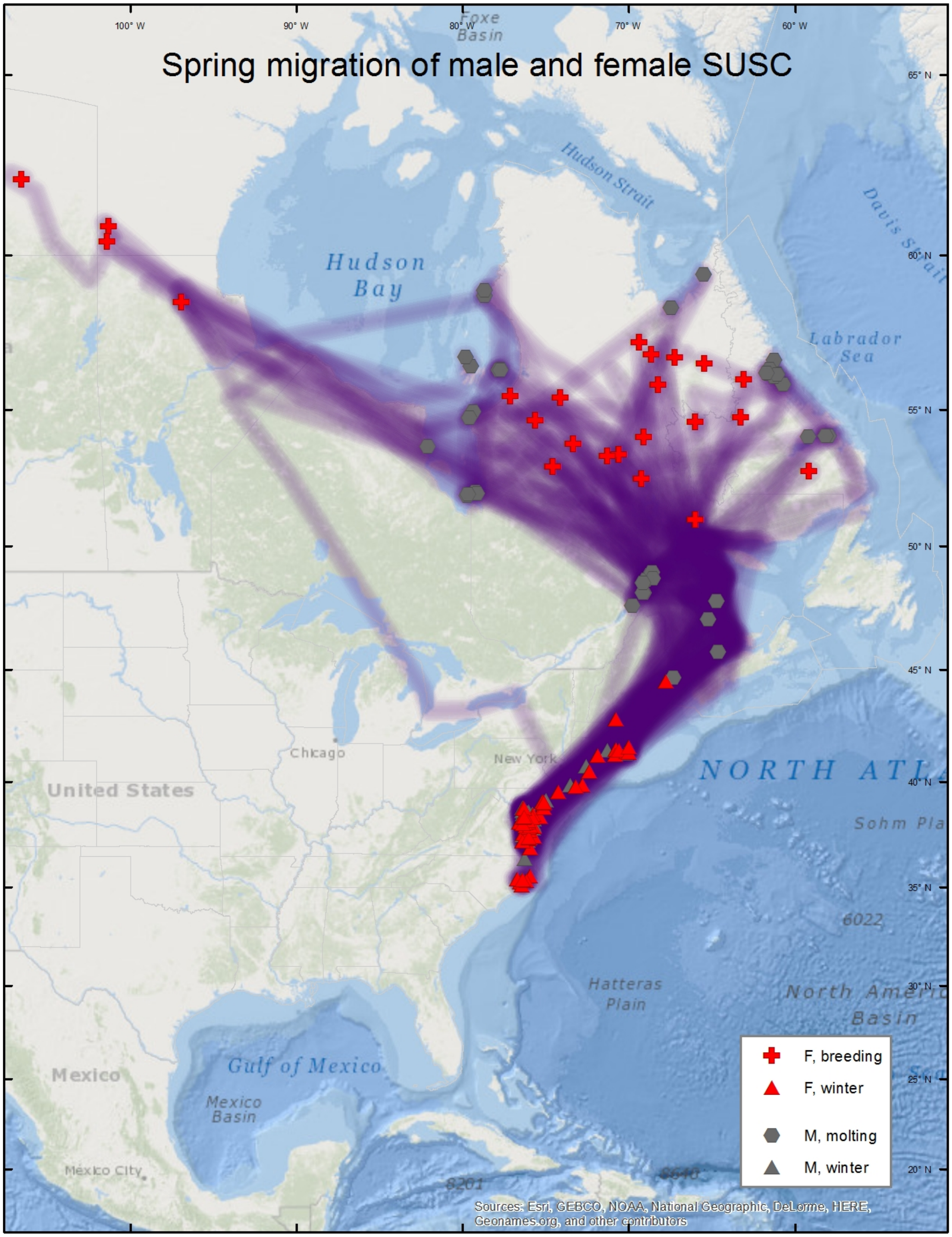


**Figure 20-7. Spring migration routes of all satellite-tagged Surf Scoters throughout the study area.** Lines do not necessarily represent direct flight paths. Point locations represent mean locations; one per animal per period.

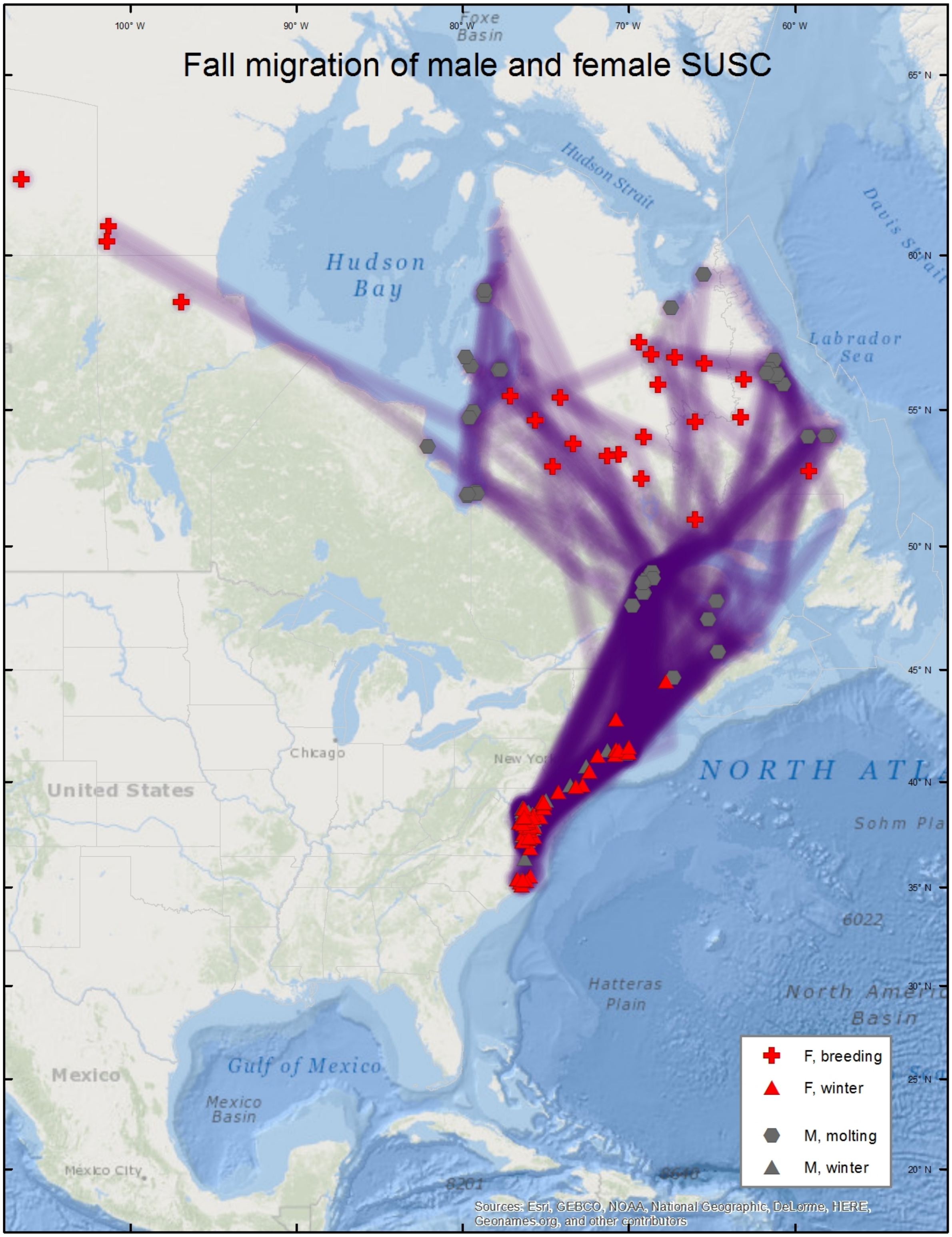


**Figure 20-8. Fall migration routes of all satellite-tagged Surf Scoters throughout the study area.** Lines do not necessarily represent direct flight paths. Point locations represent mean locations; one per animal per period.

**Table 20-1. Life stage criteria used to analyze seasonal migration and distribution data for Surf Scoters (Sea Duck Joint Venture 2014).**

| Life Stage | Criteria |
|---|---|
| **Breeding** | • Stay for ≥ 14 days<br>• Arrive between late May and June<br>• Depart between July and August |
| **Molting** | • Stay for ≥ 21 days<br>• Arrive between July and September<br>• Depart between August and October |
| **Wintering** | • Arrive between October and November<br>• Depart between late March and early April |
| **Staging** | • Along migration, bird stays in same location for ≥ 15 days |

**Table 20-2. Number of satellite-tagged Surf Scoters tracked within the study area during the wintering periods between 2000 and 2014, listed by capture site, age, and sex.** Ages are categorized as second-year (SY), after-second-year (ASY), or unknown (Unk). Sexes are male (M) and female (F).

| Capture Site | Winter Period | Capture Timing | SY | | ASY | | Unk | |
|---|---|---|---|---|---|---|---|---|
| | | | M | F | M | F | M | F |
| Chesapeake Bay | 2000-2001 | Winter | | | 1 | | | |
| | 2002-2003 | Winter | 4 | | 1 | | | |
| | 2004-2005 | Winter | | | | 3 | | |
| | 2011-2012 | Winter | | | 10 | 2 | | |
| | 2012-2013 | Winter | | | 8 | 1 | | 2 |
| | 2013-2014 | Winter | | | 9 | 3 | | |
| Chaleur Bay, NB | 2004-2005 | Spring Staging | | | | 3 | | |
| Labrador | 2006-2007 | Molting | | | 5 | | | |
| Rhode Island | 2011-2012 | Winter | | | 1 | | | |
| Pamlico Sound, NC | 2011-2012 | Winter | | | 1 | 1 | | |
| | 2012-2013 | Winter | | | | 1 | | |
| Delaware Bay | 2012-2013 | Winter | | | 2 | | | |
| | 2013-2014 | Winter | | | 1 | | | |
| Forestville, QC | 2012-2013 | Fall Staging | | | | 12 | | |
| | 2013-2014 | Fall Staging | | | | 30 | | |
| | | **Total** | **4** | | **95** | | **2** | |

**Table 20-3. Range, mean ($\bar{x}$) and standard error (SE) of habitat variables at satellite-derived locations for Surf Scoters in the utilization distributions (available; 0.95 isopleth) and core-use areas (0.5 isopleth).**

| Habitat Variable | Available | | Core Use | |
|---|---|---|---|---|
| | Range | $\bar{x}$ ± SE | Range | $\bar{x}$ ± SE |
| ***Long-term*** | | | | |
| Chlorophyll-*a* (mg/m$^3$)- 10 yr Mean | 2.0 - 44.8 | 9 ± 0.02 | 2.5 - 27.8 | 13.5 ± 0.02 |
| Sea Surface Temperature ($^0$C) - 10 yr Mean | 3.9 - 14.5 | 6.4 ± 0.01 | 3.9 - 13.3 | 5.1 ± 0.004 |
| Sea Surface Salinity (psu) - 6 yr Mean | 25.7 - 33.4 | 32.4 ± 0.005 | 16.8 - 33.4 | 29.7 ± 0.03 |
| ***Physical*** | | | | |
| Depth (m) | 0.1 - 104.4 | 10.2 ± 0.03 | 0.1 - 46.6 | 7.5 ± 0.02 |
| Slope (degrees) | 0.0 - 13.7 | 0.28 ± 0.001 | 0.0 - 9.4 | 0.27 ± 0.002 |
| Distance to Shore (km) | 0.1 - 22.2 | 4.5 ± 13.4 | 0.1 - 15.5 | 4.5 ± 14.8 |

**Table 20-4. Proportion of use of different sediment types at satellite-derived locations for Surf Scoters in the utilization distributions (available; 0.95 isopleth) and core-use areas (0.5 isopleth).**

| Grain Size (mm) | Sediment Type | Available | | Core Use | |
|---|---|---|---|---|---|
| | | n | % Use | n | % Use |
| 0.00 - 0.03 | Silt/Mud | 80129 | 13 | 6112 | 12 |
| 0.03 - 0.17 | Sand | 20362 | 25 | 14122 | 27 |
| 0.17 - 0.35 | Sand | 20557 | 26 | 24604 | 47 |
| 0.35 - 0.36 | Sand | 7778 | 10 | 1952 | 4 |
| 0.36 - 0.48 | Sand | 5649 | 7 | 2992 | 6 |
| 0.48 + | Course Sand - Gravel | 15437 | 19 | 3109 | 6 |

**Table 20-5. Logistic model selection results examining the effects of water depth (WD), distance to shore (DS), sediment type (SED), seafloor slope (SL), long-term mean chlorophyll-*a* concentration (2004-2014)(CH10), long-term mean winter sea surface temperature (2004-2014)(SST10), and long-term mean sea surface salinity (2008-2014)(SSS6) on the resource selection of Surf Scoters wintering in the mid-Atlantic U.S.** Models are ranked according to Akaike Information Criterion adjusted for small sample size ($AIC_c$). The table shows the variables included in the model, number of estimated parameters (K), difference between selected model and top-ranked model $AIC_c$ values ($\Delta AIC_c$) and $AIC_c$ weights ($W_i$).

| Model Parameters | K | $AIC_c$ | $\Delta AIC_c$ | $w_i$ |
|---|---|---|---|---|
| WD, SED, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 14 | 50404 | 0 | 0.97 |
| WD, SED, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 13 | 50411 | 7 | 0.03 |
| WD, SED, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10) | 13 | 50517 | 113 | 0.0 |
| WD, SED, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10) | 12 | 50520 | 116 | 0.0 |
| $WD^2$, SED, $DS^2$, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 14 | 50543 | 139 | 0.0 |
| $WD^2$, SED, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 14 | 50703 | 299 | 0.0 |
| WD, SED, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6) | 12 | 51328 | 924 | 0.0 |
| WD, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 9 | 51553 | 1149 | 0.0 |
| SED, DS, CH10, SST10, SSS6, SL, (SST10 x SSS6), (CH10 x SST10), (CH10 x SSS6) | 13 | 51570 | 1166 | 0.0 |
| Null | 1 | 183036 | 132632 | 0.0 |

# Chapter 21: Wintering movements and habitat use of Red-throated Loons (*Gavia stellata*) in the mid-Atlantic U.S.

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 21 Highlights

*Wintering movements and habitat use of Red-throated Loons (Gavia stellata) tracked via satellite telemetry in the mid-Atlantic U.S.*

### *Context*[1]

Location data collected from satellite transmitters can be used to estimate home-range size and other features of the utilization distribution of target species. Kernel density estimation is a standard technique for characterizing and visualizing animal home ranges, and the utilization distribution estimates the intensity or probability of use by an animal throughout its home range. From these data, resource selection functions can be used to predict habitat use across landscapes or to understand the influence of certain habitat characteristics on a species' distribution. We used this approach to analyze location data collected from satellite-tagged Red-throated Loons captured at sea on their wintering grounds in the mid-Atlantic U.S. to determine occurrence and movement patterns in relation to potential offshore wind energy areas in the region. Similar analyses were conducted to examine movements of Surf Scoters (Chapter 20) and Northern Gannets (Chapter 22), and time variant kernel density models were produced to examine the broad scale movement patterns of all three species throughout the year (Chapter 23).

### *Study goal/objectives*

Improve our understanding of wintering Red-throated Loon movements and seasonal use of space in the mid-Atlantic study area.

### *Highlights*

- Composite kernel density estimate (KDE) maps calculated in GIS, using location data for each loon, provide information on core use areas within the study area during winter.
- We identify key habitats for Red-throated Loons on their wintering grounds and during migration, as well as the timing of their use.
- Wintering loons in our sample used tidal rivers, bays, and ocean locations, and core use areas were generally 5-10 km from shore. Most ocean locations did not exceed 5 km from shore and were decidedly in the "nearshore" environment. Accordingly, water depth associated with use was shallow, with more than half the locations not exceeding 13 m.

### *Implications*

Quantifying the habitat characteristics within core use areas produced reliable information regarding Red-throated Loon habitat use in the mid-Atlantic region. This information can be paired with distribution and abundance data derived from surveys (Parts II-IV of this report) to more thoroughly describe the distribution patterns of this species in the region and determine potential exposure to future offshore wind energy development.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

The Red-throated Loon (*Gavia stellata*) is listed by the U.S. Fish and Wildlife Service (USFWS) as a species of conservation concern in much of its Arctic breeding range and wintering grounds in the Atlantic Flyway. However, data gaps exist regarding this species' wintering distributions, including concentration and timing of use, as well as migratory routes and stopover areas. In 2012 and 2013, we captured Red-throated Loons at sea on their wintering grounds in the Mid-Atlantic region and tagged them with platform terminal transmitters (PTTs) as part of a project focused on offshore wind energy development and diving birds. We used satellite tracking to map winter use movements in the mid-Atlantic study area and migratory routes of individuals (N = 23) to and from breeding locations in Quebec, Nunavut, Northwest Territories, and Greenland. We calculated composite kernel density (KDE) maps using GIS from movement data for each loon. In each winter season, kernel density rasters were generated as a composite of sub-sampled points from one year's worth of tracking for each animal (for all animals surviving more than 60 days), to generate a composite wintering utilization distribution.

Areas of heaviest habitat use were observed around the mouth of the Chesapeake Bay, south along the coast to the Virginia/North Carolina border, and the central eastern portion of Pamlico Sound and coastal waters off Cape Hatteras National Seashore in North Carolina. Small areas in central and northern Chesapeake Bay, as well as areas along the Delaware coast and Delaware Bay, were also heavily used by loons in our sample. Spring migration stopover sites included Long Island Sound, Narragansett Bay, Nantucket Sound, Buzzards Bay, Bay of Fundy, Gulf of St. Lawrence, and the St. Lawrence River; autumn stopover sites included Hudson Bay, James Bay, Hudson Strait, Ungava Bay, Gulf of St. Lawrence, and Lake Ontario. These results identify key habitats for Red-throated Loons on their wintering grounds and during migration, as well as the timing of their use.

## Introduction

Increased interest in renewable energy has led to the identification of offshore Wind Energy Areas (WEAs) for potential development in U.S. coastal waters. Wind energy is associated with fewer environmental degradation issues than fossil fuels; however, offshore wind energy development may pose multiple direct and indirect adverse effects to bird populations in the offshore environment (Fox et al. 2006, Goodale and Milman 2014). Specifically, Fox et al. (2006) described three primary factors: the potential for collision mortality, displacement from key foraging areas and migration and feeding flight pathways, and physical habitat loss associated with construction. In European studies, several species have exhibited a behavioral response of avoiding offshore wind facilities, resulting in changes in the local distribution, abundance, and flight patterns of birds (Petersen et al. 2006). For example, Red-throated Loons (*Gavia stellata*) and Arctic Loons (*G. arctica*) were present in average densities prior to construction of a wind farm in coastal waters off Denmark, but showed complete avoidance of the area during the construction phase and 3-yr post-construction period (Petersen et al. 2006). Further, this avoidance effect extended to a distance of 2 km around the wind farm. Indeed, multiple papers have identified the Red-throated Loon as a species vulnerable to this type of disturbance or displacement (Petersen et al. 2006, Halley and Hopshaug 2007, Percival 2014, Furness et al. 2013). Bird species with a high proportion of their biogeographic population occurring in offshore development areas, and those with high adult survival, are considered most vulnerable to population declines (Desholm 2009). In particular, displacement from wintering habitat may result in increased energy expenditure and competition for food resources, which has the potential to affect overall population fitness through the reduction of annual adult survival (Fox et al. 2006). Small changes in adult survival can result in significant population declines for species, such as loons, with low reproductive capacity and a slow maturation rate (Johnsgard 1987, Mitro et al. 2008).

Red-throated Loons have a circumpolar breeding distribution, nesting primarily on remote ponds in coastal tundra habitat. Similar to other loons, they are long-lived (likely 25-30 years) and experience high adult survival (≈ 0.92; Barr et al. 2000, Schmutz 2014). In North America, they winter along both coasts, as far south as northern Georgia on the Atlantic Coast, and northern Baja California and the Sea of Cortez on the Pacific Coast (Barr et al. 2000). Data related to population estimates and trends of Red-throated Loons are limited; some populations are considered stable, while others appear to have declined, and the trajectories of many other populations are unknown. Dickson and Beaubier (2011) detected very little change in the number of breeding pairs in the Canadian Beaufort Sea region between the 1985-1989 and 2007-2008 survey periods. The Alaska-Yukon Waterfowl Breeding Population Surveys, however, showed that the Alaska Red-throated Loon population declined by 53% from approximately 21,000 birds in 1977 to 10,000 in 1993 (Groves et al. 1996). Recent surveys in the Baltic Sea documented another alarming decline of 84% in wintering loon populations between 1988-1993 and 2007-2009 survey periods, the majority of which were estimated to be Red-throated Loons (Skov et al. 2011).

Approximately 70-100,000 Red-throated Loons are estimated to winter along the eastern U.S. coast, and the core of this wintering range is reported to occur in the mid-Atlantic region (New Jersey to North Carolina), with the largest concentrations reported off the coasts of Delaware and North Carolina (Forsell 1999, Root 1988). Although the development of offshore wind energy facilities are being considered for both U.S. coastlines and the Great Lakes, most of the initial development interest and planning activity overlaps with the Red-throated Loon's core wintering range, and is concentrated in the state and federal waters offshore of the mid-Atlantic region (Musial and Ram 2010). In order to assess the effects of disturbance and the potential population impacts of offshore wind development on Red-throated Loons, basic information must be collected on their distribution and behavior, including flight pathways and timing of habitat use, within proposed WEAs. The distribution and abundance of marine birds in federal waters slated for offshore wind energy development have been the focus of ongoing aerial and boat-based survey efforts in recent years (Parts II-III of this report; Geo-Marine, Inc. 2010, NFSC and SFSC 2012). Few studies in North America have tracked individual marine birds, however, to determine migratory and local within-season movements to evaluate potential interactions with proposed offshore wind energy developments.

Location data collected from satellite transmitters can be used to estimate home-range size and other features of the utilization distribution of target species. Kernel density estimation is a standard technique for characterizing and visualizing animal home ranges, and identifying the utilization distributions is a probability density function that can be extended to quantify the relative frequency distribution of an animal's occurrence in space and time (Silverman 1986, Keating and Cherry 2004). From these data, resource selection functions can be used to predict habitat use across landscapes or to understand the influence of certain habitat characteristics on a species distribution (Long et al. 2009). Used resources are a subset of available resources, and a key factor in resource selection studies is determining what and how much will be included as "available" (Buskirk and Millspaugh 2006). The probability that particular habitats will be used by a species can be examined with a logistic regression model; however, decisions regarding sampling design, the underlying probability model, and assumptions must be carefully considered to ensure correct interpretation of results (Keating and Cherry 2004).

Red-throated Loons are an important study species for gathering information on potential risk associated with wind energy development in the mid-Atlantic region because: (1) they are considered vulnerable to collision mortality and displacement from offshore wind energy facilities (Garthe and Huppop 2004, Furness et al. 2013, Robinson Willmott et al. 2013), (2) a large proportion of their biogeographic population winters in this region (Forsell 1999, Root 1988), (3) they are designated as a US Fish and Wildlife Service (USFWS) "species of conservation concern" on their wintering grounds along the New England and Mid-Atlantic Coast (USFWS 2008), and (4) they have previously been successfully tracked using satellite transmitters (Schmutz et al. 2009). Consequently, we initiated a satellite tracking study of Red-throated Loons captured at sea on their wintering grounds in the mid-Atlantic U.S. to determine the species' fine-scale occurrence and local movement patterns in the study area during winter and migration.

## Methods

### *Study area*

The priority study area included federal waters off the mid-Atlantic U.S. coast from southern New Jersey to the southern border of North Carolina. Three winter capture regions adjacent to the priority study area were selected: northern (New Jersey and Delaware), central (Maryland and Virginia), and southern (North Carolina; Figure 21-1). In 2012, capture efforts for Red-throated Loons were focused on the following waterbodies: Pamlico Sound, NC; Chesapeake Bay, MD and VA; Chincoteague Bay, MD and VA; Assawoman Bay and Isle of Wight Bay, MD; Indian River Bay, DE; and Delaware Bay, DE and NJ. Capture efforts in 2013 were focused in areas where high concentrations of target species were observed during the previous years' field efforts, including: Pamlico Sound, offshore of Hatteras, NC; Chesapeake Bay, offshore of Cape Charles, VA; Delaware Bay, offshore of Lewes, DE; and the Atlantic Ocean, offshore of Cape Henlopen, DE and Chesapeake Bay Bridge area, VA (Figure 21-1).

### *Capture and PTT deployment*

We used satellite telemetry to track the locations of Red-throated Loons on their wintering grounds in the mid-Atlantic study area. Loons were deployed with abdominal-implanted platform terminal satellite transmitters (PTTs) from several capture locations during 2012 and 2013 (Table 21-1). Capture efforts were conducted between late January and late March in each year using night-lighting techniques, i.e., birds were approached at night by boat with a spotlight and captured with a dip net. Loons were aged according to criteria established by Pyle (2008) and those determined to be "after third year" (ATY) and "after second year" (ASY) were considered primary candidates for PTT implantation. All captured birds were banded with US Fish & Wildlife Service aluminum bands. Individuals deemed fit for transmitter surgery were administered 2 mg/kg of mild sedative, midazolam hydrochloride IM, prior to transfer to the surgery location to minimize stress related to capture and handling. Surgical implantations of intra-abdominal PTTs with external antenna were conducted by qualified veterinarians following techniques described by Korschgen et al. (1996). Satellite transmitters weighed approximately 49 g and comprised <4% of the average body mass of birds deployed.

Satellite transmitters were programmed with a duty cycle of 4 hours on and 13 hours off during the period of 1 November to 31 May, in order to detect fine-scale movement patterns when birds were expected to be on migration or wintering in the mid-Atlantic study area. Longer duty cycles were employed for the remainder of the year, when birds were outside of the study area, to maximize battery life: 2 hours on and 5 days off from 31 May to 31 August, and 4 hours on and 24 hours off from 31 August to 01 November. All location data collected within 14 days of deployment were excluded from analysis to reduce bias associated with surgery (Esler et al. 2000).

### *Satellite telemetry*

Telemetry data from PTTs were received via the Argos system of satellites. Data were archived, compiled, and filtered to remove redundant data and errant points using the Douglas Argos-Filter 2012 (DAF). The DAF is a threshold filter that has several user-defined parameters to flag improbable locations in satellite tracking data (Douglas et al. 2012). The parameters are adjusted based on species' movement behaviors and the scale of the area under observation. With the DAF, data are retained if they pass (1) a spatial redundancy test, and/or (2) a movement rate and turning angle test. Since bird data contain both short-distance local movements, and long-distance migratory events, we applied the hybrid filter of both the distance, angle, and rate (DAR) and minimum redundant distance (MRD) filters. The combination achieves the best of both filters and in practice has produced very clean data with few erroneous points (Douglas et al. 2012). Using DAF, we also chose to identify the best representative point per duty cycle for each animal to reduce redundant daily positional information. The DAF also generates estimates of the distance between successive points (as best representative point per duty cycle) and the amount of time between these points. Since the time between points varied between duty cycles and individuals, we also calculated the rate of distance moved by dividing distance moved by the number of hours between each duty cycle.

A custom script in Python 2.7 was developed so locations of each individual could be mapped in ArcGIS 10.2.2 (Environmental Systems Research Institute, Inc., Redlands, CA) with the DAF-filtered data. We chose to represent only one year's worth of data per life period per animal, in order to avoid biasing the dataset towards individuals with more years of data due to tag longevity or mortality. Based on evidence that the movement patterns of birds may be adversely affected in the first year following capture and implant of a satellite tag (S. G. Gilliland, unpublished data) and that tags seldom survived for an entire third year of deployment, we chose Year 2 data preferentially over Year 3 data, and Year 1 data if only one year of data existed. Only birds that transmitted >60 days after release were included, to reduce bias from birds that could have been negatively affected by transmitter implantation and handling.

Movement maps were created to show migratory movements to and from the wintering grounds. Period locational means for winter, breeding, and molting locations were used for these maps, calculated from all best locations per duty cycle available for those periods. Movement lines were created from all intermediate points between "book-end" periods such as winter and breeding, and show the link between these periods. Increasing line density where tracks were overlaid indicates areas used by multiple birds. Lines were straight-line paths between points and do not necessarily reflect the true path of the animal; the broadness and semitransparency of the lines used in these maps was intended to illustrate general migratory paths.

### *Individual home range estimation*

We calculated the home range sizes of individuals during winter by estimating the minimum convex polygon (mcp) areas ($km^2$) using package adehabitat HR version 0.4.13 (Calenge, 2006) in R version 3.1.1 (R Core Team 2014), removing 5% outliers. The first two weeks of data for every tag (the period immediately following release) was excluded from home range estimations, as were animals that transmitted for <30 days. Winter periods were identified for each individual based on arrival and departure from the wintering area.

### *Kernel density estimation of winter use*

We produced kernel density maps that showed the broad-scale utilization distribution (0.95 isopleth) and core-use areas (0.50) of satellite transmitter-tagged Red-throated Loons, to determine potential wintering range overlap with proposed WEAs in the mid-Atlantic region. Kernel density estimation involves the use of point data from telemetry to estimate relative spatial use during specified time intervals. For each location, the bird's habitat use is estimated to be greatest directly on the point, and to decrease with distance from the point (reaching zero at a bandwidth specified by the user, after Worton 1989). Following Loring et al. (2014), we used the composite KDE method (with Gaussian kernel and Likelihood cross-validation bandwith estimator), where S random points are selected for n individuals and pooled for a single composite KDE representing the utilization distribution of all animals. Fifty-four DAF-filtered locations were randomly selected from each individual's winter period. The winter period was defined for Year 1 data as 2 weeks after deployment until the individual departed the study area for spring migration and for Years 2 and 3 data as the individual's arrival date in the study area until the end of the transmitter's battery life. The number of locations randomly selected per animal ($n = 54$) was based on the 25th percentile value of the distribution of the number of location data points available for each animals. The 25th percentile value was chosen for minimization of weighting by individuals with larger numbers of available location data compared to individuals with fewer available location data due to transmitter failures or mortality events. A bootstrapping procedure was used to generate a mean utilization distribution from 100 runs of re-sampled points as described above. For maps of wintering and staging in this report, composite maps were developed consisting of all pooled, randomly sampled points generating a single composite utilization distribution.

### *Habitat selection*

We examined third-order resource selection of Red-throated Loons to determine characteristics of preferred habitats that can be used to predict probability of use across the study area. The evaluation of resource preference pertains to the measurement of habitat components an animal "uses" in its environment compared to what is "available" (Johnson 1980). We were interested in determining core use areas within the home range, which is referred to as a third-order resource selection (Johnson 1980). Using the composite kernel density estimates, we defined "used" or core use areas as telemetry point locations within the 0.50 isopleth, and compared the habitat characteristics of those points to randomly-generated points throughout the 0.95 utilization distribution, defined as "available" locations (sampling protocol-A; Manly 2002). Habitat characteristics were chosen based on *a priori* knowledge of marine habitat and availability of spatial data within the study area, and included water depth (m), distance to shore (km), long-term (10-yr winter mean) and seasonal mean sea surface temperature (°C), long-term (10-yr winter mean) and seasonal chlorophyll *a* (mg/m$^3$), long-term (6-yr winter mean) and seasonal sea surface salinity (practical salinity units, psu), seafloor slope (°), and sediment grain type. We measured water depth and slope using the NOAA National Geophysical Data Center 3 arc-second Coastal Relief Model for the United States (NOAA 2014a). To estimate distance to shore, we calculated the Euclidean distance between Red-throated Loon locations and the nearest segment of the NOAA Medium Resolution Digital Vector Shoreline (1:70,000) using the Near Tool in Arcmap 10.2.2 (ESRI, Redlands, CA; NOAA 2014b). Estimates of seafloor slope were obtained from the Nature Conservancy's Northwest Atlantic Marine Ecoregional Assessment data portal. The benthic habitat layer contains an estimate of slope, which is calculated as the difference in elevation between two neighboring raster cells, expressed in degrees, and were grouped accordingly: (1) 0-0.015° = level flat; (2) 0.015-0.05° = flat; (3) 0.05-0.80° = gentle slope; 0.80-8.0° = slope; and > 8.0° = steep slope (Greene et al. 2010). Sediment grain size categories were also obtained from the Nature Conservancy's Northwest Atlantic Marine Ecoregional Assessment data portal. Size categories were grouped by the Nature Conservancy according to correlations with benthic habitat communities, and are not necessarily related to Red-throated Loon habitat requirements. For dynamic variables with a temporal component, we relied on Marine Geospatial Ecology Tools to create seasonal and long-term winter mean climatology rasters for sea surface temperature, chlorophyll a, and sea surface salinity (Roberts et al. 2010). We used smoothed daily Sea Surface Temperature (SST; °C) estimates based on optimal interpolation of data derived from high resolution satellite imagery and floating buoys (Stark et al. 2007). These data were produced by the UK Met Office on a global scale at a spatial resolution of 0.054 degrees latitude and longitude. Sea surface salinity (SSS; psu) was estimated using the Hybrid Coordinate Ocean Model (HYCOM) produced by the National Ocean Partnership Program at a spatial scale of 1.5 degrees latitude and longitude (Chassignet et al. 2009). To measure ocean productivity, we obtained monthly estimates of Chlorophyll *a* concentration (mg/m$^3$) produced by the NASA Goddard Space Flight Center's Ocean Data Processing System. These data had a spatial resolution of 4 km and were derived from radiometric measurements of chlorophyll fluorescence made by the Aqua sensor aboard the MODIS satellite system (Mueller et al. 2003).

We used logistic regression to model habitat covariate effects on used versus available locations over three winter periods (2011-2012, 2012-2013, and 2013-2014) within the study area. Candidate models were developed for all years combined and for each winter period. Development of models was exploratory, but, based on a priori knowledge of Red-throated Loon habitat, we predicted that used locations would be related to nine habitat covariates: depth, distance to shore, long-term and seasonal chlorophyll *a*, long-term and seasonal sea surface temperature (SST), long-term and seasonal sea surface salinity (SSS), and seafloor slope. Correlations between pairs of habitat variables were quantified using a Pearson product-moment correlation matrix. Multicollinearity among covariates was assessed by calculating variance inflation factors (VIF). Highly correlated variables included: (1) long-term SST to seasonal SST, (2) long-term to seasonal SSS, and (3) long-term to season chlorophyll-a concentration. Therefore, models included the long-term or seasonal term for each of those variables but not both. All other pairwise comparisons had correlations of < 0.60 and VIF values of < 2.3 and were retained as variables in the modeling process, resulting in six variables considered for inclusion in each model. The square of both water depth and distance to shore was also included in order to examine the possibility that their relationship to the log odds of habitat use was curvilinear rather than linear. Additionally, the product of each pair of continuous variables was included in order to assess possible interactions among the predictors. Candidate models were ranked with Akaike Information Criterion adjusted for small sample size ($AIC_c$). The model with the lowest $AIC_c$, and those having $\Delta AIC_c \leq 2$ had the most statistical support, values between 4 and 7 had considerably less support, and those > 10 had virtually no support (Burnham and Anderson 2002). The Akaike weight was also considered when determining the relative amount of statistical support for each model.

## Results

### *Capture and PTT deployment*

Forty-three Red-throated Loons were released with PTTs during the winters of 2012 (*n* = 17) and 2013 (*n* = 26), of which there were 24 females, 13 males, and six of unknown sex. Body mass of females ranged from 1,400-2,000 g and for males from 1,900-2,200 g. In each year, eight loons died within the immediate 14-day post-release period, which is the time period in which mortality can confidently be attributed to surgery (Mulcahy and Esler 1999). This corresponds with mortality rates of 41% for 2012 and 31% for the 2013 season.

### *Satellite telemetry*

We observed local movements in the bays where individuals were captured and released, as well as larger scale movements in the near shore environment along the coast. More movement along the coast, between Delaware Bay and Pamlico Sound, was observed during the winter months among birds tagged in 2013 as compared to 2012 birds. The difference in loon movement between years is likely the result of more 2013 captures targeted in the mouths of Chesapeake Bay and Delaware Bay and offshore areas, as compared to the 2012 captures, which largely occurred in more interior locations of the bays.

Distances moved between duty cycles while in the study area were calculated for the winter period following deployment, which generally reflected movements during late winter and early spring, i.e., March, April, and May. These estimates ranged from 0 – 427 km for all animals combined (n = 22), and mean distance moved per animal ranged from 1.1 – 129 km. The resulting movement rates ranged from 0 – 15 km/hr moved for all tagged birds combined (n = 23) and the mean per animal ranged from 0.1 – 3.3 km/hr. Movement data was also available the winter after deployment for 16 individuals following their return from the breeding grounds, and generally reflected movements occurring in early winter, i.e., November, December and January. Distance moved between duty cycles for those birds combined ranged from 0.3 – 647km, and the mean distance moved per animal ranged from 8.5 – 118 km. The rate of distance moved for all birds combined during the second winter season ranged from 0.01 – 34 km/hr and the range in means for distance moved per individual bird was 0.4 – 7.3 km/hr.

Satellite tracking also showed the spring migration routes of 22 loons and the fall migration routes of 17 loons, information that was not previously available for this geographic region. Spring migration departure for the breeding grounds occurred between 23 March and 01 May (Figure 21-2). Most movements were along the coast, with the greatest offshore movements observed during the departure from the study area for spring migration. The majority of the birds followed the east coast of the Atlantic northward, and spent periods of time in Raritan Bay, NJ; Lower New York Harbor and the south shore of Long Island, NY; Narragansett Bay, RI; Nantucket Sound and Cape Cod Bay, MA; the Gulf of Maine, including the Maine coastline and Bay of Fundy; the Gulf of St. Lawrence, NB and QC and the St. Lawrence River, QC; James Bay, ON and QC; and Hudson Bay, ON, QC, MB, and NU. The southward fall migration included long layovers in Hudson Bay (4-8 weeks) before moving to the Great Lakes and the Gulf of St. Lawrence. Individuals that utilized the Great Lakes moved to Delaware Bay and down the coast to their wintering areas, while those that moved to the St. Lawrence followed the Atlantic coastline southward. Red-throated Loons in our sample returned to their wintering grounds in the mid-Atlantic study area between 15 November and 29 December (Figure 21-3).

### *Home ranges and kernel density estimation of winter use*

Kernel density estimation maps were generated based on the highest quality daily location per duty cycle for Red-throated Loons with sufficient data ($n$ = 22). Utilization distributions of Red-throated Loons included areas of: Delaware Bay and the Delaware Atlantic coastline; coastal Maryland and Virginia near Chincoteague Bay; Chesapeake Bay from Annapolis south to the mouth of the bay; and the Atlantic coastline from the Chesapeake Bay Bridge to the southern end of the Outer Banks of North Carolina (Figure 21-4). Core use areas within this UD were located in: Delaware Bay offshore of Fortescue Fish and Wildlife Management Area; Indian River Bay and coastal areas offshore of Rehoboth Beach; upper Chesapeake Bay near Annapolis, Tangier Sound; central Chesapeake Bay; Potomac and Rappahannock Rivers; southern Chesapeake Bay; coastal Atlantic from Bay Bridge south to Albemarle Sound; and mid- to eastern Pamlico Sound and offshore Outer Banks, North Carolina.

Individual home ranges of satellite-tagged Red-throated Loons following deployment ranged from 14.8 – 62,648 km$^2$ ($\bar{x}$ = 6,980 ± 14,132 km$^2$; n = 22) during their first winter period and from 23 – 12,491 km$^2$ ($\bar{x}$ = 3496 ± 4290 km$^2$; n = 16) for their second winter period. Individuals in the 2012 sample were captured in more interior bay locations and had smaller home ranges ($\bar{x}$ = 631 ± 1366 km$^2$) compared to birds captured in more coastal locations in 2013 ($\bar{x}$ = 11,741 ± 17,437 km$^2$).

### *Habitat selection*

Across all three winter periods, little variation was observed between habitat characteristics associated with core use areas and available areas within the home range (Table 21-2). The greatest differences observed were in seasonal mean chlorophyll *a* concentrations ($\bar{x}$ = 7.73 ± 0.17 mg/m$^3$ in core use areas vs. $\bar{x}$ = 10.15 ± 0.17 mg/m$^3$ in available areas) and distance to shore ($\bar{x}$ = 5.81 ± 0.91 km in core use areas vs. $\bar{x}$ = 6.92 ± 0.16 km in available areas). Model results for all years combined showed a high amount of unexplained variation indicating a need to incorporate other potential habitat variables. Similarly, models examining resource selection during individual winter periods exhibited high unexplained variation for the 2011-2012 and 2013-2014 model periods. Models for the 2012-2013 winter period, however, performed well and the top model, which accounted for 0.97 of the Akaike weight, indicated a positive effect on habitat use related to increasing slope, sea surface salinity, and the compound effect of sea surface salinity and sea surface temperature, and a negative effect on use related to increasing sea surface temperature, distance to shore, and water depth; all other coefficients included in the top model had confidence intervals that included zero and were not considered reliable (Table 21-3 and Table 21-4). Among sediment types, core use areas in the top 2012-2013 model more frequently consisted of fine sand bottoms with grain sizes that ranged from 0.03-0.35 mm (Table 21-5). Considerable variation was observed in the mean habitat values associated with used and available locations between years (Table 21-6). In particular, seasonal and long-term chlorophyll *a* concentrations, water depth, and slope associated with Red-throated Loon core use areas were considerably greater during the 2012-2013 winter period compared to the 2013-2014 winter period. Conversely, mean values for long-term and seasonal sea surface temperatures and distance to shore were smaller for the 2012-2013 winter period compared to the 2013-2014 winter period.

### Discussion

The heaviest use was observed around the mouth of the Chesapeake Bay, south along the coast to the Virginia/North Carolina border, and the central eastern portion of Pamlico Sound and coastal waters off Cape Hatteras National Seashore. Small areas in central and northern Chesapeake Bay, as well as areas along the Delaware coast and Delaware Bay, were also heavily used by loons in our sample. Red-throated Loon winter use near potential wind energy lease areas was greatest along the western edge of North Carolina off the Outer Banks; spring and fall migration movements also overlapped with that area. Migration trajectories through the New Jersey and Delaware lease blocks were heavier during the spring period than in fall. This seasonal difference was likely the result of more birds using an overland migration strategy in the fall, by pushing through the Great Lakes directly to Delaware Bay and Chesapeake Bay, whereas spring migration typically involved more movement along the Atlantic coast. Satellite tracking showed that loons in our sample arrived in the study area between mid-November and late December. These arrival dates are in keeping with observations from the Cape May Bird Observatory’s annual coastal count of birds passing southern New Jersey, which indicated that the majority of Red-throated Loons (1993 to 1997 mean: $\bar{x}$ = 57,679) wintering in the mid-Atlantic arrive between 15 October and 15 December (Forsell 1999). Departure from the study area for spring migration began in late March, and all birds had left by the beginning of May. In general, the greatest chance for interaction between Red-throated Loons in our sample and potential WEAs occurred during the migration periods rather than the winter period.

Winter habitat used by Red-throated Loons in Europe has been described as nearshore, sandy, shallow marine waters (Guse et al. 2009, O’Brien et al. 2008). However, limited information is available in the literature regarding measured habitat characteristics. Our model results examining the resource selection of Red-throated Loons in our study indicated that wintering birds predominantly used waters that were near-shore, shallow, and over flat, sandy substrate in the bays and coastal areas of the mid-Atlantic region. However, none of our tested models did an adequate job of explaining habitat use for

Red-throated Loons tracked in this study, indicating the need to explore additional habitat variables in future modeling efforts. Small pelagic fish species, such as American Sand Lance (*Ammodytes americanus*), are important food source for seabirds within the mid-Atlantic region and are likely a driving force for resource selection by Red-throated Loons. Spatial data on the distribution of small pelagic fisheries are limited, however, because these species are typically missed during traditional bottom trawl surveys (Greene et al. 2010). In the absence of such critical information, we were limited in our ability to determine a probability function for Red-throated Loon resource selection in the study area. The model for the winter period of 2012-2013 performed well and highlighted several important habitat characteristics, including water depth, distance to shore, chlorophyll *a* concentration, sea surface temperature, and salinity. Interpretation of the coefficients from this model and how they relate to Red-throated Loon habitat use should be conducted with caution, however, given the variation observed in these covariates between years. Quantifying the habitat characteristics within core use areas produced the most reliable information regarding Red-throated Loons use in the mid-Atlantic region. Loons in our sample used tidal rivers, bays, and ocean locations, and core use areas were generally 5-10 km from shore. Most ocean locations did not exceed 5 km from shore and were decidedly in the "nearshore" environment. Accordingly, water depth associated with use was shallow, with more than half the locations not exceeding 13 m. Among Red-throated Loons in the Baltic Sea, the greatest densities of birds were observed within an area with a water depth zone of 5-30 m (Skov et al. 2011). Similar to our findings, Warden (2010) found that the majority of Red-throated Loons taken as bycatch in mid-Atlantic waters occurred in waters less than 8 m deep (84%), while just 16 % occurred in 8-12 m deep water, and none were observed in water ≥ 12 m. Further, 43% of takes were in water that was < 8°C, 44% occurred in 8–12 °C water, and just 13 % were in water ≥ 12 °C. Our results indicated considerable use of waters that exceeded 12 °C; however, mean winter sea surface temperature associated with core use areas was below 12 °C.

Estimates of individual home range size varied widely, and were potentially affected by multiple factors. Capture locations in 2012 were limited to more interior locations of Chesapeake and Indian River Bays, and while these areas provide important habitat to wintering Red-throated Loons, data obtained from individuals captured at the sites may not fully represent the species' habitat use in more marine locations in the study area. Specifically, the mean home range size of loons captured near the mouths of Delaware and Chesapeake Bays and Pamlico Sound were on average larger compared to birds captured in more interior locations, suggesting that there may be large differences in movements and site fidelity among loon populations wintering in the region. A closer examination of home range size between winters for individuals that provided more than one season of data, however, shows great inter-annual variation in home range size. For example, one individual that was captured within Indian River Bay, DE in late February 2012 utilized a 110 km$^2$ home range between February 29 and April 8 that first year, and had a 5,708 km$^2$ home range between November 19 and February 4 the following winter. Many factors likely contributed to the observed variation in this metric, including:

(1) Differences in time of year. Specifically, first season winters occurred immediately after capture during the months of February, March and April, whereas second season winters included the

period of November to January, representing the period between an individual's return to the breeding ground and the transmitter battery reaching its maximum life span (≈ 1 year);

(2) Differences in length of time that individuals were tracked within a winter season, which was largely based on bird survival, number of quality data transmissions, and battery life; and

(3) Intra-and inter-annual variation in behavior, potentially associated with multiple factors, such as food resources, weather events, seasonal temperatures, age and condition of the bird, etc. The health of the bird, in particular, is an important consideration when considering home range size of the first winter period following surgical implantation of a satellite transmitter.

The range of mean rates of distance moved between duty cycles was greater for the second winter period compared to the winter period following deployment; however, it is difficult to determine whether differences in time of year and food availability better explain these differences than condition of the bird. The significance of these effects will be explored more thoroughly in future modeling efforts that incorporate additional years of data and can potentially provide a more accurate description of Red-throated Loon home range size on its wintering grounds in the mid-Atlantic region.

Our initial attempts to incorporate the utilization distributions of Red-throated Loons and habitat data associated with sampled locations into a resource selection function were not successful. However, data from 43 additional Red-throated Loons satellite-tagged during the winters of 2014 and 2015 will be available in early 2016, and are expected to improve the performance of the models. Model results can be used to predict probability of use of resource units, such as WEA lease blocks, based on the measured habitat characteristics of those units. Further, adjusting the ratio of the used to available points in the sampling scheme to Manly's (2002) recommended 2:1 design, and exploring other available habitat variables, may further improve the models (Manly et al. 2002). The ability to produce robust resource selection functions is critical to confidently identifying core use habitats within marine spatial planning areas.

The relatively large winter ranges of Red-throated Loons documented by our study increases the likelihood of displacement due to offshore WEAs compared to other seabirds with smaller ranges. Displacement of Red-throated Loons from several kilometers around wind farm footprints in Europe has been estimated to result in 89-94% reduction in loon densities at some locations (Petersen et al. 2006, Percival 2014), with no sustained evidence of habituation or return to the areas post-construction. Significant stressors already exist for this species on the wintering grounds, including exposure to pesticides and other contaminants, such as mercury and PCBs (Eriksson et al. 1992, Schmutz et al. 2009), oil spills, habitat degradation, and mortality from fishing nets (Zydelis et al. 2013). Red-throated Loons are particularly susceptible to mortality in the mid-Atlantic region via bycatch in gillnet fisheries. For example, of the 2,387 birds observed dead in gillnets in the mid-Atlantic region between February and April 1998, 68% were Red-throated Loons and 21% were Common Loons (*Gavia immer*; Forsell 1999). Another 825 dead Red-throated Loons were found washed ashore on beaches that winter. In total, Forsell (1999) estimated that this represented 1.2 to 2.4 % of the total number of fall migrating birds counted at Cape May earlier that season. Warden (2010) calculated that each year between 620 and 1,297 Red-throated Loons were taken as bycatch in commercial gillnet fisheries in the mid-Atlantic region, totaling an estimated 10,758 birds between 1996 and 2007. Additional mortalities on the wintering grounds, or indirect effects to wintering loons via displacement from important habitat areas, could have detrimental effects on the wintering population of Red-throated Loons in this region, and therefore it is critical that important foraging areas and movement pathways of loons and other vulnerable species are identified and considered during the planning and development phases of offshore wind energy development. We will continue to pursue this analysis to achieve these objectives, and results will be forthcoming in an anticipated manuscript publication in 2016.

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## Figures and tables

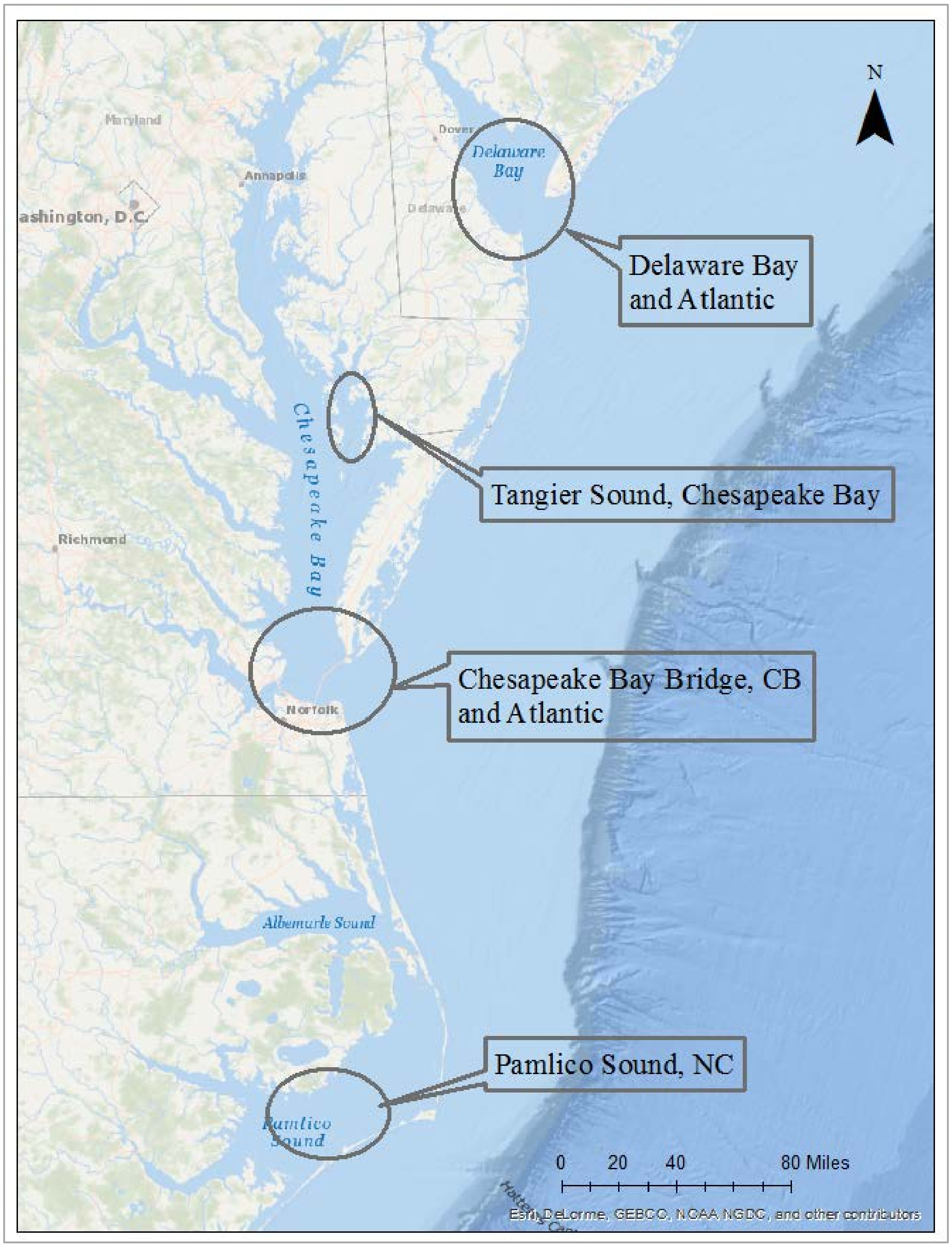
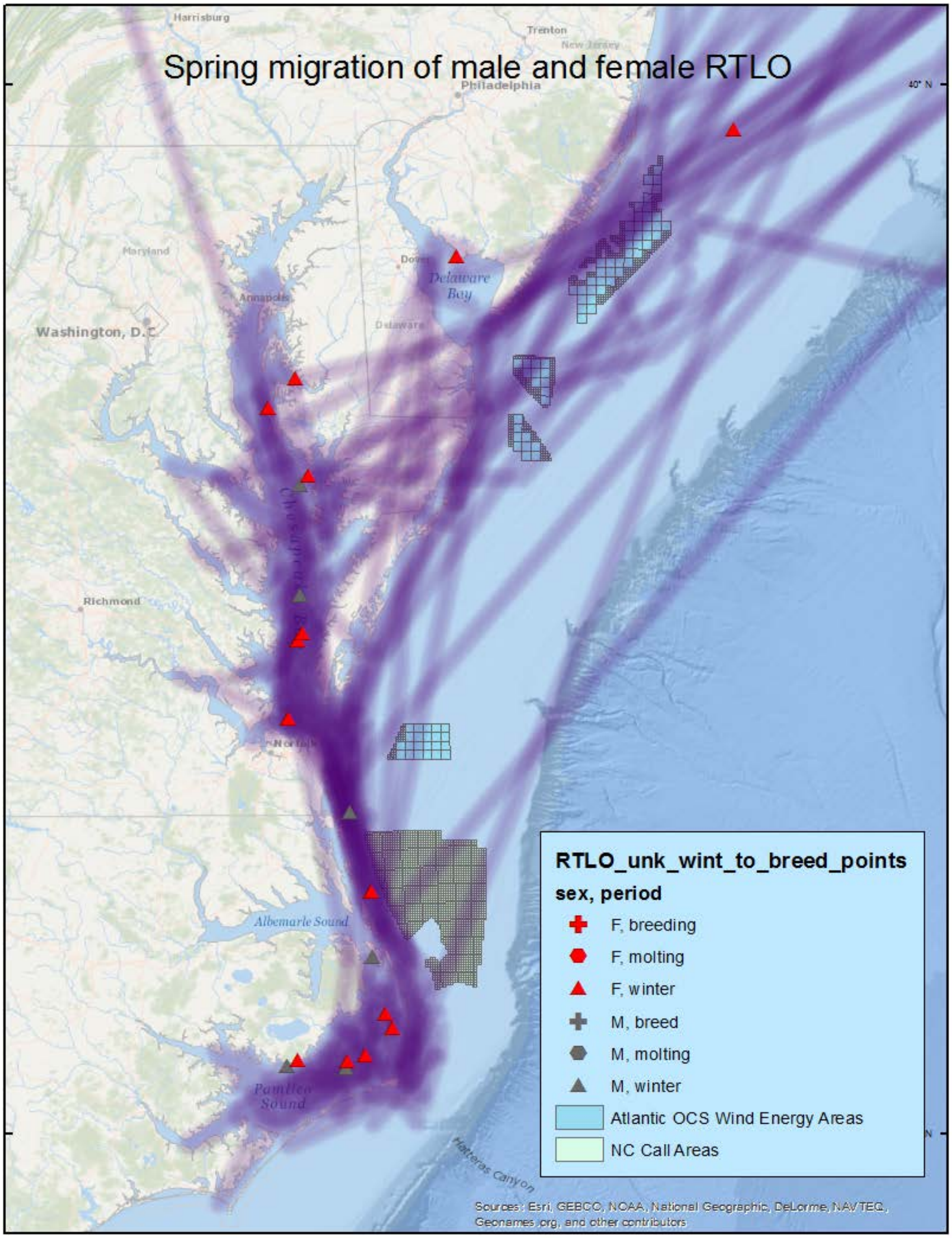
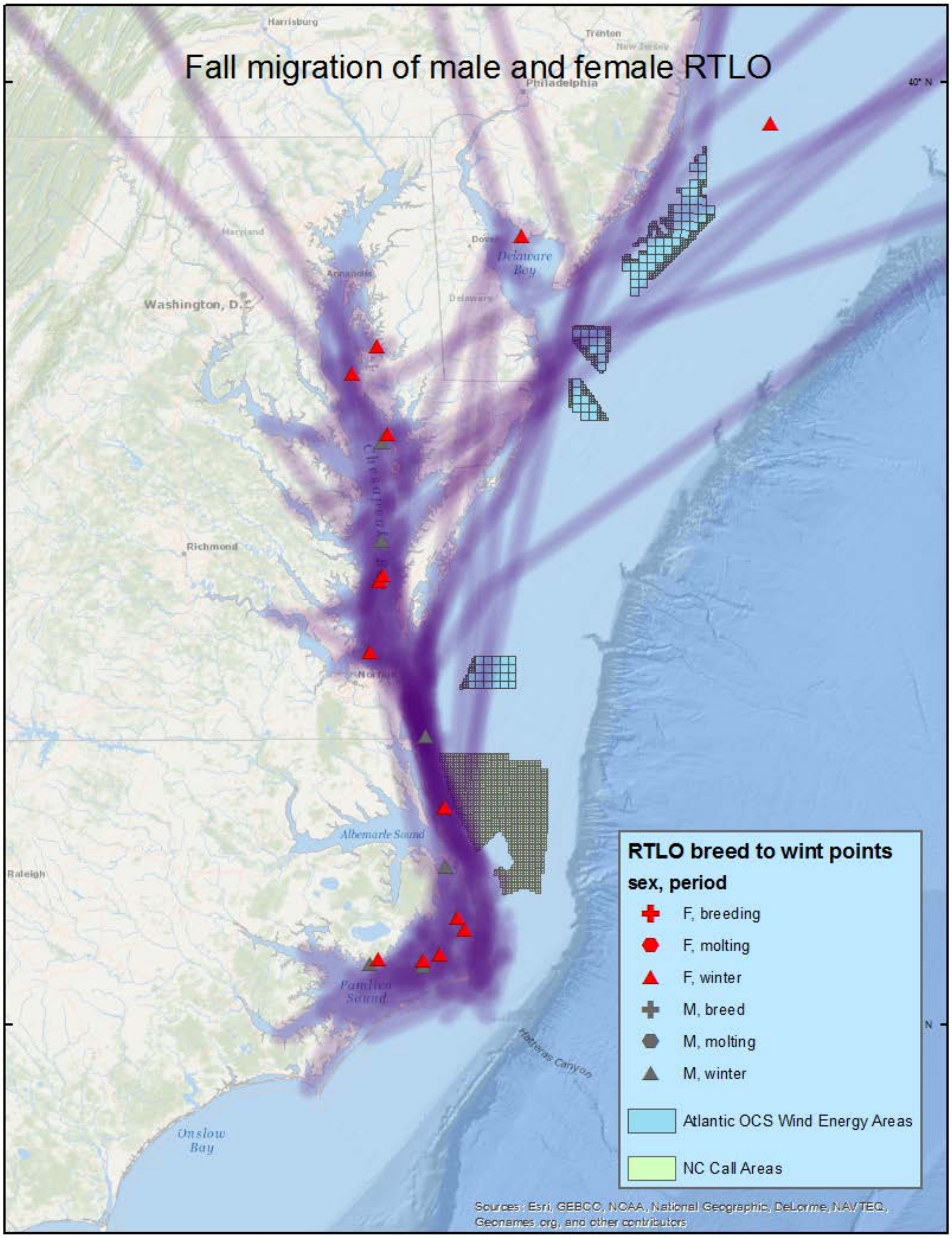


**Figure 21-1. Red-throated Loon satellite-tracking study area.** Capture efforts were concentrated in Delaware Bay (DE & NJ), Chesapeake Bay (MD & VA), and Pamlico Sound, NC.



Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, NAVTEQ, Geonames.org, and other contributors

**Figure 21-2. Spring migration movement map of Red-throated Loons departing the mid-Atlantic study area between late March and early May, 2012 – 2013.** Points are locational means for each individual and life history period.


Fall migration of male and female RTLO

Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, NAVTEQ, Geonames.org, and other contributors

**Figure 21-3. Fall migration movement maps of Red-throated Loons arriving in the mid-Atlantic study area between mid-November and late-December, 2012 – 2013.** Points are locational means for each individual and life history period.

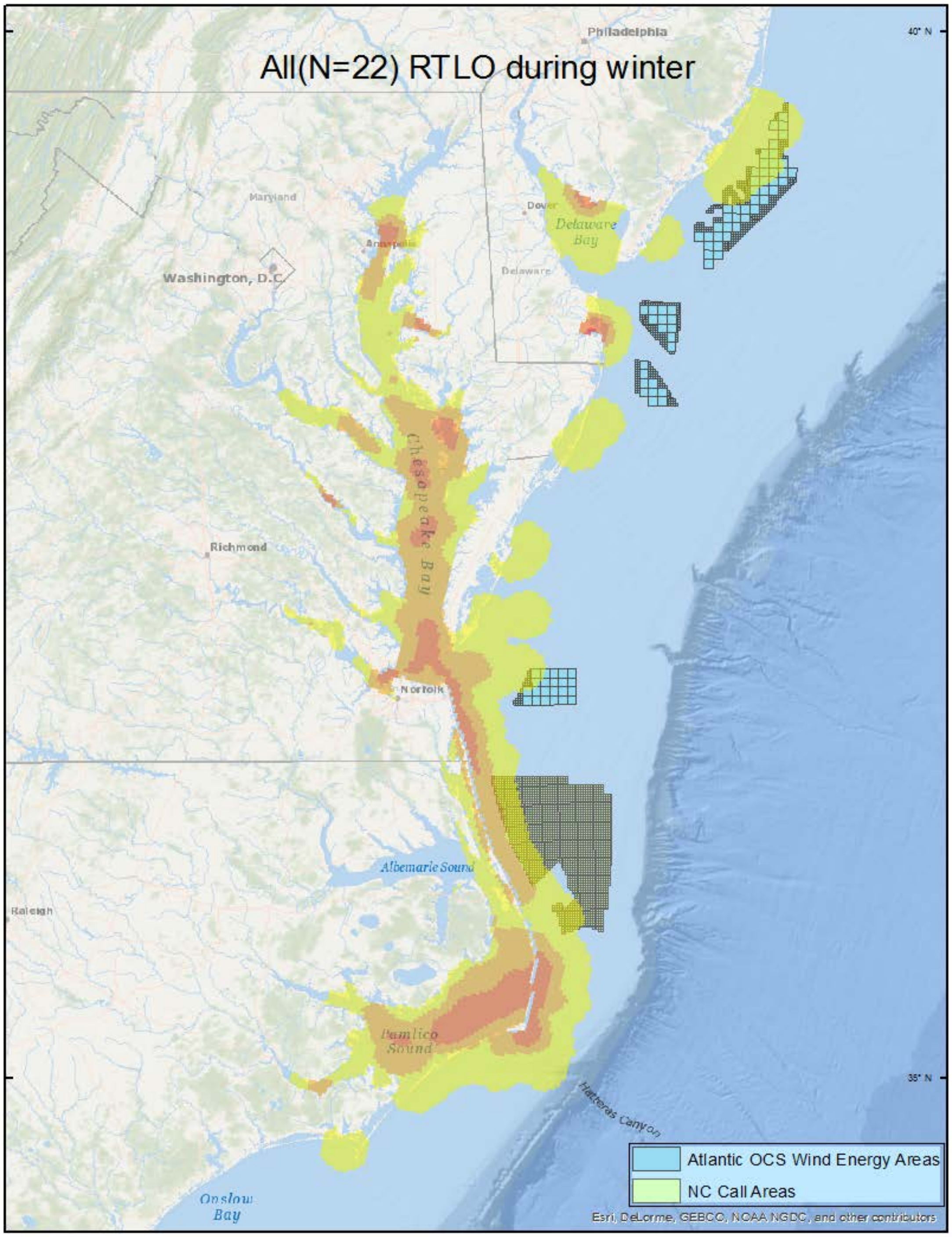


**Figure 21-4. Winter use by Red-throated Loons of areas within the mid-Atlantic study area in relation to proposed offshore WEAs.** Intensity of use ranges from lowest areas of use (yellow) to greatest areas of use (red).

**Table 21-1. Number of Red-throated Loons implanted with satellite transmitters by capture location during the winters of 2012 and 2013.**

| Capture Location | 2012 | 2013 | *Total* |
|---|---|---|---|
| Chesapeake Bay, MD | 11 | — | ***11*** |
| Chesapeake Bay, VA | — | 6 | ***6*** |
| Delaware Bay, DE | 1 | 9 | ***10*** |
| Indian River Bay, DE | 3 | — | ***3*** |
| Pamlico Sound, NC | 2 | 11 | ***13*** |
| ***Total*** | ***17*** | ***26*** | ***43*** |

**Table 21-2. Range, mean ($\bar{x}$) and standard error (SE) of habitat variables at satellite-derived locations for Red-throated Loons in the core use area (0.50 isopleth) and for random points across the utilization distribution (available; 0.95 isopleth).**

| Habitat Variable | Available | | Core Use | |
|---|---|---|---|---|
| | range | $\bar{x}$ ± SE | range | $\bar{x}$ ± SE |
| ***Long-term*** | | | | |
| Chlorophyll *a* (mg/m$^3$) - 10 yr Mean | 0.91 – 44.85 | 8.34 ± 0.13 | 0.92 – 44.85 | 8.56 ± 0.13 |
| Sea Surface Temperature (°C) – 10 yr Mean | 3.71 – 18.09 | 9.98 ± 0.11 | 3.78 – 18.86 | 9.89 ± 0.09 |
| Sea Surface Salinity (psu) – 6 yr Mean | 17.65 – 35.13 | 32.57 ± 0.04 | 17.65 – 35.35 | 32.67 ± 0.04 |
| ***Physical*** | | | | |
| Depth (m) | 0.10 – 128.40 | 9.00 ± 0.20 | 0.10 – 127.00 | 8.68 ± 0.15 |
| Slope (degrees) | <0.0001 – 3.86 | 0.16 ± 0.006 | <0.0001 – 5.38 | 0.20 ± 0.007 |
| Distance to Shore (km) | 0.50 – 29.77 | 6.92 ± 0.16 | 0.50 – 40.52 | 5.81 ± 0.91 |
| ***Short-term*** | | | | |
| Seasonal Chlorophyll *a*(mg/m$^3$) | 1.04 – 62.17 | 10.15 ± 0.22 | 1.08 – 95.04 | 7.73 ± 0.17 |
| Seasonal Sea Surface Temperature (°C) | 2.54 – 19.40 | 9.55 ± 0.11 | 0.00 - 20.46 | 9.76 ± 0.11 |
| Seasonal Sea Surface Salinity (psu) | 16.87 – 35.38 | 32.72 ± 0.05 | 17.39 – 35.56 | 32.82 ± 0.04 |

**Table 21-3. Logistic regression model selection results examining the effects of slope (°), sediment grain size category (mm; sed), mean winter sea surface salinity (psu; sssw), mean winter sea surface temperature (°C; sstw), distance to shore (km; dist), depth (m; dep), and mean winter chlorophyll-a concentration (mg/m$^3$; chlorw) on winter habitat use versus availability by Red-throated Loons in the Mid-Atlantic region during the winter of 2012 - 2013.** The table shows the variables included in the model, number of estimated parameters (K), model Akaike Information Criterion (AIC$_c$), differences between model AIC$_c$ and the top model (ΔAIC$_c$), and AIC$_c$ weights ($w_i$). Only models with ΔAIC$_c$ of <7 are included in the table.

| Model parameters | K | AIC$_c$ | ΔAIC$_c$ | $w_i$ |
|---|---|---|---|---|
| slope+sed+sssw+sstw+(sstw*sssw)+dist+(sstw*dist)+dep$^2$+ (dep$^2$*dist)+chlorw+(sstw*chlorw) | 14 | 787.456 | 0.000 | 0.973 |

**Table 21-4. Coefficients (β) and 95% confidence intervals (lower and upper) of best-fit resource selection model for Red-throated Loons wintering in the mid-Atlantic region in 2012-2013.**

| Variable | β | Lower | Upper |
|---|---|---|---|
| Slope (°) | 0.667 | -0.078 | 1.56 |
| sediment grain size (mm) | | | |
| 0.00 - 0.03 clay/silt | 0.21 | -0.6 | 1.04 |
| 0.03 - 0.17 very fine sand | 0.16 | -0.29 | 0.61 |
| 0.17 - 0.35 fine sand | -0.15 | -0.48 | 0.18 |
| 0.35 - 0.36 sand | 0.39 | -0.23 | 1.02 |
| 0.36 - 0.48 medium sand | -0.32 | -0.83 | 0.2 |
| seasonal sea surface salinity (psu) | 0.74 | 0.61 | 0.88 |
| seasonal sea surface temperature (°C) | -1.85 | -2.21 | -1.52 |
| seasonal sea surface salinity*seasonal sea surface temperature | 0.43 | 0.32 | 0.55 |
| distance to shore (km) | -0.39 | -0.51 | -0.27 |
| depth$^2$ (m) | 0.02 | 0.003 | 0.03 |
| depth$^2$ (m)*distance to shore (km) | 0.001 | 0.0009 | 0.002 |
| seasonal chlorophyll-*a* concentration ( mg/m$^3$) | -0.88 | -1.06 | -0.72 |
| seasonal sea surface temperature*seasonal chlorophyll-*a* concentration | -0.21 | -0.26 | -0.17 |

**Table 21-5. Proportion of use of different sediment types at satellite-derived locations for Red-throated Loons in the core use area (0.50 isopleth) and for random points across the utilization distribution (available; 0.95 isopleth).**

| Grain Size (mm) | Sediment Type | Available | | Core Use | |
|---|---|---|---|---|---|
| | | n | % Use | n | % Use |
| 0.00 - 0.03 | Silt/Mud | 157 | 10 | 287 | 13 |
| 0.03 - 0.17 | Very Fine Sand | 447 | 30 | 736 | 32 |
| 0.17 - 0.35 | Find Sand | 559 | 37 | 773 | 34 |
| 0.35 - 0.36 | Sand | 76 | 5 | 126 | 5 |
| 0.36 - 0.48 | Medium Sand | 116 | 8 | 136 | 6 |
| 0.48 + | Coarse Sand - Gravel | 148 | 10 | 218 | 10 |

**Table 21-6. Mean ($\overline{x}$) and standard error (SE) of habitat variables at satellite-derived locations for Red-throated Loons in the core use areas (used; 0.50 isopleth) and for random points across the utilization distribution (available; 0.95 isopleth) by winter period.**

| Habitat Variable | Point Type | n | 2011 – 2012 | n | 2012 - 2013 | n | 2013 -2014 |
|---|---|---|---|---|---|---|---|
| ***Long-term*** | | | | | | | |
| Chlorophyll *a* - 10 yr Mean (mg/m$^3$) | Used | 111 | 17.40 ± 0.72 | 1068 | 8.31 ± 0.16 | 781 | 7.64 ± 0.11 |
| | Available | 403 | 9.34 ± 0.38 | 626 | 8.01 ± 0.21 | 356 | 7.80 ± 0.17 |
| Sea Surface Temp – 10 yr Mean (°C) | Core Use | 206 | 5.31 ± 0.25 | 1377 | 9.40 ± 0.12 | 795 | 11.93 ± 0.13 |
| | Available | 459 | 8.98 ± 0.17 | 707 | 10.34 ± 0.17 | 384 | 10.53 ± 0.19 |
| Sea Surface Salinity – 6 yr Mean (psu) | Used | 13 | 32.82 ± 0.47 | 681 | 32.73 ± 0.04 | 279 | 32.51± 0.05 |
| | Available | 226 | 32.53 ± 0.11 | 302 | 32.64 ± 0.06 | 184 | 32.51± 0.05 |
| ***Physical*** | | | | | | | |
| Depth (m) | Used | 209 | 5.20 ± 0.71 | 1366 | 9.70 ± 0.20 | 795 | 7.85 ± 0.17 |
| | Available | 452 | 10.39 ± 0.48 | 701 | 8.40 ± 0.29 | 382 | 8.48 ± 0.25 |
| Slope (degrees) | Used | 197 | 0.32 ± 0.03 | 1320 | 0.23 ± 0.01 | 722 | 0.12 ± 0.01 |
| | Available | 423 | 0.18 ± 0.02 | 653 | 0.16 ± 0.01 | 361 | 0.16 ± 0.01 |
| Distance to Shore (km) | Used | 215 | 2.92 ± 0.30 | 1392 | 4.75 ± 0.11 | 795 | 8.67 ± 0.15 |
| | Available | 457 | 6.40 ± 0.21 | 709 | 6.98 ± 0.15 | 385 | 7.44 ± 0.21 |
| ***Short-term*** | | | | | | | |
| Seasonal Chlorophyll *a* (mg/m$^3$) | Used | 0 | — | 1141 | 8.58 ± 0.26 | 785 | 6.49 ± 0.10 |
| | Available | 421 | 12.93 ± 0.58 | 655 | 10.36 ± 0.34 | 369 | 6.60 ± 0.14 |
| Seasonal Sea Surface Temperature (°C) | Used | 0 | — | 1377 | 9.93 ± 0.11 | 795 | 11.99 ± 0.16 |
| | Available | 459 | 7.08 ± 0.17 | 707 | 10.66 ± 0.16 | 384 | 10.48 ± 0.23 |
| Seasonal Sea Surface Salinity (psu) | Used | 0 | — | 681 | 32.69 ± 0.04 | 279 | 33.15 ± 0.07 |
| | Available | 226 | 32.56 ± 0.12 | 302 | 32.59 ± 0.06 | 184 | 33.13 ± 0.09 |

# Chapter 22: Wintering movements and habitat use of Northern Gannets (*Morus bassanus*) in the mid-Atlantic U.S.

### Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 22 Highlights

*Wintering movements and habitat use of Northern Gannets (Morus bassanus) in the mid-Atlantic U.S.*

### Context[1]

Location data collected from satellite transmitters can be used to estimate home-range size and other features of the utilization distribution of target species. From these data, resource selection functions can be used to predict habitat use across landscapes or to understand the influence of certain habitat characteristics on a species distribution. We used this approach to analyze location data collected from satellite-tagged Northern Gannets captured at sea on their wintering grounds in the mid-Atlantic U.S., to determine occurrence and movement patterns in relation to potential offshore wind energy areas in the region. Similar analyses were conducted to examine movements of Surf Scoters (Chapter 20) and Red-throated Loons (Chapter 21), and time variant kernel density models were produced to examine the broad scale movement patterns of all three species throughout the year (Chapter 23). A companion chapter (Chapter 24) utilizes Northern Gannet telemetry data to understand more detailed behaviors, in addition to the movements and general habitat use information that is presented here. Northern Gannet interactions with offshore wind energy development are hypothesized to largely occur as a result of their foraging behaviors, so being able to differentiate between foraging and other behaviors in telemetry data will allow us to better determine areas of potential conflict between offshore wind energy development and Northern Gannet habitat use. Chapter 24 identifies locations that were consistently used by Northern Gannets for foraging, and what habitat characteristics defined these foraging areas.

### Study goal/objectives

Investigate the spatial patterns, temporal patterns, and environmental variation in Northern Gannet migration and winter habitat use through a combination of satellite telemetry data and remotely collected environmental covariate information.

### Highlights

- Northern Gannet core habitat from our sample included the protected inshore waters of the major bays and bay mouths. Although capable of roaming widely (up to 50 km from shore), most ocean locations were within 10 km of shore and in depths of around 20 m or less.
- Northern Gannets in our sample also used Outer Continental Shelf waters widely throughout the period in which they were present in the mid-Atlantic region. Individual Northern Gannets displayed rapid and extensive movements up and down the eastern seaboard over the course of the winter season, between the core use areas.

### Implications

Broad-scale movements of Northern Gannets within the study area increase the likelihood that they would encounter offshore wind developments repeatedly throughout the winter, depending on size and the number of offshore facilities. Based on European studies, this exposure may lead to displacement of gannets from wind farm footprints or to collision mortality from turbines.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

The Northern Gannet (*Morus bassanus*) is the largest seabird to breed in the North Atlantic Ocean. It exhibits a pelagic distribution and is emblematic of continental shelf waters on both sides of the North Atlantic. In North America, they breed at six colonies: three in the Gulf of St. Lawrence, Québec, and three off the eastern and southern coasts of Newfoundland. On migration, Northern Gannets move widely down the eastern seaboard to winter in the shelf waters of the mid-Atlantic region, the South Atlantic Bight, and the northern Gulf of Mexico (Nelson 1978, Fifield et al. 2014). In Europe, the Northern Gannet is highly ranked for collision risk with offshore wind turbines, and has also shown displacement from areas with offshore wind facilities (Johnston et al. 2014, Vanermen et al. 2015).

In the winters of 2011-12 and 2012-13, we captured 35 Northern Gannets and outfitted them with satellite tags, tracking their daily movements within the study area in winter, and their broader migratory movements throughout the remainder of the annual cycle. In each winter season, kernel density rasters were generated as a composite of sub-sampled points from one year's worth of tracking for each animal (for all animals surviving more than 60 days), to generate a composite wintering utilization distribution. Northern Gannets arrived in the wintering area between August 29 and December 17 (median date = November 3, *n* = 22), and departed for spring migration between February 23 and May 1 (median date = April 13, *n* = 33). Across all years, the extent of the winter composite (*n* = 17 pooled individuals) utilization distribution was 229,874 km$^2$, with the core use area totaling 42,360 km$^2$. Relative to the utilization distribution (i.e., the broader available area), Northern Gannet core-use areas were located in highly productive, shallower waters, with lower sea surface salinities, especially areas closer to shore, and over fine sandy substrate. In general, the core use areas included Delaware Bay, Chesapeake Bay, and Pamlico Sound, but Northern Gannets also used Outer Continental Shelf waters widely throughout the period in which they were present in the mid-Atlantic region.

Given the Northern Gannet's ubiquitous nature and its ability to range widely across the entire shelf region, the utilization distribution and core use areas observed in this study were large, increasing the likelihood of displacement due to offshore wind energy areas compared to other seabirds with smaller ranges.

## Introduction

The global demand for sustainable energy sources, driven largely by the impacts of climate change, has led to the rapid development of wind energy in many countries. The United States has invested heavily in terrestrial wind energy generation, having installed over 46,000 turbines, generating over 62 gigawatts (GW) of energy (AWEA 2014). There is also substantial interest in developing offshore wind energy generation, particularly along the eastern seaboard, close to high population densities. The region also supports substantial natural resources, however, with unique ecological communities, including some 600,000 breeding pairs of 31 marine bird species (Nisbet et al. 2013).

The Northern Gannet (*Morus bassanus*) is the largest seabird to breed in the North Atlantic Ocean. It exhibits a pelagic distribution and is emblematic of continental shelf waters on both sides of the North Atlantic. Northern Gannets are also opportunistic and effective predators, foraging on small to mid-sized surface-schooling fishes in dramatic plunging dives as well as diving directly from the surface (Garthe et al. 2000, Montevecchi 2007). Northern Gannets are highly gregarious breeders, nesting in dense colonies on remote rocky islands and sea stacks. In the northwest Atlantic, they breed at six well-known colonies in southeastern Canada: three in the Gulf of St. Lawrence, Québec, and three in the Atlantic, off the eastern and southern coasts of Newfoundland. The North American breeding population, which represents ~27% of the global population, has experienced a healthy rate of growth since 1984 (4.4% per year), although that appears to have slowed in recent years (Chardine et al. 2013). On migration, Northern Gannets move widely down the east coast of Canada and the U.S. to winter in the shelf waters of the mid-Atlantic region, the South Atlantic Bight, and the northern Gulf of Mexico (Nelson 1978, Fifield et al. 2014). A very small proportion of the Newfoundland breeders are known to cross the North Atlantic to winter off of West Africa (Fifield et al. 2014). Like many seabirds, Northern Gannets display a low rate of reproduction and delayed maturation, initiating breeding at around 5 years of age and laying only one egg per year (Nelson 1978, Mowbray 2002). They are long-lived, with some birds known to have survived at least 20 years (Nelson 1978), and exhibit high adult survival (~95%; Chardine et al. 2013). These particular characteristics make the population vulnerable to any increase in adult mortality.

Northern Gannets are considered to be vulnerable to fatal collisions with offshore wind developments in European waters, due to their relatively poor maneuverability, and their habit of flying and plunge-diving from heights within the rotor-swept zone of wind turbines (Langston 2010). The Northern Gannet was ranked 8th in terms of collision risk in a recent study that compared the flight heights of 26 marine bird species with the heights of offshore wind turbines around the U.K., the North Sea, and along the Danish and Dutch coasts (Johnston et al. 2014). In addition to direct collision, Northern Gannets are likely vulnerable to other recognized hazards of offshore wind development, such as avoidance/displacement, and physical habitat modification (Fox et al. 2006). A recent study has shown avoidance by several marine bird species, including Northern Gannets, from a wind farm in the Belgian North Sea (Vanermen et al. 2015). Localized changes in prey populations and distributions could also affect Northern Gannets, as seen in other species (Perrow et al. 2011).

In order to assess the effects of disturbance and the potential population impacts of offshore wind development on Northern Gannets, basic information must be collected on their distribution and behavior, including flight pathways and timing of habitat use, within wind energy areas (WEAs). The distribution and abundance of marine birds in federal waters slated for offshore wind energy development have been the focus of ongoing aerial- and boat-based survey efforts in recent years (Parts II-III of this report, Geo-Marine, Inc. 2010, NFSC & SFSC 2012). However, few studies in North America have tracked individual marine birds to determine migratory and local within-season movements to evaluate potential interactions with proposed offshore wind energy developments. Location data collected from satellite transmitters can be used to estimate home-range size and other features of the utilization distribution of the target species. Kernel density estimation is a standard technique for characterizing and visualizing animal home ranges, and the utilization distribution is a probability density function that can be extended to quantify the relative frequency distribution of an animal’s occurrence in space and time (Silverman 1986, Keating and Cherry 2004). Resource selection functions can be used to predict habitat use across landscapes or to understand the influence of certain habitat characteristics on a species distribution (Long et al. 2009). Used resources are a subset of available resources, and a key factor in resource selection studies is determining what and how much will be included as “available” (Buskirk and Millspaugh 2006). The probability that particular habitats will be used by a species can be examined with a logistic regression model; however, decisions regarding sampling design, the underlying probability model, and assumptions must be carefully considered to ensure correct interpretation of results (Keating and Cherry 2004).

The objectives of this study were to track fine-scale winter movements of Northern Gannets in the mid-Atlantic region, identify regions of consistent use, and describe the timing of their annual migrations in and around the study area. Specifically, we captured adult Northern Gannets at a single breeding colony in eastern Canada in 2012 and in the mid-Atlantic region of the U.S. in winter in 2012 and 2013, and outfitted them with satellite tags (platform terminal transmitters, known as PTTs). Tags were attached either externally by taping them to the tail feathers (a tried and tested method for this species) or internally via surgical implantation (a novel method for this species). We then tracked their daily movements within the study area in winter, and their broader seasonal movements throughout the year.

## Methods

### *Study area*

The priority study area included federal waters off the mid-Atlantic U.S. coast from northern Delaware to the southern border of Virginia. Three winter capture regions adjacent to the priority study area were selected: northern (New Jersey and Delaware), central (Maryland and Virginia), and southern (North Carolina). In 2012, capture efforts for Northern Gannets were attempted in Pamlico Sound, NC, Chesapeake Bay, MD and VA, Chincoteague Bay, MD and VA, Assawoman Bay and Isle of Wight Bay, MD, Indian River Bay, DE, and Delaware Bay, DE and NJ. Capture efforts in 2013 were focused in areas where high concentrations of target species were observed during the previous years’ field efforts, including: Pamlico Sound, offshore of Hatteras, NC; Chesapeake Bay, offshore of Cape Charles, VA; Delaware Bay, offshore of Lewes, DE; and the Atlantic Ocean, offshore of Cape Henlopen, DE and Chesapeake Bay Bridge area, VA.

### ***Satellite transmitters***

Two satellite tracking devices were used in this study: Telonics IMPTAV-2630 (29 g, <1% of adult body weight) were used as tail-mounted devices in 2012, and Telonics IMPTAV-2640 (42 g) were implanted in birds in 2012 and 2013. Pre-surgery preparation of the implanted tags increased their weight to approximately 49 g (<2 % of adult body weight).

#### *At-sea capture*

In February and March of 2012 and 2013, Northern Gannets were captured at three sites in their wintering area in the mid-Atlantic region: the Chesapeake Bay, MD and VA (*n*=15), Delaware Bay, DE and NJ (*n*=5), and Pamlico Sound, NC (*n*=6). These birds were captured using a night-lighting technique, where birds were approached at night in a small boat, briefly disoriented with a powerful spotlight, and then captured with a dip net (Figure 22-1). All captured birds were weighed and banded with a standard U.S. Fish and Wildlife Service (USFWS) metal band. A blood sample (2-4 ml) was taken from the metatarsal vein of each individual for genetic determination of sex; only a small amount was required for this analysis, and the remainder was archived.

In 2012, three Northern Gannets captured in the wintering area had satellite tags taped to their central tail feathers using a combination of self-amalgamating tape (Tesa Tape, Inc., Charlotte, NC) and cable ties, and three other individuals had satellite tags surgically implanted (Figure 22-2). Birds with tail-taped tags were released immediately at the site of capture. In 2013, all birds captured in the wintering area received implanted transmitters (*n*=20). They were administered the sedative Midazolam upon capture, then transported to shore in adapted pet-carriers (plastic dog crates or totes). Surgical implantations of intra-abdominal PTTs with external antenna were conducted by qualified veterinarians, following standard techniques (Korschgen et al. 1996, Maulcahy & Esler 1999). After surgery, when birds were cleared for release by the veterinarian, they were returned to the general area of capture and released onto the water during daylight hours (Figure 22-3).

#### *Colony capture*

In late September 2012, adult Northern Gannets (*n* = 9) were captured on the nest at Cape St. Mary's, Newfoundland, during the fledging period using a telescoping noose pole. Satellite tags were attached to tail feathers using a combination of tape and cable ties (Figure 22-2). All captured birds were banded with a standard Canadian Wildlife Service (CWS) metal band, and a small (0.5 ml) blood sample was taken from the brachial vein for genetic sexing. Due to the need to release them as quickly as possible, not all were weighed.

#### *Transmitter duty cycles*

Tail-mounted tags were programmed to transmit intensively because Northern Gannet tail feathers are molted within a few months of capture. The tags deployed at the colony were programmed with a single duty cycle: 9 hours on and 4 hours off. The tail-mounted tags deployed at sea in winter were programmed with two distinct duty cycles: 4 hours on and 9 hours off (Feb 1-May 20) and 2 hours on and 24 hours off (May 21-Sept 30), designed to provide the greatest resolution of movement while birds were expected to be in the mid-Atlantic region.

The batteries in the implanted satellite tags were expected to last up to a year or more, and these tags were programmed to transmit approximately daily throughout the winter months and migration periods, and approximately weekly during the breeding season. These tags were programmed with two distinct duty cycles: 4 hours on and 13 hours off (Sept 24-May 20) and 2 hours on and 120 hours off (May 21-Sept 23), designed to provide the greatest resolution of movement data during winter months, while prolonging the battery life for future transmissions.

### *Data processing and analysis*

Satellite data were compiled and filtered using the Douglas Argos Filter[2] (DAF). The DAF is a threshold filter that has several user-defined parameters to flag improbable locations in satellite tracking data (Douglas et al. 2012). The parameters are adjustable based on species' movement behaviors, time of year, and the scale of the area under observation. With the DAF, data are retained if they pass a spatial redundancy test and/or a movement rate and turning angle test. Since our bird data contain both short-distance, local movements and long-distance migratory events, we employed the hybrid filter of both the distance, angle, and rate (DAR) and minimum redundant distance (MRD) filters. Using DAF, we also identified the best representative point per duty cycle for each animal to reduce redundant daily positional information. All location data collected within 14 days of PTT deployment were excluded from analysis to reduce bias associated with surgery (Esler et al. 2000). The DAF also generates estimates of the distance between successive points (best representative point per duty cycle) and the amount of time between these points. Since the time between points varied between duty cycles and individuals, we also calculated the rate of distance moved by dividing distance moved by the number of hours between each duty cycle. These were calculated for the second winter period following deployment, which generally reflected movements from September to April.

A database of deployment and life-history data was built for each PTT tag deployed. Based on a close examination of the data, and the extent and direction of movement by each individual, we identified the following periods: breeding, molting, fall-staging, wintering, and spring-staging for every year that the animal was alive and/or tag transmitted.

Sensor data (e.g., temperature, voltage) could sometimes be used to determine the final fate of birds and tags. We assessed sensor data for every tag to identify confirmed mortality (by internal temperature sensor or mortality sensor) or battery/tag failure due to low voltage, and recorded the last known date alive for all birds confirmed dead during the period of tag activity. For all animals with tags that stopped transmitting, due to either low voltage or unknown reasons, we recorded the day after the last transmission as the last date of disposition. Tag duration dates were then calculated from the deployment start date to the final disposition date.

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[2] http://alaska.usgs.gov/science/biology/spatial/douglas.html

### *Kernel Density Estimation*

We chose to represent only one year's worth of data per life period per animal. Each bird then only contributed one sample to each map, preventing any bias towards animals with more years of data due to tag longevity or mortality. We chose to include Year 2 data preferentially over Year 3 data, as few individuals provided a full third year of winter location data, and Year 1 data if only one year of data existed. We included only birds that transmitted >60 days after release to reduce bias from birds that could have been negatively affected by transmitter implantation and handling.

Kernel density maps were created for the wintering period. Kernel density estimation involves the use of point data from telemetry to estimate relative spatial use during specified time intervals. For each location, the bird's habitat use was estimated to have been greatest directly on the point, and to decrease with distance from the point (reaching zero at a bandwidth specified by the user, after Worton 1989). Following Loring et al. (2014), we used the composite KDE method (with Gaussian kernel and Likelihood cross-validation bandwith estimator), where $S$ random points are selected for $n$ individuals and pooled to create a single composite KDE representing the utilization distribution of all animals. We defined $S$ by the lower quartile of the number of points per animal represented in the winter period ($S$ = 32, $n$= 17). The 0.95 utilization distribution and 0.5 core habitat use isopleths were generated from the composite KDEs. These isopleths were then used for further analyses.

### *Individual Home Range Estimation*

We calculated the home range sizes of individuals during winter by estimating the minimum convex polygon (mcp) areas ($km^2$) using package adehabitatHR version 0.4.13 (Calenge 2006) in R version 3.1.1 (R Core Team 2014) and removing 5% outliers. Only animals transmitting for at least 30 days were used.

## ***Environmental covariates and habitat selection***

We examined third-order resource selection of Northern Gannets to determine characteristics of preferred habitats that can be used to predict probability of use across the study area. Using the composite kernel density estimates, we defined "used" or core use areas as telemetry point locations within the 0.50 isopleth, and compared the habitat characteristics of those points to randomly-generated points throughout the 0.95 utilization distribution, defined as "available" locations (sampling protocol-A; Manly 2002). Habitat characteristics were chosen based on *a priori* knowledge of marine habitat and availability of spatial data within the study area, and included water depth (m), distance to shore (km), long-term (10-yr winter mean) and seasonal mean sea surface temperature (degrees Celsius), long-term (10-yr winter mean) and seasonal chlorophyll *a* ($mg/m^3$), long-term (6-yr winter mean) and seasonal sea surface salinity (practical salinity units; psu), seafloor slope (degrees), and sediment grain type. We measured water depth (m) and slope (degrees) using the NOAA National Geophysical Data Center 3 arc-second Coastal Relief Model for the United States (NOAA 2014a). To estimate distance from shore, we calculated the Euclidean distance (m) between Northern Gannet locations and the nearest segment of the NOAA Medium Resolution Digital Vector Shoreline (1:70,000) using the Near Tool in Arcmap 10.2.2 (ESRI, Redlands, CA; NOAA 2014b). Estimates of seafloor slope were obtained from the Nature Conservancy's Northwest Atlantic Marine Ecoregional Assessment data portal. The benthic habitat layer contains an estimate of slope, which is calculated as the difference in elevation between two neighboring raster cells, expressed in degrees, and were grouped accordingly: (1)

0-0.015° = level flat; (2) 0.015-0.05° = flat; (3) 0.05-0.80° = gentle slope; 0.80-8.0° = slope; and >8.0° = steep slope (Greene et al. 2010). Sediment grain size categories were also obtained from the Nature Conservancy's Northwest Atlantic Marine Ecoregional Assessment data portal. Size categories were grouped by the Nature Conservancy according to correlations with benthic habitat communities, and are not necessarily related to Northern Gannet habitat requirements. For the dynamic variables with a temporal component, we used Marine Geospatial Ecology Tools to create seasonal and long-term winter mean climatology rasters for sea surface temperature, chlorophyll *a*, and sea surface salinity (Roberts et al. 2010). We smoothed daily surface temperature estimates based on optimal interpolation of data derived from high resolution satellite imagery and floating buoys (Stark et al. 2007). These data were produced by the UK Met Office on a global scale at a spatial resolution of 0.054 degrees latitude and longitude. Sea surface salinity was estimated using the HYbrid Coordinate Ocean Model (HYCOM) produced by the National Ocean Partnership Program at a spatial scale of 1/12 degrees latitude and longitude (Chassignet et al. 2009). To measure ocean productivity, we obtained monthly estimates of chlorophyll *a* concentration (mg/m$^3$) produced by the NASA Goddard Space Flight Center's Ocean Data Processing System. These data had a spatial resolution of 4 km and were derived from radiometric measurements of chlorophyll fluorescence made by the Aqua sensor aboard the MODIS satellite system (Mueller et al. 2003).

We used logistic regression to model habitat covariate effects on used versus available locations over three winter periods (2011-2012, 2012-2013, and 2013-2014) within the study area. Candidate models were developed for each winter period and for all years combined. Development of models was exploratory, but, based on a priori knowledge of Northern Gannet habitat, we predicted that used locations would be related to nine habitat covariates: depth, distance to shore (dist), long-term (chlor10) and seasonal (chlorw) chlorophyll *a*, long-term (sst10) and seasonal (sstw) sea surface temperature, long-term (sss6) and seasonal (sssw) sea surface salinity, and seafloor slope. Correlations between pairs of habitat variables were quantified using a Pearson product-moment correlation matrix. Multicollinearity among covariates was assessed by calculating variance inflation factors (VIF). Highly correlated variables included: (1) long-term to seasonal sea surface temperature, (2) long-term to seasonal salinity, and (3) long-term to season chlorophyll *a* concentration. Therefore, models included the long-term or seasonal term for each of those variables, but not both. All other pairwise comparisons had correlations of < 0.50 and VIF values of < 2.0 and were retained as variables in the modeling process, resulting in six variables considered for inclusion in each model. The square of both water depth and distance to shore were also included in order to examine the possibility that their relationship to the log odds of habitat use was curvilinear rather than linear. Additionally, the product of each pair of continuous variables was included to assess possible interactions among the predictors. Candidate models were ranked with Akaike Information Criterion adjusted for small sample size ($AIC_c$). The model with the lowest $AIC_c$, and those having $\Delta AIC_c \leq 2$ had the most statistical support, values between 4 and 7 had considerably less support, and those > 10 had virtually no support (Burnham and Anderson 2002). The Akaike weight was also considered when determining the relative amount of statistical support for each model.

## Results

### *Capture and PTT deployment*

A total of 35 Northern Gannets were satellite tagged in 2012 (*n*=15) and 2013 (*n*=20), of which 15 were female, 19 were male, and 1 was undetermined. Body mass of females ranged from 3,050 to 3,700 g ($\bar{x}$ = 3,329 ± 240 g, *n* = 14), and for males ranged from 2,675 to 4,050 ($\bar{x}$ = 3,258 ± 358 g, *n* = 14).

In all, 12 of these Northern Gannets were equipped with tail-taped tags, and 23 had tags surgically implanted. Nine were captured at the breeding colony at Cape St. Mary's, Newfoundland, in 2012, and 26 were captured while wintering at sea in the mid-Atlantic study area (6 in 2012 and 20 in 2013).

### *General migratory movements*

In the latter half of March, satellite tagged birds were generally located inshore in the mid-Atlantic region, or in the broad shelf waters of the Gulf of Mexico (Figure 22-4). April marked the beginning of movements north. Migrating Northern Gannets generally departed the Mid-Atlantic area in early to mid-April and arrived in the region of their breeding colony in late April/early-May. Migration was highly asynchronous, however, as some individuals reached their breeding colony before others had even left the wintering area.

All of the Northern Gannets tagged in the winters of 2011-12 and 2012-13 that reached the breeding grounds appeared to breed at colonies in Québec, particularly Bonaventure Island, off the eastern tip of the Gaspé Peninsula, and Bird Rocks in the Magdalen Islands archipelago. Some individuals made extensive foraging trips from these colonies, ranging from the upper Gulf of St. Lawrence to the Labrador Sea. Owing to a pervasive broad scale warm-water perturbation in the western Atlantic during the summer of 2012, many Northern Gannets at Bonaventure Island and at Cape St. Mary's made inordinately long foraging trips from their breeding sites and experienced very low reproductive success (Montevecchi et al. 2013).

PTT-equipped Northern Gannets began their southerly migration in September (Figure 22-5). One adult from Cape St. Mary's, and another captured in the study area in the winter of 2012, passed through the mid-Atlantic study area to winter in the Gulf of Mexico. The birds that reached the Gulf of Mexico remained there through the early winter. The birds that departed the breeding area later in the fall largely spent their early winter offshore in the mid-Atlantic region. By mid- to late-winter, activity was once again concentrated in the mid-Atlantic region.

In general, Northern Gannets showed a high use of relatively shallow, coastal waters (<30 m) during their migrations. There is some indication that males tended to exhibit a more northerly distribution in their wintering range, and, as a result, may spend more time in the mid-Atlantic area than females.

### *Winter area use*

Northern Gannets arrived in the wintering area between August 29 and January 1 (median date = November 3, *n* = 22), and departed for spring migration between February 23 and May 1 (median date = April 13, *n* = 33). Data from the second winter after tagging provides some indication that males may arrive in the study area later than females (male median date = November 22, range = Nov 20-Jan 3, *n* =

4; female median date = November 13, range = Oct 25-Dec 17, $n$ = 6) and depart earlier (male median date = February 13, range = Jan 3-Mar 18, $n$ = 4; female median date = March 11, range = Dec 27-Mar 26, $n$ = 6), although sample sizes are admittedly small at this point. The average length of stay in the wintering area for both sexes combined was 85 days ($n$ = 12; range = 37-152 days). This is likely an underestimate, however, since Northern Gannets were caught mid-winter, and we did not get data for a full winter until one year later, at which point the satellite tags were reaching their expected lifespan.

Wintering movements of individual birds in the study area varied considerably. The minimum distance moved by a bird between the best points in consecutive 17-hour duty cycles was 0.5 km, while the maximum distance moved was 1,059 km. On average, birds moved 22-370 km ($\bar{x}$ = 87.5 ± 118.9 km, $n$=11) between duty cycles. These estimates were based on the best quality location per duty cycle derived from the DAF filtered data for each animal, and the amount of time between these points varied between duty cycles and individuals; therefore, we also calculated the rate of distance moved by dividing distance moved by the number of hours between each point. The minimum distance traveled per hour during the second winter season was 0.01 km/hr, the maximum was 10.9 km/hr. On average birds traveled 0.4-2.1 km/hr ($\bar{x}$ = 1.4 ± 0.5 km/hr, $n$ = 11) between duty cycles.

Northern Gannet wintering home ranges also varied widely. During the second winter period following deployment, individual home ranges varied from 3,961-128,590 $km^2$ ($\bar{x}$ = 28,734 ± 38,630 $km^2$; $n$ = 9). There is some suggestion that females may have roamed more widely than males (females $\bar{x}$ = 17,353 ± 10,386 $km^2$, range = 7,091-31,756 $km^2$, $n$ = 9; males $\bar{x}$ = 15,498 ± 16,315 $km^2$, range = 3,961-27,034 $km^2$, $n$ = 9), although sample sizes are too small to examine this quantitatively. This may be a result of males potentially being present in the study area for shorter periods in winter than females.

### *Winter habitat selection*

The core use areas of wintering male and female Northern Gannets included Delaware Bay, Chesapeake Bay, and Pamlico Sound, but individual Northern Gannets roamed widely across the area in each wintering period, showing low site fidelity within seasons. Although individual Northern Gannets ranged widely, and often visited several core use areas (even on a daily basis), the general locations used by wintering Northern Gannets were relatively consistent. Across all years, the extent of the winter composite ($n$ = 17 pooled individuals) utilization distribution was 229,874 $km^2$, with the core use area totaling 42,360 $km^2$ (Figure 22-6).

Relative to the utilization distribution (i.e., the broader available area), Northern Gannet core-use areas were located in highly productive, shallower waters, with lower sea surface salinities, especially areas closer to shore (Table 22-1), and over fine sandy substrate (Table 22-2). The highest-ranked habitat selection model differed slightly between years (Table 22-3). The top model for all years of data combined accounted for 0.67 of Akaike weight, and indicated a positive effect of long-term chlorophyll *a* on habitat use, and a negative effect of depth, distance to shore, and long-term salinity (Table 22-3). Other coefficients included in the top model had confidence intervals that included zero and were not considered reliable (Table 22-3 to Table 22-4). Among sediment types, the greatest frequency of use associated with core use areas consisted of very fine to fine sandy bottoms with grain sizes that ranged from 0.03-0.35 mm (Table 22-2).

## Discussion

The results from this study confirm the importance of the mid-Atlantic region for wintering and migrating Northern Gannets. Birds first arrived in the mid-Atlantic in August, with peak arrival time around November, departing again mostly in April. While in the study area, Northern Gannets used the Outer Continental Shelf waters broadly in winter, with core habitat including the protected inshore waters of the major bays and bay mouths, including the Delaware Bay and Chesapeake Bay areas. Satellite tracking to date also suggests that Northern Gannets show relatively weak winter site fidelity from year to year. Individual Northern Gannets also displayed rapid and extensive movements up and down the eastern seaboard over the course of the winter season, between the core use areas, which increases the likelihood that they would encounter offshore wind developments repeatedly throughout the winter.

This study has also confirmed that Bonaventure Island, Québec (the Northern Gannets' largest North American colony; Chardine et al. 2013), and Cape St. Mary's, Newfoundland (the species' southernmost colony in North America; Montevecchi et al. 2013) are colonies of origin for Northern Gannets using mid-Atlantic waters, although birds from all six North American colonies most likely use the region in winter. As they migrate south from breeding colonies during autumn, Northern Gannets generally move into coastal waters (Fifeld et al. 2014). There is some suggestion in the tracking data that this inshore movement is more pronounced among females than males, and that females may move further south than males, but this will be examined further in future analyses. The greater movement south could explain why females departed the study area slightly later, or this difference may be driven by the males need to return to the breeding colony early in order to reclaim ownership of their nest site (nest sites are known to be established and held by males; Mowbray 2002).

Important foraging and habitat use areas for Northern Gannets appear to be defined by a wide variety of habitat characteristics. Thus, development activities anywhere within the study area could potentially affect gannets, but this is most likely in the western half of the study area, where gannets were more abundant. Overall, Northern Gannets used shallow (<20 m) areas around 10 km from shore, although they also ranged well out onto the Outer Continental Shelf, up to 50 km. Northern Gannets are capable of frequent and long-distance movements in search of ephemeral epipelagic prey (Hamer et al. 2007, Wakefield et al. 2013), and can easily move between the core wintering areas in the mid-Atlantic region. Not surprisingly, the utilization distribution and core use areas observed in this study were large compared to other marine birds wintering in the area, such as Surf Scoters (Chapter 20) and Red-throated Loons (Chapter 21). Placement of offshore wind energy facilities in shallow (<30 m) inshore areas near the mouths of large bays, in the designated WEAs, or elsewhere within the study area, could potentially affect Northern Gannets in several ways: by increasing mortality due to direct collision (Furness et al. 2013), displacing them from high-quality foraging areas (Vanermen et al. 2015), or increasing energy expenditures through added flight costs to avoid turbines (Masden et al. 2010).

There was wide variation in estimates of individual home range size, distances moved, and the rate of distance moved by individuals between duty cycles. This variability was potentially driven by multiple factors, such as the specific locations and dates of capture, as well as true variations in movement patterns between individuals. The significance of these effects will be explored more thoroughly in future modeling efforts that incorporate additional years of data and can potentially provide a more accurate description of Northern Gannet home range size on its wintering grounds in the mid-Atlantic region.

The large winter ranges of Northern Gannets documented in this study, however, probably increase the likelihood of displacement due to development within offshore WEAs compared to other seabirds with smaller ranges. Significant stressors already exist for this species on the wintering grounds, including exposure to pesticides and other contaminants, such as mercury and PCBs (Champoux et al. 2015), oil spills (Montevecchi et al. 2012), mortality from bycatch in fishing nets (Lanza 1998, Forsell 1999, Žydelis et al. 2013), and changes to fish distributions due to climate change (Nye et al. 2009). Additional mortalities in the wintering area, or indirect effects such as displacement, could have detrimental effects on the North American breeding population of Northern Gannets, and it is critical that important foraging areas and flight pathways of this and other sensitive bird species are identified and considered during the planning and development phases of offshore wind energy development. We will continue to pursue this analysis to achieve these objectives, and our results will be forthcoming in an anticipated manuscript publication in 2016.

### *Next steps*

This preliminary analysis includes data from two years of a broader four-year study. Future analyses will include additional data gathered from satellite tagged birds during the 2013-14 and 2014-15 winter periods. The inclusion of these additional data will also allow for more thorough and comprehensive analysis of wintering movements, home range sizes, and habitat selection and use. Habitat covariate data used in modeling will also be updated to include long-term mean values that match the duration of the tracking data. Ultimately, this is hoped to improve model fit and/or uncover new interactions between covariates not highlighted in the existing analysis.

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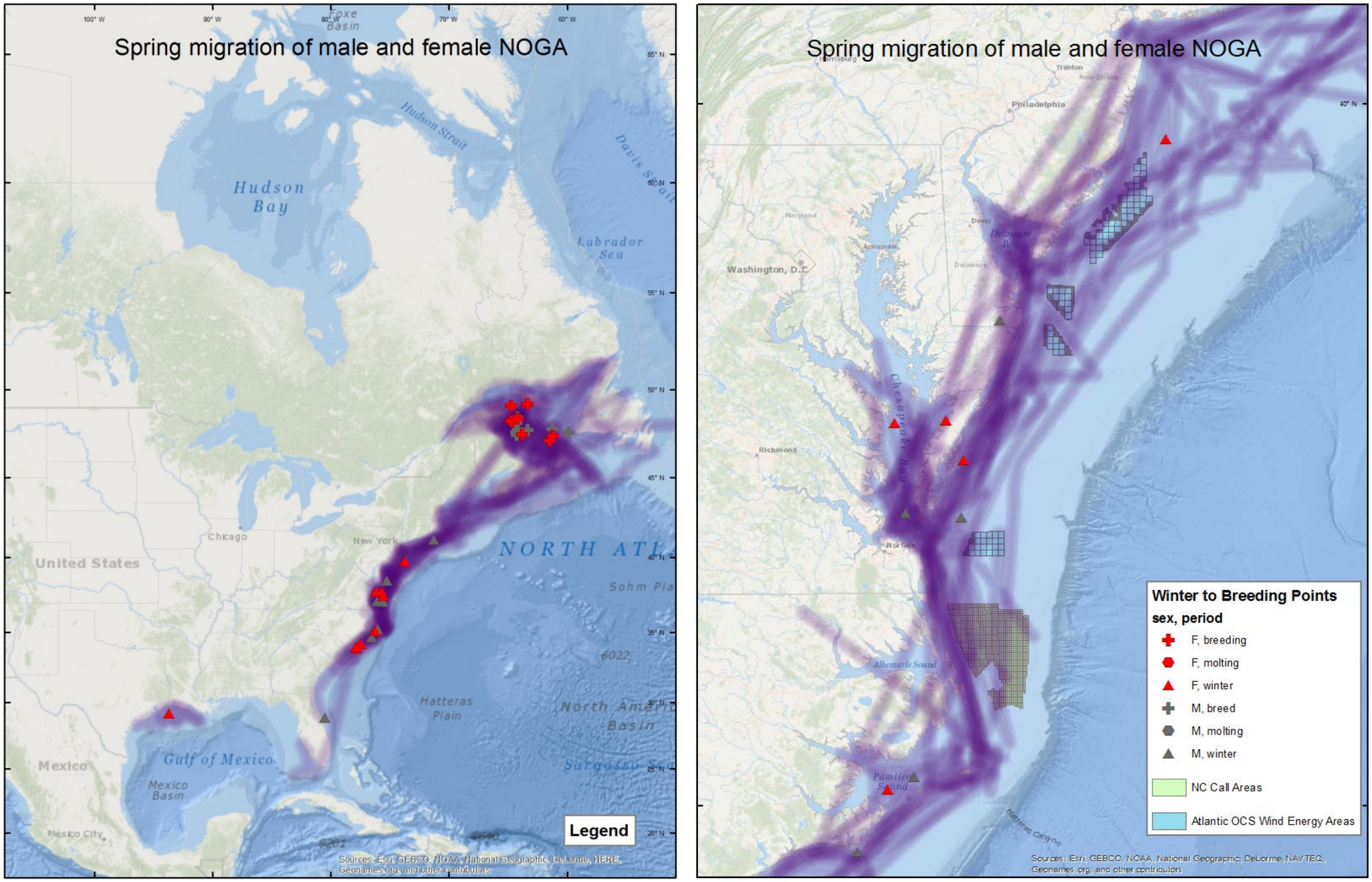
## Figures and tables

**Figure 22-1. Basic night-lighting technique used to capture Northern Gannets at sea in their wintering area.** (Photograph: J. Fiely/BRI)

**Figure 22-2. Satellite transmitter attachment in Northern Gannets – externally taped to the underside of the central tail feathers (*left*) and surgical implantation (*right*).** (Photographs: J. Fiely/BRI)

**Figure 22-3. A Northern Gannet with an implanted satellite transmitter (the protruding antenna clearly visible on the lower back) being released in Delaware Bay, MD, in 2013.** (Photograph: J. Fiely/BRI).



Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, NAVTEQ, Geonames.org, and other contributors

**Figure 22-4. The spring migration of Northern Gannets across the eastern seaboard (*left*) and within the study area (*right*).** Lines do not necessarily represent direct flight paths. Symbols indicate the endpoints for breeding in the Gulf of St. Lawrence (+), and the general wintering areas in the mid-Atlantic and the Gulf of Mexico (Δ) for males (grey) and females (red).

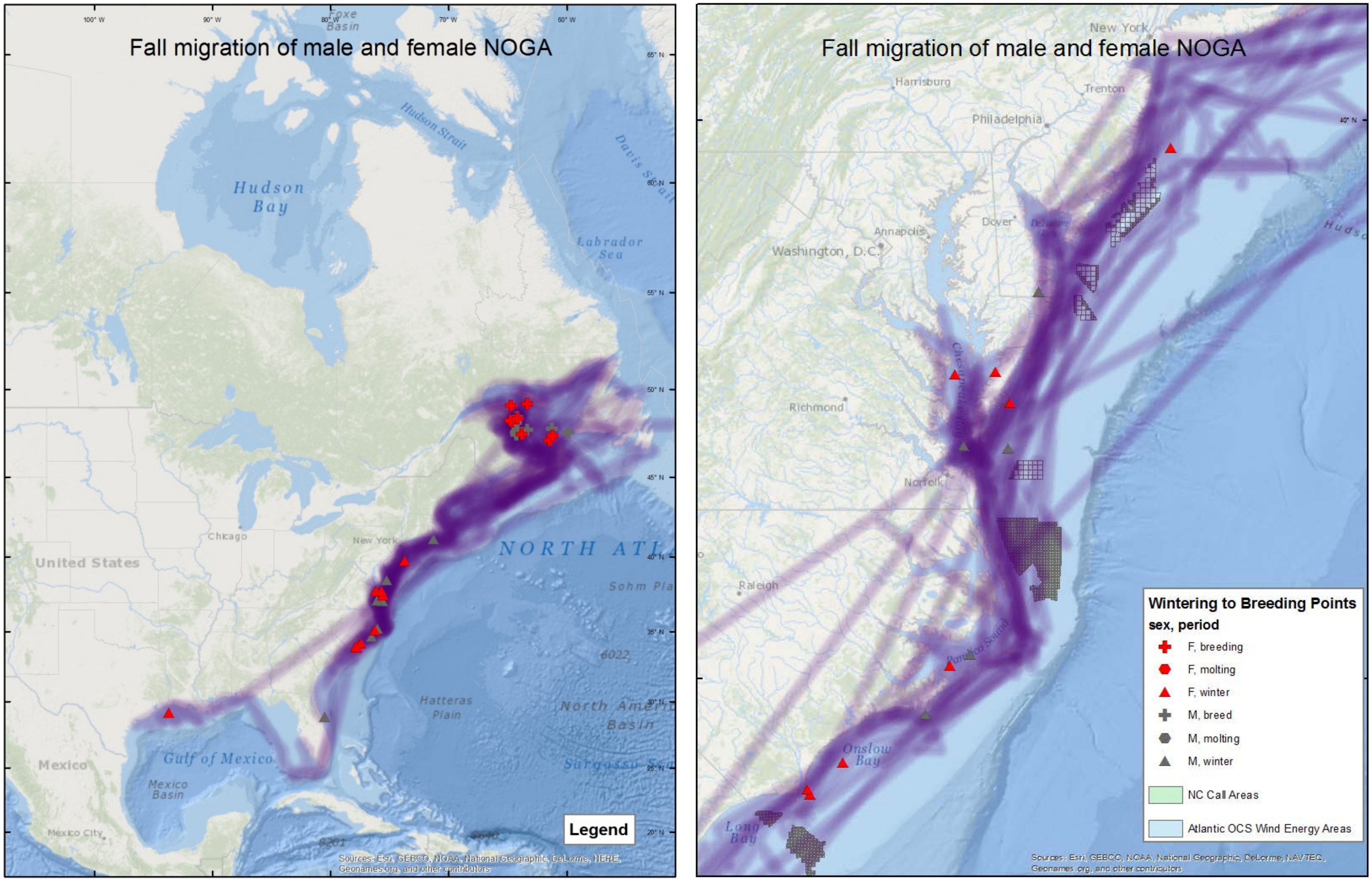
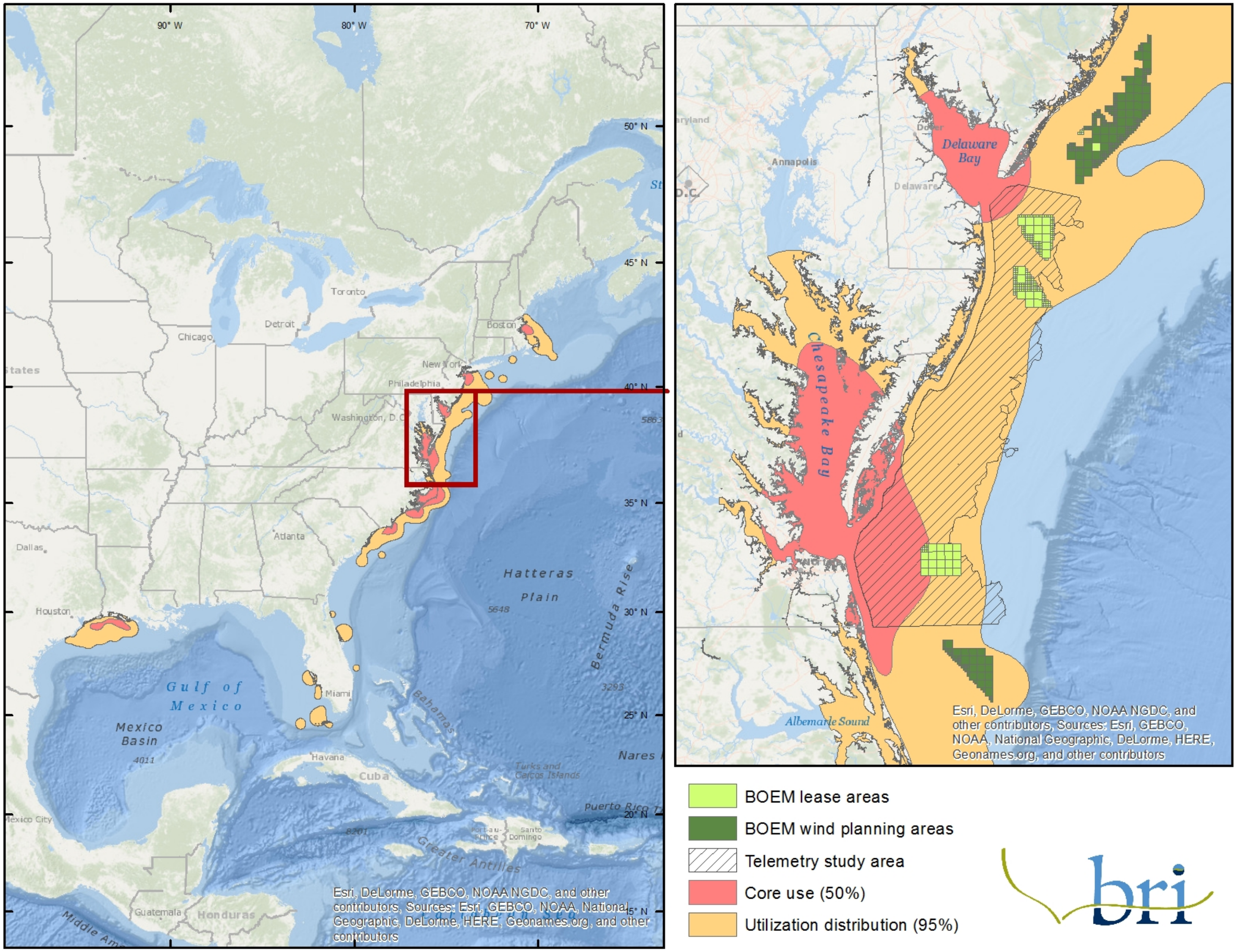


**Figure 22-5. The autumn migration of Northern Gannets across the eastern seaboard (*left*) and within the study area (*right*).** Lines do not necessarily represent direct flight paths. Symbols indicate the endpoints for breeding in the Gulf of St. Lawrence (₊), and the general wintering areas in the mid-Atlantic and Gulf of Mexico (Δ) for males (grey) and females (red).



Esri, DeLorme, GEBCO, NOAA NGDC, and other contributors, Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

Esri, DeLorme, GEBCO, NOAA NGDC, and other contributors, Sources: Esri, GEBCO, NOAA, National Geographic, DeLorme, HERE, Geonames.org, and other contributors

**Figure 22-6. The overall utilization distribution (orange) and core use (red) areas based on satellite-derived locations for Northern Gannets, and in relation to the BOEM lease areas and wind planning areas (inset).** Within the broader mid-Atlantic study area, core use includes Delaware Bay and Chesapeake Bay.

**Table 22-1. Range, mean ($\overline{x}$) and standard error (SE) of habitat variables at satellite-derived locations for Northern Gannets in the core use area (0.5 isopleth) and for random points across the utilization distribution (available; 0.95 isopleth).**

| Habitat Variable | Abbr. | Units | Available | | Core Use | |
|---|---|---|---|---|---|---|
| | | | range | $\overline{x}$ ± SE | range | $\overline{x}$ ± SE |
| ***Long-term*** | | | | | | |
| Chlorophyll *a* - 10 yr Mean | chlor10 | mg/m$^3$ | 0 - 33.6 | 2.9 ± 0.07 | 0 - 32.9 | 6.4 ± 0.16 |
| Sea Surface Temperature – 10 yr Mean | sst10 | °C | 0 - 24.9 | 13.1 ± 0.11 | 0 - 18.7 | 10.0 ± 0.17 |
| Mean Sea Surface Salinity | sss6 | psu[1] | 18.8 - 36.3 | 34.3 ± 0.03 | 16.8 - 35.8 | 32.9 ± 0.12 |
| ***Physical*** | | | | | | |
| Depth | depth | m | 0.1 – 2,491 | 37.3 ± 2.22 | 0.1 - 98.1 | 12.4 ± 0.38 |
| Slope | slope | degrees | <0.001 - 12.8 | 0.19 ± 0.011 | <0.001 - 3.1 | 0.17 ± 0.009 |
| Distance to Shore | dist | km | 0.1 – 133.2 | 32.8 ± 0.53 | 0.1 – 50.5 | 10.1 ± 0.31 |
| ***Short-term*** | | | | | | |
| Chlorophyll *a* - seasonal | chlorw | mg/m$^3$ | 0.12 - 56.2 | 2.93 ± 0.07 | 0.95 - 38.9 | 6.2 ± 0.16 |
| Sea Surface Temperature - seasonal | sstw | °C | 2.8 - 25.6 | 13.9 ± 0.11 | 3.1 - 20.8 | 10.3 ± 0.17 |
| Sea Surface Salinity - seasonal | sssw | psu[1] | 19.0 - 36.5 | 34.4 ± 0.03 | 16.5 - 36.0 | 33.0 ± 0.12 |

[1] psu = practical salinity units

**Table 22-2. Proportion of use of different sediment types at satellite-derived locations for Northern Gannets in the core use area (0.5 isopleth) and for random points across the utilization distribution (available; 0.95 isopleth).**

| Sediment Grain | | Available | | Core Use | |
|---|---|---|---|---|---|
| **Size (mm)** | **Type** | **n** | **% Use** | **n** | **% Use** |
| 0.00 - 0.03 | Silt/Mud | 81 | 7 | 32 | 5 |
| 0.03 - 0.17 | Very Fine Sand | 209 | 17 | 154 | 26 |
| 0.17 - 0.35 | Fine Sand | 282 | 23 | 209 | 36 |
| 0.35 - 0.36 | Sand | 154 | 13 | 38 | 6 |
| 0.36 - 0.48 | Medium Sand | 152 | 13 | 54 | 9 |
| 0.48 + | Coarse Sand - Gravel | 329 | 27 | 98 | 17 |

**Table 22-3. Logistic model selection results examining the annual effects of water depth (dep), distance from shore (dist), sediment type (sed), slope (slope), long-term mean chlorophyll *a* concentration (2004-2014; chlor10), long-term mean winter sea surface temperature (2004-2014; sst10), long-term mean sea surface salinity (2008-2014; sss6), seasonal mean winter chlorophyll *a* concentration (chlorw), seasonal mean winter sea surface temperature (sstw), and seasonal mean winter sea surface salinity (sssw) on the resource selection of Northern Gannets wintering in the mid-Atlantic U.S.** Models are ranked according to Akaike Information Criterion adjusted for small sample size (AICc). The table shows the variables included in each model, differences between model AICc values (ΔAICc), and AICc weights ($w_i$). Only models with ΔAICc values < 7 were included in the table.

| Winter Period and Model | $AIC_c$ | $\Delta AIC_c$ | $w_i$ |
|---|---|---|---|
| *2011 - 2012* | | | |
| sed+chlor10+sstw+slope | 34.587 | 0.000 | 0.374 |
| sed+chlor10+sss6+slope | 35.161 | 0.574 | 0.280 |
| sed+chlor10+sstw+slope+(chlor10*sstw) | 37.034 | 2.447 | 0.110 |
| sed+chlor10+sssw+slope | 37.037 | 2.450 | 0.110 |
| sed+dist$^2$+chlor10+sss6+slope | 37.717 | 3.130 | 0.078 |
| sed+chlorw+sstw+slope | 40.407 | 5.820 | 0.020 |
| sed+dist$^2$+chlor10+sss6+slope+(chlor10*sss6) | 40.411 | 5.824 | 0.020 |
| *2012 - 2013* | | | |
| dep+sed+dist$^2$+chlor10+sss6 | 462.933 | 0.000 | 0.496 |
| dep+sed+dist$^2$+chlor10+sss6+(chlor10*sss6) | 464.306 | 1.373 | 0.250 |
| dep+sed+dist$^2$+chlorw+sssw | 464.816 | 1.883 | 0.194 |
| dep+sed+dist$^2$+chlorw+sssw+(chlorw*sssw) | 467.305 | 4.372 | 0.056 |
| *2013 - 2014* | | | |
| sed+dist$^2$+chlor10+sss6+slope | 420.539 | 0.000 | 0.591 |
| sed+dist$^2$+chlor10+sss6 | 423.638 | 3.107 | 0.125 |
| dep+sed+dist$^2$+chlor10+sss6 | 423.280 | 2.749 | 0.150 |
| sed+dist$^2$+chlor10+sss6 | 423.638 | 3.107 | 0.125 |
| *All Years* | | | |
| dep+sed+dist$^2$+chlor10+sss6 | 1017.190 | 0.000 | 0.671 |
| dep$^2$+sed+dist$^2$+chlor10+sss6 | 1020.200 | 3.010 | 0.149 |
| dep+sed+dist$^2$+chlor10+sssw | 1021.170 | 3.980 | 0.092 |
| dep+sed+dist$^2$+chlorw+sss6 | 1022.830 | 5.640 | 0.040 |

**Table 22-4. Coefficients (β) and 95% confidence intervals (lower and upper) of best-fit resource selection model for Northern Gannets wintering in the mid-Atlantic region in 2012-2013.**

| Model Years and Variable | β | Lower | Upper |
|---|---|---|---|
| ***2011 - 2012*** | | | |
| Long-term chlorophyll *a* (mg/m$^3$) | 1.576 | 0.313 | 3.63 |
| Seasonal sea surface temperature (°C) | -5.733 | -13.809 | -2.613 |
| Slope (°) | -19.135 | -43.389 | -6.24 |
| Sediment grain size (mm) and type | | | |
| 0 - 0.03 silt/mud | -28.332 | -65.765 | -11.300 |
| 0.03 - 0.17 very fine sand | 5.468 | 1.123 | 14.926 |
| 0.17 - 0.35 fine sand | 5.08 | 1.798 | 12.020 |
| 0.35 - 0.36 sand | 3.157 | -2.494 | 11.271 |
| 0.36 - 0.48 medium sand | 8.217 | 2.302 | 21.966 |
| ***2012 – 2013*** | | | |
| Long-term chlorophyll *a* (mg/m$^3$) | 0.051 | -0.041 | 0.148 |
| Long-term sea surface salinity (psu) | -0.148 | -0.335 | -0.011 |
| Distance to shore$^2$ (km) | $-1.449*10^{-9}$ | $-2.584*10^{-9}$ | $-5.529*10^{-10}$ |
| Depth (m) | -0.118 | -0.164 | -0.077 |
| Sediment grain size (mm) and type | | | |
| 0 - 0.03 silt/mud | -0.819 | -2.41 | 0.587 |
| 0.03 - 0.17 very fine sand | 0.481 | -0.047 | 1.018 |
| 0.17 - 0.35 fine sand | 0.14 | -0.338 | 0.629 |
| 0.35 - 0.36 sand | 1.131 | 0.407 | 1.868 |
| 0.36 - 0.48 medium sand | 0.133 | -0.517 | 0.779 |
| ***2013 – 2014*** | | | |
| Long-term chlorophyll *a* (mg/m$^3$) | -0.003 | -0.118 | 0.117 |
| Seasonal sea surface temperature (°C) | -0.373 | -0.672 | -0.079 |
| Seasonal sea surface salinity (psu) | 0.691 | 0.167 | 1.218 |
| Seasonal sea surface temperature* seasonal sea surface salinity | 0.172 | 0.044 | 0.302 |
| Distance to shore$^2$ (km) | $-1.662*10^{-9}$ | $-2.387*10^{-9}$ | $-1.045*10^{-9}$ |
| Depth (m) | -0.0113 | 0.027 | -0.0001 |
| Sediment grain size (mm) and type | | | |
| 0 - 0.03 silt/mud | 0.547 | -0.585 | 1.742 |
| 0.03 - 0.17 very fine sand | -0.025 | -0.625 | 0.565 |
| 0.17 - 0.35 fine sand | 0.457 | -0.043 | 0.961 |
| 0.35 - 0.36 sand | -0.539 | -1.487 | 0.327 |
| 0.36 - 0.48 medium sand | -0.332 | -1.005 | 0.316 |
| ***All Years*** | | | |
| Long-term chlorophyll *a* (mg/m$^3$) | 0.072 | 0.011 | 0.137 |
| Long-term sea surface salinity (psu) | -0.106 | -0.219 | -0.013 |
| Distance to shore$^2$ (km) | $-1.899*10^{-9}$ | $-2.449*10^{-9}$ | $-1.417*10^{-9}$ |
| Depth (m) | -0.022 | -0.035 | -0.011 |
| Sediment grain size (mm) and type (mm) | | | |
| 0 - 0.03 silt/mud | -0.091 | -0.963 | 0.744 |
| 0.03 - 0.17 very fine sand | 0.313 | -0.043 | 0.669 |
| 0.17 - 0.35 fine sand | 0.205 | -0.099 | 0.512 |
| 0.35 - 0.36 sand | 0.220 | -0.264 | 0.695 |
| 0.36 - 0.48 medium sand | -0.219 | -0.645 | 0.197 |

# Chapter 23: Incorporating temporal variation in seabird telemetry data: time variant kernel density models

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** www.briloon.org/mabs

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**Acknowledgments:** This chapter includes data from several collaborative field studies focused on the movement patterns of Red-throated Loons, Northern Gannets, and Surf Scoters in the Atlantic flyway, carried out by numerous principal investigators and agencies.

The work reported in this chapter constitutes part of two multi-year collaborative projects: 1) determining Offshore Use of Diving Bird Species in Federal Waters of the Mid-Atlantic United States Using Satellite Tracking, developed by the Bureau of Ocean Energy Management (BOEM) and the U.S. Fish & Wildlife Service (USFWS) and 2) the Atlantic and Great Lakes Sea Duck Migration Study, developed by the USFWS, Canadian Wildlife Service (CWS), and other state and federal partners. The first study was designed in partnership with the U.S. Geological Survey (USGS)-Patuxent Wildlife Research Center, Biodiversity Research Institute (BRI), and Memorial University of Newfoundland (MUN), and funded by BOEM, the Department of Energy (DOE; Award Number DE-EE0005362), BRI, MUN, and The Bailey Wildlife Foundation. The second project was primarily funded by the USFWS through the Sea Duck Joint Venture, DOE, and BOEM. In addition to these funding entities and the authors of this chapter, other partners that provided logistical or financial support include the Maryland Department of Natural Resources, Virginia Department of Game and Fisheries, University of Rhode Island, Delaware Division of Fish and Wildlife, and North Carolina Wildlife Resource Commission. The authors would also like to thank Dr. Scott Ford (Avian Specialty Veterinary Services), Dr. Glenn Olsen (USGS), Dr. Darryl Heard (University of Florida) and Stephane Lair and Ariane Santamaria-Bouvier (University of Montreal) for performing countless implant surgeries. We also thank Doug Howell (North Carolina Wildlife Resources Commission) for logistical support and area knowledge, Rob Hossler and Brian Rogers (Delaware Division of Fish and Wildlife) for logistical support and Stephen Earsom (USFWS) for aerial survey reconnaissance. We are indebted to Dr. Jim Woehr (BOEM), Jocelyn Brown-Saracino (DOE), and Scott Johnston, Caleb Spiegel, and Kirsten Luke (USFWS) for their assistance in project management. This report covers all data through the end of 2013, including the first two years of the BOEM/USFWS study and all prior years of the SDJV study.

# Chapter 23 Highlights

*Incorporating temporal variation into utilization distributions using seabird telemetry data: time variant kernel density models*

### *Context*[1]

Satellite telemetry is an effective tool for understanding individual animal movement patterns that can yield insight into habitat use over a species' spatio-temporal range. Chapters 20-22 explored spatial patterns and movement of three target species using satellite telemetry data: Red-throated Loon, Northern Gannet, and Surf Scoter. The utilization distribution approach applied in these chapters does not use the temporal component of the movement data in modeling animal habitat use, but rather collapses the data into a single period (defined by collaborators' identification of each individual's transition between life history stages). In this chapter, we explore the use of time variant kernel density models as a way to improve understanding of species' use of the landscape through time. A more data-intensive approach, focused on individual rather than group utilization distributions, is explored for Peregrine Falcons in Chapter 25.

### *Study goal/objectives*

Incorporate temporal variation into kernel density models to 1) better understand space use through time, and 2) create utilization distribution maps at fine temporal scales, and generate movement videos.

### *Highlights*

- Time variant kernel density maps proved to be an effective tool for showing fine-scale temporal variation in use of the study area.
- Analysis showed limited overlap of seabird habitat use with wind energy areas for the three species examined. The most overlap occurred for Northern Gannets.

### *Implications*

Time variant kernel density models are an effective tool for generating dynamic maps (as video), demonstrating temporal variation in utilization distributions in the environment.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

A key component of the Mid-Atlantic Baseline Studies project was tracking the individual movements of focal marine bird species (Red-throated Loon [*Gavia stellata*], Northern Gannet [*Morus bassanus*], and Surf Scoter [*Melanitta perspicillata*]) through the use of satellite telemetry. This element of the project was a collaborative effort with the Department of Energy (DOE), Bureau of Ocean Energy Management (BOEM), the U.S. Fish and Wildlife Service (USFWS), and Sea Duck Joint Venture (SDJV), among other organizations. Satellite telemetry is an effective and informative tool for understanding individual animal movement patterns, allowing researchers to mark an individual once, and thereafter follow the movements of the animal in space and time. Aggregating telemetry data from multiple individuals can provide information about the spatial use and temporal movements of populations.

Tracking data is three dimensional, with the first two dimensions, X and Y, ordered along the third dimension, time. GIS software has many capabilities to store, analyze and visualize the location information, but little or no support for visualizing the temporal data, and tools for processing temporal data are lacking. We explored several ways of analyzing the movement patterns using the spatiotemporal data provided by satellite tags. Here, we present the results of one promising method: time-variant kernel density analysis (Keating and Cherry, 2009). The goal of this chapter is to demonstrate new methods in spatial analysis to visualize and interpret tracking data for a large number of individual birds across time in the mid-Atlantic study area and beyond. In this chapter, we placed greater emphasis on analytical methods than on the behavior and ecology of the animals tracked. For more detailed examinations of the ecology and wintering habitat use of the focal species in the mid-Atlantic, see Chapters 20-22.

## Introduction

Van Winkle's (1975) seminal work on utilization distribution (UD) of animals has been variously adapted, but the basic two-dimensional probability surface showing space use for a period of time can overly simplify the dynamic structure of location data and the complexities of animal movements and spatial use. The utilization distribution, as defined by Van Winkle (1975), is simply a probability surface representing the habitat use of an animal throughout its home range, and is generated from location-based studies. Reducing point data to a UD has the advantage of estimating probability of use by an animal in areas where it has not been located, which can also be disadvantageous when assigning probability to an area where an animal does not occur (e.g., inland areas for marine birds). The kernel estimators are non-parametric in that there are no assumptions about the underlying distribution (Millspaugh et al., 2006; Seaman et al., 1999). The UD is dependent on sample size, however, and extrapolating UD to the population level may be biased by capture location and how well the sample of tagged animals represents the population of interest. In this study, we were unable to sample animals in an unbiased way, given the constraints of time and budgets. We were most interested in movements and habitat use along the mid-Atlantic coast of the U.S., however, and are hopeful that this capture bias did not negatively affect our estimates for this region. For Surf Scoters, where we captured birds in Québec and Labrador as well as in the mid-Atlantic region, we were able to confirm that there was negligible difference in winter use of the mid-Atlantic between these groups (A. Gilbert, unpub. data).

While we do have data for some Northern Gannets captured on the breeding grounds, these tags were short-lived and did not provide enough data to generate comparable UDs. Nevertheless, we pooled all data for each species irrespective of capture location.

In kernel based estimation of the UD, spatial smoothing parameters, $h_{xy,}$ in X (longitude) and Y (latitude) dimensions dictate the degree to which points contribute to the overall probability surface. These parameters extend the surface to the space between locations and beyond the edges of measured use, and appropriate assignment of $h_{xy}$ is important for generating accurate UDs for the data. Thus, a larger value of $h_{xy}$ will better account for uncertainty in the UD estimate, but can also smooth out any real small-scale variation in space use (Millspaugh et al., 2006). A variety of smoothing parameter estimation routines have been proposed, with various benefits among them (Millspaugh et al., 2006; Worton, 1989), but we do not examine their use here. Rather, we explore the kernel method for UD estimation to assess its usefulness in summarizing space use across the study area.

Typical two-dimensional UDs, generated to understand home ranges, collapses time into the estimate of the UD probability surface. This method simplifies the estimation of the surface, but reduces our ability to understand and demonstrate the temporal variability inherent in animal movements. Recently, Keating and Cherry (2009) extended the work of Van Winkle (1975) by adding time as another dimension by which to estimate UD. This extension to the basic time-insensitive UD can improve our understanding of migratory or other cyclical animal movements. Their approach uses a product kernel method (Silverman, 1986) with a wrapped Cauchy distribution for continuous circular variables, allowing kernel density estimation with Julian date as a circular time variable. The resulting time-variant kernel density model can represent the UD throughout the year, depending on the time in which the distribution is centered and the distribution's temporal bandwidth, $h_t$ (time over which points are integrated to create a density surface). This temporal probability surface is analogous to the spatial probability surface governed in part by spatial bandwidth $h_{xy}$, but in the temporal dimension. Changing the size of $h_t$ can have similar effects to that of UD estimates at the edge of the range, where animal use can be overestimated, depending on the size of the smoothing parameter ($h_{xy}$). Likewise, in the temporal dimension, this appears as lingering distribution during a time when animals may not actually occur. This is, in effect, a temporal smoothing on the spatial dimension that, if wide enough, results in a UD "ghosting" and an inaccurate representation of space use at that time. Unfortunately, the software used to calculate time-variant kernels does not have a method for estimating appropriate $h_t$; therefore, we explored multiple values for $h_t$ to understand the model's sensitivity to $h_t$ in the resulting maps.

## Methods

Satellite telemetry data for focal species were collected from transmitters implanted in Red-throated Loons and Northern Gannets in the mid-Atlantic region (Delaware Bay, Chesapeake Bay, and Pamlico Sound), tags implanted in Surf Scoters by the SDJV elsewhere in eastern North America (see Chapter 20), and tail-taped tags on Northern Gannets from the colony at Cape St. Mary's, Newfoundland (see Chapter 22). Capture methods and locations, and PTT implantation methods, are discussed in Chapters 20-22. Satellite data were compiled and filtered using the Douglas Argos Filter[2] (DAF; Douglas et al.,

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[2] http://alaska.usgs.gov/science/biology/spatial/douglas.html

2012). The DAF is a threshold filter that has several user-defined parameters to flag improbable locations in satellite tracking data. The parameters are adjustable based on species' movement behaviors, time of year, and the scale of the area under observation. With the DAF, data are retained if they pass a spatial redundancy test and/or a movement rate and turning angle test. Since our bird data contain both short-distance, local movements, and long-distance migratory events, we employed a hybrid of the distance, angle, and rate (DAR) filter, and the minimum redundant distance (MRD) filter. Using DAF, we also identified the best representative point per duty cycle for each animal, to reduce redundant daily positional information and spatial-autocorrelation. We used the best point per duty cycle in all estimations of time-variant UD. We aggregated all data across years within species (2000-2013 for Surf Scoter, and 2012-2013 for all others) for this analysis. We chose to include points from all years for all birds, to represent the variability in spatial use regardless of whether that variation was between animals or years.

The resulting output files from DAF were processed in R version 3.1.2 (R Core Team, 2014) for modeling and mapping. We subsampled locations using the package ‘adehabitatHR’ version 0.4.12 (Calenge, 2006) for modeling the UDs. We filtered data from implanted birds to exclude the first two weeks of data, to reduce the effects of implantation surgery on movement, and to exclude those that transmitted for fewer than 30 days. To reduce the “temporal ghosting” effect inherent in this method, in which any location where an animal has been recorded is identified as a possible current use area, we binned location data by 60 day intervals over the annual cycle. For each time interval, we selected a random sample of locations from each individual to prevent animals with large numbers of locations from disproportionately influencing the resulting pooled UD. The size of each resample was determined by the lower quartile (or fewer) of the number of locations available for all animals (see Table 23-1). We derived a composite estimate of the kernel UDs using a bootstrapping procedure. For each species, we resampled locations from all animals and calculated the UD for the resample. The resampling procedure was repeated for 100 iterations and the bootstrapped mean and standard error surface served as the composite UD for each time interval.

The likelihood cross-validation procedure that was used by Keating and Cherry (2009) was not implemented in the R package ‘adehabitatHR’ version 0.4.12 (Calenge, 2006). Therefore, we subjectively chose an appropriate bandwidth to suit the data. We used temporal bandwidth $h_t$=0.05, and spatial bandwidths $h_{xy}$=0.25. Values of $h_t$ could vary between 0-1, but without good guidance on appropriate values for this smoothing parameter, smaller values generally produced results that better represented the actual movement and distribution patterns of the animals. However, more work is needed to explore this parameter. Spatial bandwidth was also subjectively explored in this exercise, but we found that the smoothing parameter of 0.25 was reasonable without losing multimodal detail of high use areas or extending the probability surface into areas where animals are unlikely. There are other objective methods for determining the spatial bandwidth not implemented here (Millspaugh et al., 2006), but it has also been suggested that there are many situations where subjective selection of $h_{xy}$ is appropriate (Kie et al., 2010).

Map production was accomplished in R using package ‘OpenStreetMap’ version 0.3.1 (Fellows and Stotz, 2013) to create background maps and package ‘raster’ version 2.3 (Hijmans, 2015) to crop the UDs to exclude over-land areas and overlay the UD output on each background map. Videos of the sequence of time variant kernel maps were produced in mp4 format using the open-source audio-video editing software Ffmpeg[3]. We generated maps and corresponding videos for each focal species scaled to the mid-Atlantic Outer Continental Shelf (the study area of the BOEM telemetry project, which extends from the New York Bight to the South Carolina border, and fully encompasses the mid-Atlantic study area described in other chapters of this report). Twenty-four static maps were produced, eight for each species (Figure 23-1 to Figure 23-12). Maps include overlays of the mid-Atlantic study area, BOEM renewable energy leasing areas, wind planning areas, and wind energy areas[4] (collectively referred to hereafter as Wind Energy Areas, or WEAs), and mid-Atlantic capture locations.

## Results and discussion

The time-variant kernel density maps depict broad movement patterns throughout the year (see Figure 23-1 to Figure 23-12) and do an excellent job revealing the temporal patterns of movement throughout the mid-Atlantic study area and beyond. The “temporal ghosting” effect inherent to this method, in which any location where an animal had been recorded was identified as a possible current use area, was limited by the use of a 60-day moving time window around each date. The small temporal bandwidth ($h_t$=0.05) also reduced the effect of apparent occupancy in an area during a period when they do not occur. We were able to determine a reasonable $h_t$ parameter by running a series of models with decreasing temporal bandwidth. The challenge was to smooth data temporally over the period of estimation, but not to the extent that the kernel density model was homogenized across the entire time period (in a sense removing the temporal component) such that we lost temporal detail of occurrence in the kernel density maps. This was an inexact method which we hope to explore more objectively, e.g., likelihood cross-validations estimators used by Keating and Cherry (2009). Similarly, we used a subjectively determined spatial bandwidth ($h_{xy}$=0.25), which was also determined by running a series of model runs while changing $h_{xy}$. We arrived at the $h_{xy}$ used in this work because it provided a more realistic continuous surface while not generating the “spurious structure” warned against by Keating and Cherry (2009). Again, we plan to explore a more objective method for bandwidth estimation, such as the likelihood cross-validations estimator.

### *Red-throated Loons*

The Red-throated Loons tracked in this study wintered primarily in bays and relatively nearshore areas of the mid-Atlantic, with the UD largely focused near initial capture locations. Areas of highest use in January-April were in Delaware Bay, Chesapeake Bay, and Pamlico Sound (Figure 23-1 and Figure 23-2). Birds began migrating north in April-May, with a migratory stopover area apparent offshore of Cape Cod, Massachusetts in May (Figure 23-2). Birds began returning to the mid-Atlantic by November, to spend the non-breeding period in the region where they had been captured the previous winter (Figure 23-4). There was little apparent overlap between UDs and WEAs, with loons remaining largely inshore of these areas. For a more detailed discussion of the movement and habitat use of tracked Red-throated Loons, see Chapter 21.

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[3] http://ffmpeg.org/

[4] http://www.boem.gov/Renewable-Energy-GIS-Data/

### *Northern Gannets*

In winter, Northern Gannets ranged much farther than the other focal species, moving from Cape Cod to well south of the study area, primarily staying nearshore and somewhat around large bays, particularly at the mouths of Chesapeake and Delaware Bays (Figure 23-5 to Figure 23-6). Spring migration began in April with birds returning from southern locations, moving north to Cape Cod (Figure 23-6). Birds began arriving back in the study area and points south by November (Figure 23-8). At this time, Northern Gannets were most widely dispersed. From the maps, it is clear that there is the potential for Northern Gannet UDs to overlap WEAs, particularly off of Delaware and New Jersey, but primarily birds appear to be inshore of these areas. For a more detailed discussion of the movement and habitat use of these tracked Northern Gannets, see Chapter 22.

### *Surf Scoters*

Like Red-throated Loons, Surf Scoters wintered in bays and nearshore, primarily Chesapeake Bay, Delaware Bay, Pamlico Sound, and off of Cape Cod (Figure 23-9 to Figure 23-10). Spring movements north began in April and extended into May. However, some Surf Scoters remained in the Chesapeake Bay area as late as May (Figure 23-10). By November, most Surf Scoters had returned to mid-Atlantic wintering areas (Figure 23-12). There was very little overlap between the UD and WEAs, the birds being primarily inshore of these areas, though it is important to note the potential bias introduced by the sampling design, as noted in the Introduction. Specifically, some wintering areas may be under-represented by including Surf Scoters marked in the mid-Atlantic study areas, particularly those wintering areas located further north in the Gulf of Maine. Comparisons of UDs calculated separately for birds captured within and outside of the study area suggest, however, that this potential bias is unlikely to be affecting our conclusion about overlap between the UD and WEAs. For a more detailed discussion of the movement and habitat use of these tracked Surf Scoters, see Chapter 20 and a recent seaduck migration study report by the Seaduck Joint Venture (SDJV, 2015).

### *Utility of time-variant wildlife data for offshore wind energy planning*

Spatiotemporal movement patterns are difficult to portray with static maps that represent entire years or seasons, but when depicted as an animation, using shorter time-steps, the life-history [or temporal migratory] connections of these animals are revealed. This was not easily accomplished prior to the advent of satellite telemetry tags and the more sophisticated computer modeling techniques. We believe that this technique has great utility in depicting complex spatiotemporal patterns not immediately obvious in multiple static maps. The time-variant kernel method allowed us to bring temporal information into our estimation of the density surface depicting utilization distributions. Normal kernel density estimation methods collapse the temporal dimension, losing some of this important detail. We believe this method has great potential for understanding and visualizing the annual cycles of these species.

Understanding the temporal variation in animal movements and habitat use is essential for minimizing the effects of offshore wind energy development on wildlife populations. For example, certain development activities may be timed to minimize impacts to potentially vulnerable populations in those locations, or possible mitigation strategies can be targeted to life history periods of greatest need. This type of temporal data is a key component of Strategic Environmental Assessments (SEA) in Europe and

Environmental Impact Assessments (EIS) in the United States (Allison et al., 2008; Fox et al., 2006). All three focal species in this satellite telemetry study are of conservation interest to the USFWS, specifically in relation to offshore wind energy development (ACJV, 2008; USFWS, 2008), and we expect that these data will be useful to both regulators and developers as the offshore wind energy development expands in the United States.

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## Figures and tables

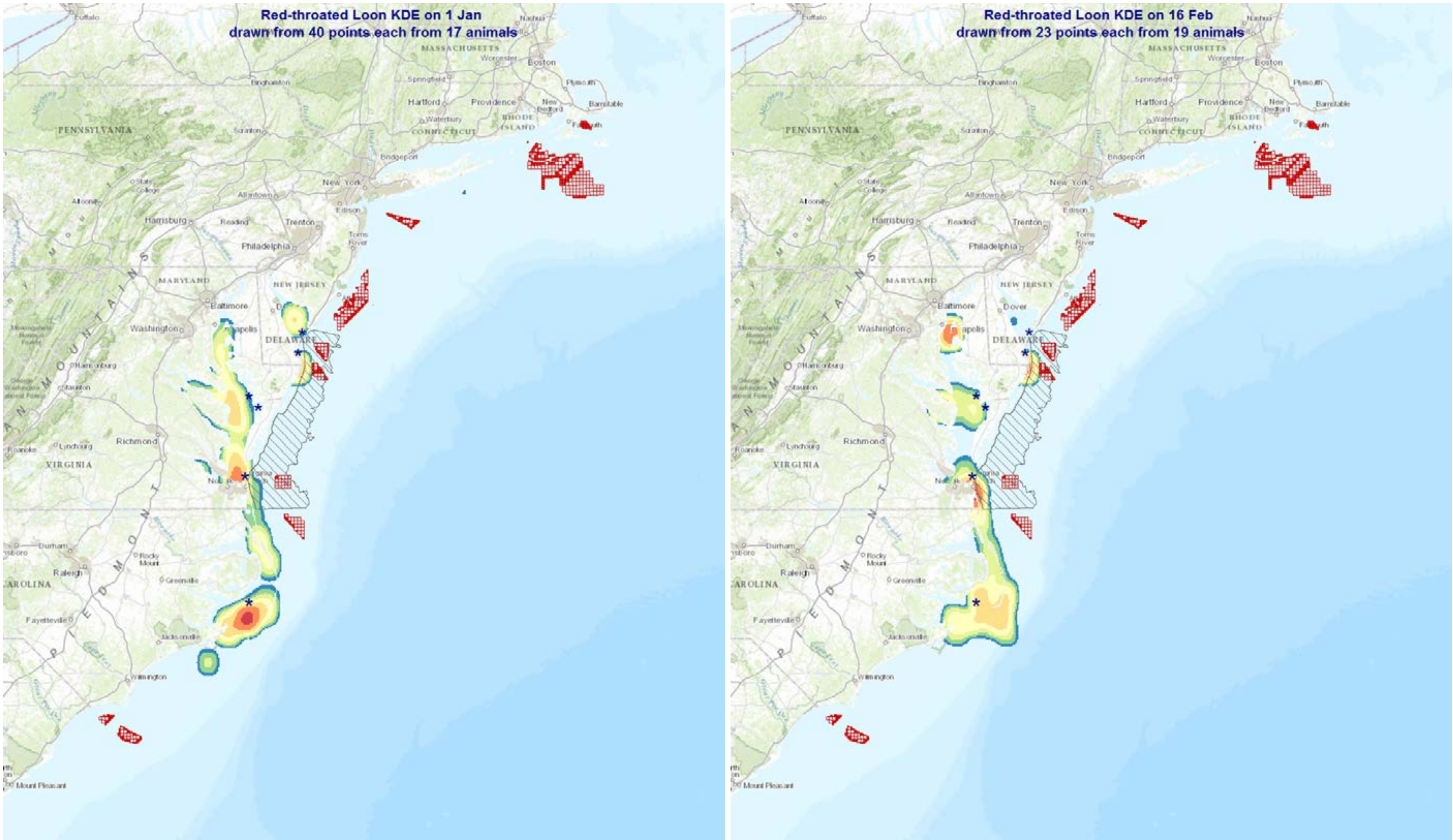


**Figure 23-1. Time variant kernel density (KDE) maps for Red-throated Loons on January 1 (left) and February 16 (right) for 2012-2013 in the mid-Atlantic study area.** Forty points from 17 animals (left) and 23 points from 19 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

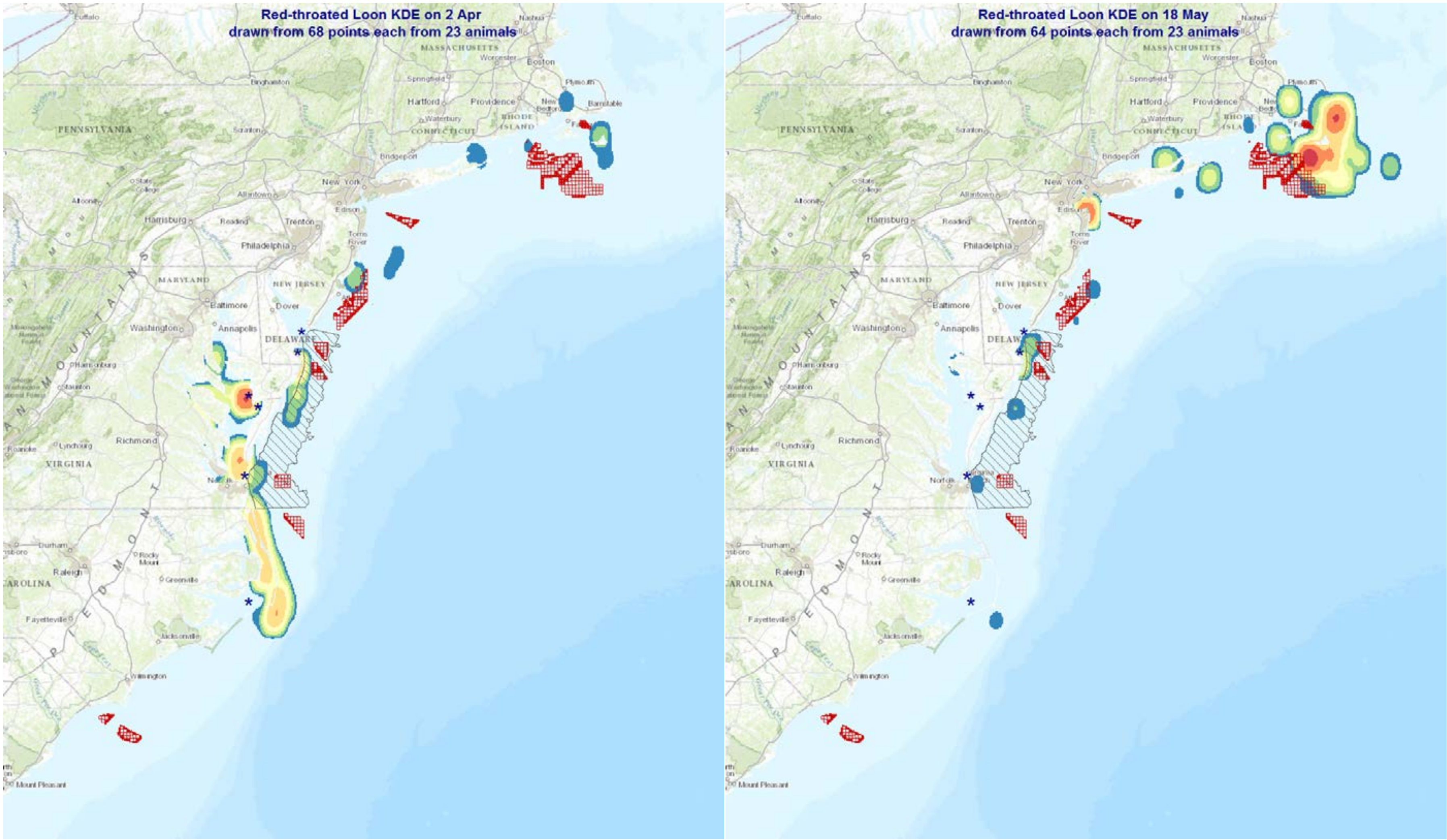
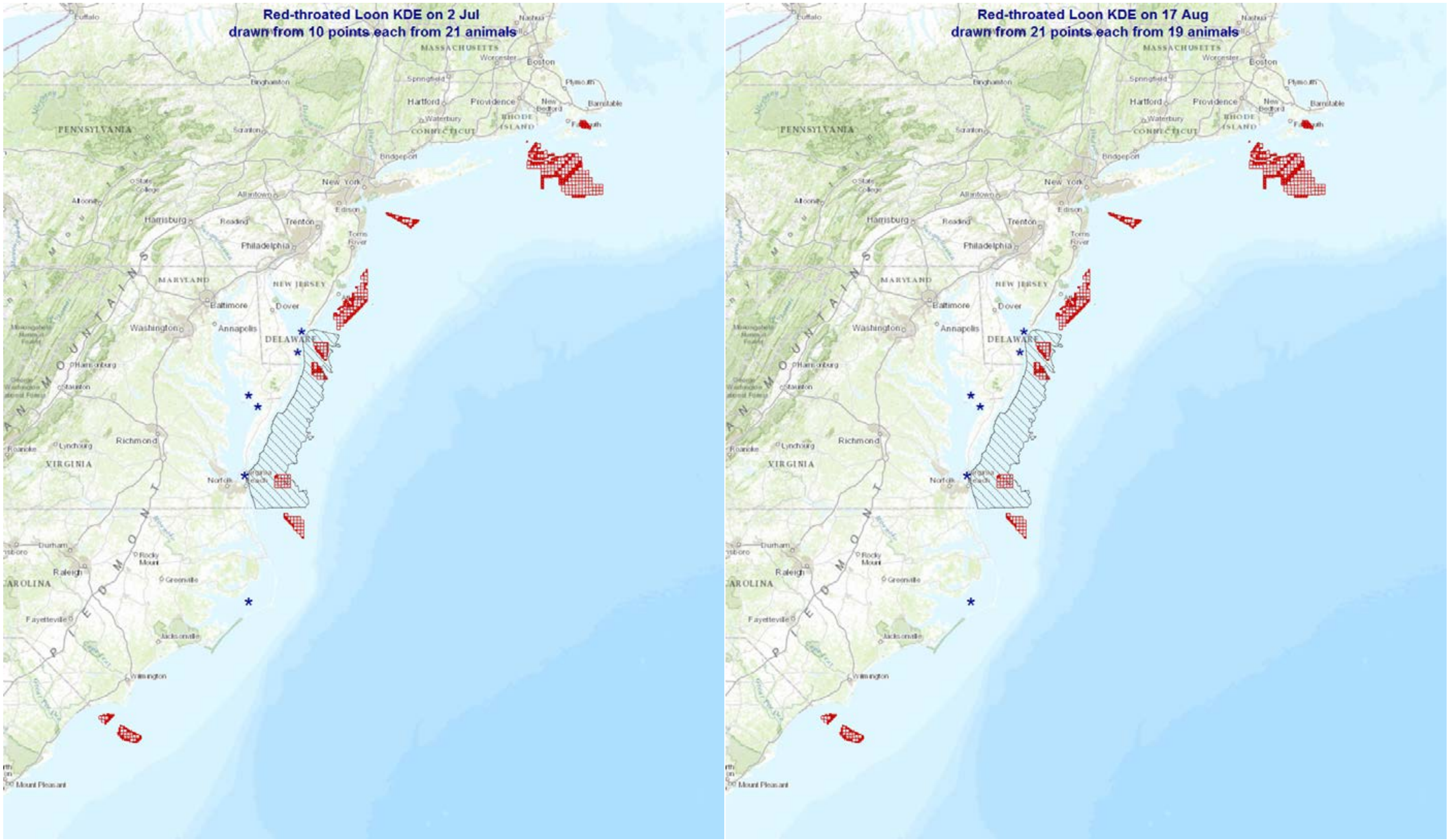


**Figure 23-2. Time variant kernel density (KDE) maps for Red-throated Loons on April 2 (left) and May 18 (right) for 2012-2013 in the mid-Atlantic study area.** Sixty-eight points from 23 animals (left) and 64 points from 23 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.



Red-throated Loon KDE on 2 Jul
drawn from 10 points each from 21 animals

Red-throated Loon KDE on 17 Aug
drawn from 21 points each from 19 animals

**Figure 23-3. Time variant kernel density (KDE) maps for Red-throated Loons on July 2 (left) and August 17 (right) for 2012-2013 in the mid-Atlantic study area.** Ten points from 21 animals (left) and 21 points from 19 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

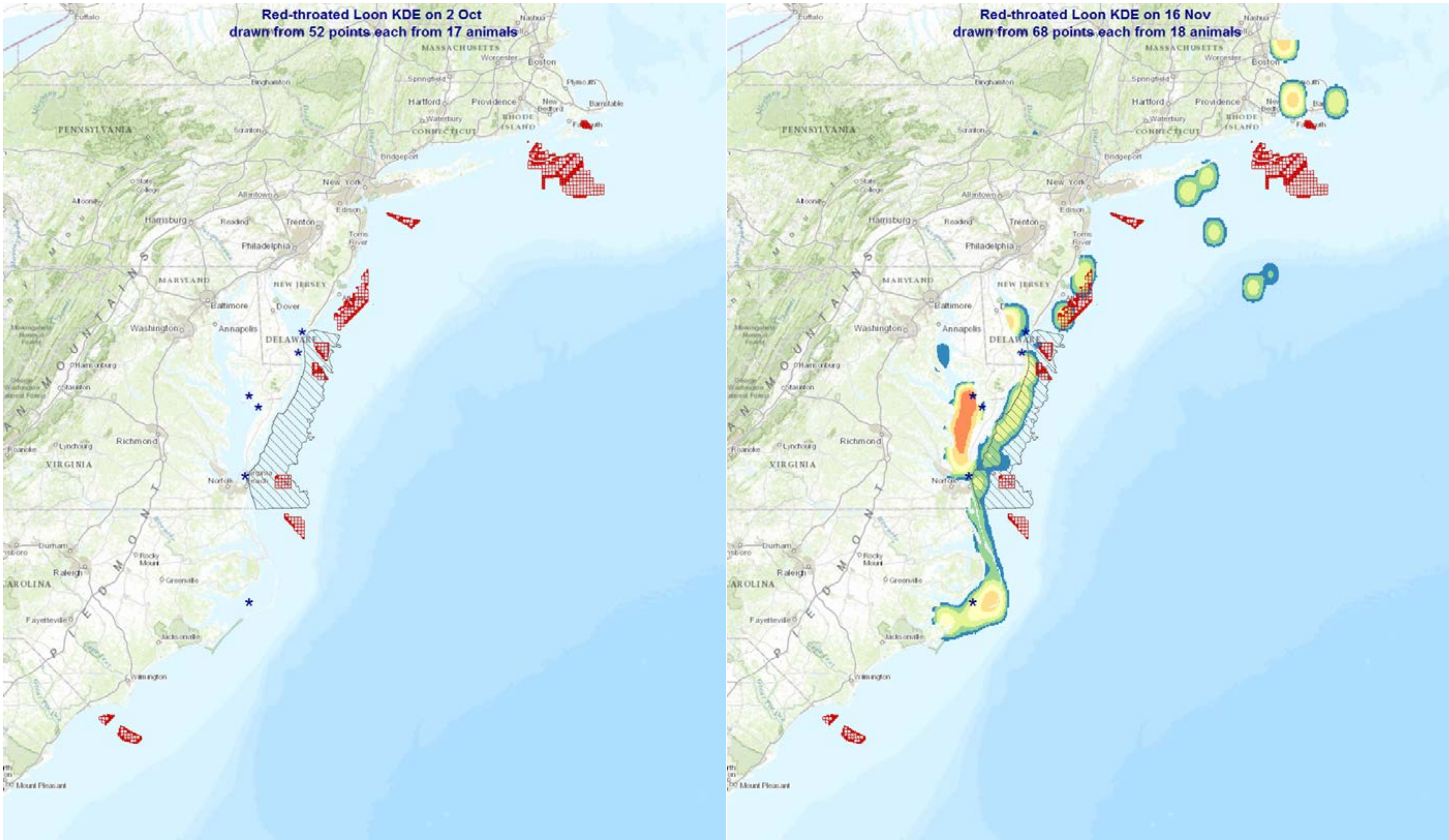


**Figure 23-4. Time variant kernel density (KDE) maps for Red-throated Loons on October 2 (left) and November 16 (right) for 2012-2013 in the mid-Atlantic study area.** Fifty-two points from 17 animals (left) and 68 points from 18 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

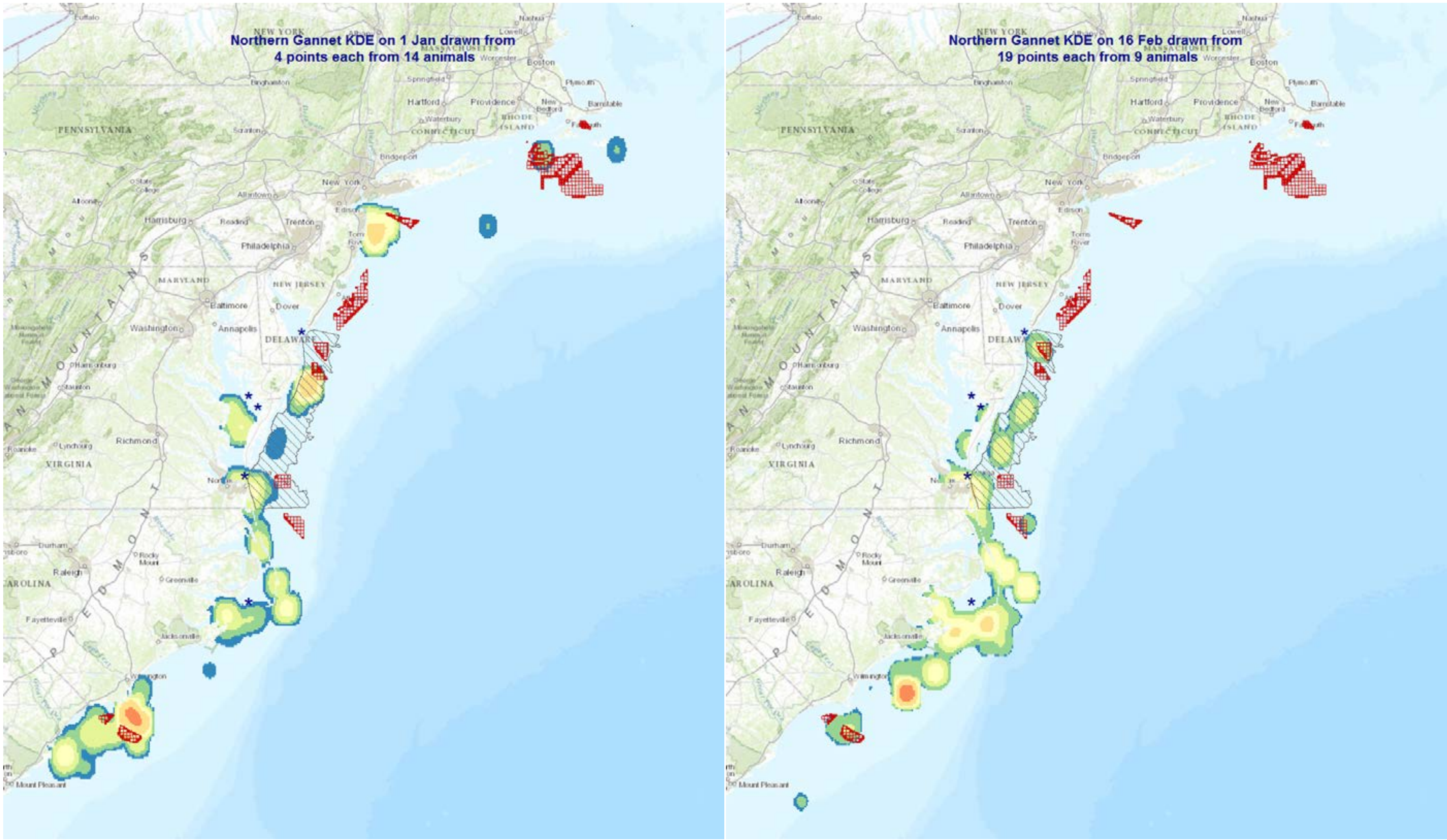


**Figure 23-5. Time variant kernel density (KDE) maps for Northern Gannets on January 1 (left) and February 16 (right) for 2012-2013 in the mid-Atlantic study area.** Four points from 14 animals (left) and 19 points from 9 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

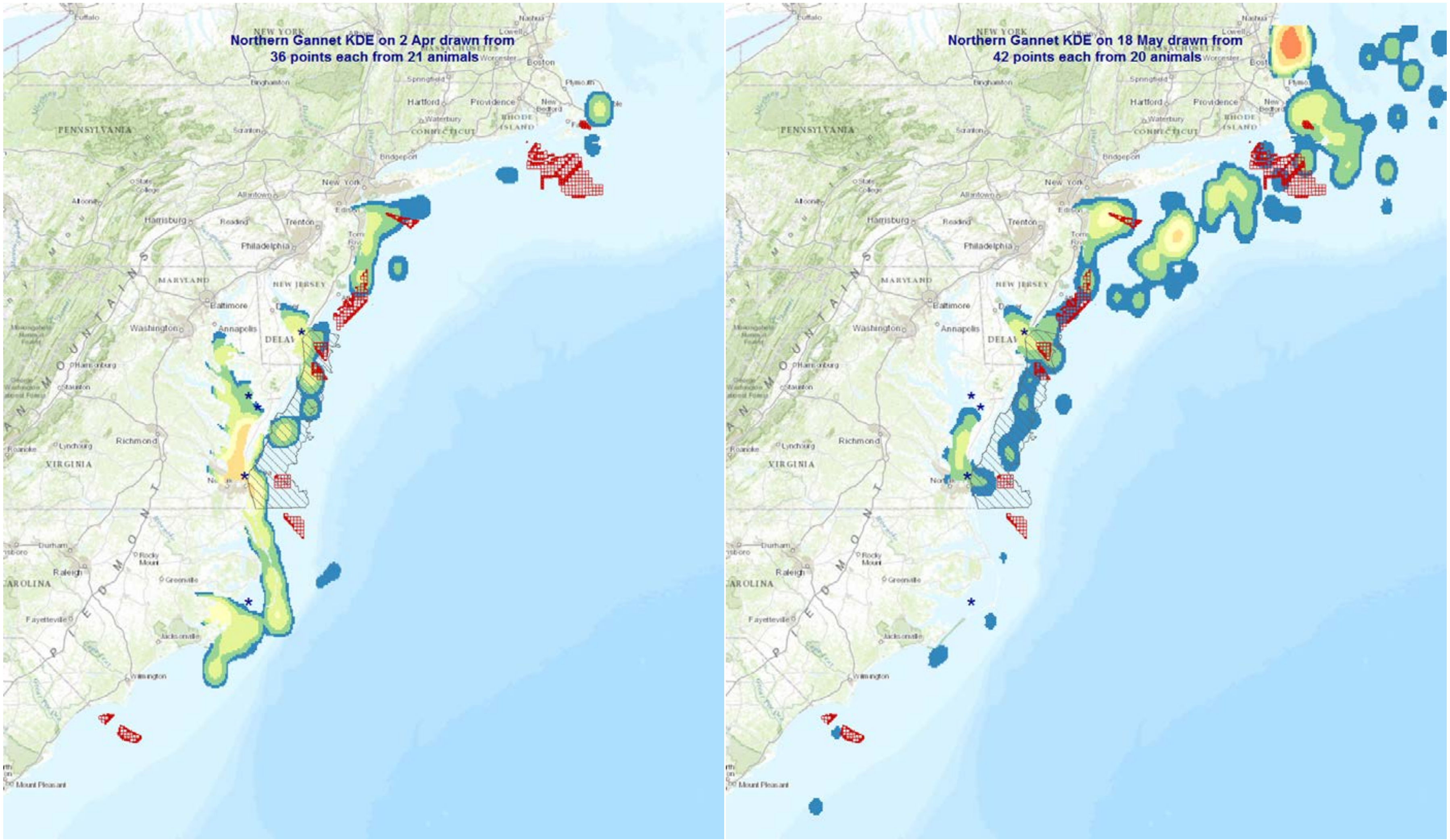


**Figure 23-6. Time variant kernel density (KDE) maps for Northern Gannets on April 2 (left) and May 18 (right) for 2012-2013 in the mid-Atlantic study area.** Thirty-six points from 21 animals (left) and 42 points from 20 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

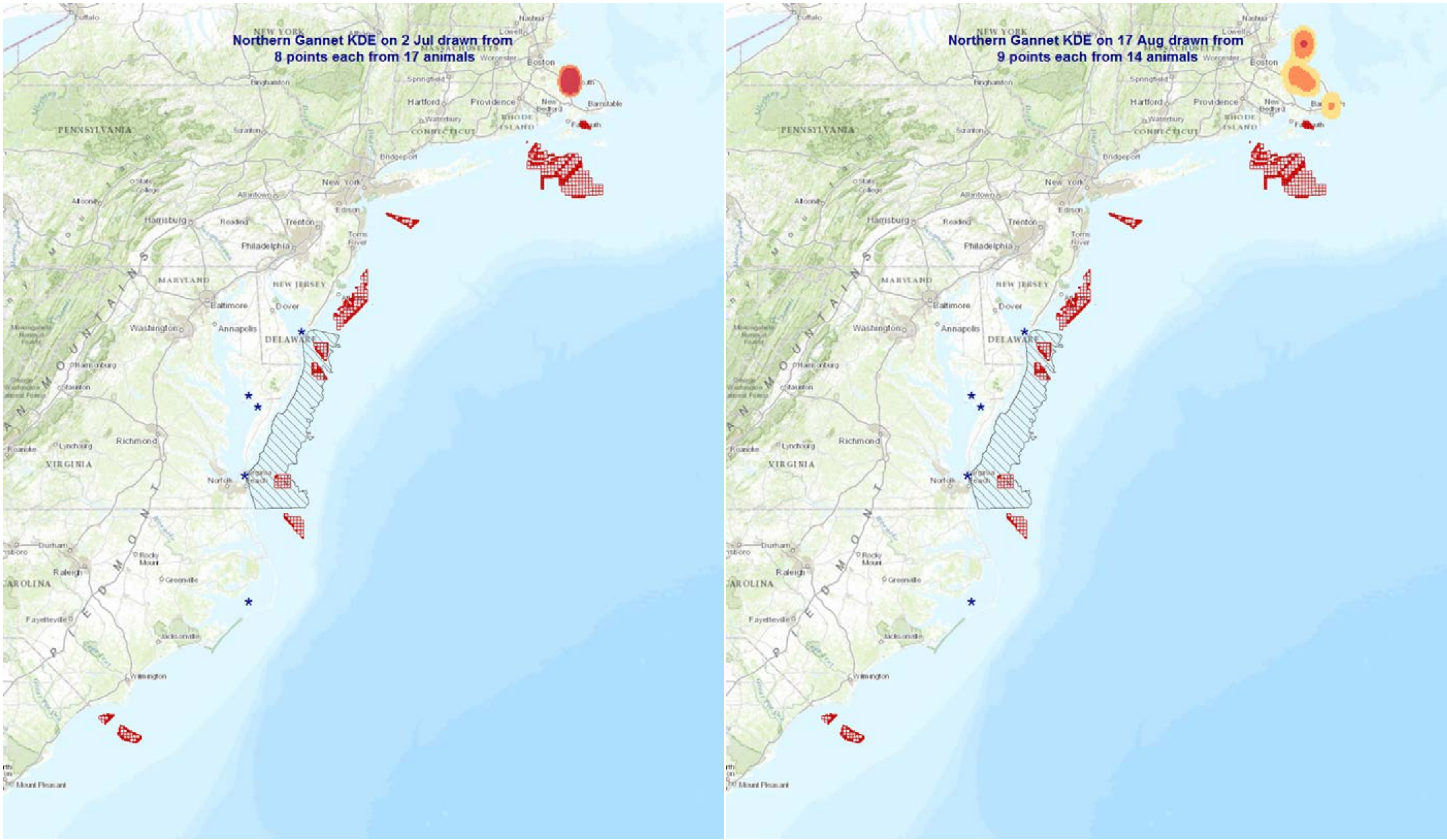


**Figure 23-7. Time variant kernel density (KDE) maps for Northern Gannets on July 2 (left) and August 17 (right) for 2012-2013 in the mid-Atlantic study area.** Eight points from 17 animals (left) and 9 points from 14 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

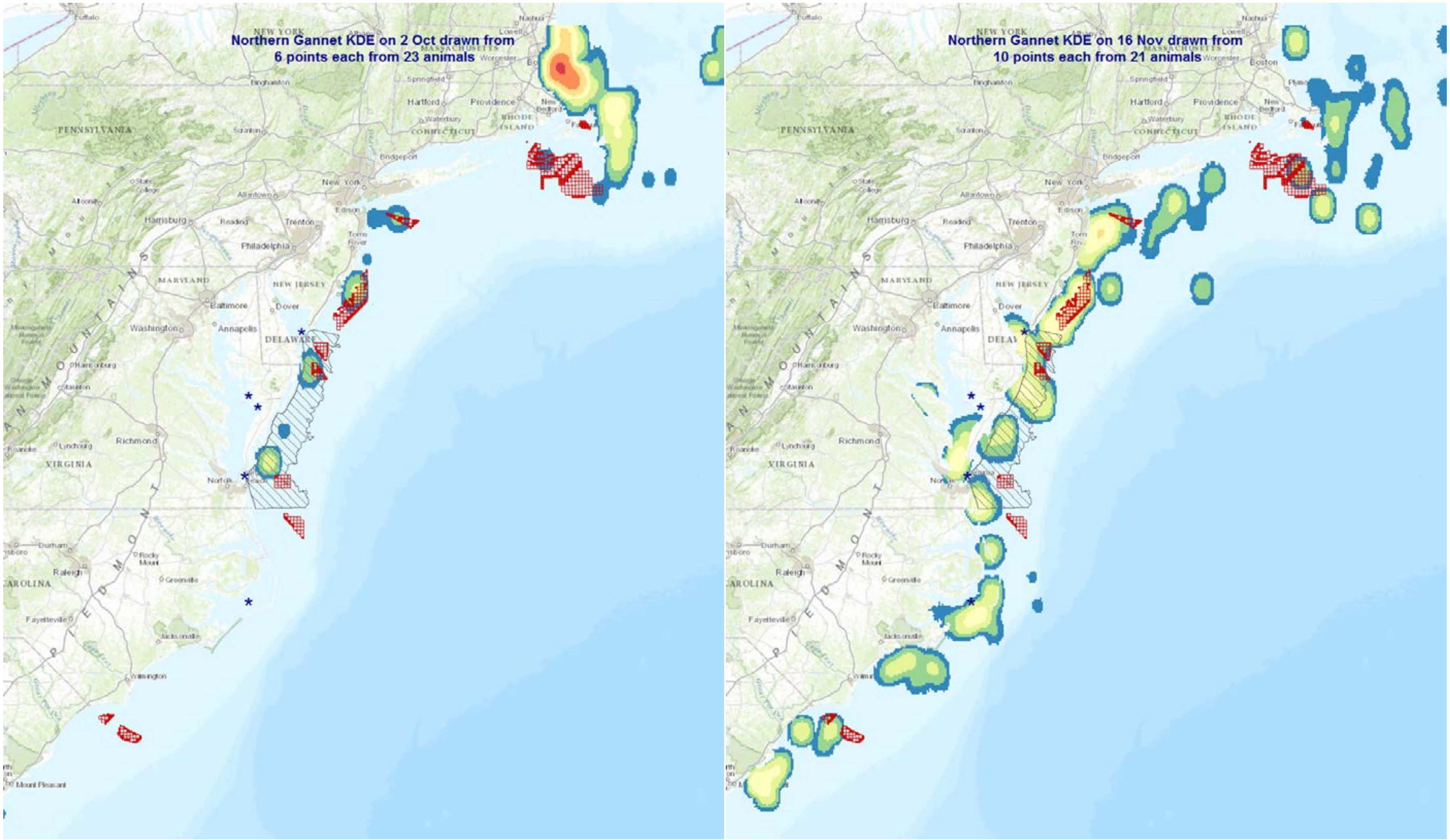


**Figure 23-8. Time variant kernel density (KDE) maps for Northern Gannets on October 2 (left) and November 16 (right) for 2012-2013 in the mid-Atlantic study area.** Six points from 23 animals (left) and 10 points from 21 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

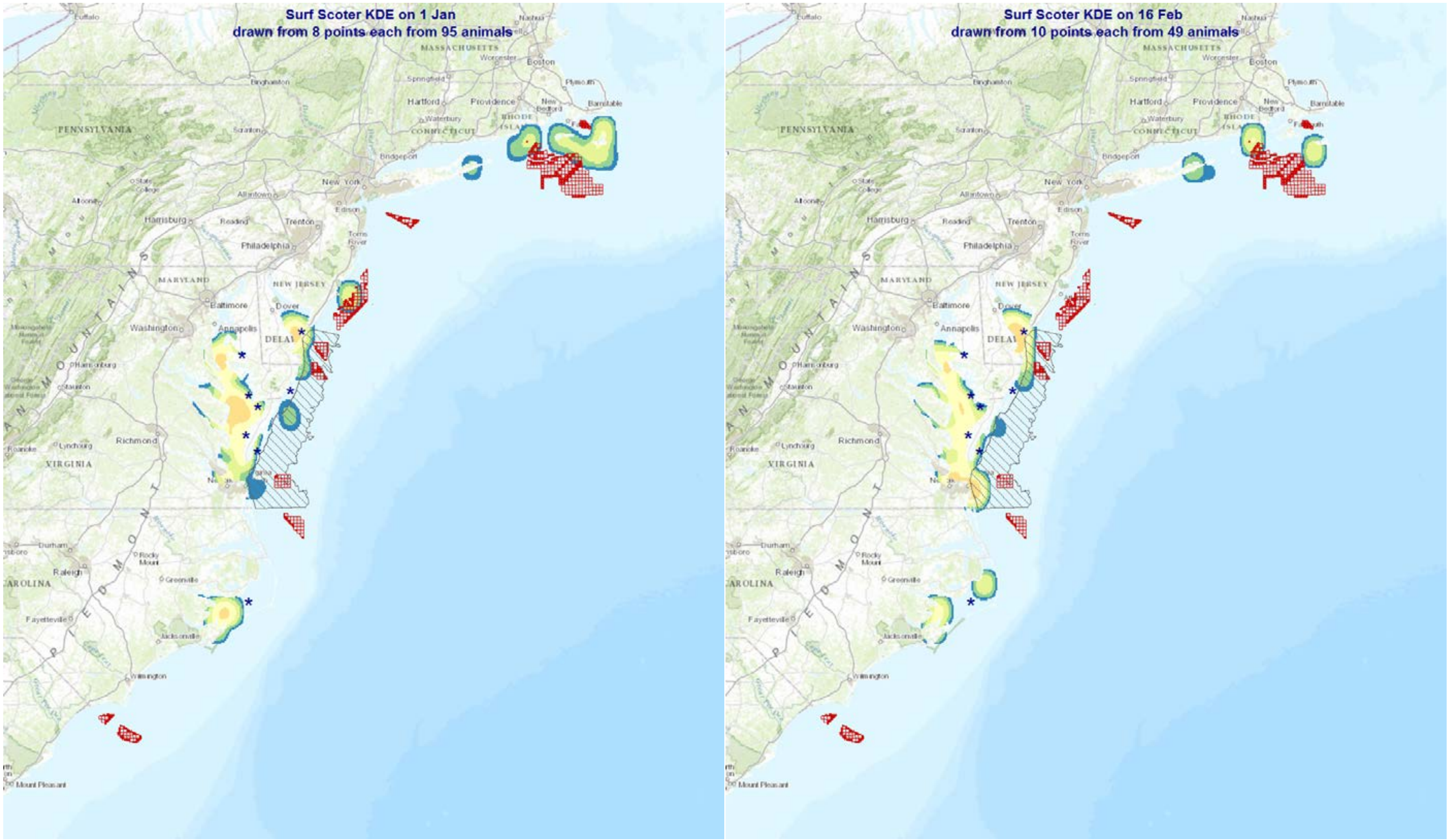


**Figure 23-9. Time variant kernel density (KDE) maps for Surf Scoters on January 1 (left) and February 16 (right) for 2012-2013 in the mid-Atlantic study area.** Eight points from 95 animals (left) and 10 points from 49 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

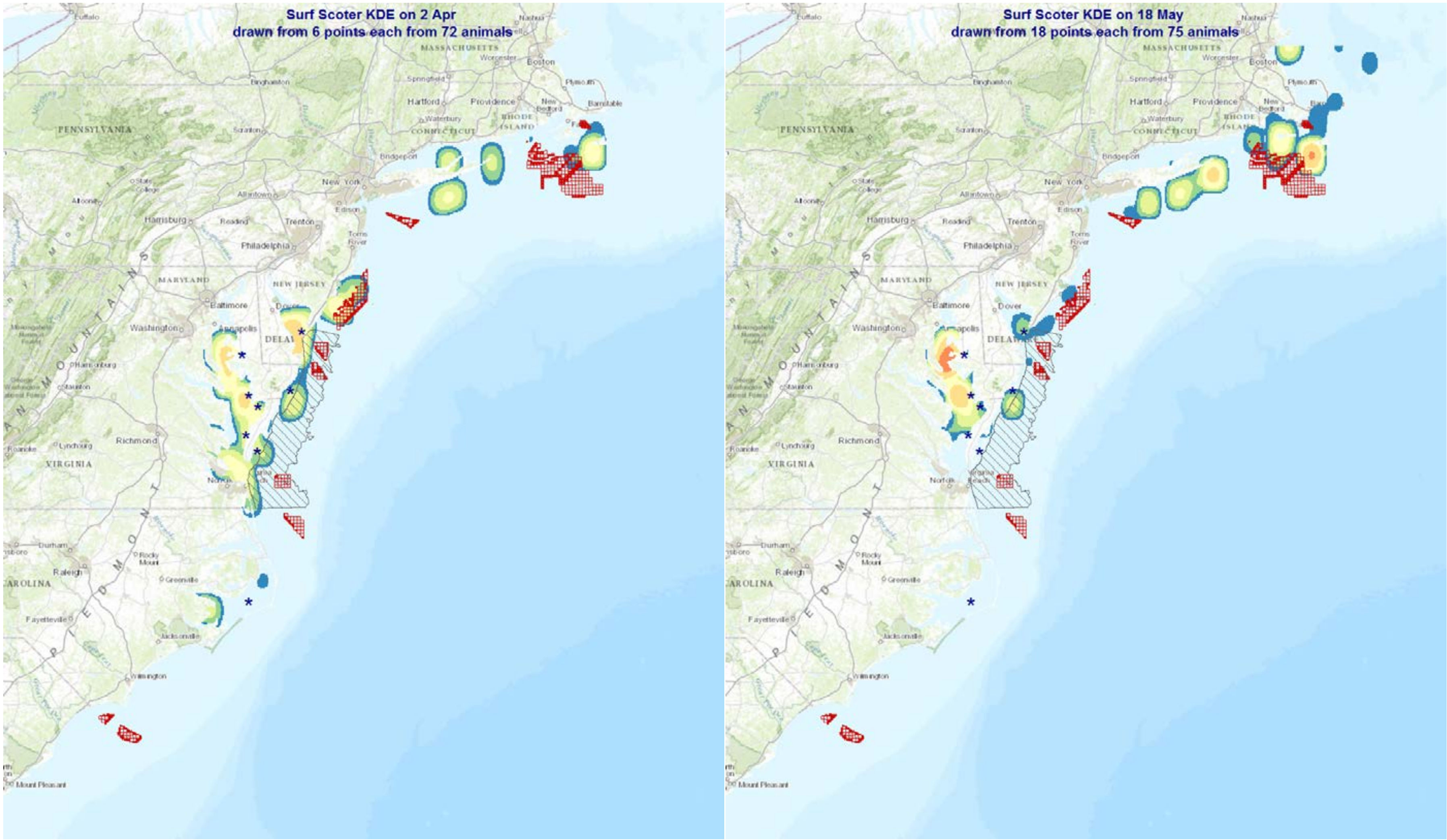


**Figure 23-10. Time variant kernel density (KDE) maps for Surf Scoters on April 2 (left) and May 18 (right) for 2012-2013 in the mid-Atlantic study area.** Six points from 72 animals (left) and 18 points from 75 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

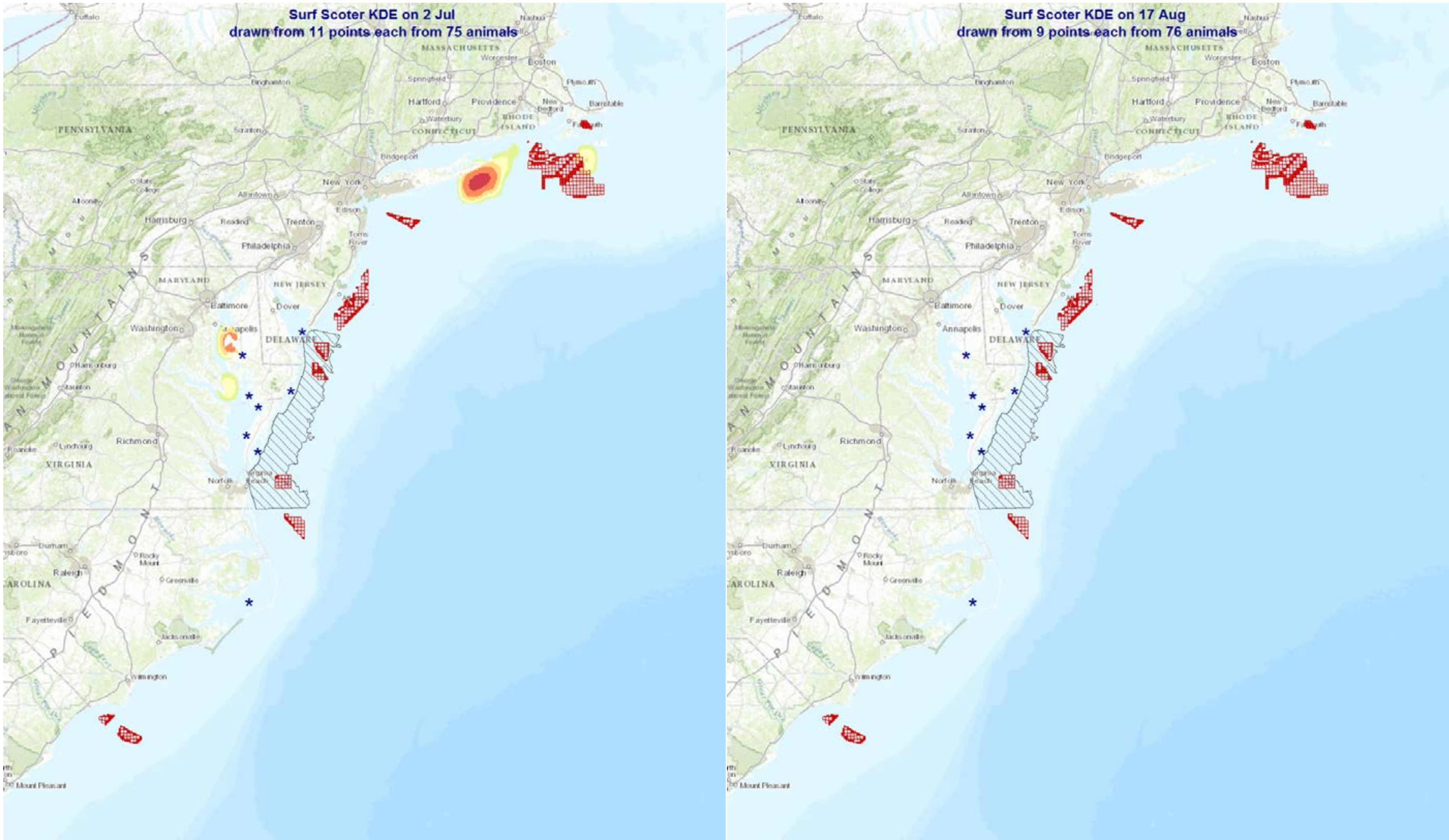


**Figure 23-11. Time variant kernel density (KDE) maps for Surf Scoters on July 2 (left) and August 17 (right) for 2012-2013 in the mid-Atlantic study area.** Eleven points from 75 animals (left) and 9 points from 76 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

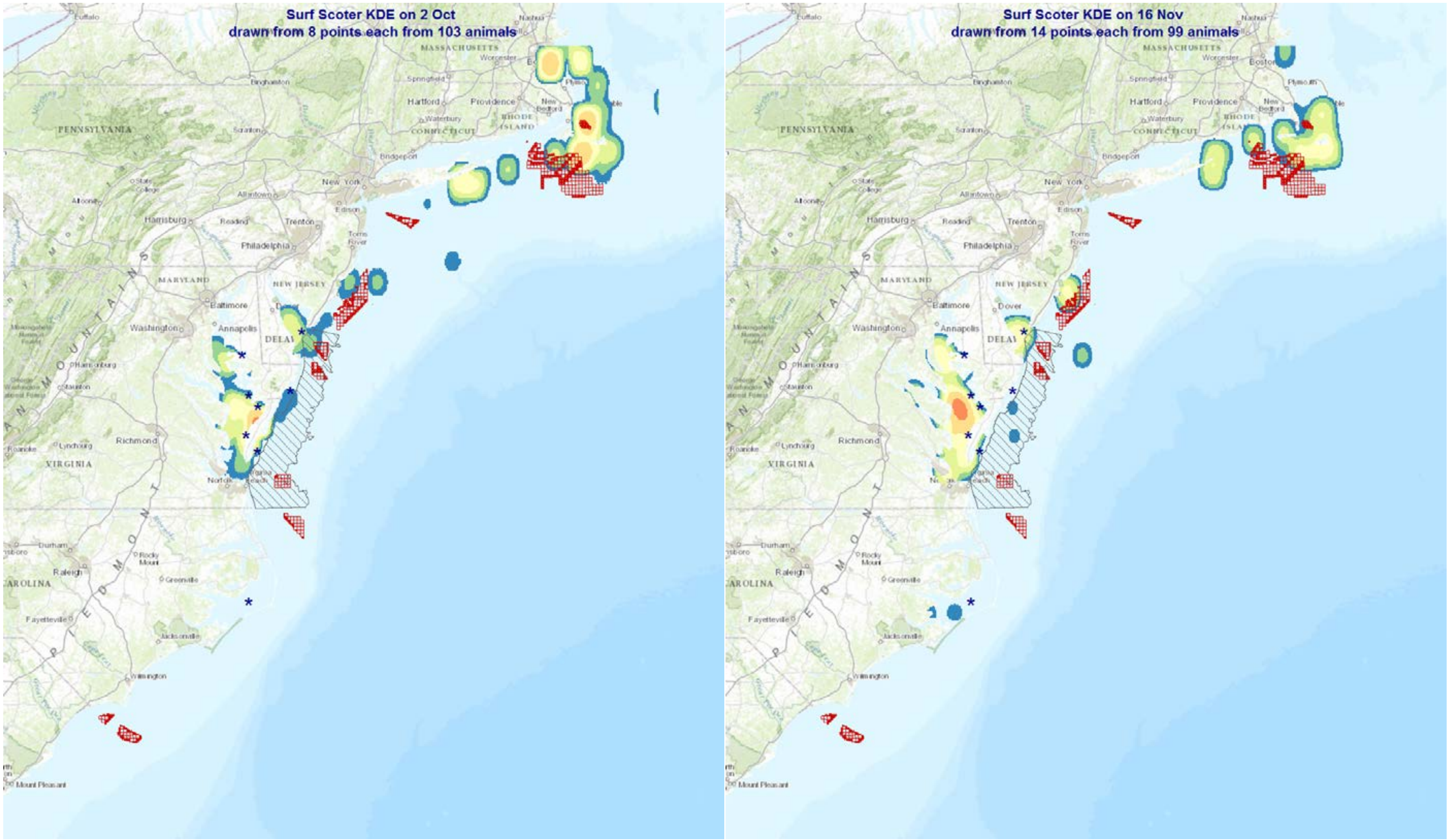


**Figure 23-12. Time variant kernel density (KDE) maps for Surf Scoters October 2 (left) and November 16 (right) for 2012-2013 in the mid-Atlantic study area.** Eight points from 103 animals (left) and 14 points from 99 animals (right) were used for calculating the composite KDEs for these dates. Composite KDE is depicted in quantile gradations from lowest (blue) to highest (red) probabilities of use. The mid-Atlantic study area is in light gray, the BOEM WEAs in red, and capture locations are indicated by blue asterisks.

**Table 23-1. Number of animals and (samples sizes per individual) on each date the time variant kernel was estimated.** Samples sizes are the number of locations for each animal used in the composite kernel estimates.

| Date | Red-throated Loon | Northern Gannet | Surf Scoter |
|---|---|---|---|
| **January 1** | 17 (40) | 14 (4) | 95 (8) |
| **February 16** | 19 (23) | 9 (19) | 49 (10) |
| **April 2** | 23 (68) | 21 (36) | 72 (6) |
| **May 18** | 23 (64) | 20 (42) | 75 (18) |
| **July 2** | 21 (10) | 17 (8) | 75 (11) |
| **August 17** | 19 (21) | 14 (9) | 76 (9) |
| **October 2** | 17 (52) | 23 (6) | 103 (8) |
| **November 16** | 18 (68) | 21 (10) | 99(14) |

# Chapter 24: Using state-space models to identify areas of persistent winter activity and their associated environmental covariates in Northern Gannets

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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**Project webpage:** www.briloon.org/mabs

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# Chapter 24 Highlights

*Using telemetry data to determine environmental drivers of foraging activity in Northern Gannets*

### *Context*[1]

The project team investigated the spatial patterns, temporal patterns, and environmental variation in offshore bird migration and winter habitat use through a combination of satellite telemetry data and remotely collected environmental covariate information. Northern Gannet telemetry data are analyzed in three chapters: Chapter 22 (focused on wintering habitat use of Northern Gannets in the mid-Atlantic), Chapter 23 (inclusion of a temporal component in models of habitat use, to improve our understanding of the species' use of the landscape through time), and this chapter, which is focused on individual-scale movements and foraging patterns. Northern Gannet interactions with offshore wind energy development are hypothesized to largely occur as a result of their foraging behaviors, which include a large proportion of time spent soaring at or near the altitude of the rotor-sweep zone for offshore wind turbines. Being able to differentiate between foraging and other behaviors in telemetry data will allow us to better determine areas of potential conflict between offshore wind energy development and Northern Gannet habitat use, and could inform the siting and permitting of offshore wind energy development.

### *Study goal/objectives*

Identify the locations of foraging Northern Gannets during the non-breeding season, and determine how bathymetry and sea surface temperature fronts are related to these patterns.

### *Highlights*

- Used satellite tracking data in a state-space model to describe locations where Northern Gannets were conducting foraging behavior in the mid-Atlantic.
- Foraging locations were more common in and around bays, but tended to be in deeper areas.
- The number of nearby sea surface temperature fronts in the week prior to a Northern Gannet observation was the strongest predictor of foraging behavior. As the number of fronts in the area increased, so did the chances of foraging. These fronts are likely aggregating food for Northern Gannets.
- Coastal sea surface temperature fronts showed both annual and daily variability. To predict locations of gannet foraging activity in the future, we will need to be able predict the locations of these fronts.

### *Implications*

Predictions of the effects of offshore wind energy development on Northern Gannets in the mid-Atlantic must consider exposure (e.g., whether they are present in an area in large numbers) as well as well as whether they are foraging in the area. Within their range, gannets are preferentially foraging in deeper waters with high numbers of temperature fronts. Sea surface temperature front locations change on a daily and annual basis, and exposure to offshore development activities may vary at those same scales.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

Satellite telemetry data provide an understanding of the spatial ecology as well as the individual movement behavior of birds. In seabird ecology, movement behavior can be an indicator of whether an animal is foraging at a location and even what kind of prey the animal is pursuing. In this study, we used three years (2012 – 2014) of satellite telemetry data from Northern Gannets (*Morus bassanus*) wintering on the Atlantic coast of North America to build a state-space model in the Bayesian modeling environment. This model was designed to: (1) use all telemetry data acquired to determine the most likely daily position of an individual, and (2) employ a correlated random walk model with a behavioral switching component to identify whether a given location was part of a transient or stationary behavioral pattern. The objective was to identify areas where gannets are likely to be foraging (i.e. conducting area restricted searching, or ARS), and determine the environmental covariates that were associated with that behavior.

Using data from 34 individuals tracked over three winters, positions were classified as foraging behavior 67% of the time. ARS locations were associated with deeper water and locations with persistent sea surface temperature fronts, suggesting that both static and dynamic environmental variables can influence foraging behavior. These behavioral – environmental associations are likely related to increased prey availability. Movement behavior was highly variable among individuals, with some birds moving over long distances, while others focusing on foraging at a single site. Given the strong relationship between temperature fronts and Northern Gannet foraging behavior, accurate projections of ocean temperatures and fronts with climate change will be critical to successfully predicting the locations of Northern Gannet foraging activity.

## Introduction

Understanding animal associations with dynamic environmental conditions is an important behavioral and ecological concern (Nathan 2008), and these relationships can be difficult to quantify in marine environments (Weimerskirch 2007). There are some known general distributional patterns, such as greater abundance in nearshore areas, and greater productivity in estuaries and upwellings (Small and Menzies 1981), but spatial and temporal variance at small spatial scales and short temporal scales can also alter animal distributions and behavior and are not as well understood. Water temperature likely influences animal distributions in several ways. Many marine animal distributions are strongly limited by temperature because ectothermic species like fishes and sea turtles cannot function outside of a given temperature range (Houde 1989, Portner 2001). Changes in ocean temperature can also affect distributions more indirectly, as sea surface temperature (SST) fronts, or abrupt thermal gradients, are mechanisms for generating currents that can aggregate plankton and other passively propelled marine life (Bost et al. 2009), which in turn can aggregate species at higher trophic levels (Benoit-Bird and McManus 2012, Woodson and Litvin 2015). Because ocean temperature can change over multiple temporal scales, it plays a large role in determining the dynamism we see in marine predator behavior and distribution.

Seabirds are adapted to this environmental dynamism and have developed behavioral strategies to deal with the issues of highly clustered and volatile foraging opportunities (Weimerskirch 2007, Miramontes et al. 2012), including methods to predict the occurrence of prey (Pettex et al. 2010). Due to this, seabird foraging behaviors and population parameters have proven to be reliable indicators of prey distributions in the marine environment (Cairns 1987, Santora et al. 2009), and even can be diagnostic to prey species (Elliot et al. 2008). Many seabird species employ wide-area searches followed by area restricted search (ARS) patterns once a location with suitable foraging opportunities is identified (Weimerskirch et al. 2007, Hamer et al. 2009, Pavia et al. 2010). Thus, seabird foraging activity can indicate areas of high prey abundance.

Satellite telemetry provides opportunity to understand the space use and movement behavior of seabirds at scales useful for elucidating relationships with dynamic environmental conditions (Camphuysen et al. 2012). Analytical approaches have been developed to translate spatial information into an understanding of rules that govern the movement of individuals. For example, sequential position data can be used in a correlated random walk model in a Bayesian state-space modeling framework (Jonsen et al. 2003, 2005, 2006, 2007, Patterson et al. 2008). In these models, vector velocity and direction dictates movement from one point to the next. By using more complex correlated random walk models that allow the individual to switch among different behavioral patterns, we can identify how that animal is using marine habitats for different purposes (Jonsen et al. 2007). For example, a bird employing ARS foraging behaviors would be expected to show a high turn rate and little forward movement, as they try to use a limited amount of space in the ocean to maximize foraging success (Hamer et al. 2009, Pavia et al. 2010).

For a seabird species like the Northern Gannet (*Morus bassanus*), foraging behavior is also related to anthropogenic development and activity in the marine environment. Northern Gannet interactions with offshore wind energy development are hypothesized to largely occur as a result of their foraging behaviors, which include a large proportion of time spent soaring at or near the altitude of the rotor-swept zone of offshore wind turbines (Garthe et al. 2000, Langston 2010, Johnston et al. 2014). These interactions have been suggested to include collision risk during foraging (e.g., Furness et al. 2013, Johnston et al. 2014), though such has not been observed to date. However, displacement or avoidance behaviors have been observed in this species at several offshore wind energy facilities in Europe (Lindeboom et al. 2011, Vanermen et al. 2014). Being able to differentiate between foraging and other behaviors will allow better determinations of areas of potential conflict between offshore wind energy development and Northern Gannet habitat use, and could inform the siting and permitting of offshore wind energy development.

The objectives of this study are 1) to determine how Northern Gannets use marine habitat during the non-breeding season and 2) identify the environmental conditions associated with changes in an individual’s movement strategy. Using positional data from satellite transmitters deployed on Northern Gannets during the nonbreeding season, we used a behavioral state switching state-space model (SSSM) in a Bayesian modeling framework to identify when these seabirds were employing ARS foraging behavior versus transient behavior in the mid-Atlantic U.S. This analysis provides information on locations that are consistently used by Northern Gannets for foraging, and the habitat characteristics (e.g., water depth, SST front density) that define these foraging areas. Understanding marine habitat most important to foraging Northern Gannets will provide key conservation information for the species, and better inform species distribution models as well as marine spatial planning.

## Methods

Satellite transmitters were deployed on Northern Gannets in the winters of 2012-14 at several locations in the mid-Atlantic U.S., as well as in autumn 2012 at a breeding colony in Canada (Chapter 22). We include a third year of data (2014 deployments) that are not included in Chapter 22, which were funded by the Bureau of Ocean Energy Management (BOEM) as part of the project, *Determining Offshore Use of Diving Bird Species in Federal Waters of the Mid-Atlantic United States Using Satellite Tracking*.

### *Satellite transmitter deployment*

We deployed 55 satellite transmitters on Northern Gannets at several locations. These techniques are described in detail in Chapter 22. Most individuals were captured on the wintering grounds in the mid-Atlantic, using a night-lighting technique in which birds were approached at night in a small vessel, briefly disoriented with a spotlight, then captured with a dip net. A metal USGS bird band was attached and morphometric measures were taken. Northern Gannets were also captured at a breeding colony in Newfoundland during the fledging period, using a telescoping noose pole. Individuals captured at this location were given a USFWS metal band.

Satellite transmitters were deployed either by external attachment or internal implantation. At the breeding colony, transmitters were attached by tape to the underside of the central retrices of nine birds (Chapter 22). On the wintering grounds, transmitters were either taped as above at the capture location (n=3 in 2011-12) or were surgically implanted in the abdominal cavity by a veterinarian onshore (n=3 in 2011-12, n=20 in 2013, n=20 in 2014). After being cleared by the veterinarian, birds were released on the water the following morning.

Two different duty cycles were used during the winter period. The tail-mounted transmitters were set to be on for 4 hours then off for 9, while the implanted transmitters were on for 4 hours and off for 13. Tail-taped transmitters were expected to not last as long as implanted devices and so we increased the data collection period per day trading finer temporal scale for decreased battery life. Satellite transmitter data were collected from the ARGOS service and included the estimated position of the individual and the quality of that positional estimate (ranked in qualitative categories). The beginning and end of winter were defined by assessing each individual track for rapid transient movements to or from the breeding grounds. Once each bird stopped making such long distance flights, the winter period was said to begin.

### ***State-space modeling***

Using the raw position data from 30 of the satellite transmitters (not all deployed transmitters collected useful data during the winter), including three tail-mounted and 31 implanted transmitters, we parameterized a SSSM to predict movement behavior and daily position Two main components of the state-space model include a first order correlated random walk with a behavioral state switching function that describes the overall movement pattern of the animal, and a positional estimation model that gives the most likely daily location for the animal. This analysis is modeled after similar work with marine mammals that was effective at determining the most likely position of the animal and identifying periods where the animal was exhibiting foraging behavior versus transiting behavior (Jonsen et al. 2007). Here we predefined two behavioral states: (1) an ARS state where spatial autocorrelation is high, turning angle is high and distance traveled is low, and (2) a transient state where spatial autocorrelation is low, turning rate is low and distance traveled is high. While more details are provided in Jonsen et al. (2007), we briefly describe how this type of model works. The movement behavior portion of the model is described by a switching correlated random walk:

$$\boldsymbol{d}_t \sim N_2[\gamma_{bt}\boldsymbol{T}(\theta_{bt})\boldsymbol{d}_{t-1}\boldsymbol{\Sigma}]$$

Where, $\mathbf{d}_t$ is the difference between locations $x_t$ and $x_{t-1}$, $\mathbf{d}_{t-1}$ is the difference between locations $x_{t-1}$ and $x_{t-2}$, $\mathbf{T}(\theta)$ is a transition matrix that provides the rotational component ($\vartheta$) required to move from $\mathbf{d}_{t-1}$ to $\mathbf{d}_t$, where $\theta$ is the mean turning angle, and $N_2$ is a bivariate Gaussian distribution with the covariance matrix $\boldsymbol{\Sigma}$, with $\gamma$ controlling variability in the autocorrelation of direction and speed. The subscript b designates which behavioral state the animal is currently using (in this case, either ARS or transient) and the subscript *t* designates the time step. This model is based on our estimates of position from this equation:

$$\boldsymbol{y}_{t,i} = (1 - j_i)x_{t-1} + j_i x_t + \boldsymbol{\varepsilon}_t$$

Where, $\mathbf{y}_{t,i}$ is the *i*th observed position during the time interval *t*-1 to *t*, $j_i$ is the fraction of time spent at the location for the *i*th observation, and $\boldsymbol{\varepsilon}_t$ is a random variable representing the error involved in estimating the positions of the individuals using Doppler-based satellite telemetry.

This model was fit using a Bayesian hierarchical framework using JAGS (Just Another Gibbs Sampler; mcmc-jags.sourceforge.net) and package 'rjags' in the R Statistical Computing Environment (R Core Team 2015). To achieve convergence, we ran the model in adaptive mode for 200,000 iterations, ran a 100,000 iteration burn-in, then analyzed the last 100,000 iterations, thinning one out of every five iterations to reduce autocorrelation, while monitoring three separate chains. We used visual evaluation and the Heidel test—a method for determining if sufficient model iterations had been run to achieve convergence for each parameter—to determine that chains were appropriately converging for each model and could be included in the next stage of analysis. In order to map the locations where individuals were exhibiting ARS or transient behavior, we mapped all points where that behavioral state was observed in greater than 75% of the model runs, then conducted a kernel density estimation (KDE) analysis in ArcGIS 10.3.1 (ESRI, Redlands, CA) to document the locations where many individuals expressed these behaviors. If the point did not meet those criteria then the behavioral state at that location was said to be uncertain. To control for the effect of variable individual sample sizes on KDE, we subsampled the data for each individual down to 20 total points (for individuals with fewer than 20 points we left their data as is) to generate an activity plot for each behavioral state over the mid-Atlantic portion of the study area.

### *Deriving environmental covariates*

Water depth data was used from standard ocean bathymetry layers and sea surface temperature (SST) using the Marine Geospatial Ecology Tools (MGET[2]) in ArcGIS 10.3.1. SST was collected from the NASA JPL satellite across the globe. For each of the daily position estimates generated by the state-space model, we related both water depth and a daily SST values.

In addition to SST, we also used the NASA JPL dataset to import 1° Centigrade Cayula-Cornillon (1992) sea surface temperature fronts and then MGET to calculate Cayula-Cornillon SST fronts (an algorithm designed to identify 1° C in SST changes over space) across the entire study area for each day of the study. To make these more useful for our analysis, we created a composite raster (with 1.2 $km^2$ grid cells) for all the temperature fronts for the seven days around the day of interest. When a raster pixel was identified as having a front for one day in the study week it was assigned a score of one; if it was a front in two days, then it was assigned a score of two, etc. Finally, an equal weight spatial smoother was used such that each raster pixel was assigned the sum total of all the front scores in an 11 by 11 grid (149 $km^2$). So, for each day, we smoothed the data over space and time to create a weekly SST front density variable that reduces the chance that strong fronts might not match perfectly with the satellite tracking data and to emphasize locations with consistent fronting.

### *Correlates to gannet behavioral state*

Using the state-space model output of behavioral state, we specified a generalized linear mixed model to determine how behavioral state was influenced by environmental factors. First, we defined when the animal was in the ARS foraging state as any point where it was identified as such in at least 75% of the JAGS model runs. This was treated as a categorical variable, where all points that were defined as ARS were given a one, and all other points (both transient and uncertain) a zero. The model also accounts for

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[2] https://code.env.duke.edu/projects/mget

individual variation in behavioral state as a random variable, which creates a repeated measures analysis. Lastly, we added year (e.g., winter of 2011-12, 2012-13, or 2013-14), capture year (a categorical variable with two designations: First Winter, which are data from the winter of capture for birds caught at the wintering grounds and the winter after capture for birds caught at the breeding grounds; and Second Winter, which only applies to birds caught at the wintering grounds are data from the next full winter after the winter of capture), water depth, and sea surface front density to the model as fixed effects. We included capture location (a categorical variable that divided individuals into one of seven broad locations of capture on both the wintering and breeding grounds), attachment type (a binary categorical variable that covers the two attachment methods), and SST (a continuous variable) into early modeling efforts, but these variables were not found to improve model fit and were removed from the final model for clarity and to increase the precision of the model parameter estimates. This analysis was conducted using package “lme4” in the R Statistical Computing Environment (Bates et al. 2014). Confidence intervals of model coefficients were calculated using the “profile” method from this package.

## Results

### *Assessment of behavioral state*

Movement models from 30 individuals passed the convergence tests and were used in subsequent analyses. Among all 30 individuals, ARS behavior occurred 69% of the time, uncertain behavior 26% of the time, and transient behavior 5% of the time. Northern Gannets appeared to be utilizing local marine resources the majority of time during winter, punctuated by periods of transient behavior in which they made linear movements to the next location (e.g., Figure 24-1). Individual Northern Gannets varied in their behavioral assessment; while the median individual showed ARS behavior 75% of the time, the first quartile was at 33%, and the third quartile at 93%. Similar variability was seen in transient behavior, with the median percentage at 5%, the first quartile at 0%, and the third quartile at 11%. In the mid-Atlantic region, we found that many individuals were in the ARS behavioral state when they were located in large bays, such as Chesapeake Bay (Figure 24-2A). Transient behavior, while still occurring in many of the bays (Figure 24-2B), was more equitably distributed, with areas on the continental shelf outside of the bays also being consistently used for this behavior.

### *Environmental correlates to foraging behavior*

The generalized linear model was a significant improvement over the null model with a random effect ($\chi^2$=32.3, $p$<0.001), with most modeling terms significantly contributing to an improved overall fit (below). The random effect of individual was important to predicting foraging behavior. Overall, the term had a variance estimate of 3.5 (SD=1.9), which suggests large variation in how often individuals utilized ARS or transient behaviors compared to the fixed effects portion of the model.

Each fixed effect was found to be both statistically and ecologically important. The effect of capture year was large, with birds in their first winter showing less ARS behavior than birds in their second winter (β=0.8, 95% CI: 0.2, 1.4; Figure 24-3). There was no significant effect of year itself once we accounted for this variable ($F_2$=1.9). As water depth increased, the probability of ARS foraging behaviors also increased (β= 0.47, 95% CI: 0.17, 0.94; median depth = 15 m and maximum depth = 862 m; Figure 24-4). The largest effect was that of SST front density, which increased the probability of Northern Gannets using ARS foraging behavior (β=0.28, 95% CI: 0.09, 0.49; Figure 24-5). There was considerable variation in the locations of SST fronts at the annual scale during our study, as relatively higher front density extended further from shore in the winter of 2012-13 compared to the other years (Figure 24-6). The correlation between SST front density and ARS behavior suggests that Northern Gannet foraging locations were also varying on an annual scale.

## Discussion

Our model indicates that Northern Gannets often engaged in ARS behaviors during winter, punctuated by transient movements to the next primary foraging area. Similar behavior has been found in Northern Gannets during the breeding season when they exhibit central place foraging (Hamer et al. 2009, Scales et al. 2014, Thiebault et al. 2014), but this is the first time that this behavior has been documented during the non-breeding season. Water depth and the density of SST fronts were both correlated with behavioral state; individuals were most likely to engage in ARS behaviors where the waters were relatively deep and frontal density was relatively high. Sea surface temperature fronts have been long known to correlate with seabird densities (Haney and McGillivary 1985), but our results suggest that Northern Gannets were not only occupying these spaces, but using them consistently for foraging opportunities. Temperature front density was a very strong predictor of foraging activity, which indicates that Northern Gannets were dynamically responding to either the change in water temperature itself, or to the increase in prey availability that is likely occurring in areas with high front density (Bost et al. 2009, Benoit-Bird and McManus 2012).

Passively moving marine species, such as plankton, are often aggregated in the frontal areas between two water masses (Bost et al. 2009), which propagates up through the pelagic community and results in greater relative abundance of higher trophic level species (Benoit-Bird and McManus 2012). Predatory fish, such as tuna, are seen in high abundance at areas of cold water intrusion into warmer waters (Andrade 2003). Capture rates for some species (Pacific Herring, *Clupea pallasii*, and Northern Anchovies, *Engraulis mordax*) have been found to be greater close to SST fronts on the U.S. Pacific coast (Reese et al. 2011). In the South Atlantic Bight, such fronts are associated with increased plankton and planktivorous birds, though forage fish are also likely to occur in such conditions (Haney and McGillivary 1985). Fronts in different locations vary in their capacity to aggregate marine biomass (Bost et al. 2009), and we lack direct evidence regarding the species that SST fronts are aggregating in the mid-Atlantic. Further research on the distribution of plankton and pelagic forage fishes in this region, particularly around temperature fronts, is warranted. During the breeding season, Northern Gannets primarily forage on near-surface schooling pelagic mackerel, capelin, sand lance, squid, saury, and herring (Montevecchi and Myers 1997). Given the size and ecology of those species, we suspect menhaden and shad constitute large proportions of gannets' diet in the non-breeding season, but little is known about their diet during this period. This uncertainty makes it even more difficult to confirm that SST fronts are aggregating prey fish for this species, though there is strong overall evidence suggesting that SST fronts in the region may act in this manner.

The results of this study suggest slightly different environmental drivers of gannet distributions than we have seen in other chapters in this report, for several reasons. The first is a matter of scale: unlike boat or aerial surveys (Parts II-IV) or other telemetry analyses (Chapters 20-23) in this report, this analysis focuses on individual foraging decisions, which are necessarily made at an individual scale, rather than a population scale. We do not attempt to describe the overall distribution of Northern Gannets, but rather to understand, within that broader distribution where the species has been observed, why individuals forage in some locations over others. Utilization distributions and survey data for Northern Gannets indicated that while they were wide-ranging across the Outer Continental Shelf (OCS), they were most common in shallower waters and closer to shore (Chapters 12, 17, and 22). This analysis suggests that, within the areas close to shore where they were abundant, Northern Gannets preferred to use ARS behaviors in areas of deeper water. This likely indicates the habitat preferences of their prey or increased competition in nearshore areas. Sea surface temperature fronts were not assessed in previous studies in this report, making it difficult to draw further comparison, though we should note that this chapter assesses front densities at a daily scale (using a sliding weekly window) while Chapter 22 looks at annual and seasonal patterns in climate data (climatologies) for inference. A third year of tracking data (from birds captured in the winter of 2013-2014) was also included in this analysis, which may have further affected the observed relationships with environmental covariates relative to the analysis in Chapter 22.

Several limitations of this study should be noted. First, we have based this analysis on data from only three winters. Given the variance described in this study, it seems likely that winter conditions could vary even more than we observed, which could influence the accuracy of our predictions. Second, our modeling methodology allowed us to assess ARS and foraging behavior at the daily scale in this study. Previous studies on Northern Gannets and other seabirds that used higher-accuracy GPS data loggers have assessed ARS at finer temporal and spatial scales. While our results have some similarity to these other studies (Hamer et al. 2009, Pavia et al. 2010), our satellite telemetry technology did not allow inference at a sub-daily scale, and any differences in ARS behavior that may be occurring at smaller time scales could not be analyzed here.

There are several potential explanations for the effect of capture year on our results. Birds could employ ARS behavior less often in their first winter as an effect of satellite tag deployment. A recent meta-analysis suggests a range of effects to birds caused by transmitter attachment, including changes in foraging duration and food consumption (Barron et al. 2010) and diving behavior in King Penguins (Ropert-Coudert 2000). Implanted transmitters have been shown to cause mortality in seabirds (reviewed in Burger and Shaffer 2008) but few studies using implants have specifically examined foraging behavior. A study on Adélie Penguins found no effect of transmitter implantation on foraging duration (Ballard et al. 2001) but this study’s relevance specifically to the Northern Gannet foraging behavior is tenuous. While we think it is unlikely that birds would respond to capture by exhibiting more transient behavior, it remains possible that this is a short-term response to capture and handling. Alternatively, the SSSM could be under-predicting rare transient behavior in the second winter, when we typically had fewer data points per individual. A third possibility is that foraging behavior may vary seasonally; winter capture efforts were centered in the middle of the winter season, so data from the winter of capture was limited to late winter periods, while data from the winter after capture were generally from early winter, before tags began to fail.

Northern Gannets spent considerable time using ARS foraging behavior in relatively deep parts of the OCS, where there are high densities of SST fronts. Given the strong relationship to SST fronts that we observed, we would expect that as the locations of SST fronts change over time, the locations of consistent foraging activity in Northern Gannets will change with them. Frontal patterns have changed during periods of climate change in the past (Chavez et al. 2003, Hoegh-Guldberg and Bruno 2010), but the scientific community is currently uncertain about how SST fronts will change specifically on the mid-Atlantic OCS in the coming years or decades. The variation could look similar to differences between El Niño and La Niña years that are currently observed, which would make predictions of future changes relatively simple. However, the possibility of generating no-analog communities—and their difficult to predict conditions—also exists (Williams and Jackson 2007). Under this scenario, climate change modifies environmentally conditions in an ecosystem to an arrangement that has not been documented by science, thus making ecosystem difficult to impossible to predict.

Offshore wind energy development could potentially affect Northern Gannets in several ways: by increasing mortality due to direct collision, displacing them from high-quality foraging areas, or increasing energy expenditures through added flight costs to avoid turbines (Fox et al. 2006, Furness et al. 2013, Johnston et al. 2014, Vanermen et al. 2014). Foraging birds are suggested to be at greater risk to collision with wind turbines (Furness et al. 2013). Thus, for assessing the potential for both collisions and displacement of Northern Gannets in relation to offshore wind energy development, it will be important to understand the locations of foraging areas, and to develop accurate predictions of SST fronts, both now and in the future, when changing ocean temperatures could alter the distribution of seabird foraging locations.

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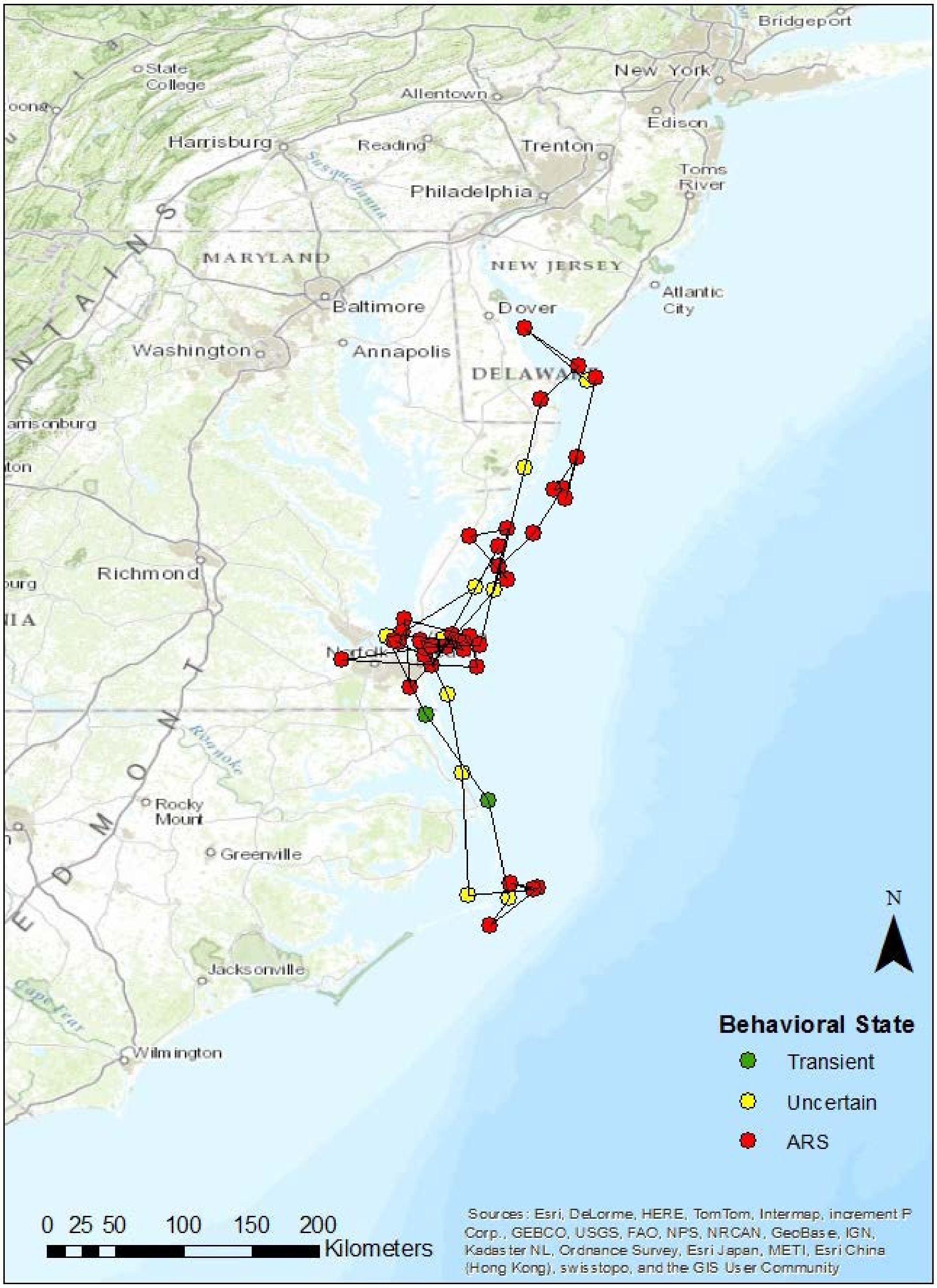
## Figures



Sources: Esri, DeLorme, HERE, TomTom, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, and the GIS User Community

**Figure 24-1. Examples of ARS and transient movements from a Northern Gannet with a satellite transmitter.** Points represent the most likely daily position of one Northern Gannet estimated by the state-space model, and the lines connect points sequentially in time. The color of the point indicates what type of behavior was occurring at that point: transient behavior (green), uncertain (yellow), or area-restricted search behaviors (red).

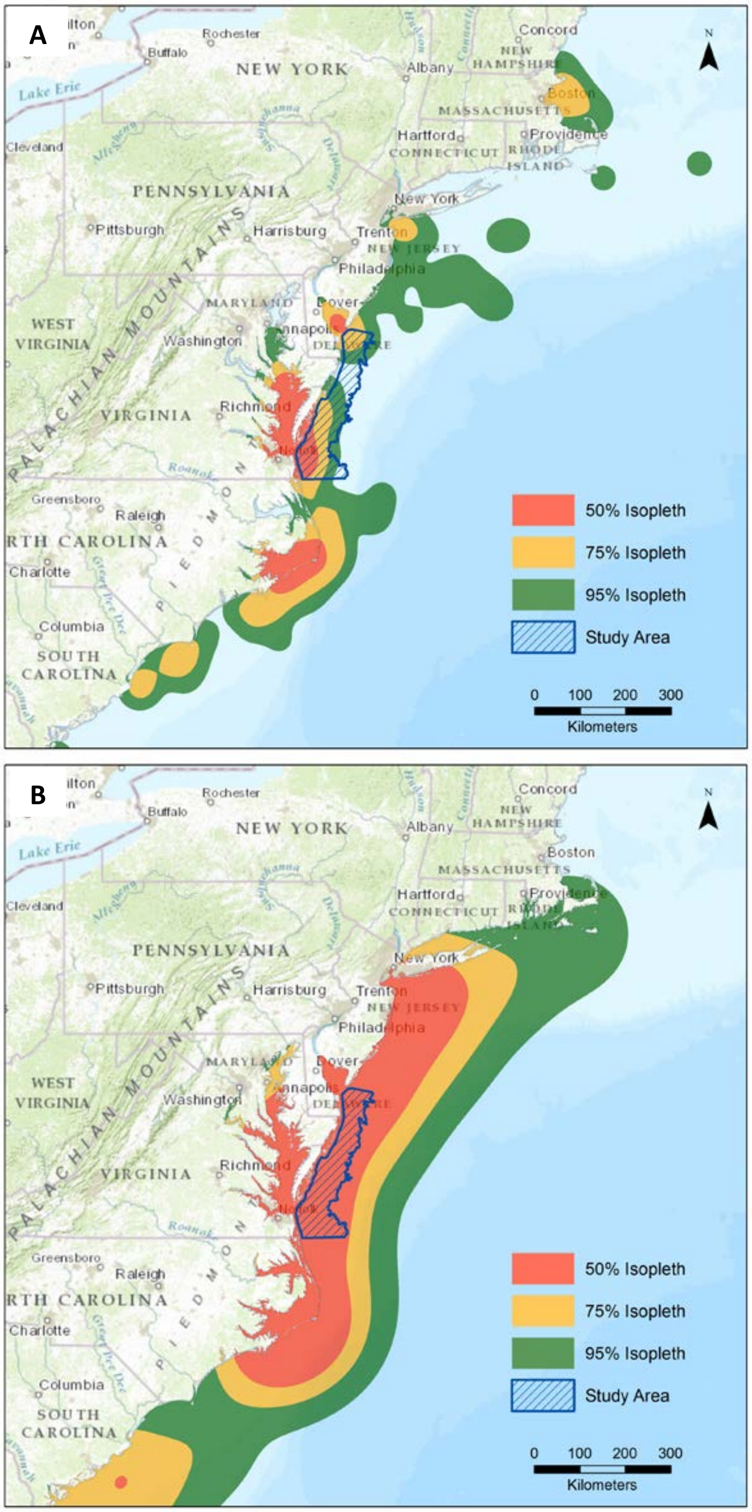


**Figure 24-2. Kernel Density Estimates of Northern Gannets exhibiting area-restricted search (ARS) foraging behavior (A) and transient behavior (B) in the mid-Atlantic.** Redder colors (and high density levels) indicate that more individuals in that area are exhibiting the respective behavioral state. This analysis sums activity across years and individuals.

**Figure 24-3. The predicted effect of capture year on the probability of Northern Gannets being in an ARS state during the winter they were captured (First Winter) and the winter immediately after (Second Winter).** Individuals demonstrated a significantly higher ratio of ARS to non-ARS behaviors in the second winter. Error bars indicate the 95% confidence interval of the mean.

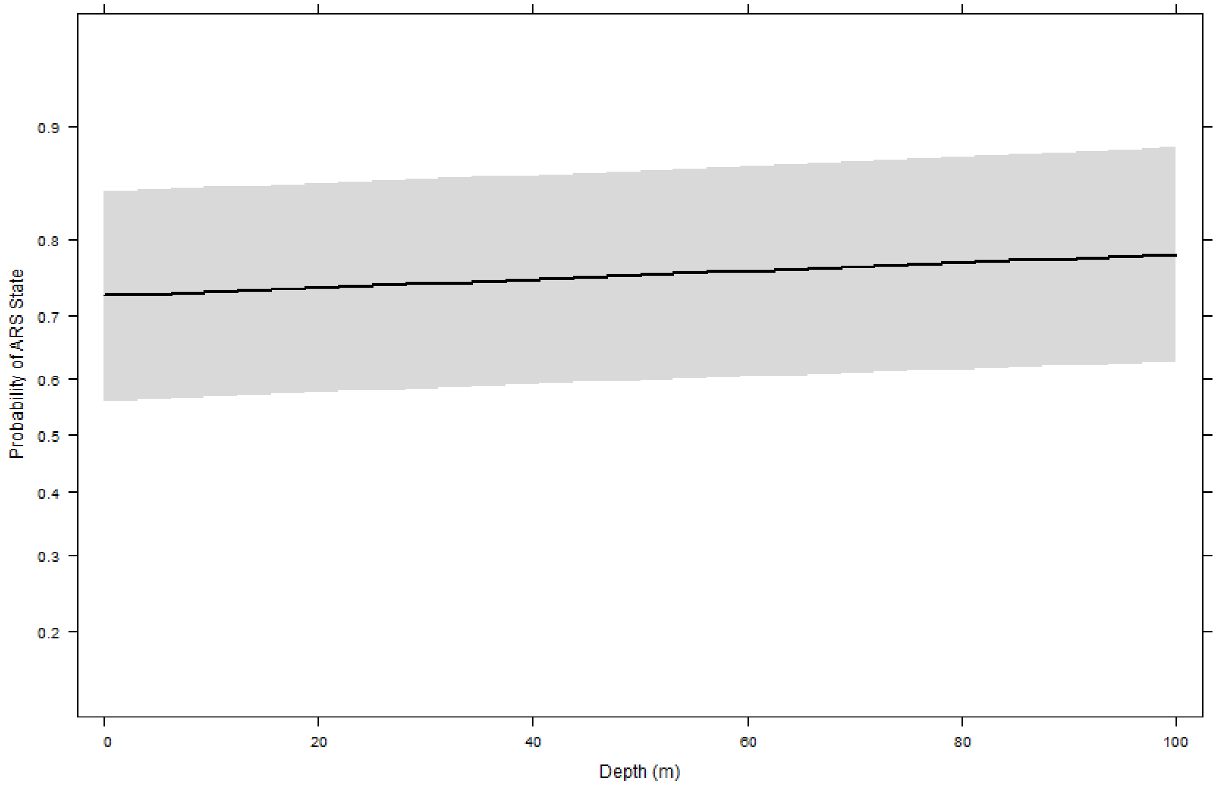


**Figure 24-4. The predicted effect of water depth (m) on the probability of Northern Gannets being in an ARS state during winter (across all years and individuals).** The shaded area indicates the 95% confidence interval of the prediction.

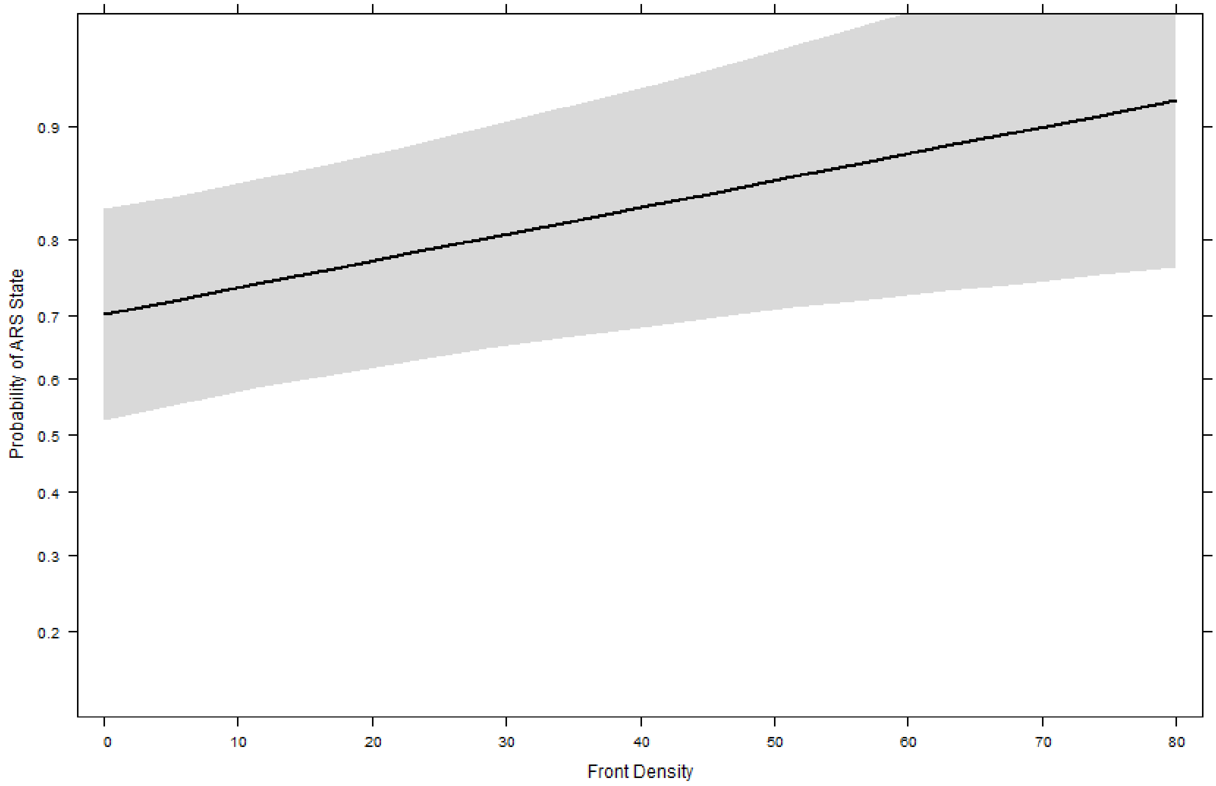


**Figure 24-5. The effect of sea surface temperature front density (number of fronts for each position estimate) on the probability of Northern Gannets being in ARS state during winter (across years and individuals).** The shaded area indicates the 95% confidence interval of the prediction.

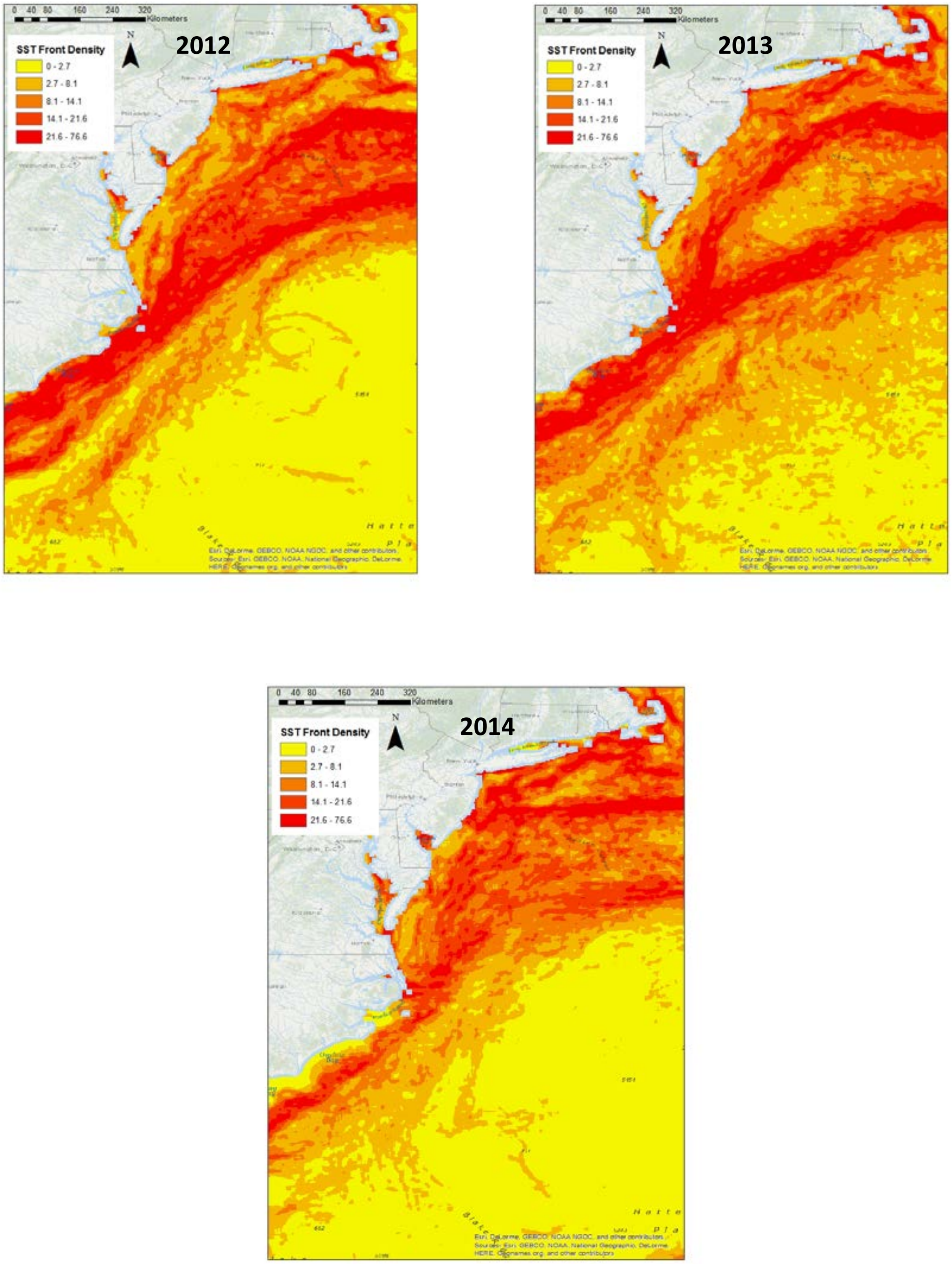


**Figure 24-6. Maps of daily SST front density averaged for three winters (2012-14) in the mid-Atlantic region.** Color shifts from yellow to red as the number sea surface temperature fronts in the past week around each point increases.

# Chapter 25: Offshore migration of Peregrine Falcons (*Falco peregrinus*) along the Atlantic Flyway

### Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 25 Highlights

*Characterizing fall migration patterns of Peregrine Falcons using the Atlantic flyway*

### Context[1]

Numerous studies have characterized risks that wind turbines pose to raptors in terrestrial settings, and Peregrine Falcon (*Falco peregrinus*; peregrine hereafter) fatalities have been documented at terrestrial wind facilities. As peregrines feed and migrate offshore, they may also encounter offshore wind energy developments. While many survey techniques used for raptors in terrestrial studies are not feasible in the offshore environment, animal tracking technologies are increasingly improving the ability of researchers to document long-distance movement and behavior patterns of migratory animals.

We used satellite telemetry to document space use for fall migrant peregrines to improve our understanding of peregrine ecology and to evaluate potential exposure to offshore development. We used a Dynamic Brownian Bridge Movement Model to develop Utilization Distributions (UDs) for individuals, and compared UD overlap with the mid-Atlantic Outer Continental Shelf study area. This modelling technique, while data-intensive, improves upon traditional (i.e., fixed kernel) approaches used in Chapters 20-22, because it accounts for the order in which locations were fixed, the time interval between them, and location error, and thus generates space use estimations that more accurately depict high and low use areas and movement corridors.

### Study goal/objectives

Evaluate potential exposure of peregrines to offshore development by characterizing the migration of peregrines along the Atlantic U.S. coast, including peregrine use of the mid-Atlantic study area.

### Highlights

- Peregrines commonly ventured into offshore habitats throughout the Atlantic Flyway.
- Peregrine migration routes were more concentrated in the northern portion of the U.S. Atlantic flyway (RI to NC) compared to the southern portion (NC southward), at which point migration routes were more diffuse.
- The majority of peregrines initiated transoceanic flights from in North Carolina.
- Proportions of peregrine UDs falling in the mid-Atlantic study area ranged from 0 – 59% (mean ± SD: 21 ± 21%). Peregrines varied widely in their probabilistic use of the mid-Atlantic study area.
- Peregrine locations recorded over water were at significantly higher altitudes than over land.

### Implications

Peregrines from a broad geographic range use the Atlantic flyway during fall migration, and study findings suggest they use offshore habitats regularly. However, the proportion of the population that passes through with the mid-Atlantic study area, the frequency with which individuals might encounter turbines, and the behavioral responses of peregrines to offshore turbines remain unknown.

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[1] For more detailed context for this chapter, please see the introduction to Part V of this report.

## Abstract

Peregrine Falcons (*Falco peregrinus*; peregrines hereafter) are among a limited number of raptors capable of enduring extended journeys over open water. This characteristic may increase their exposure to offshore wind energy facilities being considered for development in both state and federal waters along the Atlantic Flyway. We fitted 16 migrant peregrines with satellite transmitters in Maine and Rhode Island to characterize their migration patterns, and to evaluate their space use relative to the mid-Atlantic Outer Continental Shelf study area (mid-Atlantic study area hereafter) containing Delaware, Maryland, and Virginia Wind Energy Areas (WEAs). Peregrine tracks were more concentrated in the northern portion of the U.S. Atlantic Flyway (Block Island, RI, to Cape Hatteras, NC) compared to the southern portion (Cape Hatteras, NC southward), at which point southbound peregrines migrated over a broader front. The 120-km stretch of shoreline spanning between Cape Lookout and Cape Hatteras, North Carolina was significant to the migratory ecology of peregrines along the Atlantic flyway: nearly all peregrines initiated transoceanic flights from this stretch of coastline. Proportions of peregrine utilization distributions (UDs) coinciding with the mid-Atlantic study area ranged from 0 – 59% (mean ± SD 21 ± 21%). Thirty-three percent of peregrines entering the mid-Atlantic study area had <10% of their UDs in the study area, while remaining individuals generally: (a) travelled down the Delmarva Peninsula and crossed over the southern portion of the mid-Atlantic study area, (b) ventured, often repeatedly, from the Atlantic coastline into the mid-Atlantic study area and returned, or (c) chose an offshore migration route either within or to the east of the study area. Peregrines spent between <1 d and 14 d in the latitudinal zone of the mid-Atlantic study area, and roughly half of this time (mean ± SD: 56 ± 34%) was spent over water vs. over land. Altitude estimates from two instrumented peregrines indicated peregrines flew significantly higher when over water compared to when they were over land. Peregrine flight altitudes suggested peregrines fly above, within, and below the rotor sweep area when offshore. Findings from this study emphasize the close association between peregrines and open water habitats during migration. Peregrines may have increased collision risks with offshore turbines if they are attracted to them for perching and foraging, as has been observed for other lighted structures (i.e., oil drilling platforms, offshore vessels) or if they encounter them during times of limited visibility.

## Introduction

Peregrine Falcons (*Falco peregrinus;* peregrines hereafter) are one of the most widely distributed raptors worldwide (White et al. 2013). Due primarily to adverse effects of persistent synthetic chlorinated hydrocarbons (DDT and others) on reproduction, peregrine populations declined precipitously throughout large portions of their range in the mid-20th century (Ratcliffe 1980, Cade et al. 1988, White et al. 2002). Populations residing in the eastern U.S. (*F. p. anatum*) were fully extirpated, while notable declines were observed in migratory *F. p. anatum* and *F. p. tundrius* from arctic and subarctic regions (Kiff 1988, Henny et al. 2009). Federal protection under the Endangered Species Act (transferred from the Federal Endangered Species Conservation Act of 1969) the banning of DDT, and unprecedented population reintroduction and recovery efforts were largely successful in reestablishing breeding populations throughout much of their former range, prompting removal of *F. p. anatum* from the U.S. Federal Endangered Species List in 1999 (Cade and Burnham 2003, USFWS 2003). Resident Peregrine populations continue to recover but remain listed on threatened and endangered species lists in many eastern U.S. states. Data from long-term hawk count migration stations along the Atlantic

Flyway, a major migration corridor for peregrines, has shown increasing population trends in migrating peregrines (Farmer et al. 2008), and notable declines in DDT exposure have been observed (Henny et al. 1996, 2009). Fall migrant peregrines using the Atlantic Flyway likely represent a combination of reestablished local residents (formerly comprised by the *anatum* subspecies) and far greater numbers of peregrines originating from arctic regions across North America and Greenland (primarily comprised of the *tundrius* subspecies; White et al. 2013).

Interest in renewable energy development has increased in the United States and internationally over the last several decades. State and federal mandates for generating a portion of energy production from renewable resources have encouraged the development of offshore wind energy technologies in the U.S. Wind energy facilities are associated with positive environmental benefits, particularly lessening greenhouse gas emissions associated with fossil fuel combustion. In 2010, the U.S. Department of the Interior launched the "Smart from the Start" initiative to promote the siting and construction of wind energy projects on the Atlantic Outer Continental Shelf, an area under federal jurisdiction (>3 nautical miles from shore; USDOE and USDOI 2011). The Bureau of Ocean Energy Management designated Wind Energy Areas (WEAs) in specific locations along the Atlantic coast to minimize conflicts with other marine uses such as shipping and naval activities. Four mid-Atlantic WEAs off the coasts of New Jersey, Delaware, Maryland, and Virginia were the first proposed areas for potential development under this initiative. Other offshore lease and planning areas are in various stages of development in both state and federal waters along the Atlantic coast (BOEM 2015*a*). Completion of an offshore wind energy facility off the coast of Block Island, Rhode Island is anticipated in 2017, which would make it the first operational offshore wind project in the U.S.

Wind energy facilities can have adverse effects on birds, particularly if they are located in areas of high bird use; however, proper citing of facilities through preconstruction planning can reduce risks to individuals (Langston and Pullan 2003, Smallwood and Thelander 2008, Smallwood 2013, Miller et al. 2014). The extent to which offshore wind energy facilities might impact wildlife populations remains poorly understood for many species. Wind energy facilities are associated with numerous risks to bird populations including those related to collision, increased energy expenditure due to avoidance behaviors, and displacement from foraging areas (Langston and Pullan 2003, Chamberlain et al. 2006, Drewitt and Langston 2006, Fox et al. 2006). The potential for adverse effects of wind turbines on birds have prompted several efforts to develop species vulnerability assessments to be used in future wind energy planning efforts and risk evaluations (Garthe and Hüppop 2004, Watts 2010, Furness et al. 2013, Willmott et al. 2013). These studies have focused almost exclusively on waterbirds, with the exception of Willmott et al. (2013), whose authors largely discounted risk for migratory landbirds on the assumption that they spend minimal time over the Atlantic Outer Continental Shelf.

Negative impacts on survivorship, particularly adult survival, can have significant and long-term consequences on the stability of raptor populations, particularly for long-lived or endangered species (Newton 1979, Grier 1980, Carrete et al. 2009). Thus, further investigations of the potential risks offshore wind facilities may pose to raptors are warranted. The primary negative effects of wind turbines associated with raptors in terrestrial settings are related to displacement (primarily focusing on foraging raptors) and collision (Madders and Whitfield 2006, De Lucas et al. 2008, Smallwood and

Thelander 2008, Garvin et al. 2011). A study in Spain suggests collision fatalities are not related to raptor abundance (De Lucas et al. 2008). Topography, species-specific flight behaviors, turbine height and local weather patterns have been found to influence raptor collision risks with wind turbines (De Lucas et al. 2008, Garvin et al. 2011). Compared to other raptors such as Golden Eagles (*Aquila chrysaetos*) and Red-tailed Hawks (*Buteo jamaicensis;* Smallwood and Thelander 2008), reports of peregrine collisions with wind turbines in terrestrial settings are relatively uncommon. Peregrine fatalities have been documented at multiple terrestrial wind facilities in Europe and the United States (Meek et al. 1993, Hötker et al. 2006, Mizrahi et al. 2009, Dürr 2011). To date, peregrine fatality risks are difficult to assess in marine settings because most survey techniques used in terrestrial settings are not feasible in the offshore environment. Peregrines are known to travel over water during migration, and large numbers of individuals may commonly use some migratory corridors offshore.

Rapid advances in the field of animal tracking technologies over the last two decades are increasingly improving researchers' abilities to document movement and behavior patterns of widely roaming species (Seegar et al. 1996, Walls and Kenward 2007, Kie et al. 2010, Lanzone et al. 2012). Several tracking technologies including satellite telemetry and high resolution GPS telemetry now enable animals to be tracked at a global scale for consecutive years (Mellone 2013, Sokolov et al. 2014, Watson et al. 2014). In parallel to the rapid advances in the field of animal tracking, our ability to model animal movements has notably improved in recent years (Kie et al. 2010, Fischer et al. 2013). Traditional home range estimation methods such as fixed kernel approaches have limitations because they consider each animal location independently and they do not generate utilization distributions based upon movement paths. In 2007, Horne et al. introduced the Brownian Bridge Movement Model (BBMM) to model animal movements, and the approach has since been widely used (Farmer et al. 2010, Fischer et al. 2013, Watts et al. 2015). The BBMM improves upon traditional approaches because it does not consider location estimates independently, it incorporates location accuracy information, and it employs a probabilistic estimation of an animal's path from data recorded at intervals (Fischer et al. 2013). Recently, Kranstauber et al. (2012) proposed the 'Dynamic Brownian Bridge' approach to modelling animal movements with Brownian Bridges, which allowed the Brownian movement variance to vary with time and space.

Unlike the majority of non-piscivorous North American raptors, peregrines have a close association with aquatic habitats, particularly marine ecosystems. Long, pointed wings and favorable wing loading characteristics enable peregrines to use of powered flight to cross large expanses of open water where thermal formation is generally poor (Kerlinger 1985, Newton 2008). As a result, peregrines are commonly observed foraging or perching far from shore at offshore islands, oil drilling platforms, and large offshore vessels (Voous 1961, Cochran 1975, 1985, Russell 2005, Johnson et al. 2011, DeSorbo et al. 2012). Limited efforts to track migrant peregrines along the Atlantic coastline have documented that peregrines may commonly use offshore habitats, and that they are capable of flying for several consecutive days across large expanses of open water (Cochran 1975, Fuller et al. 1998, Desorbo et al. 2012). Peregrines are possibly the most commonly observed raptor flying in inclement weather, and several accounts have also documented nocturnal soaring (Enderson 1965, Cochran 1975, Russell 1991) and nocturnal foraging on birds disoriented by lights on offshore structures (Johnson et al. 2011). These cumulative observations raise concerns that peregrines may have elevated collision risks with offshore wind turbines, particularly if they are attracted to them for perching or foraging.

In this study, we instrumented peregrines with satellite transmitters to: (1) characterize their migration patterns along the Atlantic flyway, and (2) evaluate potential for exposure to offshore wind development by characterizing use of the mid-Atlantic Outer Continental Shelf study area and the three WEAs (DE, MD, VA) within it.

## Methods

### *Study area*

We characterized the movements of peregrines at two spatial scales (Figure 25-1): (1) the Atlantic coast, spanning from Block Island, RI (our primary deployment site) to the southern tip of Florida, and (2) the mid-Atlantic Outer Continental Shelf study area (mid-Atlantic study area hereafter) and the three WEAs within it (Figure 25-1).

### *Peregrine capture, PTT instrumentation and programming*

We captured migrant peregrines at two demonstrated fall migration stopover sites: (1) Monhegan Island, 16 km off the mid-coast of Maine (2010), and (2) Block Island, 21 km off the southern Rhode Island coast (2012-2014). Block Island was chosen as our primary capture site because it was presumed to be a peregrine migration stopover site, was logistically attractive, and it is approximately 350 km north of the northern periphery of the mid-Atlantic study area.

Migrant peregrines were captured using standard methodologies in which dho gaza nets, mist nets, and bow nets were arranged around lures (Hull and Bloom 2001). Upon capture, we removed peregrines from traps, banded them using U.S. Geological Survey (USGS) leg bands, and collected standard morphometric data (flat wing cord, tail, culmen, body mass) following standard protocols. We attached satellite transmitters (Platform Transmitter Terminals, or PTTs) to a subset of captured individuals, prioritizing peregrines that were visibly healthy and heavier, such that the PTT package remained ≤3.5% of bird body mass. Transmitters were instrumented to individuals with a backpack-style harness made of 0.63 mm (0.25 in) Teflon® ribbon (Bally Ribbon Mills, Bally, PA) sewn with Teflon® thread through a Teflon® kangaroo leather patch centered on the breast (Kenward 2001, Steenhof et al. 2006, Walls and Kenward 2007, Fair et al. 2010).

We used the Argos satellite system to track peregrine movements (CLS 2015). We used two types of satellite transmitters (manufactured by North Star Science and Technology, King George, VA). Nine female peregrines (7 captured on Block Island, 2 captured on Monhegan Island) were instrumented with 22 g solar GPS PTTs programmed to fix 4 (Monhegan) to 11 (Block Island) GPS locations distributed evenly throughout daylight hours, plus one location at midnight EDT (04:00 GMT). Additional locations were estimated by Doppler-shift (Argos locations hereafter) during 8-hour transmission cycles following a 30-hour off cycle. Two females and six males were instrumented with 12 g solar PTTs programmed to fix Argos locations during all daylight hours as unit charging permitted.

Argos locations were estimated by CLS America (CLS) using the least squares method. Argos locations are classified by CLS into seven location classes (LCs), generally associated with increasing levels of accuracy: Z, B, A, 0, 1, 2 and 3 (CLS 2015). Given that low-quality Argos locations (LCs Z, B, A, and 0) are often associated with high errors, a filtering strategy is commonly adopted to exclude outliers. We used the merged minimum-redundant-distance and distance-angle-rate tests within the Douglas-Argos Filter (DAF) to remove implausible locations from our dataset (Douglas et al. 2012). The user defined MAXREDUN parameter within the DAF has a strong influence on the extent to which locations are excluded or retained. A lower MAXREDUN setting (i.e., 5 km) produces outputs with higher overall accuracy, but more locations are excluded from analysis. Conversely, a higher MAXREDUN setting (i.e., 15 km) retains more locations, with less stringent requirements on overall location accuracy. We used moderately conservative filtering criteria (MAXREDUN = 10 km) to remove implausible low-quality Argos locations from our peregrine data. Given our filtering parameter settings, 68th percentile location error estimates for each low-quality post-filtered Argos LC have been reported as (in km): Z (4.7), B (7.6), A (4.1), 0 (6.8; Douglas et al. 2012). Location errors reported by Douglas et al. for unfiltered LC 1, 2, and 3 locations were 2.5, 1.0, and 0.4 km, respectively (see also CLS 2015). Accuracy of GPS locations fixed by units used in this study generally range between 5 – 15 m (K. LeSage, Geotrak, Inc., pers. comm.). Rare instances of GPS locations with large errors were identified by implausibly high movement rates, and were excluded.

### *Characterizing movements of peregrines along the Atlantic coast*

To characterize the general flight paths of fall migrant peregrines using the U.S. Atlantic flyway, we mapped fall and spring tracks of individuals between Block Island, Rhode Island to southern Florida and qualitatively described migration patterns. We characterized the distance fall migrant peregrines travelled from shore during migration by calculating the distance between each location estimate and the Atlantic shoreline along the U.S. Atlantic coast. The Euclidean distance (m) was calculated between overwater peregrine location estimates and the nearest segment of the NOAA Medium Resolution Digital Vector Shoreline (1:70,000; NOAA 2014) using the Near Tool in ArcMap 10.2.2 (ESRI 2013). We then calculated the mean daily distance that each individual travelled from shore. Based on observations that Cape Hatteras, NC was significant geographically to the distance migrants travelled from shore, we compared distance measures between individuals in two broad geographic regions: Block Island to Cape Hatteras, and Cape Hatteras to the southern tip of Florida.

#### *Altitude*

Two GPS transmitters deployed in 2013 were programmed to collect altitude data. The accuracy of altitude estimates generated by PTTs vary with each satellite fix according to numerous factors. Altitude error for units used in this study typically range from 10 – 30 m (K. LeSage, Geotrak, Inc., pers. com.). To provide perspectives on peregrine flight altitude relative to the height of offshore wind turbines, we categorized location estimates between Block Island and the southern tip of Florida as either over land or over water, and then characterized flight heights into the following generalized rotor height categories: (1) < 20 m; below the rotor swept zone; (2) 20 – 200 m; within the rotor swept zone, and (3) >200 m; above the rotor swept zone. The rotor swept zone of offshore wind turbines varies with manufacturer, turbine type and tides. Rotor sweep zone categories used in our analyses are based on those used in Willmott et al. (2013), considered to cover a variety of possible turbine types and tidal effects.

### *Peregrine use of the mid-Atlantic study area and WEAs*

To characterize movements of fall migrant peregrines within the mid-Atlantic study area, we used a dynamic Brownian Bridge Movement Model (dBBMM; Horne et al. 2007, Kranstauber et al. 2012) to generate individual utilization distributions (UDs; Worton 1989) for the 15 instrumented peregrines that crossed the northern latitude of the mid-Atlantic study area. Traditional approaches to generating UDs (i.e., fixed Kernel methods) are limited because they do not account for the order in which location estimates were fixed, the time interval between them, or location error. The dBBMM accounts for these factors, and thus generates UDs that are more accurate in depicting high and low use areas and identifying migratory corridors (Kernohan et al. 2001, Kie et al. 2010, Kranstauber et al. 2012, Fischer et al. 2013). We quantified space use of fall migrant peregrines relative to potential wind energy development within the study area by calculating the proportion of each animal's UD that intersected the mid-Atlantic study area boundary and the Delaware, Maryland and Virginia WEA boundaries. Since we were only interested in relative use as peregrines pass through the study area in the fall, we limited our analysis of the UDs to the approximate northern (38.8940°) and southern (36.5461°) latitudinal boundaries of the mid-Atlantic study area (Figure 25-1).

#### *Time spent over water vs. over land*

To estimate the amount of time peregrines spent between the northern and southern latitudinal boundaries of the mid-Atlantic study area, we input peregrine location data between Block Island, RI and Cape Hatteras, NC, into a Continuous-time Correlated Random Walk Model (CTRCW) developed for animal telemetry data (Johnson et al. 2008). A primary function of the CTRCW model is to convert telemetry data, typically collected at irregular intervals, into a time-series of location estimates that are uniformly spaced in time. We parameterized the CTRCW model using the 68% error percentiles presented in Douglas et al. (2012) for location accuracy of Argos locations, and 28 m for GPS location accuracy. We provisioned the CTRCW to predict locations for each peregrine migration at 1-hr intervals. We then categorized the predicted locations as either 'over water' or 'over land' by noting location relative to the NOAA Medium Resolution Digital Vector Shoreline (1:70,000; NOAA 2014). Times were calculated for each peregrine track using predicted locations closest to the respective point of entry and exit for each track. We considered this approach to be an improvement over using raw PTT data, which occasionally harbors large time intervals between 'raw' locations for some individuals. We considered two consecutive annual fall migration tracks for one peregrine (HYF02) independent in analyses.

### *Data analysis*

We used the package 'move' (7 July, 2015) in R (R Core Team 2014)to calculate dBBMM probabilities and contours. We used the package 'crawl' (19 February, 2015) in R to run the Continuous-time Correlated Random Walk model used to estimate the amount of time individuals spent within the latitudinal boundaries of the mid-Atlantic study area to estimate time individuals spent over land vs. over water. Animal movements were mapped using ArcMap 10.2.2 (ESRI 2013). Animal UDs were summarized using the 'summarize by zones' tool in ArcMap 10.2.2. Data summaries were performed in JMP 9.0 (SAS 2010).

## Results

### *Peregrine captures and PTT deployments*

We captured 157 peregrines on Block Island and 35 on Monhegan Island. Of these, the vast majority (99%) were young of the year (hatching year; HY hereafter). We instrumented 14 peregrines with satellite transmitters on Block Island and two peregrines on Monhegan Island. Thus, data from 16 instrumented peregrines were available for analysis. Of these individuals, two were adult females, six were HY males, and eight were HY females.

### *Characterizing movements of peregrines along the Atlantic coast*

After filtering, a total of 3,044 location estimates fixed between Block Island, RI and southern Florida remained in our dataset. Sixty percent of these locations (n = 1,814) were Argos locations, while the remaining 40% (n = 1,230) were GPS locations. The higher proportion of Argos locations was due to the use of non-GPS 12g solar transmitters on six males and two females. Argos location estimates in location classes 1 – 3 comprised 22% of all locations received, while 'lower-quality' location classes (0, A, B, Z; Douglas et al. 2012, CLS 2014) comprised the remaining 38% of all locations.

The majority of individuals instrumented with satellite transmitters (87%; 14 of 16) migrated southward following departure from capture sites (Figure 25-2). One exception to this pattern, HYM02, travelled approximately 1,500 km eastwards before travelling another 1,300 km south and then returned to the general vicinity of the capture site prior to travelling another 900 km south to a location >200 km offshore east of Pamlico Sound, NC where it ceased transmitting (Figure 25-3). This individual is considered an outlier, and was presumed to be perching on offshore vessels intermittently during this period based on observed daily travel rates consistent with the speed of offshore vessels. The second individual that did not travel southwards following PTT instrumentation was an adult female, ADF02, that overwintered on Block Island after capture and then migrated to Greenland the subsequent spring. One individual (HYM05), migrated as far south as the VA / NC state border, before reversing direction and heading 170 km north to Assateague Island (a barrier island in coastal MD and VA; a well-known stopover for migrant peregrines; Seegar et al. 2012), where it was recovered near the base of a Bald Eagle (*Haliaeetus leucocephalus*) nest. In general, however, peregrines exhibited more typical north-south migratory movements. We obtained a single fall migration dataset for most birds. We did record two fall migration tracks from one HY female (HYF02) that migrated from the Hudson Strait during its second fall, destined for a second winter in the Bahamas.

Our sample of instrumented peregrines suggested that the 530 km stretch of coastline between Cape Charles, VA and Cape Fear, NC is of strategic importance to peregrines using the Atlantic flyway. Instrumented peregrines reaching this shoreline zone in the fall either: (a) initiated a significant transoceanic flight, or (b) continued along the Atlantic coastline to Florida. Of thirteen peregrines continuing migration beyond the Mid-Atlantic coast, 92% (12 of 13; all HYs) initiated transoceanic flights from locations between Cape Charles, VA and Cape Fear, NC (Figure 25-2). Of these 12 peregrines, 69% (6 HY females, 3 HY males) departed from points along the 120 km stretch of coastline between Cape Hatteras and Cape Lookout (5 from Cape Lookout, 3 from Cape Hatteras; Figure 25-4 to Figure 25-8). One individual, HYF02, was tracked during two consecutive fall migration seasons, departed from the

Cape Fear area of NC during her first migration, and Ocracoke Island, NC during her second journey (Figure 25-4). The only individual that did not venture on a transoceanic flight was an adult female, ADF01, who continued down the Florida coastline (Figure 25-6). Excluding ADF02 that did not migrate, 46% (7 of 15) instrumented individuals stopped at Assateague Island along the coast of Maryland and Virginia.

*Distance to shore*

Excluding outliers HYM02 and ADF02, individual pooled overwater locations between Block Island and Cape Hatteras ranged from <1 to 1,495 km from shore (mean ± SD; 67 ± 196, n = 1,153). Individual pooled locations between Cape Hatteras and southern Florida ranged from <1 to 2,081 km from shore (271 ± 376 km, n = 376). Individuals had mean daily distances from shore of <1 to 299 km (30 ± 78 km, n = 14) in the northern region and 16 – 976 km (228 ± 262 km, n = 13) in the southern region.

Within the latitudes of the mid-Atlantic study area, individual pooled overwater location estimates ranged from <1 to 222 km from shore (n = 311). The mean daily distance peregrines in this subgroup travelled from shore ranged from <1 km – 57 km (16 ± 19 km, n = 14). The inclusion of the offshore-dwelling HYM02 had a strong influence on these measures (range: <1 km – 790 km, mean ± SD: 68 ± 200, n = 15).

*Altitude*

Two instrumented Peregrine Falcons acquired 1,884 total altitude estimates between October 2013 and June 2014. In total, 1,642 points were over land and ranged from 0.3 – 4469.2 m (mean ± SD: 105.5 ± 265.6) and 242 points were over water, which were significantly greater than land points which ranged from 0.3 – 4811.8 m (mean ± SD: 390.5 ± 747.9; $\chi^2 = 76.5$, $p < 0.0001$). The error of altitude estimates ranged from 4.9 – 85.8 m (mean ± SD: 20.4 ± 13.8 m). All altitude estimates between Block Island, RI and southern Florida (n = 349) occurred in October 2013. Fifty percent (n = 175) of altitude estimates within this area were fixed over water off the Atlantic coast, while the remaining 50% (n = 174) were fixed over land. Peregrine HYF06 ranged in altitude from 0.03 – 465.9 m (mean ± SD: 55.9 ± 100.3, n = 60) while over land along the coast between Block Island and southern Florida and ranged between 0.33 – 3313.9 m (mean ± SD: 338.7 ± 705.9, n = 106) while over water. Peregrine HYF07 ranged in altitude from 0.24 – 783.6 m (mean ± SD: 77.6 ± 152.1 m, n = 114) over land and between 4.1 – 1890.9 m (mean ± SD: 361.9 ± 337.8, n = 69) while over water. No significant differences in altitude were observed between the two birds for points occurring over land between Block Island and southern Florida; however, altitude estimates for HYF07 were significantly greater than HYF06 for points over water in this area ($\chi^2 = 22.5$, $p < 0.0001$). Forty percent of points that occurred over water between Block Island and Florida ( n = 242) were located above 200 m (Figure 25-9, Figure 25-10). Thirty-one and 29% of estimates occurred within the 0 - 20 m and 20 – 200 m zone, respectively.

*Spring migrants*

Of the 16 peregrines instrumented in this study, three provided information on spring migration routes. One of these three migrants (HYF02) used the Atlantic coast during spring migration and provided information about spring migration patterns relative to the mid-Atlantic study area (Figure 25-4). The other two spring migrants used overland migration routes from Florida to breeding sites in Manitoba and Saskatchewan, Canada.

### *Peregrine use of the mid-Atlantic study area and WEAs*

In total, 586 location estimates fell between the northern and southern latitudinal boundaries of the mid-Atlantic study area. GPS locations comprised 53% of location estimates in this area. Estimates in location classes 1 – 3 comprised 22% of all locations, while low quality location classes (0, A, B, Z) comprised the remaining 27% of locations. Of the 16 peregrines instrumented in this study, all but one (ADF02) reached the northern latitude of the mid-Atlantic study area (Figure 25-4 to Figure 25-8). One peregrine, HYF02, completed two fall migrations through the Mid-Atlantic study area.

Migrant peregrines displayed widely varying use patterns within the mid-Atlantic study area. Of the 15 peregrines, the proportions of their UDs falling in the mid-Atlantic study area ranged from 0 – 59% (mean ± SD 21 ± 21%; Table 25-1). Only one peregrine, HYM02, had a UD entirely outside (to the east) of the study area. However, 33% (5 out of 15 individuals) of instrumented peregrines entering the mid-Atlantic study area had <10% of their UDs in the study area. In general, peregrines with <10% of their UDs in the study area (i.e., HYF03 – HYF05, ADF01; Figure 25-5 to Figure 25-6) appeared to be following the Atlantic shoreline during migration. The remainder of individuals can be loosely characterized as: (a) travelling down the Delmarva Peninsula and crossing over the southern portion of the mid-Atlantic study area (i.e., HYF06, HYF07, HYM06), (b) venturing, often repeatedly, from the Atlantic coastline into the mid-Atlantic Outer Continental Shelf and returning (HYM05, HYF02), or (c) choosing an offshore migration route either within or to the east of the study area (i.e., HYM01, HYF01). The mean percentage of UDs falling within the mid-Atlantic study area showed a tendency to be higher in males compared to females (Table 25-1).

#### *Time spent over land vs. over water*

The amount of time peregrines spent within the northern and southern bounding latitudes of the mid-Atlantic study area varied substantially among individuals. Peregrines spent between 18 – 357 hrs within this latitudinal zone (0.8 – 14.9 d; mean ± SD: 67 ± 82 hrs, n = 16 tracks, 15 individuals). Of the 10 tracks intersecting the mid-Atlantic study area, time spent within this zone ranged from 18 – 357 hrs (0.8 – 14.9 d; mean ± SD: 87 ± 100 hrs, n = 10 tracks, 10 individuals). These figures were notably influenced by HYF02a, a peregrine that conducted multiple foraging forays into the mid-Atlantic study area from the Atlantic shoreline (HYF02a; 357 hrs). Exclusion of this individual resulted in an average time spent ranging between 18 – 119 hrs (0.8 – 5.0 d; mean ± SD: 56 ± 31 hrs, n = 9 tracks/individuals).

Peregrines were estimated to spend between 8 – 100% (mean ± SD: 55.8 ± 34.3%, n = 16 tracks, 15 individuals) of their time within the bounding latitudes of the mid-Atlantic study area over areas of open water. Of the 10 tracks/individuals intersecting the mid-Atlantic study area at least once, peregrines spent 27.5 – 100% (mean ± SD: 70.5 ± 24.6%, n = 10 tracks/individuals) of their time over water. Seventy percent (7 of 10) individuals spent >50% of their time within the bounding latitudes of the mid-Atlantic study area over water, while 50% spent >75% of their time over water.

## Discussion

This study improves upon limited previous characterizations of peregrine migration along the Atlantic flyway. Efforts to gather baseline information on raptor migration patterns and space use in this flyway are particularly needed, as this area concentrates substantial portions of fall migrant raptors originating from a very broad geographic range. The Atlantic flyway also holds a substantial number of offshore wind energy planning and lease areas in state and federal waters (BOEM 2015*b*), and researchers are striving to understand wildlife patterns and evaluate risks prior to construction of facilities. To date, raptors have been poorly represented in efforts to evaluate the risks that offshore wind energy facilities may pose to bird populations. For peregrines, this is in part because: (a) compared to many waterbirds, peregrines are generally considered to migrate quickly through WEAs, (b) they travel individually and are rarely observed in high densities, and (c) exposure and collision risks are often assumed to be low or moderate in the offshore environment.

Peregrines and other raptors are commonly considered in risk assessments for terrestrial wind power projects (Madders and Whitfield 2006, Smallwood and Thelander 2008, Garvin et al. 2011, Miller et al. 2014). Raptors are generally considered to have higher collision risks when foraging, or when visibility is limited, but numerous factors such as topography, lighting, season, and habitat of the surrounding area are also important (Drewitt and Langston 2006, Madders and Whitfield 2006). While many species associated with collision risks at terrestrial-based wind energy facilities (i.e., Golden Eagles, *Aquila chrysaetos*) are rarely encountered offshore, peregrines, Merlins (*Falco columbarius*), and several other species are capable of enduring open water journeys, and they are commonly encountered offshore. Most survey techniques used to evaluate risks that terrestrial-based wind facilities pose to birds in are not appropriate or practical for understanding the ecology of peregrines in marine settings. At present, fitting individuals with tracking devices and modelling their movements may be the best approach available to learn about raptor space use offshore and to evaluate exposure to or interactions with offshore wind energy facilities. Animal tracking data has a wide range of additional conservation applications, such as establishing migratory connectivity among populations, identifying important habitats for conservation, and improving our understanding of migration ecology.

### *Characterizing movements of peregrines along the Atlantic coast*

In general, movement patterns of our sample of transmitter-instrumented peregrines were consistent with previous knowledge of peregrine migration within the Atlantic flyway (White et al. 2002, Cade and Burnham 2003). The majority of individuals migrated relatively quickly down the Atlantic seaboard, generally following the coastline. A portion of individuals used well-known peregrine migration stopovers such as Assateague Island (Yates et al. 1988) during migration. Migration tracks and distance to shore measurements demonstrated that the peregrine migration corridor was generally more concentrated in the northern portion of the U.S. portion Atlantic flyway (i.e., from the primary deployment site, Block Island, RI, to Cape Hatteras, NC) compared to the southern portion (i.e., southward from Cape Hatteras, NC), at which point migrants spread out over a broader front. Our findings also demonstrated that peregrines regularly ventured substantial distances offshore. This finding is consistent with seemingly regular observations of peregrines foraging or perching far from shore at offshore islands, oil drilling platforms, and large offshore vessels (Voous 1961, Cochran 1975, 1985, Russell 2005, McGrady et al. 2006, Johnson et al. 2011, DeSorbo et al. 2012), but these behaviors remain poorly quantified. Individuals in our study also revealed details of some noteworthy behaviors, as some peregrines: (a) travelled approximately 1500 km eastwards into the Atlantic while presumably resting on offshore vessels, (b) reversed direction in North Carolina to migrate 170 km northwards back to Assateague Island in the fall, and (c) remained in the mid-Atlantic coastal zone areas for 26 d while making regular presumed foraging trips offshore.

Our findings highlighted the significance of the mid-Atlantic region of the U.S. to the migratory ecology of peregrines. Of the 14 fall peregrine migration tracks (13 individuals) reaching the Mid-Atlantic study area, 69% departed the mainland on transoceanic flights from a 120 km stretch of shoreline between Cape Hatteras, NC and Cape Lookout, NC. This area is known for its use by birds as a staging area and a ‘launching area’ (UNC 2009). This finding may be important in efforts to conserve staging areas for avian migrants, as well as investigations of the potential impacts offshore wind energy facilities in this region may pose to migrant raptors.

### *Peregrine use of the mid-Atlantic study area and WEAs*

This study provided valuable perspectives on peregrine space use in the mid-Atlantic study area, which will be needed to assess potential exposure to offshore WEAs. Peregrines moving between the northern and southern latitudes of the mid-Atlantic study area exhibited an overlap of between 0 – 59% (mean ± SD: 21 ± 21%) of their UDs with the study area. Of the peregrine tracks reaching the northern latitude of the study area, only one had a UD entirely outside of the study area, and it was located farther offshore (Figure 25-7). One third of individuals had <10% of their UDs fall within the study area, and these individuals tended to be those following the Atlantic shoreline. In general, individuals with >10% of their UDs within the study area were those who: (a) travelled down the Delmarva Peninsula and crossed over the southern portion of the study area, (b) ventured on presumed ‘foraging flights’ into the study area and returned to the shoreline, or (c) chose offshore migration routes either within or to the east of the mid-Atlantic study area. The latter group tended to be those with highest proportions of their UDs in the study area; however, among individuals, higher UDs within the study area did not necessarily always correspond to an increased amount of time spent within the study area. Peregrines whose flight paths intersected the greatest proportion of the study area but moved through the area quickly could have a higher proportion of their UD in the study area than individuals that used a broader area, but displayed a longer residence time. For example, 43% of the UD for HYM01 overlapped with the mid-Atlantic study area, but this individual travelled through most of the study area in approximately 20 hours (approximately 11.3 km/hr).

Peregrines varied widely in the amount of time spent within the latitudinal boundaries of the mid-Atlantic study area. One individual spent 0.8 d (18 hrs) in this zone, while another resided there for over two weeks (357 hrs; mean ± SD: 67 ± 82 hrs). Peregrines in this zone spent over half (56 ± 34%) of their time over water. Some inaccuracies in time spent or over water estimates may have arisen due to interpolation errors generated by the CTRCW model. One-hour interval data, as selected for our analysis, may have resulted in some misclassification errors of over land vs. over water habitat types, particularly for individuals that spent large amounts of time in the intertidal zone. Nonetheless, the majority of locations were easily classified, and the model use probably improved upon estimates generated with raw, irregular spaced PTT data.

The proportion of peregrines' UDs falling within the Delaware, Maryland, and Virginia WEAs ranged from 0 – 7% and mean proportions of UDs in each of the WEAs were <1%. Low proportions of UDs within the WEAs are to be expected, given the size of WEAs relative to peregrine travel rates. For example, Fuller et al. (1998) found migrant peregrines travelled at an average rate of 172 km/d during fall migration (Fuller et al. 1998), while DE, MD and VA WEAs range in size from approximately 323 – 456 $km^2$.

### *Peregrines and offshore wind energy*

The vulnerability of raptors to collisions varies depending on flight behavior, weather, season, location and other factors (Richardson 2000, Madders and Whitfield 2006). The extent to which peregrines may be vulnerable to collisions with offshore wind turbines remains largely unknown. While peregrine collisions with wires (e.g., transmission lines) have been well-documented (Olsen and Olsen 1980, White et al. 2002), there are few accounts of direct mortalities from terrestrial wind turbine collisions. Our literature search revealed two cases of peregrine mortalities associated with terrestrial-based wind turbines in Belgium (Hötker et al. 2006), four in Germany (Dürr 2011), one in Scotland (Meek et al. 1993), and one in New Jersey, U.S.A. (Mizrahi et al. 2009). Each case is associated with site-specific circumstances that often complicate comparisons among projects, particularly with those based offshore. To date, no peregrine fatalities have been documented at European offshore wind projects, although offshore mortality monitoring methodologies are inadequate to properly detect or assess collision risks of peregrines. We identified only two sources that speculate about peregrine vulnerability at offshore wind projects. Using data from visual surveys and general impact assessment methods, including the project's magnitude of pressure (intensity, duration, and spatial range) and sensitivity of environmental factors, Jensen et al. (2014) considered peregrines to have a low collision risk at the Horns Rev 3 wind farm planned for construction off the coast of Denmark. Willmott et al. (2013) assessed and ranked collision vulnerability of various species found in the Atlantic Outer Continental Shelf using metrics such as population size, annual occurrence, flight behaviors, and displacement sensitivity. Peregrine Falcons were qualitatively ranked as "medium" in collision sensitivity in that study.

A substantial portion of peregrines, particularly *tundrius* peregrines originating from Greenland and Arctic Canada, and restored peregrine populations in many Atlantic U.S. states, migrate along the Atlantic flyway enroute to southern wintering areas. Thus, efforts to understand migration ecology and risks for individuals are warranted. Findings from this study indicating peregrines commonly use offshore habitats along the Atlantic coast are consistent with their general evolutionary-based association with water for nesting, foraging, and during migration (Ratcliffe 1980, White et al. 2002, 2013). Our limited sample suggests use of offshore habitats may be relatively common during migration. The extent to which our sample reflects movements of the broader population of fall migrant peregrines using the Atlantic flyway remains unknown. Our sample, comprised primarily of migrant *tundius* peregrines, may not reflect movement patterns of resident peregrine populations in various Atlantic U.S. states. Movement patterns of resident populations remain poorly documented for resident populations in most states along the Atlantic U.S.

Our selection of offshore islands for trapping locations may have biased our sample toward individuals with greater tendencies to travel offshore; however, wind conditions, patterns of food supply, body condition, and other factors may play an equal or larger role (Newton 2008, 2010). Age class and gender may also influence peregrine movement patterns observed in this study. Our sample was comprised by only two adults, and both behaved notably differently compared to HY individuals. Overall, migration tracks of the two adults appeared relatively deliberate and time efficient compared to many HYs, whose movements generally appeared more exploratory. First-year peregrines, still refining their hunting skills, may be more inclined to venture out over water where avian prey is more vulnerable to capture. In a radio tracking study of nine HY peregrines captured on Assateague Island and followed by aircraft, Cochran (1985) considered HY peregrines of both sexes to be 'somewhat independent of land' and he speculated that use of offshore habitats was preferred by peregrines and may increase with age and experience. Cochran (1985) also suspected adult male peregrines to be somewhat 'pelagic' (Cochran 1985). While our limited sample size for adult birds did not support this latter idea, our data do suggest that peregrine migration patterns in the Atlantic flyway are not particularly limited by proximity to land. While larger sample sizes are needed, males in our study showed a tendency to spend more time in the mid-Atlantic study area compared to females (Table 25-1), and males showed a tendency to migrate further than females to reach wintering areas.

It remains unknown how peregrines might respond to turbines encountered offshore. The two peregrines providing flight altitude data in this study demonstrated that peregrines fly above, within, and below the rotor swept zones for turbines currently proposed for use in the offshore environment. While our sample size for evaluating flight altitude is limited, local weather conditions likely have a predominant influence on flight altitude choices during migration (Newton 2007, Shamoun-Baranes et al. 2010, Mandel et al. 2011, Mellone et al. 2011, Bohrer et al. 2012, Lanzone et al. 2012). Peregrines likely fly at lower altitudes during migration when looking for prey along the water surface. A wide variety of other factors including behavior state (i.e., foraging or migrating) and patterns of prey populations also influence on the flight altitude selected by peregrines and their risks of colliding with offshore turbines. Lighted structures such as oil drilling platforms and barges may attract peregrines for foraging or resting. Johnson et al. (2011) observed peregrines feeding nocturnally on birds disoriented by lights on offshore structures. Peregrines may have an elevated collision risk with offshore wind turbines if they are attracted to them for perching and especially foraging, or if they encounter them during inclement weather during the day or at night.

Findings in this study would be strengthened by increased sample sizes, particularly within age and gender classes. It remains unknown what proportion of migrant peregrines using the Atlantic flyway travel offshore, or the extent to which our selection of offshore capture sites may have influenced our findings. Continuing advancements in animal tracking technologies, such as high resolution GPS, or 'GSM' transmitters (i.e., cellular network based; Global System for Mobile Communications; Lanzone et al. 2012) can produce location data with higher sampling rates and greater horizontal and vertical accuracy, which could improve insights gained in future investigations. Further weight reductions for all transmitter types will enable biologists to better understand the movements of peregrines, particularly males, and smaller raptors such as Merlins, American Kestrels (*Falco sparverius*) and Northern Harriers

(*Circus cyaneus*) that also use the Atlantic coast during migration. Efforts to characterize raptor movement patterns and evaluate collision risks with turbines are particularly important along the Atlantic Flyway given the substantial quantity of individuals using this region during fall migration, and the number of proposed wind energy planning and lease areas in state and federal waters in various stages of development.

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## Figures and tables

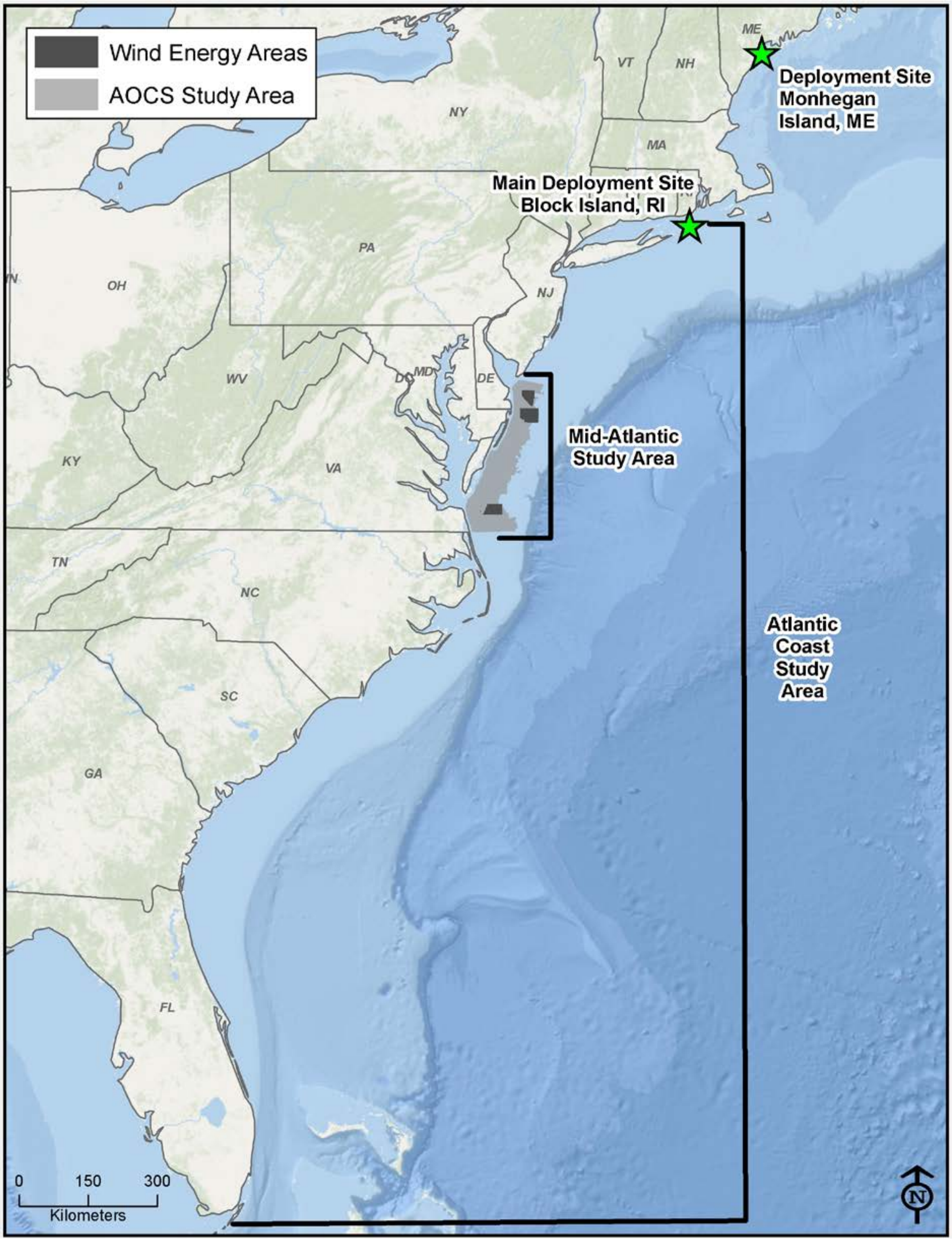

**Figure 25-1. Atlantic coast study area, Mid-Atlantic Study area and Wind Energy Areas, and two satellite transmitter deployment locations.**

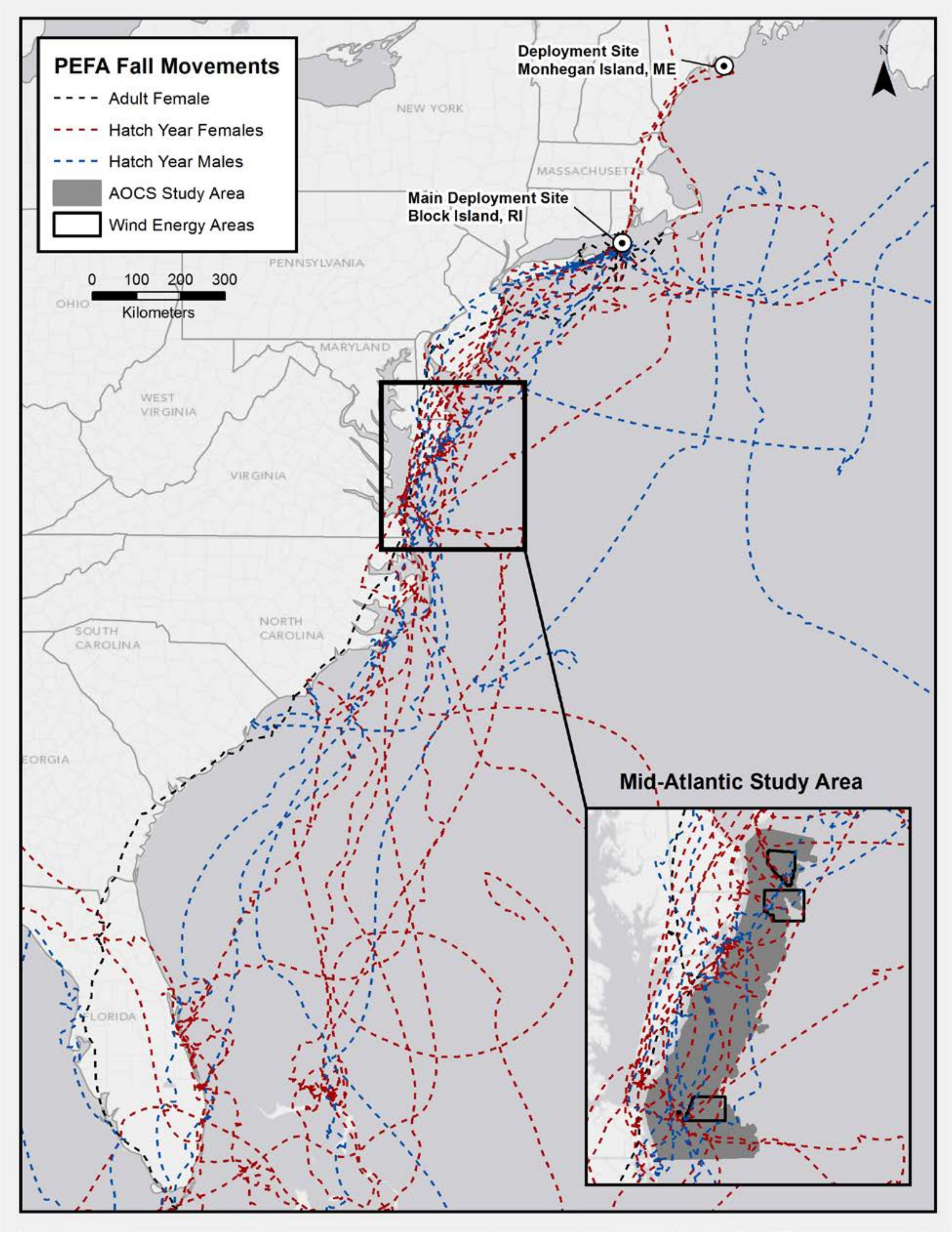

**Figure 25-2. Movement patterns of 16 satellite transmitter instrumented Peregrine Falcons migrating along the Atlantic U.S. coast, fall, 2010 - 2014.**

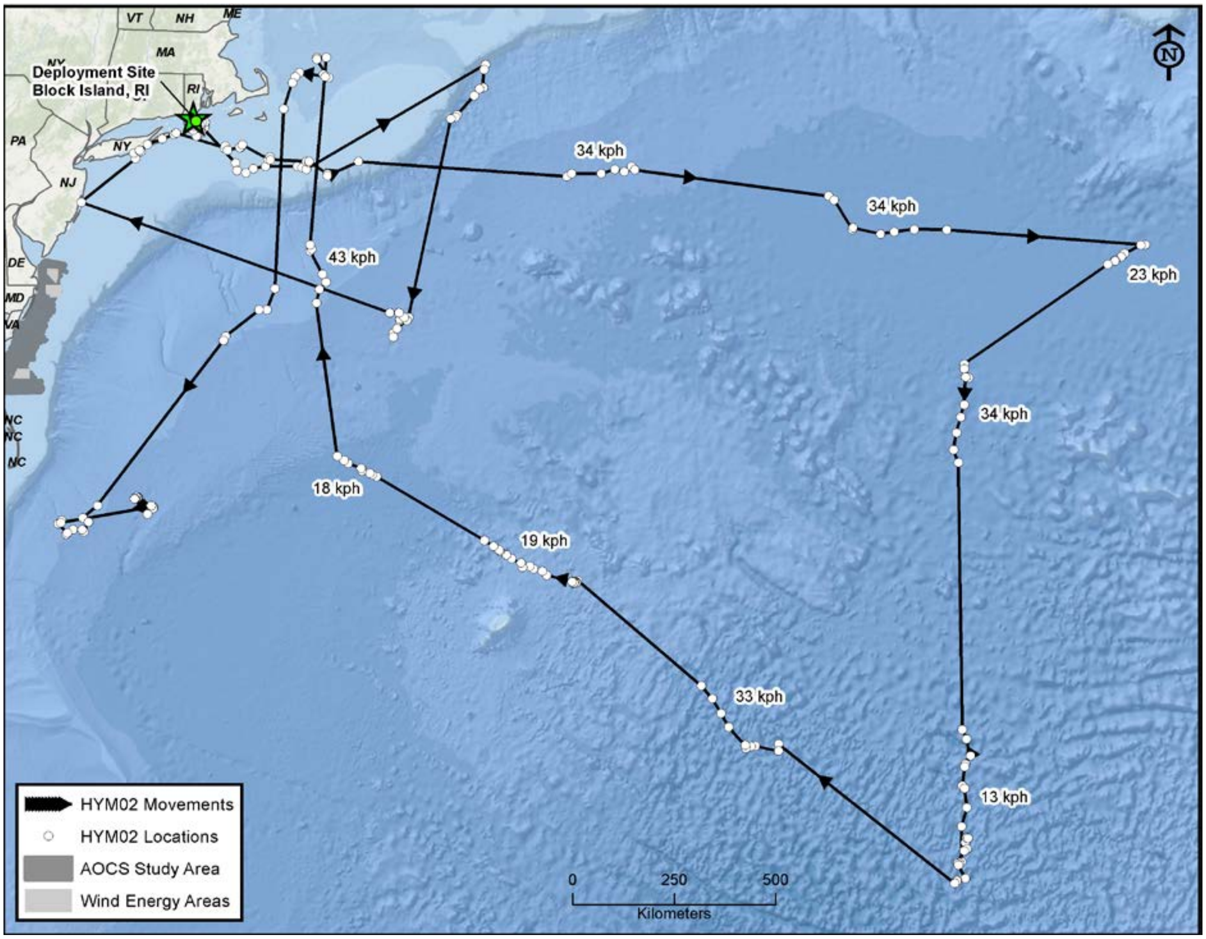

**Figure 25-3. Movements of male hatching year Peregrine Falcon instrumented with a satellite transmitter during fall, 2012 at Block Island, RI.** Rate of movement indicated for daily clusters of satellite location estimates.

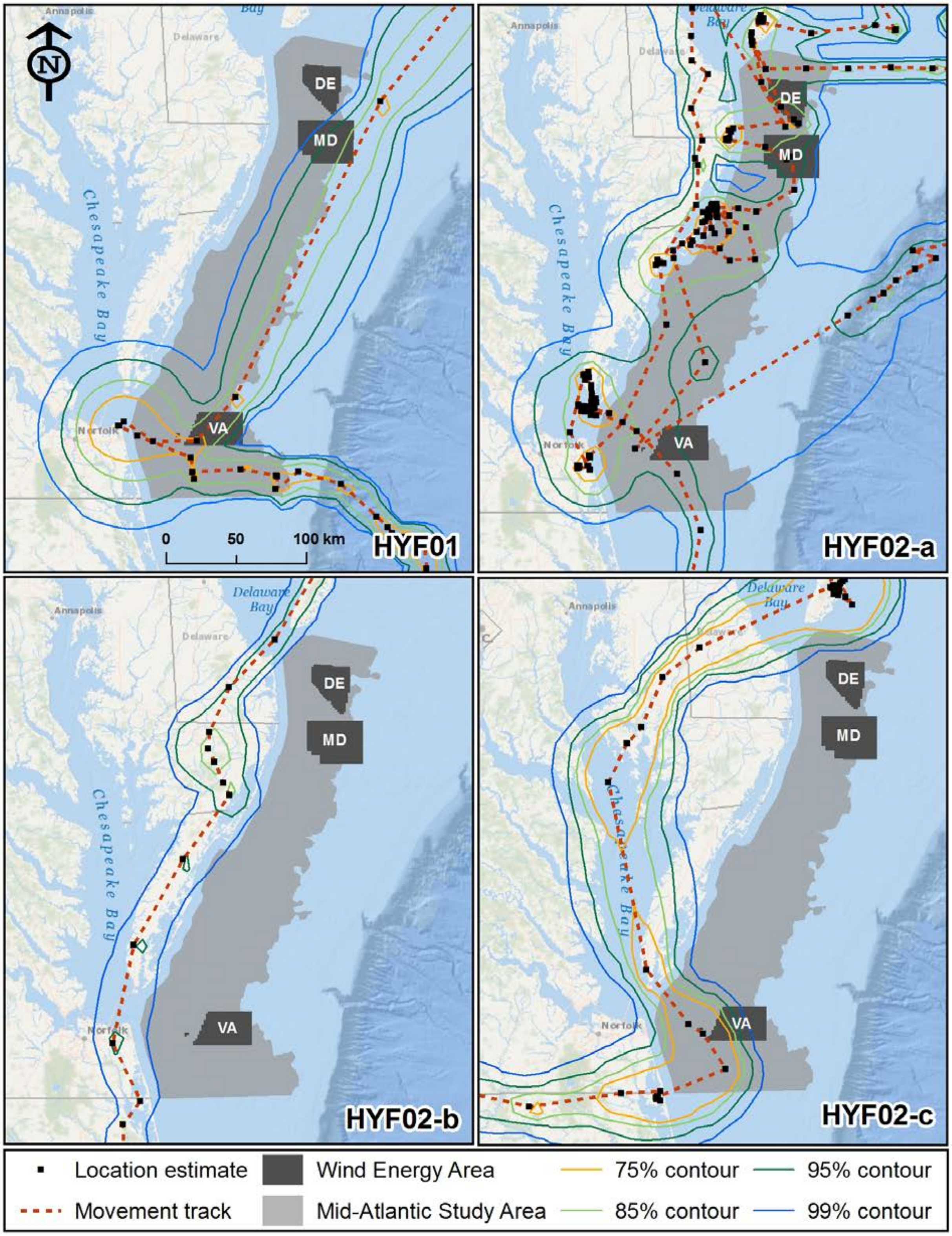

**Figure 25-4. dBBMM utilization distribution contours for two migrating hatching year (HY) female Peregrine Falcons relative to the Mid-Atlantic Study Area and three Wind Energy Areas.** Three migrations shown for HYF02 (a – 1st fall; b – 2nd fall; c – 1st spring). All maps shown at same scale; scale bar shown in first map pane.

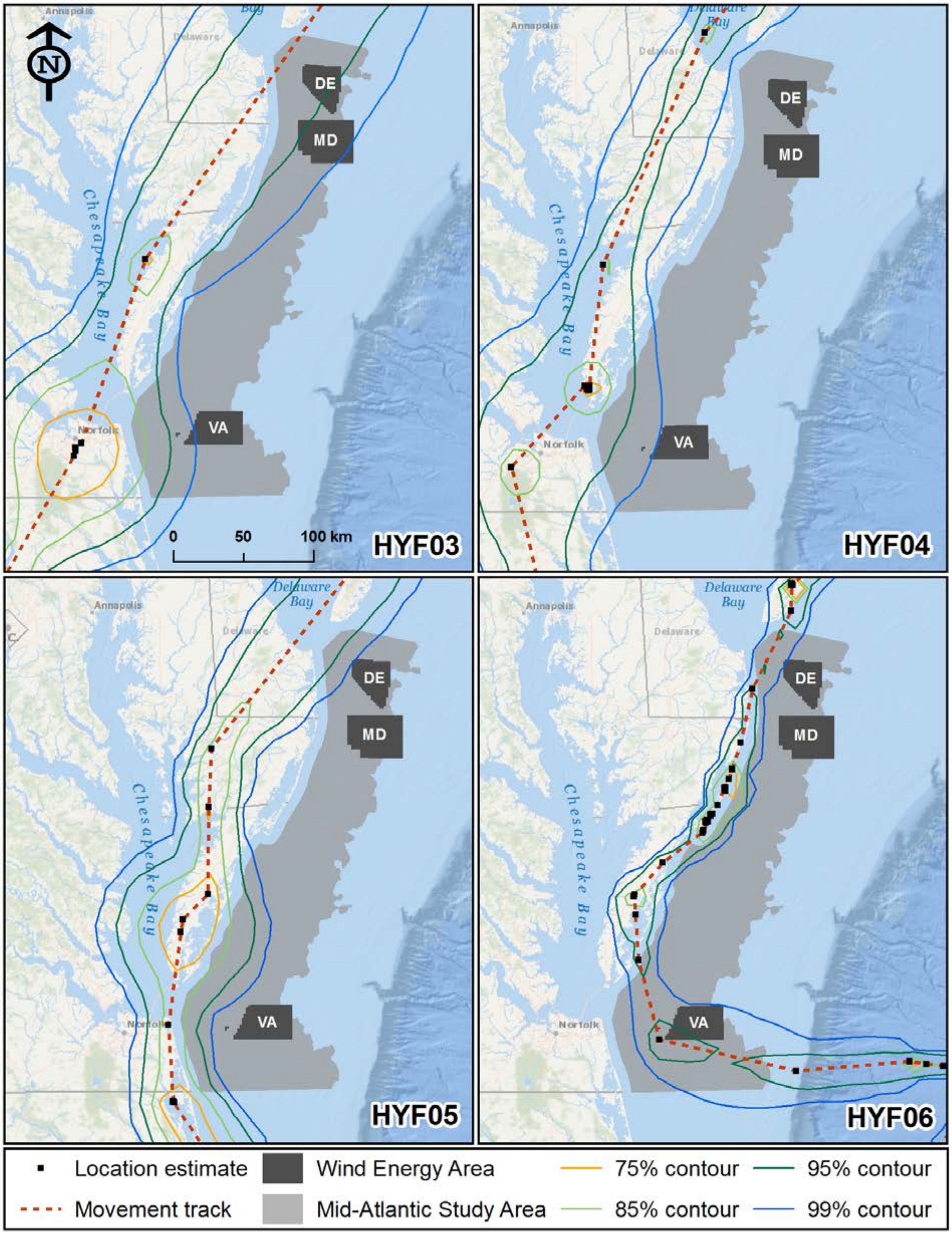

**Figure 25-5. dBBMM utilization distribution contours for four migrating hatching year (HY) female Peregrine Falcons relative to the Mid-Atlantic Study Area and three Wind Energy Areas.** All maps shown at same scale; scale bar shown in first map pane.

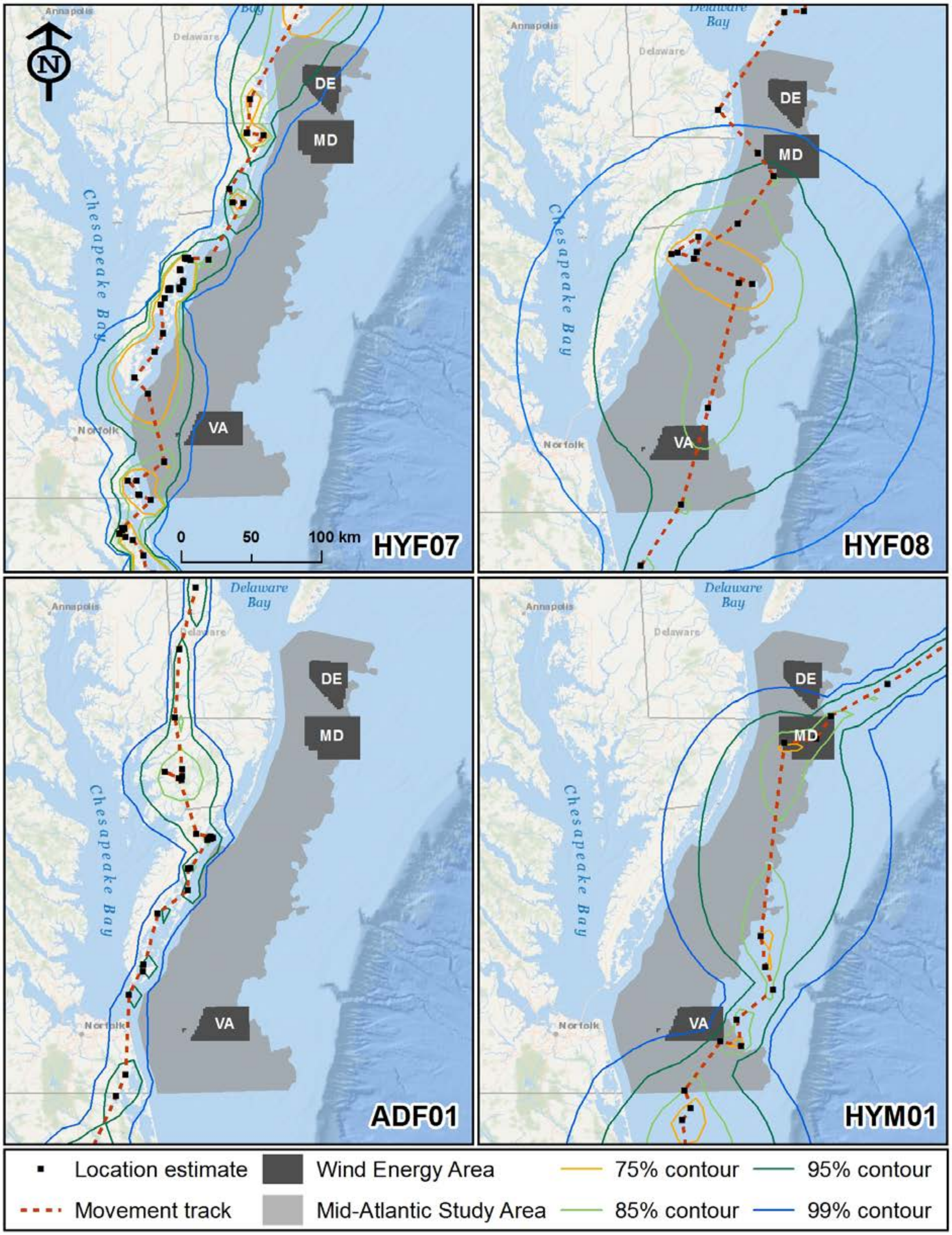

**Figure 25-6. dBBMM utilization distribution contours for two migrating hatching year (HY) female, one adult (AD) female, and one hatching year male Peregrine Falcons relative to the Mid-Atlantic Study Area and three Wind Energy Areas.** All maps shown at same scale; scale bar shown in first map pane.

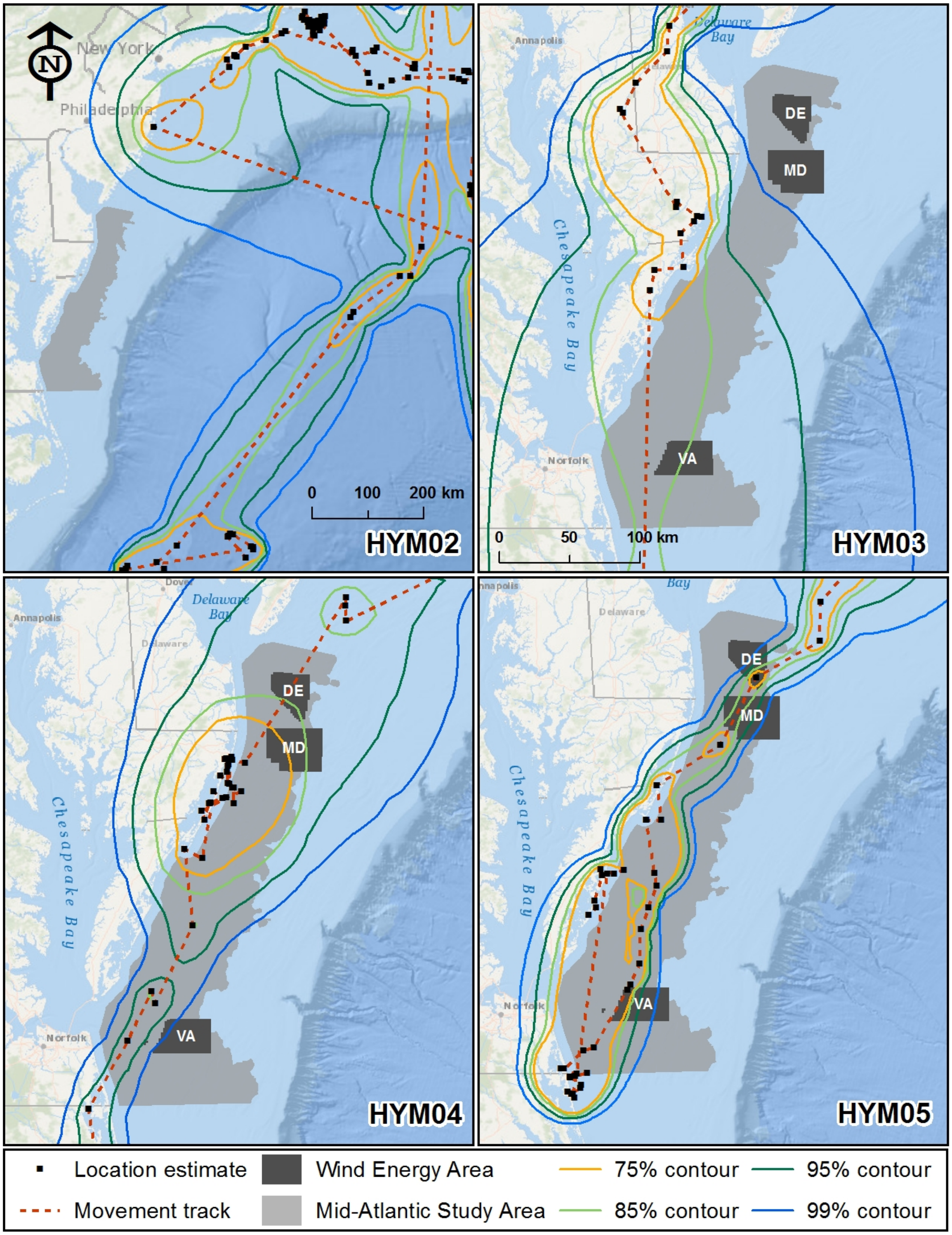

**Figure 25-7. dBBMM utilization distribution contours for four migrating hatching year (HY) male Peregrine Falcons relative to the Mid-Atlantic Study Area and three Wind Energy Areas.** First map pane shown at different scale; all others shown at same scale as second map pane.

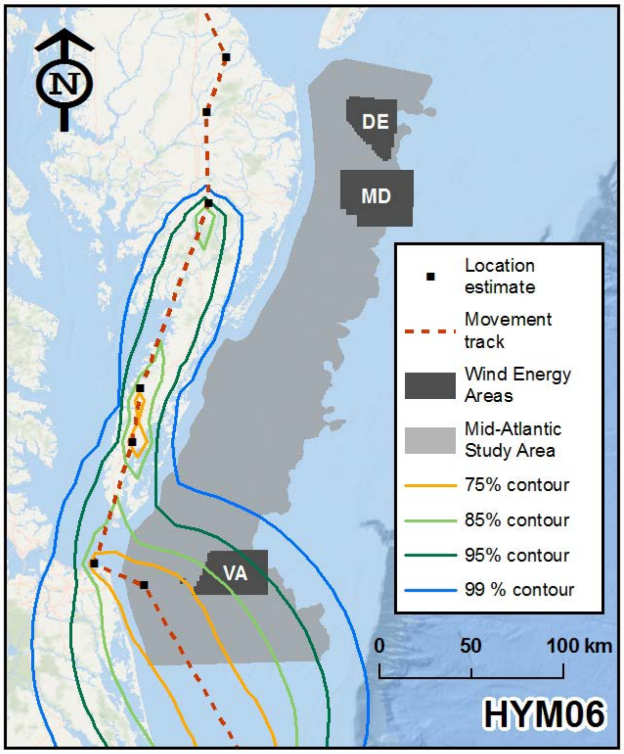

**Figure 25-8. dBBMM utilization distribution contours for one hatching year (HY) male Peregrine Falcon relative to the Mid-Atlantic Study Area and three Wind Energy Areas.**

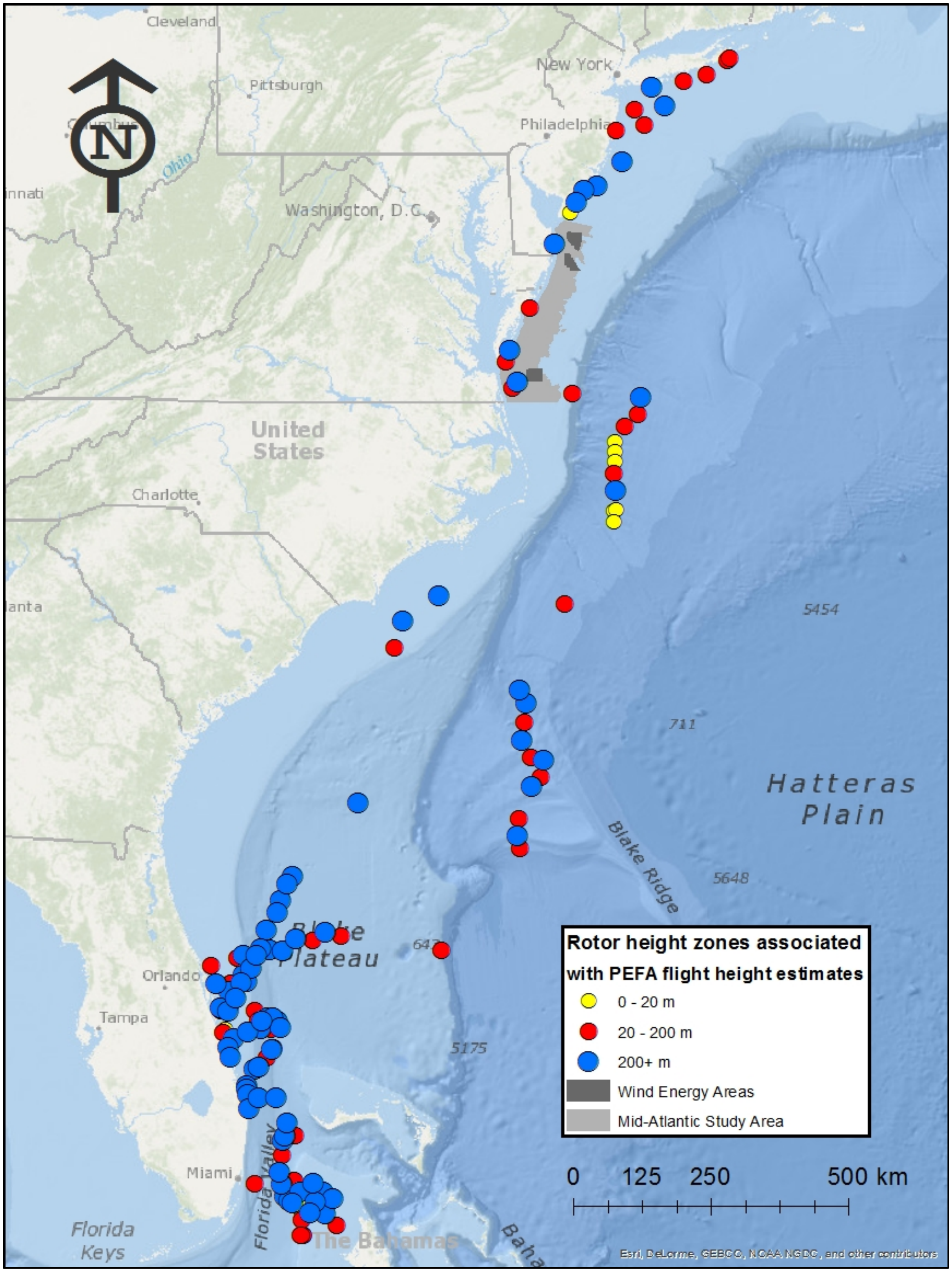

**Figure 25-9. Flight altitude of two female Peregrine Falcons tracked using GPS satellite telemetry in four height categories related to proposed height of offshore turbines: (1) 0 – 20 m, (2) 20 - 200 m, and (3) >200 m.**

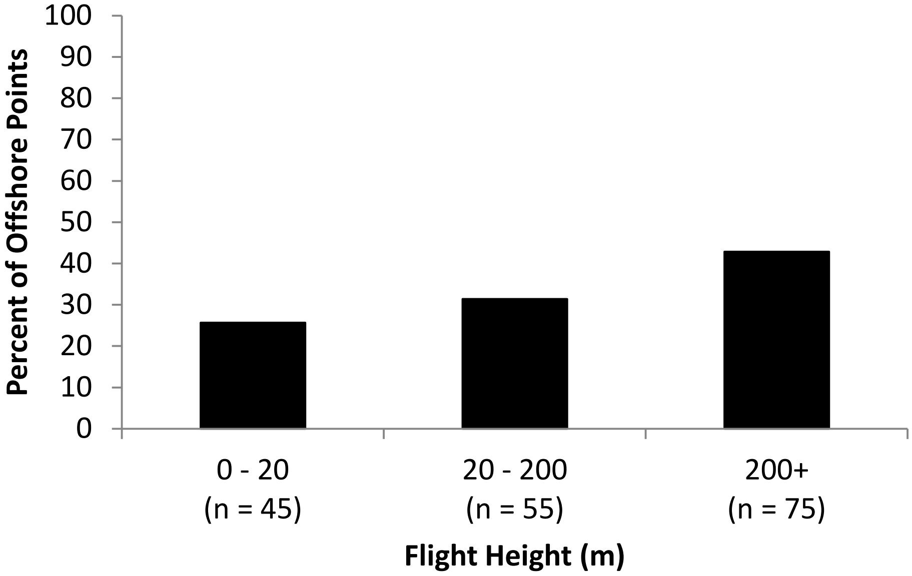

**Figure 25-10. Proportion of altitude estimates of two migrant female Peregrine Falcons falling within three height categories related to offshore wind turbines.** Categories: (1) below rotor-swept range (< 20 m), (2) within rotor-swept zone (20 – 200 m), (3) and above rotor height (>200 m).

**Table 25-1. Proportion of female (n = 9), male (n = 6) and combined gender class (n = 15) Peregrine Falcon Utilization Distributions falling within the mid-Atlantic study area and Delaware, Maryland, and VA Wind Energy Areas (WEAs).**

| Area | Group | Mean | SD | Min | Max |
|---|---|---|---|---|---|
| **Mid-Atlantic Study Area** | All individuals | 21% | 21% | 0% | 59% |
| | Females | 15% | 19% | 0% | 56% |
| | Males | 33% | 22% | 0% | 59% |
| **DE WEA** | All individuals | 0% | 0% | 0% | 1% |
| | Females | 0% | 0% | 0% | 1% |
| | Males | 0% | 0% | 0% | 1% |
| **MD WEA** | All individuals | 1% | 2% | 0% | 7% |
| | Females | 0% | 0% | 0% | 1% |
| | Males | 2% | 3% | 0% | 7% |
| **VA WEA** | All individuals | 1% | 1% | 0% | 3% |
| | Females | 1% | 1% | 0% | 3% |
| | Males | 1% | 1% | 0% | 2% |

# Introduction to Part VI
# Nocturnal avian migration monitoring

## Report structure

The chapters in this report represent a broad range of study efforts and goals. Some chapters are purely methodological in nature, while others present a variety of analyses and results. Generally speaking, however, chapters fall into two categories: efforts focused on population distributions, and those focused on individual movements (Figure I).

Part I of this report (the Executive Summary and Chapters 1-2) summarizes and synthesizes project results. The 25 subsequent chapters and their relationships to each other are shown in Figure I. In Parts II (Chapters 3-6) and III (Chapters 7-12), we describe methods and results for high resolution digital video aerial surveys and boat surveys, respectively. Part IV of this report (Chapters 13-19) combines data from both survey approaches to develop a comprehensive understanding of marine wildlife populations that use the mid-Atlantic study area. Part V (Chapters 20-25) focuses on individual movements and habitat use of focal avian species, tracked via satellite telemetry; and Part VI (Chapters 26-27) focuses on population-level migratory movements over the oceans, using several approaches for studying nocturnal avian migration. An additional study effort, which further explores statistical approaches for combining boat and aerial survey data to develop joint models of wildlife distributions and abundance, will be published as an addendum to this final report.

## Part VI: Nocturnal avian migration monitoring

Oceans and other large bodies of water can act as barriers to migrating birds. Some species will not pass over these obstacles; however, many will stop over to rest and refuel before making long overwater flights (Delingat et al. 2008, Faaborg et al. 2010). Large bodies of water like the Mediterranean Sea or the Gulf of Mexico are regularly crossed by even small songbirds (Bruderer and Lietchi 1999, Gauthreaux and Belser 1999). Within the study area for this project, Cape May and Delaware Bay are both known areas where large numbers of migrants stop over during migration (Clark et al. 1993, Moore et al. 1995). While there is some evidence of passerine migration over the northwestern Atlantic for some species (Williams et al. 1977, Faaborg et al. 2010, DeLuca et al. 2015), oceanic flyways and migrant use of offshore regions of the mid-Atlantic are poorly known.

There are two chapters in Part VI of this report, focused on the use of different monitoring technologies to understand the movement patterns and species composition of nocturnal avian migrants off the Eastern Seaboard of the United States:

| | |
|---|---|
| Chapter 26. | Passive acoustics pilot study: nocturnal avian migration in the mid-Atlantic. |
| Chapter 27. | Utility of WSR-88 weather radar for monitoring nocturnal avian migration in the offshore environment. |

The project team investigated the species composition, spatial patterns, and weather-dependent variation in offshore bird migration through a combination of acoustic and radar data collection. Both the nocturnal passive acoustic avian monitoring from the survey boat (Chapter 26) and the analysis of WSR-88 radar data, also known as NEXt generation RADar (NEXRAD, Chapter 27) were undertaken to determine the utility of these approaches for examining avian migration in the offshore environment. Acoustic monitoring techniques were tested in the marine environment and used to begin developing a list of songbird and shorebird species that migrate over the study area. NEXRAD was used to improve our understanding of patterns in migratory activity in the offshore environment on the Atlantic coast of the U.S.

### *Passive acoustics*

Passive acoustic detectors can be useful for documenting nocturnal migration of low-flying songbirds and shorebirds, which migrate at night and emit short vocalizations during flight (Farnsworth 2005). Many bird species can be identified by their vocalizations, so nocturnal acoustic monitoring stations can provide species-specific presence-absence data and indices of activity for birds that vocalize during migration. To determine whether passive acoustic detectors could be successfully deployed in the offshore environment from a boat platform, and, if so, what species are flying over the Mid-Atlantic continental shelf, we deployed an acoustic detector on the survey vessel (Chapter 26). When the boat stayed overnight on the water (seven total occasions over two years, located 25-46 km from shore), we documented what birds passed overhead. Migratory flight calls were detected from at least fifteen species, including both passerines and shorebirds, mostly during a single September survey when the boat was located 40 km off the coast. Our limited sampling effort makes inference difficult, but this pilot study suggests that a diverse range of landbirds migrate over the Mid-Atlantic continental shelf at low altitudes, at least on occasion. This design appears to be an effective means to monitor the number and type of avian species that migrate offshore, though a more extensive effort is warranted before making broader conclusions.

### *Weather radar*

Developed as a tool to monitor meteorological phenomena, weather surveillance radars regularly detect flying animals in the atmosphere, and are being increasingly incorporated into biological studies of migration patterns and movements (Bridge et al. 2011, Chilson et al. 2012). Open water areas often have poorer radar coverage than terrestrial areas of the United States. In this study, we examine the utility of WSR-88D (NEXRAD) weather radar for studying migration offshore, and specifically off the

Atlantic coast of the U.S. from New York to North Carolina (Chapter 27). We also compare migratory activity at sites over land and up to 80 km out to sea, controlling for variables that could affect measured levels of migratory activity, and we identify the environmental variables correlated with offshore activity, as well as specific geographic locations that may represent offshore migration pathways.

The high level of altitudinal overlap between our measurements and turbines heights suggests that our predictions of migratory activity in the offshore environment are highly relevant to migration occurring at rotor-sweep heights. After controlling for biases in measured levels of migratory activity due to distance from the radar, site elevation, and other factors, we found that in fall, there was no correlation between migratory activity and distance from shore, and no significant difference in predicted activity levels at offshore vs. terrestrial sites across the study area. This suggests that migration over open water areas may be quite common in the mid-Atlantic during this season. There is also a strong weather-related component to offshore fall migration; though responses varied by topographic location along the coast, offshore activity was particularly high under west winds. There were high levels of daily variation in activity at our study sites, but some offshore areas had consistently higher predicted activity levels, most notably the New York Bight (south of Long Island) and offshore of North Carolina. In spring, there still was offshore activity around North Carolina, but levels were fairly minimal in other locations. Westerly winds were less important to offshore activity during this season, with onshore activity aided by both southerly and westerly winds. These data suggest that while birds are less likely to migrate offshore in spring, during the fall, there appear to be multiple “jumping off points” along the coast for tailwind-aided overwater migrations.

### *Implications*

While there may be fewer overwater migrants in the mid-Atlantic as compared to smaller ecological barriers such as the Gulf of Mexico or Mediterranean Sea, there still appears to be both overwater songbird and shorebird migration in the region. This may be particularly true during fall migration; evidence in a variety of species (including bats as well as landbirds; Chapters 25-27; Hatch et al. 2013) suggest that overwater migration in the northwestern Atlantic is much more common in fall than in spring, when animals presumably migrate preferentially over water due to consistent tailwinds from the northwest (e.g., Morris et al. 1994, Hatch et al. 2013, DeLuca et al. 2015). Trans-oceanic migrations, once thought to be extreme events only undertaken by few individuals or species with extreme physiological adaptations (DeLuca et al. 2015), are perhaps more commonplace than previously thought in this region.

Offshore structures, particularly those with full-spectrum nighttime lighting, have been known to cause mass mortality events to migrating passerines in low visibility weather conditions (Hüppop et al. 2006). Given the levels of migratory activity predicted in offshore locations, regulators for offshore wind energy development may want to consider potential impacts to terrestrial species (passerines, shorebirds, bats, etc.) in offshore wind development scenarios. This may be particularly important in locations with consistently higher levels of migratory activity, such as the New York Bight and areas offshore of North Carolina. Predicted levels of activity in many other parts of the study area were also intermittently high, however, suggesting that offshore migration is a widespread phenomenon, and should be regarded as such during planning activities.

**Acknowledgments:** This material is based upon work supported by the Department of Energy under Award Number DE-EE0005362. Additional funding support came from the Maryland Department of Natural Resources and the Maryland Energy Administration. HiDef Aerial Surveying, Inc. made significant contributions toward the completion of this study.

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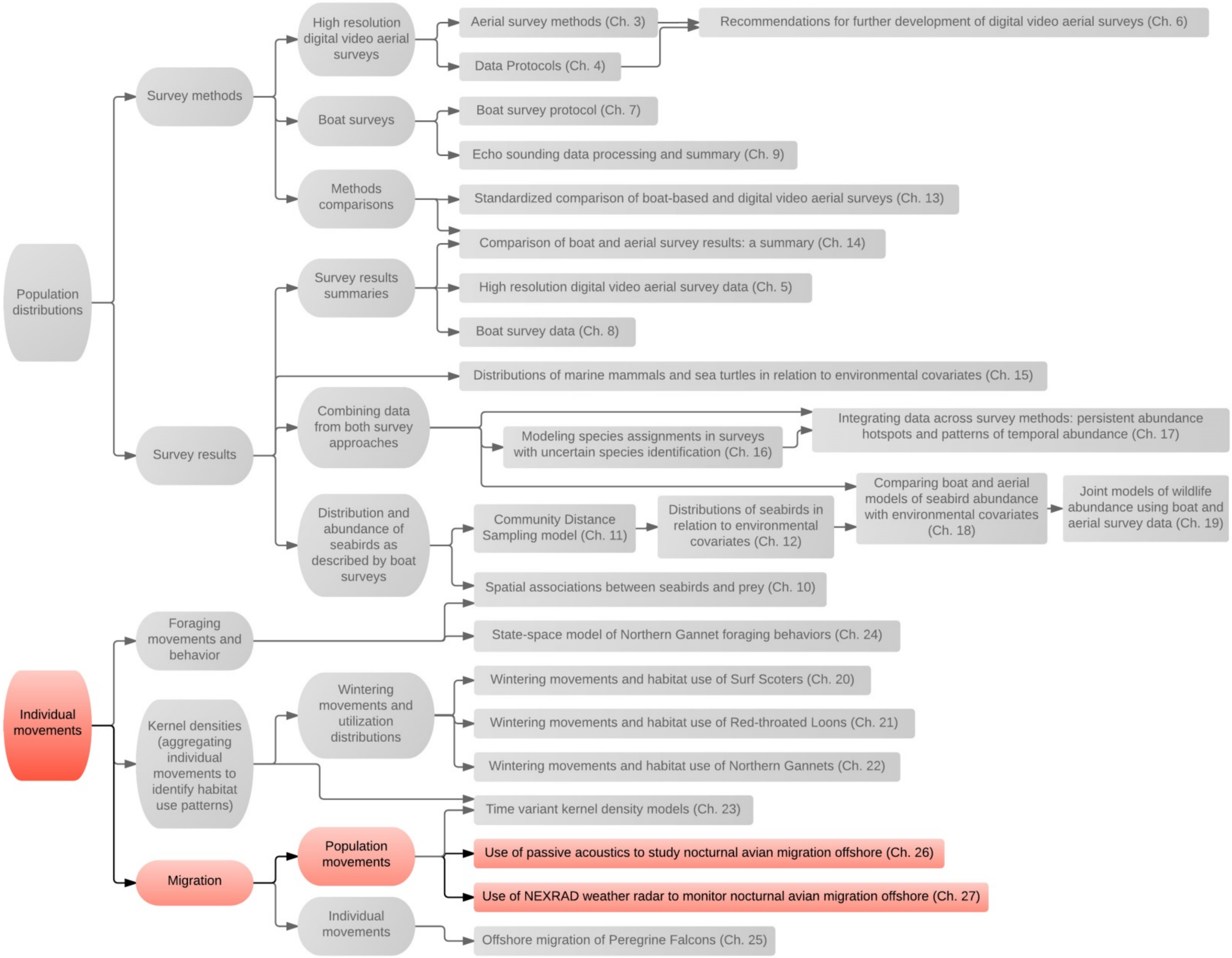

**Figure I. Organization of chapters within this final report.**

# Chapter 26: Passive acoustics pilot study: nocturnal avian migration in the mid-Atlantic

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 26 Highlights

*Using passive acoustic sensors to document bird species migrating offshore*

### *Context*[1]

Little is known about the offshore migratory movements of songbirds in the northwestern Atlantic Ocean. Large bodies of water can act as migratory barriers to birds, but many are also crossed regularly by migrants in North America and Europe. Past research has suggested that songbirds likely migrate over the Atlantic—leaving from the northeastern United States and traveling to the Caribbean or South America—but we have limited information about which species conduct these movements or how often these movements occur.

As a complement to Chapter 27, where we document large-scale patterns of offshore migration and what conditions promote it, this chapter focuses on a pilot effort to identify some of the species that conduct these migratory journeys. Here, we use passive acoustic sensors deployed on a survey boat to detect the vocalizations that birds make during migration and determine what species are migrating over the mid-Atlantic Outer Continental Shelf.

### *Study goal/objectives*

Determine whether passive acoustic sensors can be used in the offshore environment to identify bird species that migrate over the Atlantic Ocean.

### *Highlights*

- The deployment of a passive acoustic sensor from a survey vessel was effective for detecting nocturnal migrants.
- While most acoustic surveys did not detect any songbird migrants, one survey 40 km off the coast detected 123 individual calls in one night.
- The most commonly detected songbirds were Yellow-rumped Warblers (*Setophaga coronata*) and American Redstarts (*S. ruticilla*); both of these species are forest-breeding insectivores.
- Many finches and thrushes were also detected, but species-level identification was not possible.
- Several species of sandpipers were also detected in these surveys.

### *Implications*

Acoustic surveys have not always been successful in the marine environment, but this technique was effective in a pilot study. Migratory landbirds appear to be using this area at least on occasion. The effects of offshore developments on migratory passerines are not often considered and are poorly understood, and more research is needed on landbird migrations in this environment.

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[1] For more detailed context for this chapter, please see the introduction to Part VI of this report.

## Abstract

Passive acoustic detectors can be useful for documenting nocturnal migration of low-flying songbirds and shorebirds. Many of these taxa produce species-diagnostic calls during migratory flight, which can be used to identify the species composition of migrants in an area and the relative migratory activity. To test the utility of this technology in the offshore environment, and to determine what species are flying over the mid-Atlantic Outer Continental Shelf (OCS), we deployed a passive acoustic detector on the project's 55' survey boat. When the boat stayed overnight on the water (seven total occasions over two years) we documented the birds passing overhead. No flight calls were heard on five of the seven nights, but on one night in early September, we documented 123 calls at a location over 40 km away from shore. Our limited sampling effort makes inference difficult, but the large number of species documented suggests that a diverse range of landbirds migrate over the mid-Atlantic OCS at low altitudes, at least on occasion. A long history of radar studies and incidental songbird observations aboard research vessels suggest that overwater migration by terrestrial species may be fairly common. The deployment of passive acoustic detectors from mobile platforms in the offshore environment appears to be a viable approach for detecting such events and identifying species that are migrating offshore.

## Introduction

Oceans and other large bodies of water can act as barriers to migrating birds. Some species will not pass over these obstacles, however, many will stop over to rest and refuel before making long overwater flights (Delingat et al. 2008; Faaborg et al. 2010). Large bodies of water like the Mediterranean Sea or the Gulf of Mexico are regularly crossed by even small songbirds (Bruderer and Liechti 1999; Gauthreaux and Belser 1999). Within the study area for this project, Cape May and Delaware Bay are both known areas where large numbers of migrants stop over during migration (Clark et al. 1993; Moore et al. 1995). While there is some evidence of passerine migration over the northwestern Atlantic for some species (Faaborg et al. 2010; DeLuca et al. 2015), oceanic flyways and migrant use of offshore regions of the mid-Atlantic are poorly known and overwater migrations are thought to be only achieved by certain species (e.g., the Blackpoll Warbler, *Setophaga striata*).

Many bird species migrate at night and emit short vocalizations during flight ("flight calls"; Farnsworth 2005; Evans 2012). As songbirds are typically inactive at night during the non-migratory season, the combination of time of year, time of day, and flight call vocalizations provide strong evidence that the detected individuals are migrating. These flight calls are thought to be a form of communication between individuals, or possibly serve as a type of "echolocation," helping birds determine their altitude or navigate during periods of poor weather (Farnsworth 2005; Evans 2012). Many bird species can be identified by their vocalizations, so nocturnal acoustic monitoring stations can provide species-specific presence-absence data and indices of activity for birds that vocalize during migration.

The objective of this study was to determine if boat-based acoustic monitoring of nocturnal migrants was effective in describing the magnitude and complexion of offshore migration and, if so, to determine what species are moving through the offshore environment. We used an acoustic recording unit on the *Stormy Petrel II*—the boat survey platform—to passively monitor for nocturnal migrants when the boat was at sea between survey days. Here, we describe the species detected in these seven survey nights, where these calls were found, and what these data suggest about the future of this technique for measuring offshore migration.

## Methods

During the infrequent nights spent at sea in boat surveys, a Song Meter 2 acoustic recording setup (Wildlife Acoustics, Inc.) was deployed with a weatherproof microphone designed for recording distant night flight calls in the sky. The setup was operated atop a 3 m tall pole on the upper deck of the survey vessel (placing the microphone approximately 5 m above sea level), and operated nonstop between nautical sunset and nautical sunrise to record flight calls of nocturnally migrating birds. The vessel was anchored at variable locations at sea (between 25 and 46 km offshore on any given survey). Resulting flight call data were analyzed to identify bird species (when possible) and provide information on the timing and intensity of nocturnal migration activity.

Bird acoustic analysis was conducted using Program RAVEN (Cornell Lab of Ornithology). Flight call files were identified via visual analysis of audio spectrograms by experienced BRI ornithologists. Calls were identified to species or to a complex of species with similar calls (Evans 2012; Murray 2004); calls that were not definitively identified to species or species grouping but were confirmed as birds were labeled as “no ID,” and were used in analyses of nightly migratory activity (Evans 2012).

## Results

We recorded seven nights of data (every night which the survey vessel spent offshore, rather than in port). We collected nighttime acoustic data on the May, June, August, September, and November 2012 and the June and August 2013 boat surveys, primarily from areas offshore of Maryland (Figure 26-1). No data were collected in winter and early spring surveys due to short daylight hours and poor weather. Most days had fewer than three detections, with the exception of one night in early September 2012 that had 123 detections (Table 26-1). Detections of non-migratory calls from seabirds (gulls, terns, storm petrels, etc.) were common, and are not presented here. Flight calls from at least 15 species of migrants were detected offshore during the seven nights of monitoring, including at least seven different warbler species.

On the night where 123 calls were detected the boat was located 30 km offshore of Virginia. Finch species accounted for the greatest proportion of identified calls (28 total detections; Table 26-2). Other commonly identified species or species groups were thrushes (14), Yellow-rumped Warblers (*Setophaga coronata*; 14) and American Redstarts (*Setophaga ruticilla;* 8). Non-passerines were present as well with 7 Least Sandpipers (*Calidris minutilla*) and 3 Semipalmated Sandpipers (*Calidris pusilla*). On this evening most calls either occurred within 1-5 hours after sunset or 9-11 hours after sunset.

Qualitatively, recording quality overall was good enough to identify passerine flight calls (Table 26-1). Recording quality on the April 2012 survey was hampered by weather and boat engine interference and was less than ideal. With this exception, recording conditions were generally favorable, with minimal overlap in frequencies between flight calls and wind and wave action against the boat.

## Discussion

This pilot study represents one of the first times that avian passive acoustic detectors have been deployed from a moving platform in the offshore environment to monitor migratory songbirds and shorebirds. Nocturnal acoustic detectors proved to be an effective means to monitor the number and type of avian species that migrate offshore. Listening conditions were frequently above average, and after some initial adjustments to the monitoring setup to minimize engine noise, species were as identifiable as in similar projects documenting terrestrial migration. Detections of non-migratory calls from seabirds (gulls, terns, storm petrels, etc.) were common, and migratory flight calls from at least fifteen species including loons, passerines and shorebirds were also detected. These results suggest that the surveys can detect migrants, and that songbirds and shorebirds are migrate offshore at least occasionally.

Songbird flight calls were only detected on two of the seven nights monitored, both occurring during early fall migration. The low frequency of our acoustic surveys prevents us from making broader ecological inferences about how many species consistently migrate over the ocean in the mid-Atlantic. Offshore migration could be intentional (thus a common strategy) or a product of wind drift (an unintentional and perhaps less common strategy). It is unclear why birds are migrating offshore in this area and how often it occurs in a season, though other studies of nocturnal avian migration over the ocean (as well as over land) have suggested that migratory activity is highly episodic and appears to be largely driven by variations in weather (Hill et al. 2014). Most acoustic detections occurred within four hours of either sunrise or sunset, indicating that there may be more migratory activity at the beginning or end of a migratory flight (or perhaps when birds are most vocal). Overwater migration in the mid-Atlantic has historically been thought to be a relatively low probability event, since wintering grounds are accessible via terrestrial migration routes in the Americas. These data and results from Chapter 27 suggest that songbirds could use overwater migratory routes more than previously thought on the Atlantic seaboard, perhaps even preferentially over terrestrial routes (Faaborg et al. 2010; DeLuca et al. 2015).

The aural quality of the acoustic surveys in this study was relatively high. We were able to minimize the effect of engine noise and wave action by placing the detector on a tower above the boat. While acoustic surveys were effective on this offshore vessel, similar tests on other larger ships have been less successful (BRI unpubl. data), with large amounts of interference coming from the ships' navigation systems. Even once this basic technical bar is passed, nocturnal acoustic surveys still have limitations. Birds migrating over the station could remain undetected due to several factors: (1) they migrated outside of the detectable zone around the detector, or (2) they did not emit a flight call while they were inside the detectable zone. While our survey clearly can detect migrants, we are underestimating the number of species and total migratory activity in all of our surveys. Hierarchical models could be useful for disentangling these effects, and are an area of needed future research for this methodology in general.

The effect of offshore development on migratory animals is commonly considered where migratory corridors have been described. Landbird migration may occur across broad geographic areas, rather than in narrow "flyways" as have been described for some waterbirds (Faaborg et al. 2010). While there may be fewer overwater migrants in the mid-Atlantic as compared to smaller ecological barriers such as the Gulf of

Mexico, there still appears to be both songbird and shorebird migration in the region. This may be particularly true during fall migration; evidence for a variety of species (including bats as well as landbirds) suggests that overwater migration in the northwestern Atlantic is much more common in fall than in spring, when animals presumably migrate preferentially over land due to consistent tailwinds from the northwest (e.g., Morris et al. 1994; Hatch et al. 2013; DeLuca et al. 2015; Chapter 23; Chapter 25). Overwater migration might not be restricted to a few species in the mid-Atlantic either; perhaps many species migrate facultatively over water as they move down the coast, when weather and physiological conditions are ideal. Offshore structures, particularly those with full-spectrum nighttime lighting, have been known to cause mass mortality events to migrating passerines in low visibility weather conditions (Hüppop et al. 2006). More data are required to understand offshore migration in this region (see Chapter 27), but these limited acoustic samples suggest that it is occurring. Until a more accurate estimate of migratory activity is ascertained, we suggest caution when making assumptions about a lack of songbird migratory activity in the region.

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## Figures and tables

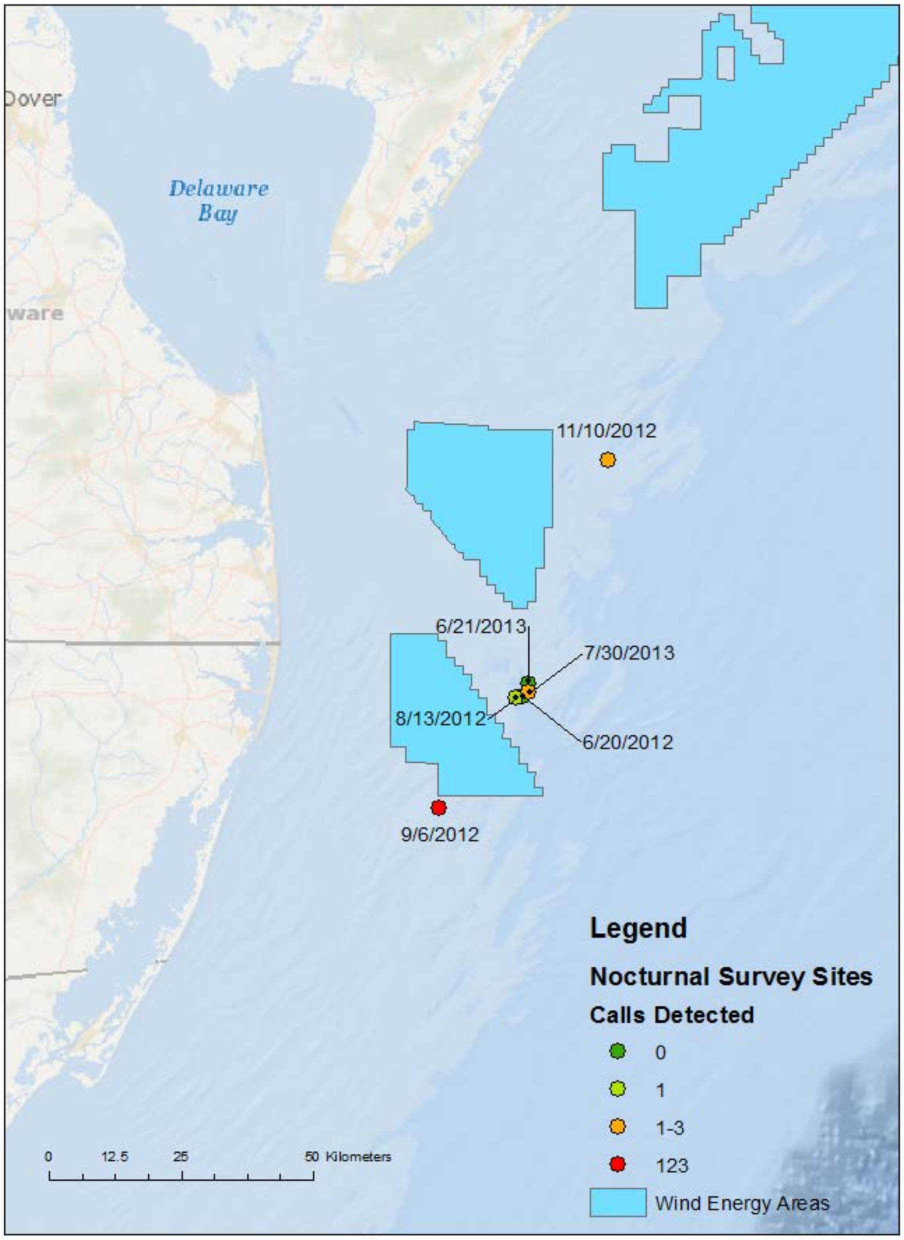

**Figure 26-1. Map of the study area with 6 of the survey locations plotted with the Wind Energy Areas for reference.** The 4/28/2012 survey was not plotted due to the boat drifting during the survey. The color of the point describes how many flight calls were detected at each of the surveys.

**Table 26-1. Summary of migratory flight calls detected during offshore acoustic surveys to date.**

| Date of Survey | # of calls detected | # of calls identified to spp. or spp. group | Conditions for recording | Comments |
|---|---|---|---|---|
| 4/28/2012 | 1 | 1 | Poor | Common Loon |
| 6/18/2012 | 0 | 0 | Good | No flight calls detected |
| 8/13/2012 | 1 | 1 | Excellent | Sparrow spp. |
| 9/06/2012 | 123 | 95 | Fair | See Table 26-2 for species summary |
| 11/11/2012 | 3 | 2 | Good | Dunlin |
| 6/21/2013 | 0 | 0 | Fair | No flight calls detected. Some wind and wave noise. |
| 7/30/2013 | 2 | 2 | Fair | Wind and wave noise throughout the file. Least Sandpiper detected twice. |

**Table 26-2. Migratory flight calls detected during the September 2012 acoustic survey.**

| Species or Group | Flight Calls Detected |
|---|---|
| American Redstart | 8 |
| Canada Warbler | 1 |
| Cape May Warbler | 1 |
| Common Yellowthroat | 2 |
| Northern Waterthrush | 3 |
| Yellow-rumped Warbler | 14 |
| Ovenbird | 2 |
| Warbler spp. | 1 |
| Song Sparrow | 2 |
| Chipping Sparrow | 1 |
| Sparrow spp. | 6 |
| Least Sandpiper | 7 |
| Semipalmated Sandpiper | 3 |
| Shorebird spp. | 1 |
| American Goldfinch | 1 |
| Finch spp. | 28 |
| Thrush spp. | 14 |
| Unknown | 28 |
| **Total flight calls detected:** | **123** |

# Chapter 27: Using WSR-88 weather radar to identify patterns of nocturnal avian migration in the offshore environment

## Final Report to the Department of Energy Wind and Water Power Technologies Office, 2015

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# Chapter 27 Highlights

*Estimating offshore nocturnal migratory activity of birds and bats using weather radar*

### *Context*[1]

While there is evidence of passerine and shorebird migration over the northwestern Atlantic for some species, oceanic flyways and migrant use of these offshore regions are poorly known. In part, this is because migration occurs over a broad spatial scale, and varies at a fine temporal scale, so efforts such as this project's boat and aerial surveys (conducted 7-8 times a year) may do a poor job of documenting migratory activity. Effective monitoring is particularly difficult for many terrestrial species that migrate at night.

An alternative approach is to use WSR-88D weather radar (also known as NEXRAD) to document migratory activity in the atmosphere. Migratory animals, from songbirds to waterfowl, are detected by these radar units over broad spatial areas. There are limitations to the distance and altitude at which migratory activity can be measured offshore, but the modeling approaches utilized in this chapter address these limitations and allow for less biased predictions of migratory activity levels at varying distances from shore. Precipitation events can interfere with the documentation of animal activities using radar, but our new analytical techniques provide the most accurate assessment of nocturnal migratory activity in the region to date.

### *Study goal/objectives*

Examine the utility of NEXRAD weather radar for studying migration off the Atlantic coast of the U.S. from New York to North Carolina, assess offshore areas for consistent migratory activity, and determine how weather influences overwater migrations.

### *Highlights*

- NEXRAD monitoring was successful in describing migratory activity at low altitudes over the ocean; often the units could detect birds 82 m above sea level. Sampling sites were located up to 80 km offshore.
- Weather interference was successfully removed from the NEXRAD data so that we could monitor offshore migration in a wider range of environmental conditions.
- Offshore activity was much higher in fall than spring. Areas south of Long Island had consistently high migratory activity in fall; activity offshore of the Carolinas was high in both fall and spring.
- Westerly winds promoted offshore activity in the fall, but not in spring. In fall, effects of wind changed with latitude: offshore activity in the northern part of the radar coverage area was highly dependent on westerlies, while offshore activity in the south was high regardless of winds.

### *Implications*

There is good evidence that birds are regularly migrating overwater up to 80 km out on the mid-Atlantic Outer Continental Shelf. Given the levels of migratory activity predicted in offshore locations, regulators for offshore wind energy development may want to consider potential impacts to migrants in development scenarios, particularly in locations with consistently higher levels of migratory activity, such as the New York Bight and areas offshore of North Carolina.

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[1] For more detailed context for this chapter, please see the introduction to Part VI of this report.

## Abstract

Developed as a tool to monitor meteorological phenomena, weather surveillance radar regularly detects volant animals in the atmosphere, and is being increasingly incorporated into biological studies of migration patterns and movements. Understanding these patterns in open water environments is of increasing interest to regulators of offshore wind energy development, but these areas often have poor radar coverage in comparison to terrestrial areas of the United States. In this study, we aimed to create a robust statistical model that can be used to predict offshore migratory activity. Initially, we examine the utility of WSR-88D (NEXRAD) weather radar for studying migration offshore, specifically off the mid-Atlantic coast of the U.S. from New York to North Carolina. Then we compare migratory activity at sites over land and up to 80 km out to sea, controlling for variables that could affect measured levels of migratory activity, and we identify the environmental variables correlated with offshore activity.

After controlling for biases in measured levels of migratory activity, we found that in fall, there was no significant difference in migratory activity at offshore vs. terrestrial sites across the radar coverage area. This suggests that migration over open water areas may be quite common during this season. During fall, offshore activity was particularly high with west winds in the northern part of the study area (e.g., New York to New Jersey). There was less migratory activity offshore in spring than in fall, though there was some in the southern part of the radar coverage area (the Carolinas). There were high levels of daily variation in migratory activity related in part to wind conditions, but some offshore areas had consistently higher predicted activity levels, most notably the New York Bight (south of Long Island) and offshore of North Carolina.

## Introduction

Weather radars such as the WSR-88Ds (also known as Next Generation Radar, or NEXRAD) are designed to detect the presence of hydrometeors such as rain drops, ice pellets, snowflakes, or hail stones in the atmosphere. These radars transmit radio waves (typically with wavelengths between about 3 and 10 cm) and receive electromagnetic energy scattered by these particles (Rinehart 2004). The time delay between when a pulse of energy is transmitted and the echo returns provides information about the distance, or range, of the hydrometeor from the radar. The power amplitude of echo signal is then used to calculate a product called the radar reflectivity factor (Z), which relies on several assumptions concerning the size, distribution, number of the hydrometeors along with their phase (liquid or solid; Chilson et al. 2012a). The bulk velocity of the collection of samples taken along the path of the transmitted and received radio wave is calculated by finding the change in phase angle of the electromagnetic signal with time.

Although developed as a tool to facilitate the monitoring of meteorological phenomena, weather surveillance radars regularly detect “bioscatter,” or reflectivity caused by biological entities in the atmosphere, such as birds, bats, and insects (Chilson et al. 2012a), and are being increasingly incorporated into biological studies (Bridge et al. 2011; Buler et al. 2012; Gauthreaux and Belser, 2003; Shamoun-Baranes et al. 2014). A discussion of how these radars detect bioscatter and how the resulting backscatter can be used to make meaningful biological inferences has occurred in the open literature (Bridge et al., 2011; Chilson et al. 2012; Chilson et al. 2012a; Robinson et al., 2010).

Migration is a difficult phenomenon to study, in part due to the large geographic scale at which it occurs. The long-distance migration of many passerines, shorebirds, and other avian taxa also occurs at night and often at high altitudes, making visual observations difficult or impossible. For these reasons, weather radar can be an attractive approach for studying large-scale migratory movement patterns in volant taxa. Monitoring can be conducted at any time of day or night (weather permitting), which is a clear advantage for studying nocturnal phenomena, including migration and bat emergence from roosts (Chilson et al. 2012; Frick et al. 2012). As weather radar measures collective reflectivity of biomass in the aerosphere, it is also an effective way of monitoring populations whose individuals are too small-bodied to effectively track via satellite telemetry or similar approaches. Though they lack the fine scale resolution of traditional marine radar (e.g., Harmata et al, 1999, Brabant et al. 2012), NEXRAD data allow for efficient monitoring of spatial and temporal patterns in migration on a broad scale (Gauthreaux and Belser,2003), and may prove useful for developing a better understanding of environmental factors affecting migration and for identifying major migration routes.

Recently, methods have been developed to analyze NEXRAD data to estimate the relative abundance, direction of movement, speed, and altitude of migrants based on radar reflectivity data (Diehl et al. 2003; Horn and Kunz 2008). When multiple animals are present, radar reflectivity values express cumulative reflectivity per volume of sampled air. The size of an individual migrant can affect cumulative reflectivity; a goose, for instance, reflects back more microwaves than a warbler, so without knowing the identities of migrants, reflectivity alone cannot be directly translated to abundance of animals. However, reflectivity is directly correlated with biomass density (a combination of the number of individuals present in a given airspace multiplied by the average body size of those individuals) in the atmosphere, and as such can be regarded as an index of migratory activity at the measured altitude (Buler and Diehl 2009; Chilson et al. 2012a). It should be noted that with distance from the radar station, the average height of the volume of air sampled by the radar beam increases in altitude, and the power of the beam weakens. Thus it becomes increasingly difficult to detect low altitude and low density bioscatter with increasing range from the radar.

NEXRAD radars are located throughout the United States in order to provide comprehensive coverage for monitoring of meteorological phenomena (Maddox et al. 2002). However, monitoring of offshore areas is limited to coastal radars whose coverage partially extends into the marine environment. Perhaps as a result, the utility of NEXRAD data for predicting migratory patterns in the marine environment is relatively unstudied, though it has been examined in the Great Lakes, where radar coverage is more comprehensive (Diehl et al. 2003). One of the only studies to examine this topic was focused on areas offshore of New Jersey, as part of a baseline study of offshore wildlife in relation to future offshore wind energy development (Geo-Marine Inc. 2010). Data from the KDIX radar, located in Mount Holly, New Jersey, indicated that migratory activity was substantially greater in coastal than offshore areas, although this study did not fully account for the relatively poor and high altitude-biased coverage available offshore (since those areas were located farther from the radar). Migratory activity in coastal areas of New Jersey was highly variable from year to year, but was two to three times greater during fall migration than in spring (Geo-Marine Inc. 2010).

Based on this and other studies of NEXRAD radar and migration, we suggest that the variables that influence migratory activity at a given site (or our ability to detect that activity using radar) include:

- Site characteristics, such as the distance and of the site from the coast, and whether the site is on- or offshore;
- Relationship of the site with the NEXRAD station, such as the distance of the site from the nearest radar unit, and which unit that site is closest to;
- Temporal variables, such as time of year, season, and year; and
- Weather, such as wind speed and direction.

In this study, we examine the utility of NEXRAD data for studying migration offshore, and specifically off the Atlantic coast of the U.S.; we compare NEXRAD migratory activity at land and water sites, controlling for variables that could affect measured levels of migratory activity; and we identify potential offshore migration pathways and timing, as well as the environmental and temporal variables correlated with these patterns. The study area for this effort, hereafter referred to as the "radar study area," includes locations from New York to North Carolina, a broader geographic range than the mid-Atlantic study area discussed in previous chapters. This expanded area was examined due to poor NEXRAD coverage of the Outer Continental Shelf (OCS) offshore of Maryland and Virginia. Because direct assessment of some habitats in the mid-Atlantic study area was impossible, we chose to describe offshore migration more broadly along the mid-Atlantic eastern seaboard. The objective of this analysis is to create a robust model that can be used to predict offshore migratory activity across the radar study area in relation to topography and environmental conditions. This model may prove useful for understanding the potential interactions between nocturnal migrants and offshore wind energy development along the Atlantic coast of the United States.

### *A short primer on weather radar and uses in aeroecology*

Radar data used for this study were stored at the National Oceanic and Atmospheric Administration's National Severe Storms Laboratory database, housed at the National Weather Center on the University of Oklahoma (OU) campus. Radar reflectivity factor (Z) data from all WSR-88D sites in the NEXRAD network for the contiguous United States (CONUS) are combined to create a mosaic GeoTIFF file for each of six subdomains (tiles)[2]. The mosaicked data are stored as both "quality-controlled" (reflectivity values believed to have originated from non-meteorological sources have been removed) and "un-quality-controlled" files (all sources of radar backscatter are present, including bioscatter). The quality-controlled data are available as 3-D fields and as 2-D raster projections onto the Earth's surface. When constructing the projections, the maximum available reflectivity value in height for a particular raster element in the 2-D projection is used. This is called a composite reflectivity (CREF). Composite reflectivity projections consisting of the un-quality-controlled data are also available ("UNQC_CREF" data), though no 3-D data are stored for the un-quality-controlled fields. CREF and UNQC_CREF have a spatial resolution of 0.01° x 0.01°. Prior to mid-2013, they were stored every 5 minutes; since that time period, CREF and UNQC_CREF have been stored every 2 minutes.

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[2] http://nmq.ou.edu/

Figure 27-1 and Figure 27-2 show UNQC_CREF data for 2000, 2300, 0200, and 0500 EDT for May 1, 2011 and September 1, 2011, respectively. For both months, 2000 EDT corresponds approximately to the time of sunset along the Atlantic coast of the United States. This can be seen by the strobes of high reflectivity directed roughly towards the west. These are "sun spurs" created by the radars' detection of radiation from the setting sun. In the September image, the sun spurs occur further west, indicating that the sun had already set along the coast at 2000 hours. Other spurious echoes can also be seen in the radar images; as the UNQC_CREF data have not been quality controlled, they contain such effects as sun spurs, radio interference, ground clutter, sea clutter, and bioscatter, as well as precipitation features. For bioscatter, one expects to see the signal diminish with range from the radar, since the biological entities are predominantly near the earth's surface. The bulk of the reflectivity data seen in Figure 27-1 and Figure 27-2, especially at night (2300 and 0200 hours) is the result of nocturnal migrants in the atmosphere.

Generally speaking, the minimum height of the atmosphere that can be probed by a fixed location radar increases as a function of range from the radar for a given antenna elevation angle. There are five primary contributing factors to the minimum height of the radar beam above ground level: 1) antenna elevation angle; 2) antenna beam width; 3) curvature of the Earth; 4) thermodynamic properties of the atmosphere, which affects refraction; and 5) the local topography. The culmination of the first four effects is depicted in Figure 27-3. Since biological scatter is expected to primarily occur near the Earth's surface, only the lowest elevation angle (0.5°) is depicted in the figure. The sample volume for weather radar is usually based on the 3-dB (half power) width of the beam (Figure 27-3). However, biological scatterers can have relatively large cross sections, making them quite "bright" to the probing radar, and thus correspondingly easily detected at lower powers (Chilson et al. 2012a). Therefore, we elected to use the 6-dB point when assigning the beam width for this study.

## Methods

### *Radar data collection*

CREF and UNQC_CREF data from the National Severe Storms Laboratory were transferred to a computing cluster at OU explicitly dedicated to processing and analysis of radar data for aeroecological applications. Six WSR-88D sites along the east coast of the United States were selected for this study (Figure 27-4):

- KOKX (Upton, NY: 40.8656 N, 72.8647 W)
- KDIX (Mt. Holly, NJ: 39.9469 N, 74.4111 W)
- KDOX (Dover Air Force Base, DE: 38.8256 N, 75.4400 W)
- KAKQ (Wakefield, VA: 36.9839 N, 77.0025 W)
- KMHX (Morehead City, NC: 34.7761 N, 76.8767 W)
- KLTX (Wilmington, NC: 33.9892 N, 78.4292 W)

Twenty-four sampling sites were chosen around each radar, for 144 sampling sites in total. Data processing for these sites was initialized according to the workflow outlined in Figure 27-5. Sites were located in three concentric rings, centered at the radar, with radii of 30, 60, and 90 km (similar to Williams et al. 2013).

Along each ring a sampling site was located on the eight cardinal and intercardinal compass positions (Figure 27-4). This configuration of sampling sites allowed us to examine potential differences in biological activity at sites that had approximately the same radar beam height (because they were the same distances from the radar), but had differing characteristics in other respects (such as location over land vs. water). The non-uniform distribution of NEXRAD units across the eastern coast of North America led to coverage gaps in the mid-Atlantic OCS, particularly off the coast of Maryland and Virginia.

For all reported data, radar reflectivity factor values (Z), which are used for hydrometeors (e.g., for weather detection), were converted to the more biologically relevant metric of reflectivity η (“eta”), or scattering area per unit of volume ($cm^2$ per $km^3$; Chilson et al. 2012b). Log transformations of reflectivity data are common practice in the field of radar aeroecology to normalize reflectivity data for analyses (Chilson et al. 2012b). When values of η are log-transformed with a reference value of $\eta_o = 1$ $cm^2$ per $km^3$, the resulting quantity is known as dB η. The log-transform follows the relationship: dB η = $10*\log_{10}(\eta/\eta_o)$, so a value η = 1000 $cm^2/km^3$ is equivalent to a dB η of 30.

We developed tools in MATLAB and open-source Geographic Information System (GRASS GIS[3]) to process four years of May, September, and October NEXRAD reflectivity radar data, weather data, and the vector images for the 144 sites (2010-2013). Data used for the analysis were the raster CREF and UNQC_CREF files. For each of the 144 sample sites, a localized collection of raster elements were extracted (Figure 27-6), and the combined data were averaged in time across a six hour time period centered around midnight for each sampling date. These averages were used when calculating the spatial univariate statistics of dB η (e.g., the minimum value to reduce the chance of spurious high reflectivity radar interference in the 5 minute interval) for both CREF and UNQC_CREF data for each site and time step. The results were written to a CSV file (Figure 27-7). Dedicated computer resources for aeroecological applications with multi-core processing capabilities were assigned to this processing-intensive task. When no other processes were running on these computer resources, it required 2-3 days to process one month of radar data for a single file type (CREF or UNQC_CREF).

To the extent that the quality control procedures developed by the National Ocean and Atmospheric Administration (NOAA) are accurate, the CREF data should only contain reflectivity signals created by weather. Since weather signals generally mask or obscure the presence of bioscatter, it is common to only use weather radar during clear conditions when conducting biological studies. Often, researchers will discount entire radar scans if any precipitation is present within any spatial region or time of the scan (e.g., Geo-Marine Inc. 2010). As an alternative, we developed a method in MATLAB which the CREF data were used to “mask” the times and locations corresponding weather from the UNQC_CREF data, allowing preservation of the remainder of these scans for analysis (Figure 27-8). The resulting combined data (univariate outputs for each site and each time step after masking) were used for the remainder of data processing.

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[3] http://grass.osgeo.org/

### Covariates

For each site, we identified four static explanatory variables (radar unit, distance from radar unit, site type, and distance from shore). Each site's latitude and longitude were included as continuous variables. Sites were located in sets of eight, at 30 km, 60 km, or 90 km from each radar, with distance band and radar unit included as a categorical variable for each. Each site was classified as to whether it was over water or over land (site type), and how far away it was from the coastline, using the National Land Cover Database (2011) in ArcGIS 10.3 (ESRI, Redlands, CA). Negative values indicated the site was over land (inland from the coast) and positive values indicated the site was over the ocean.

Weather covariate data were collected and compiled from airports and buoys located in proximity to the 144 study sites for the spring and fall migration seasons from 2010-2013. Using package 'weatherData' (Narasimhan 2014) in the R Statistical Computing Environment (R Core Team 2014) we gathered weather data from 1900-2400 EDT each day from a weather station at each of the NEXRAD units. For each day, we collected wind speed and wind direction data, and linearized the circular direction data into two vector components: U wind (East/West, with westerlies, or eastward winds, taken as positive) and V wind (North/South, with northerlies, or southward winds, taken as positive). We then used the median U and V value for each evening. The 24 sites associated with each radar unit were assigned values of dynamic variables from the weather station at that radar unit. Calendar date and year were also included in models.

Generally, we included a variable in a model because (1) it was involved in a hypothesis that explained migratory activity, or (2) it was a variable that confounded our ability to understand the biological explanations for these patterns. Variables of the second type, or "nuisance" variables, included year, calendar date, elevation (the ground can clutter radar reflectivity and alter migratory patterns), and distance from the radar unit (as previously discussed, the beam has lower reflectivity and increased elevation with distance). Site type was also included as an explanatory variable in weather models, specifically to allow weather patterns over land and water to lead to different kinds of radar interference. All other variables were used to explore how time of year, weather and distance from shore influenced migratory activity.

### Modeling bioscatter

To determine where animals migrated offshore in the mid-Atlantic region, and the conditions under which they migrated, we used a total of four different general linear mixed models (two each for the spring and fall migration seasons). Average reflectivity (dB η) was paired with static and dynamic explanatory variables for each site and date. Reflectivity is a normally distributed variable (as it is already converted to the logarithmic dB scale to normalize the data), so the Gaussian family was used with an identity link using package "lme4" in the R Statistical Computing Environment (Bates et al., 2014). Each model was tested for overall fit by withholding 20% of the data, and comparing predicted responses using the model to actual responses in the withheld data.

We used two different models to describe migratory activity: a "spatial" model designed to examine general patterns of migratory activity at sites across the radar study area (in relation to static covariates), and a "weather" model that explored the day-to-day variation in migratory activity across all sites. The spatial model was designed to quantify the geographic variation in dB η across the 144 sites. Calendar date, year, and site were used as random variables to account for unexplained variation in reflectivity across space and time. The fixed effects in the model were the radar unit, distance to shore (modeled as continuous), distance to the radar hub (a categorical variable, as sites were located in three distinct distance bands), and the average elevation of the site. We added interactions between radar and distance to shore to account for potential differences in inland versus open water migration patterns for each radar unit (e.g., to help account for differences in migration patterns related to local topography and sites' positions relative to the coast). Once we controlled for nuisance variables like distance to radar and elevation, we then used this model to predict the average dB η values for each site in both seasons.

In the weather model, we wanted to focus on how weather was influencing the amount of bioscatter over the ocean across the radar study area. Site and year were included as random effects (the inclusion of date as a random effect over parameterized this model). Radar unit, distance from the radar hub, elevation, and site type were used as fixed effects. We also included two high level interactions, and all of their parent base effects and interactions, to predict the effects of winds on offshore dB η at various latitudes in the study region: 1) two-way interactions between east/west wind, distance to shore, and latitude; and 2) two-way interactions between north/south wind, distance to shore, and latitude. These interactions allowed us to explore variation in the numbers of animals over the ocean with regards to both the local winds during migration and to sites' positions along the coast. Longitude was also included as a base effect, but was not interacted with other variables due to a correlation with latitude and distance to shore in our models. To test for significance, we calculated the 95% confidence intervals of the individual parameters using a parametric bootstrapping method in package "lme4" and a likelihood ratio test for groups of effects (e.g., radar unit).

## Results

### *Assessment of the utility of radar data for measuring migratory activity offshore*

We calculated the predicted beam height as a function of distance from a given radar unit (Figure 27-3). This plot assumes that the Earth is perfectly round (which is not accurate, but provides a reasonable approximation of the earth's curvature) and takes into account the effect of refraction as the beam propagates through the atmosphere (assuming standard atmospheric conditions). We projected predicted beam height values onto the radar study area, and used digital elevation maps with ArcGIS and GRASS to calculate the lowest detectable beam heights for all 24 points at each of our six focal NEXRAD stations (Figure 27-9). While water sites tended to be farther away than land sites (because the radar units are positioned over land), the minimum beam height did not appear to be high enough to be a limiting factor in analyses; the average minimum beam height at which we could expect to detect bioscatter at our 144 sites was 82 m (Figure 27-9), and the highest minimum beam height for our sites was 226 m. There was little variation in minimum beam height among the six units, likely because there are few major elevation changes in this region of the United States.

To test our ability to detect bioscatter at the selected NEXRAD stations, we selected a particularly active night in spring migration (May 1, 2011). On this night, migratory activity erupted at sunset and tapered off as the night progressed (Figure 27-1). Weather was mild on this evening, though rain showers appeared in the final two frames at 06 and 09 UTC (0200 and 0500 EDT). Bioscatter was clearly detectable throughout the night. Reflectivity decreases with distance from the NEXRAD unit, due to decreasing beam power, and this remains a confounding factor in this visualization. However, biological scatter was detected at least 60 km from the radar over land on this date, so we expect that bioscatter would also be detected over water if it occurred within this range (at least within the range of beam heights defined in Figure 27-3 and Figure 27-9). Indeed, during a night of fall migration (September 1, 2011), bioscatter clearly occurred over the ocean to the south of KMHX (Figure 27-2).

We compared migratory activity over land and water at sites that were the same distance from the radar unit on the same date (May 1, 2011) for the two North Carolina radars (Figure 27-1). Bays and inlets were treated as “land” in this analysis, rather than being lumped with open ocean locations. The KLTX unit had more activity overall than the KMHX unit on this date, and migratory activity was consistently higher over land than water for both these radar units. The innermost eight sampling points for each of the units showed a rapid increase of activity at sunset over land sites (also called “migratory exodus”) as migrants left terrestrial habitats to begin their nocturnal flights (Gauthreaux and Belser, 2005); this exodus activity did not occur at water sites (Figure 27-10). Where there was activity over water, it occurred later in the evening. While activity over the open water was low on this particular night in 2011, our analysis suggests that this was due to a lack of overwater bioscatter rather than to biases in detection.

### *Analysis of migratory activity offshore: Fall*

The spatial and weather models had 0.71 and 0.63 Pearson correlations, respectively, between the known response and predicted response in the withheld data, which indicated a reasonable model fit. Residuals for both models also qualitatively appeared to be randomly distributed, suggesting good overall fit.

In the spatial model, we found that radar unit and distance to the radar unit were important to predicting dB η levels, as well as the interaction between radar unit and distance of the site to the coast (Table 27-1). The random effect of date explained 17.9% of the total variance in the model, while site was less important, only explaining 5% of the variance. dB η decreased at 60 km (β=-6.6; 95% Confidence Interval=-7.0, -6.2) and 90 km (β=--6.0; 95% CI: -6.4, -5.6) away from the radar hub when compared to sites in the 30 km band (where β is the estimated effect of the variable on reflectivity and the confidence interval is the range in which we are 95% certain that this parameter value is contained). Radar unit was also important ($F_5$=6.7), with KLTX showing more activity (β=1.0; 95% CI: 0.3, 1.7) than the other units. Distance to shore was not significant as a single effect or in an interaction with radar; effectively, the model could not distinguish between sites on- or offshore in terms of reflected bioscatter during fall, suggesting that migratory activity was similar onshore and offshore within the radar coverage area. Elevation of the site had a positive effect on dB η (β=7.3E-3; 95% CI: 4.9E-5, 1.5E-2).

After correcting for biases due to elevation, radar unit, and distance to the radar, the model predicted considerable migratory activity offshore in the fall (Figure 27-11). There was no apparent decrease in bioscatter signal over the ocean on average, though there was wide variation in migratory activity levels between locations along the coast. Sites offshore of Long Island and coastal North Carolina appeared to have higher levels of migratory activity over water, on average, than locations off the New Jersey and Delaware coasts.

In the fall weather model, we found that in addition to radar unit and distance to the radar, weather and relative spatial position was also important to determining when birds were migrating offshore (Table 27-2). Site and year only made up 6% of the total variance in the model. Radar reflectivity was influenced by significant interactions between U wind and latitude (β=2.1E-3; 95% CI: 8.7E-3, 2.6E-2), U wind and distance to shore (β=1.1E-7; 95% CI: 8.1E-7, 2.3E-6), and V wind and latitude (β=-9.1E-3; 95% CI: -1.5E-2, -3.9E-3). Westerly winds led to more offshore activity at all latitudes, but particularly in the northern part of the radar study area (New York/New Jersey). In the southern part of the radar study area (the Carolinas), offshore activity was equally high in westerlies and easterlies, but activity levels varied in terrestrial areas (Figure 27-12). The positive effect of northerly winds on reflectivity at higher latitudes was primarily true onshore (Figure 27-13); offshore levels of bioscatter were predicted to be the same regardless of North-South winds. The opposite was true at lower latitudes within our radar study area, where strong northerlies led to decreases in offshore activity relative to onshore activity.

### *Analysis of migratory activity offshore: Spring*

Model fit during the spring was good, with the spatial model showing a 0.78 Pearson correlation between known and predicted response for the withheld data, and the weather model showing a 0.69 Pearson correlation. Both models were above average in predicting within-study bioscatter.

The spatial model predicted there to be little activity over the ocean during the spring in the mid-Atlantic (Figure 27-14). The one location of high activity was offshore of the border between North and South Carolina, and was primarily due to unexplained random effects. The spatial model suggested that radar unit, distance from the radar, distance to shore, elevation, and the interaction between radar and distance to shore were all important to predicting bioscatter (Table 27-3), with distance from radar by far the most important. Date and site each made up 37% of the total variance, leaving fixed effects to explain no more than 26% of the total variance. Most radar stations were similar in activity except KMHX in NC, which was below average (β=-1.9; 95% CI: -3.5, -0.33). Reflectivity was positively correlated with elevation (β=1.9E-2; 95% CI: 2.7E-3, 3.5E-2). There was a decrease in bioscatter with distance to the radar hub; sites at 60 km were 5 dB η less (95% CI: -5.9, -4.1) and sites at 90 km were 5.5 dB η less (95% CI: -6.5, -4.5) than sites located 30 km away. Bioscatter alone did not show a strong decrease with distance from shore (β=-9.5E-6; 95% CI: -4.1E-5, 2.4E-5; $F_1$=22.5), but it was important in interactions with radar unit, showing particularly significant declines at KDIX (β=-5.6E-5; 95% CI: -9.9E-5, -1.3E-5) and KDOX (β=-6.4E-5; 95% CI: -1.3E-4, -9.9E-6). In areas around Delaware and New Jersey migratory activity was highest at inland sites, and we documented higher migratory activity on or near the shoreline than in areas farther offshore.

The most important terms in the weather model, aside from nuisance variables like distance from radar and elevation, were interactions between latitude and distance to shore, both wind vectors and distance to shore, and the base effect of east/west winds (Table 27-4). As a random variable, site made up 16.7% of the total variance. The predictions from this model suggest a differing process for overwater activity in different subregions of the radar study area. Overall, there was less activity in more eastern longitudes (β=-2.3; 95% CI: -3.0, -1.6), even when controlling for distance from shore. Westerly winds generally led to more migratory activity over land but less over water (β=-1.8E-6; 95% CI: -2.8E-6, -7.9E-7; Figure 27-15). Similarly, southerly winds increased onshore activity and decreased offshore activity with no variance by geography (β= -2.0E-6; 95% CI: -3.0E-6, -1.2E-6; Figure 27-16). Unlike fall, there were significant differences in reflectivity offshore by latitude (β=-1.2E-5; 95% CI: -1.6E-5, -7.5E-6); the southern part of the region (the Carolinas)had significantly more activity over water than over land, while in the central and north regions (the mid-Atlantic and New York/New Jersey, respectively), offshore and onshore levels of migratory activity were similar (Figure 27-17).

## Discussion

By using currently available NEXRAD radar technology, we assessed the migratory activity levels of aerofauna in the mid-Atlantic region and in a broader area along the eastern seaboard, both on- and offshore. While radar coverage of areas offshore of Virginia and Maryland was limited due to the placement of NEXRAD radars in the region, this analysis represents a broad assessment of offshore migration over much of the OCS for the eastern United States. As with a previous study in the region (e.g., Geo-Marine Inc., 2010), our analysis indicated that there was more migratory activity over the ocean during the fall than during the spring. Unlike this previous study, we corrected for bias in measured values based on distance from the radar, elevation, and other nuisance variables, then quantified the patterns of migratory activity in space relative to distance from shore, and determined the influence of weather on migratory activity offshore. These corrections, along with the broader geographic range examined in this study, led to different findings from the previous work in New Jersey. In particular, after correcting for nuisance variables we found that migratory activity did not decrease with increasing distance from shore in the fall (within our radar study area, at least, which included locations up to 80 km from the shoreline). There was a decrease in migratory activity with distance from shore in the spring, but only offshore of Delaware and New Jersey. While NEXRAD has limited range offshore in some areas, and we had few sampling points in the Delaware, Maryland, and Virginia Wind Energy Areas, using multiple radar stations allowed us to survey a broad area for patterns in nocturnal migratory activity over multiple years. Pairing this study with other recent advances in our understanding of overwater migration in the region (DeLuca et al., 2015; Johnson and Connors, 2010; Williams and Williams, 1990; Chapters 19-24), these data suggest a consistent pattern of overwater aerofauna migration during fall.

### *Evaluating the usefulness of NEXRAD for detecting overwater bioscatter*

Based on beam geometry, the Earth's curvature, and standard atmospheric models of radio-wave refraction, at a range of 90 km away from the radar unit, bioscatter resulting from nocturnally migrating birds should be detected above ~300 m altitude with a quarter power beam, and above 500 m with a half-power beam. Consequently, much of the aerosphere where nocturnal avian migration occurred could be surveyed by weather radar. Not all radar units are close enough to the coast to survey over the ocean, but those that are located in coastal areas should have no issues doing so (though an increasingly large portion of the lower aerosphere will go unsurveyed in places farther from the NEXRAD unit).

Most of the 144 sites surveyed for this study had a minimum beam height similar to the altitude of the potential rotor-swept zone for offshore wind turbines. The rotor-sweep zone of offshore wind turbines varies by turbine size and type, but may include altitudes between approximately 20 m and 200 m (Willmott et al., 2013). The rotor-sweep zone for a Siemens 3.6 MW offshore turbine is approximately 22-142 m in altitude, while a larger Siemens 6 MW turbine's rotors reach between about 25 m and 179 m in altitude. Larger turbines for offshore deployment are currently in development, and the altitude of the rotor-swept zone will continue to increase as the technology develops; a prototype Vestas 8 MW turbine, for example, reaches 222 m in altitude. The highest minimum beam height in this study was 226 m, and 73% of the sites had a minimum beam height that overlapped with the height of current commercially available technology (about 180 m in height). While we corrected for distance from the radar in this study, thus theoretically negating the effects of distance from radar on predicted migratory activity levels, this assumes that bioscatter is linearly correlated across altitudes, which may not always be the case. The high level of altitudinal overlap between our measurements and turbines heights, however, suggests that our predictions of bioscatter levels in the offshore environment are likely to be highly relevant to migration occurring at rotor-sweep heights in the mid-Atlantic.

### *Patterns of migration along the Atlantic coast of the United States*

Migratory activity was highest in the fall, particularly off the coast of the Long Island and North Carolina. Topography and weather patterns could be consistently facilitating overwater migration in these areas. Alternatively, these locations around Pennsylvania and the Carolinas may be where breeding populations from northwest and northeastern North America converge (e.g., the many boreal-breeding warblers), leading to more birds being present in these areas in general during migration. Bioscatter was generally high in the southern part of the radar study area, suggesting that this second explanation may have some merit.

Tailwinds are crucial for successful migrations in birds (Alerstam et al. 1990, Jenni et al. 2003). In the present study, we found that offshore activity in fall was higher under westerlies, particularly in the northern parts of the radar study area. In the New Jersey/Long Island areas, westerly winds either caused birds to drift over the ocean or were used as a cue for birds to attempt an overwater migration. In the south, offshore activity was similar in all wind conditions. Perhaps this geographic position is so effective for beginning an overwater journey that birds are willing to make this trip under a wide variety of conditions. Migratory activity onshore showed strong influences of tailwinds as well; northerlies promoted migratory activity in the north and westerlies in the south.

In the spring, there was considerably less migration over water than in fall, with high levels of offshore activity only predicted in a small area off the coast of North Carolina. Animals still migrated over water, but our spring reflectivity mapping suggests these migrants stay closer to shore, rather than attempting the long offshore trips seen in fall. In the central and northern parts of the radar study area, we did not see more birds over water under tailwind conditions (southerlies and westerlies), so birds did not appear to be crossing the ocean during peak migratory conditions. This might indicate that birds fly overwater as a mistake during this time period, where headwinds perhaps force a mid-flight redirection. While migrants are known to cross water barriers during the spring (including a substantial migration of a wide range of species across the Gulf of Mexico; Moore et al. 1990), they did not appear to do so in the radar study area. The lack of consistent southerlies in spring may preclude this from being an evolutionarily stable strategy, as without reliable tailwinds, the risk of an overwater crossing may overwhelm the reproductive benefit of arriving early at the breeding grounds.

### *Sources of uncertainty*

Distance from the radar was associated with a non-linear decrease in reflectivity. As the reduction of beam strength and the increase in sampling volume with distance from the radar is already accounted for in the calculation of dB η (Chilson et al. 2012b), this suggests that other processes are affecting our understanding of migratory activity at longer distances from the radar unit. When interpreting the backscattered signal from a collection of scatterers, one assumes that the scatterers are uniformly distributed throughout the sampling volume. Clearly this condition becomes increasingly difficult to fulfill as the size of the sampling volume increases (as with increased distance from the radar unit). This is particularly true in the case of migration, which is not evenly distributed throughout the airspace. Although birds are relatively “bright” to a weather radar compared to hydrometeors, the number and density of birds is much less than that for precipitation. Consequently, the loss of sensitivity due to clumped distribution within the sampling volume could be a significant source of the decrease in reflectivity at 60 and 90 km from the radar unit.. There is also a potential biological explanation for this decreased reflectivity. While minimum beam height was only ~200m above sea level at 90 km from the radar unit, it is possible that large numbers of birds were migrating below that altitude, and we were unable to sample most of the migrating population in these areas. In this analysis we assume that the data from the 30 km distance band is the least biased due to the smaller sampling area and lower beam height at this distance from the radar; we correct values in the other distance bands such that they display data as if they were sampled at 30 km. This correction allows for an “apples to apples” comparison of average reflectivity values between locations (as in Figures 27-11 and Figure 27-14), but our uncertainty about the biological or physical mechanism behind these differences makes it difficult to ensure that we are completely correcting for these disparities.

Future areas of research should include a focus on the efficacy of weather radar for detecting bioscatter over open water. “Ground-truthing” the weather radar data with vertical marine radar (if possible offshore) and passive acoustic detection would go a long way towards confirming the patterns we have seen in this study. As radar technology improves, the ability to analyze overwater data by elevation band will be an interesting means of describing the altitudinal distribution of migrants, and may allow for better modeling of activity in low altitude areas far from radar units. Individual radars often have slightly different settings or other factors that could influence measured reflectivity values, and more detailed studies of the spatial and temporal variation in migratory activity between individual radar units would also be useful for understanding the limitations of each unit’s efficacy in surveying the ocean. Lastly, gathering data from marine habitat offshore of Virginia and Maryland would be helpful for confirming patterns of migratory activity in this area. While we can assume that migratory activity there is similar to areas north and south of it (as animals would have to migrate over the site to get to the next location), direct measurement of migratory activity at these locations was lacking due to the inland locations of nearby NEXRAD units.

### *Implications for development along the eastern seaboard*

While we have known for some years that birds and bats migrate over the northwest Atlantic Ocean (Williams et al. 1977, Gauthreaux and Belser 2005), this study identified areas that were consistently used by migrants in both spring and fall, as well as the conditions under which offshore migration was most likely to occur in various locations along the eastern seaboard. Our findings suggest that there is more nocturnal migratory activity offshore than predicted by past studies. We suspect the newer methods employed in this study, including controlling for distance from the radar and other nuisance variables, improved our ability to obtain accurate estimates of migratory activity offshore. Migratory activity levels were quite variable, and it is also possible that we merely chose a four-year period with higher than normal levels of offshore activity. But the geographic variability in these predicted values, and the consistency of these patterns in certain locations along the eastern seaboard, particularly in fall, suggest that there may be substantial offshore migration pathways that begin with “jumping off points” along the coast.

Overwater activity is much more widespread in the fall than in spring. Paired with data collected from our shipboard surveys (Chapter 26), these data suggest that an overwater migration strategy may be utilized by many species, particularly songbirds and shorebirds. Trans-oceanic migrations, once thought to be extreme events only undertaken by few individuals or species with extreme physiological adaptations (e.g., DeLuca et al. 2015, Delingat et al. 2008), are perhaps more commonplace than previously thought in this region. Given the levels of migratory activity predicted in offshore locations, regulators for offshore wind energy development may want to consider potential impacts to nocturnal migrants, including terrestrial species (passerines, shorebirds, bats, etc.) in offshore wind development scenarios. This may be particularly important in locations with consistently higher levels of migratory activity, such as the New York Bight and areas offshore of North Carolina. Predicted levels of bioscatter in many other parts of the radar study area were also intermittently high, however, suggesting that offshore migration is a widespread phenomenon, and should be regarded as such during planning activities.

Our work here suggests that weather radar systems can be effective in describing overwater migration, and that there are parts of the ocean that are consistently being used by migratory aerofauna. In particular, our findings demonstrate a strong pattern of overwater migration in aerofauna during fall, and the variation in overwater migration activity in relation to weather patterns suggests that overwater migration may be facultative for many taxa. More research into these areas will allow us to better describe this phenomenon, and inform decision making regarding the anthropogenic utilization of the marine environment.

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## Figures and tables

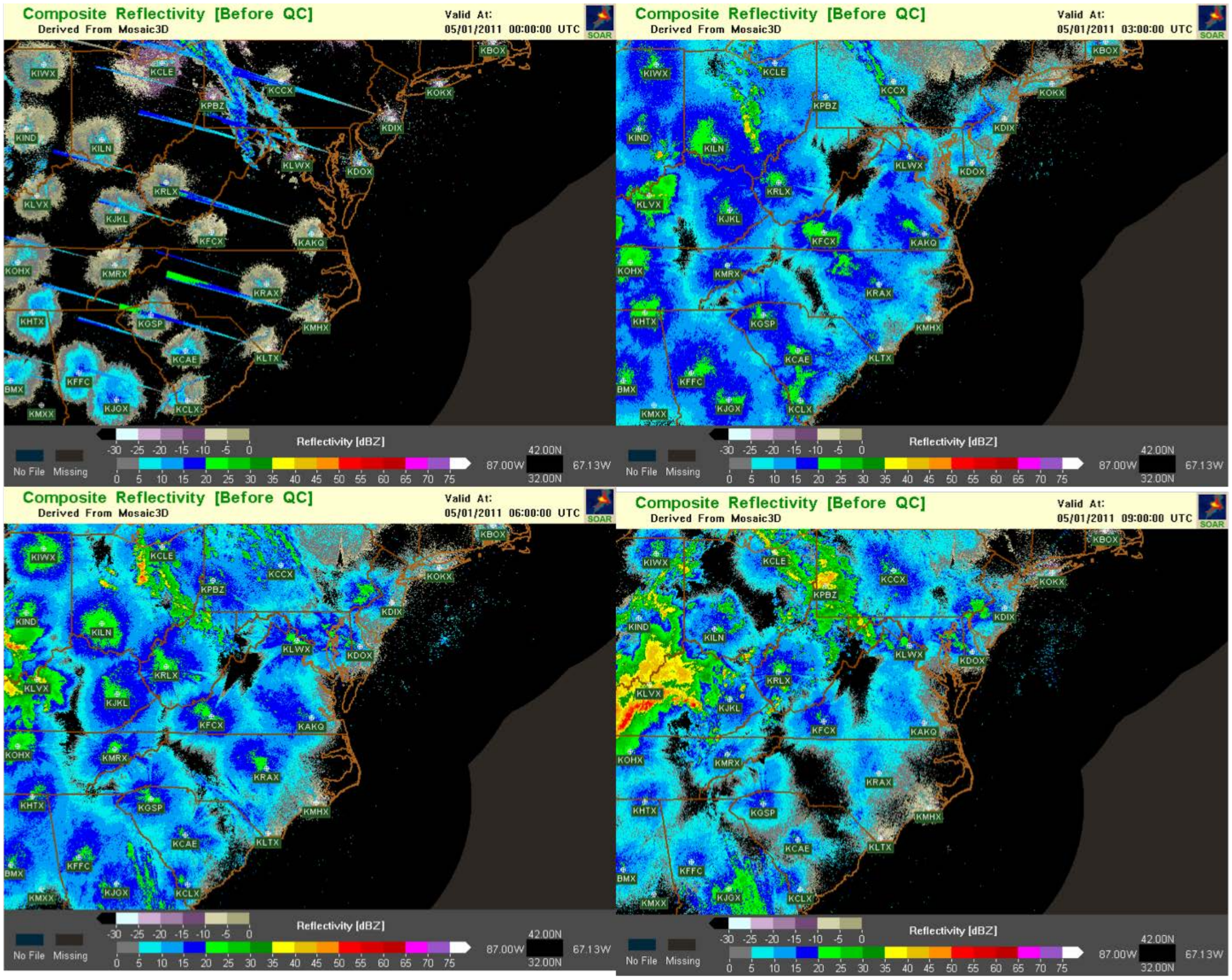

**Figure 27-1. Composite radar data (expressed in dB Z) for May 1, 2011.** The radar has not been quality controlled to remove biological scatter, radio interference, or anomalous propagation. The four panels correspond to times of 00, 03, 06, and 09 UTC (2000, 2300, 0200, and 0500 hours EDT). Typical precipitation features are seen (irregular green, yellow and red areas), as well as the onset of nocturnal migration (round “blooms” around the radar units). 00 UTC (2000 EDT) also shows “sun spurs” caused by the setting sun (see text).

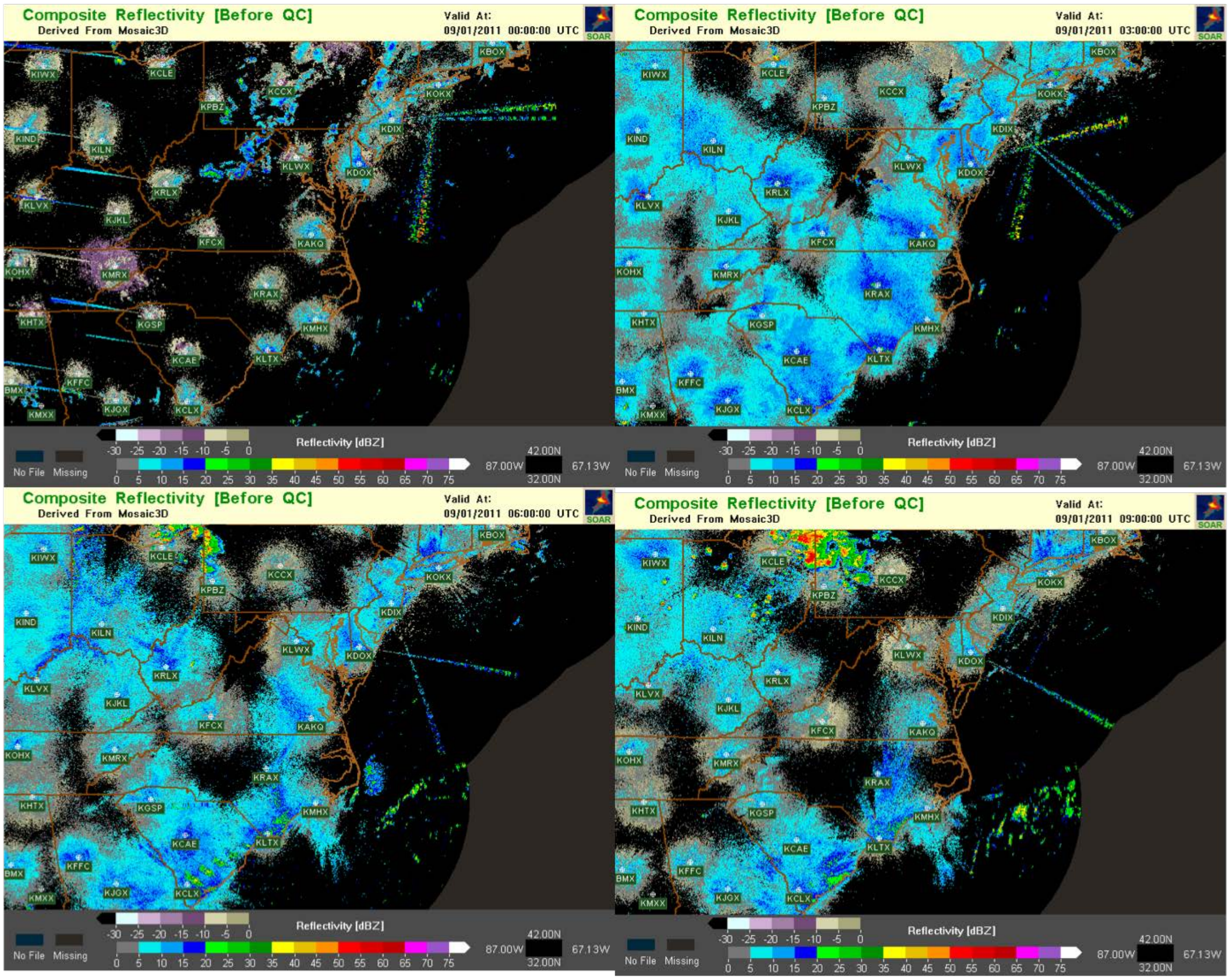

**Figure 27-2. Composite radar data (expressed in dB Z) for September 1, 2011.** See caption for Figure 27-1.

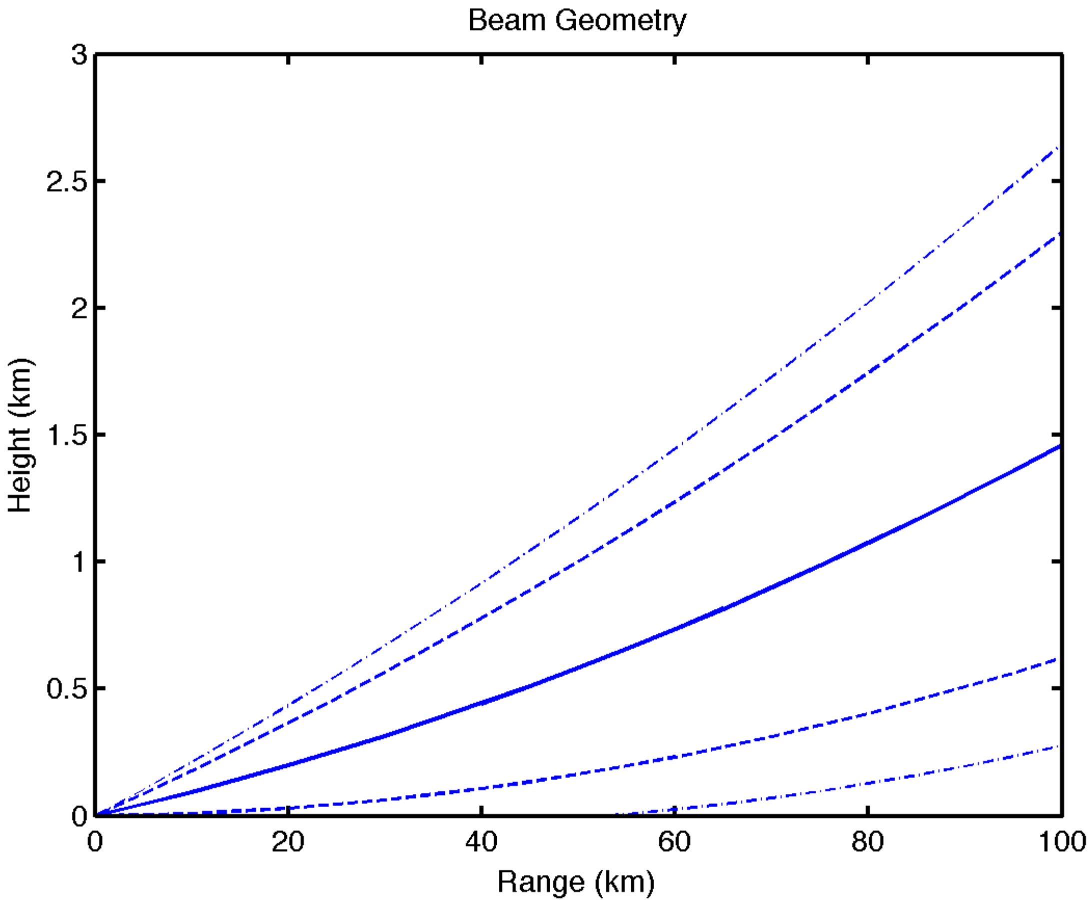

**Figure 27-3. Depiction of the sampling heights of NEXRAD as a function of range for a radar beam with an elevation angle of 0.5°.** Here the effects of the Earth curvature and the refraction of radio waves through a standard atmosphere have been taken into account. The bold solid line marks the center of the beam. The dashed line represents the beam's half-power (3 dB) point and the dashed-dotted line represents the beam's quarter-power (6 dB) point.

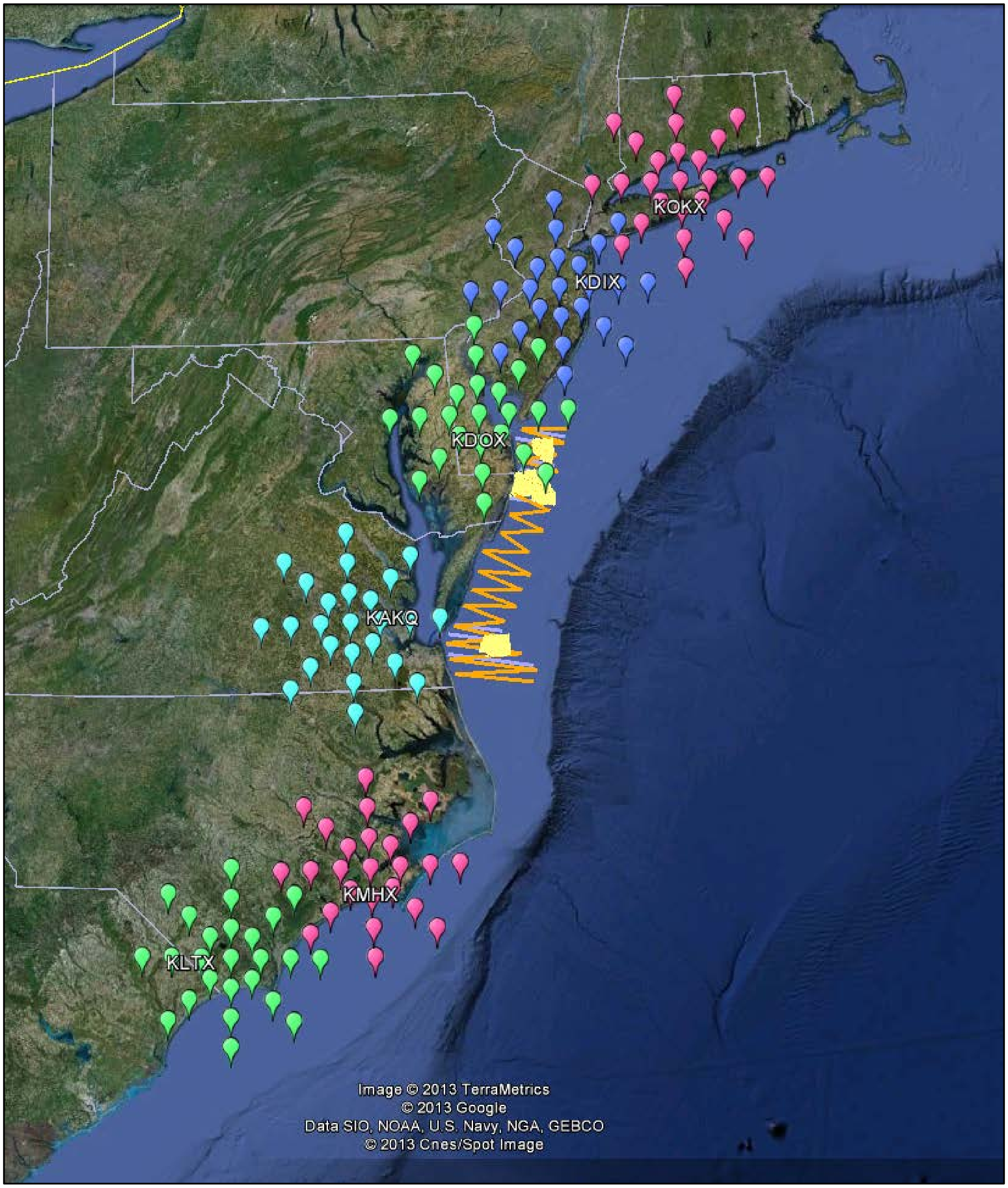

**Figure 27-4. The six NEXRAD radar units (KOKX, KDIX, KDOX, KAKQ, KMHX, and KLTX) used for analyses.** Each radar station had 24 comparison locations located at standardized distances from the unit, which were used for landscape-scale analyses of migratory activity patterns. Aerial survey transects delineating the radar study area are shown in yellow and orange.

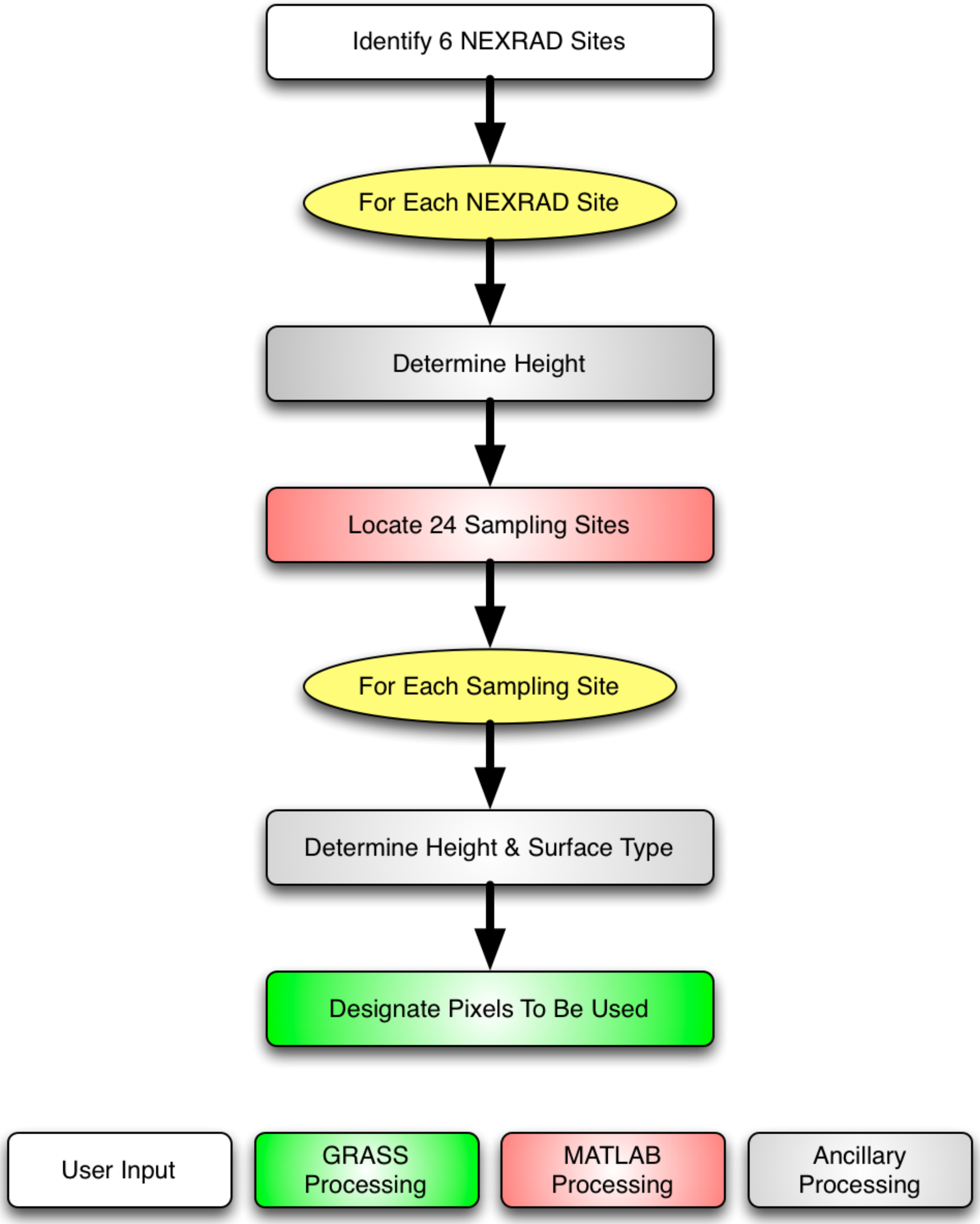

**Figure 27-5. Workflow showing the initialization steps conducted as part of the data processing.**

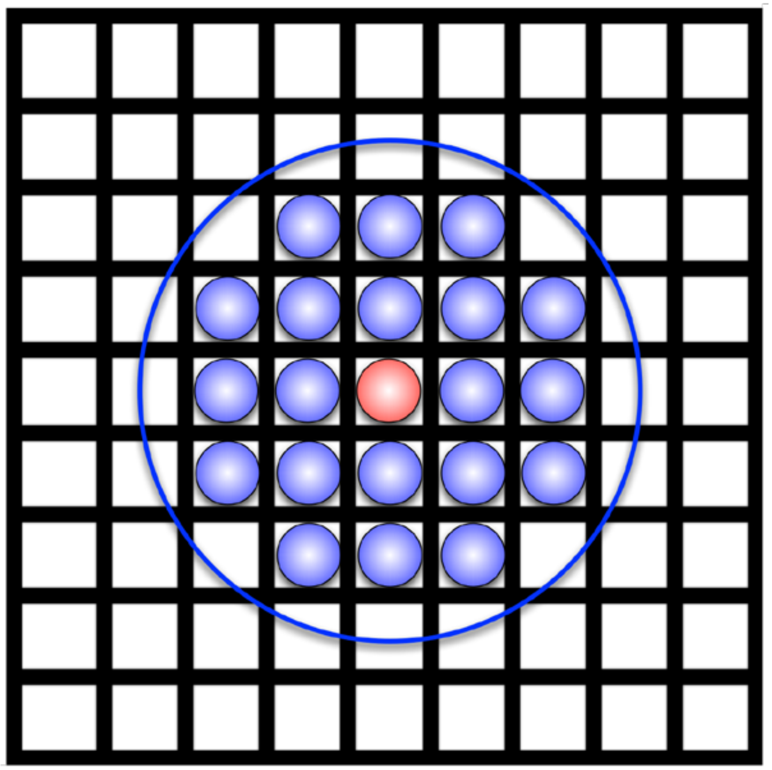

**Figure 27-6. Illustration of the process used to select the raster elements used when evaluating the univariate statistics of the CREF and UNQC_CREF data for each of the sampling sites.** The red dot corresponds to the coordinate of one of the 24 locations placed around the radar site. Using GRASS, the area to be considered is “grown” radially outward from the point. Those raster elements fully contained within the circle are designated as those to be considered. For each of the sampling sites, 21 raster elements were chosen using this process.

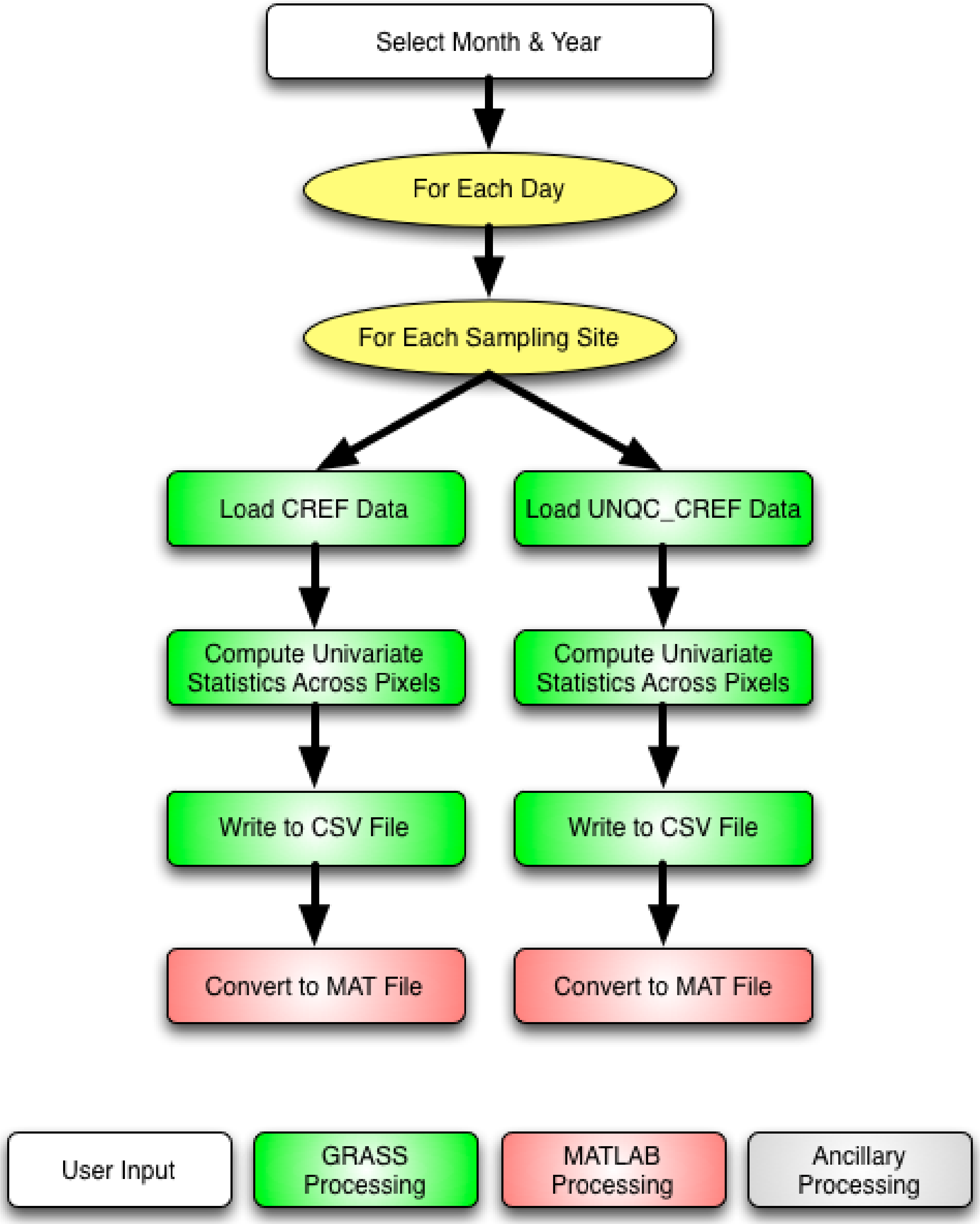

**Figure 27-7. Workflow showing the radar processing conducted primarily using GRASS when analyzing the CREF and UNQC_CREF GeoTIFF files.**

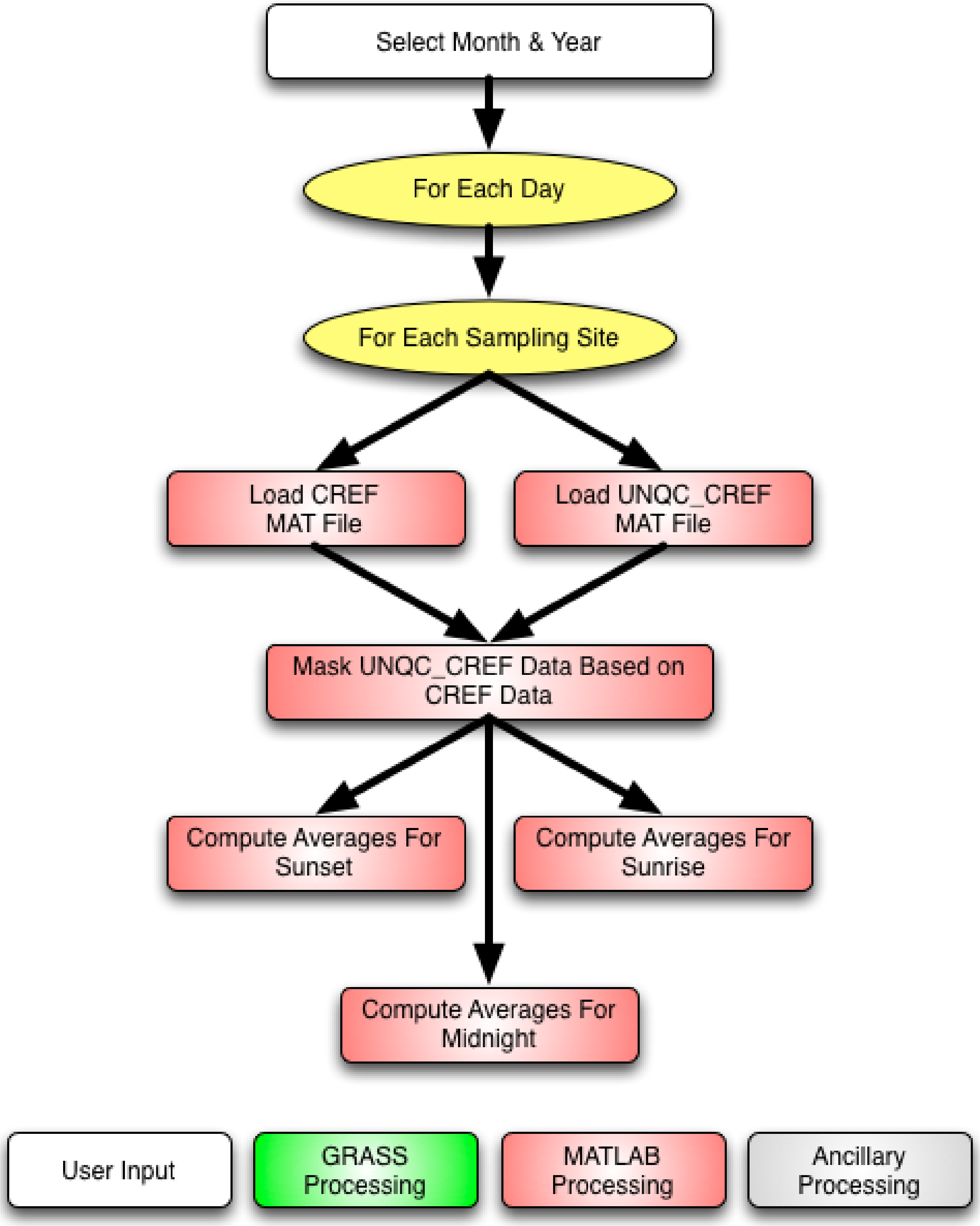

**Figure 27-8. Workflow showing the radar processing conducted using MATLAB when conducting such steps as data masking and temporal averaging.** Averages for sunset and sunrise were not used in this analysis. The "midnight" period included six hours centered around midnight for each night of sampling.

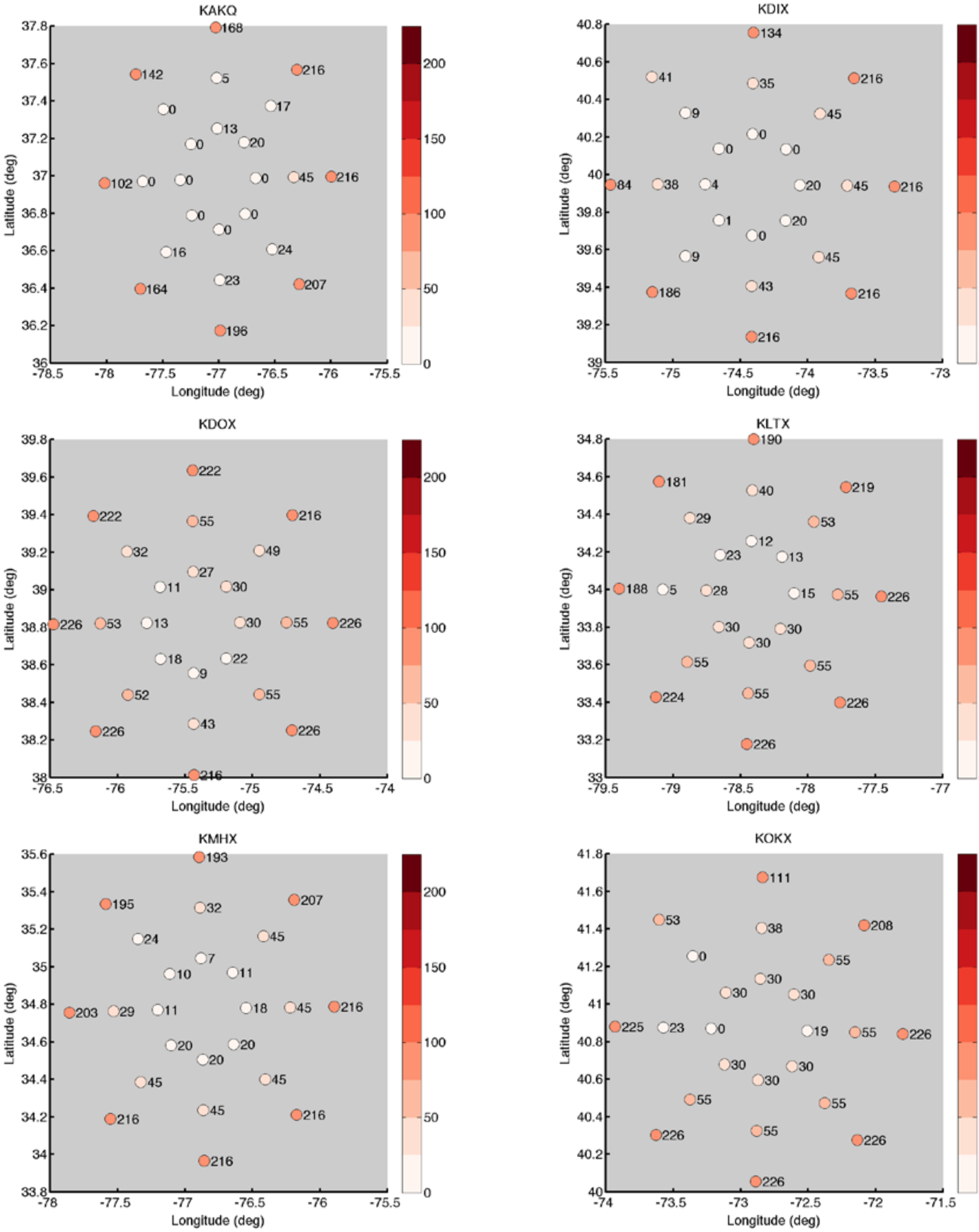

**Figure 27-9. Representation of the lowest detectable heights (m) for the 144 different sampling sites used in the study for each radar location.** Since biological entities have relatively large radar cross-sections, the 6-dB points in the beam have been used. The height of the radar and elevation of each sampling site have been factored into the calculation.

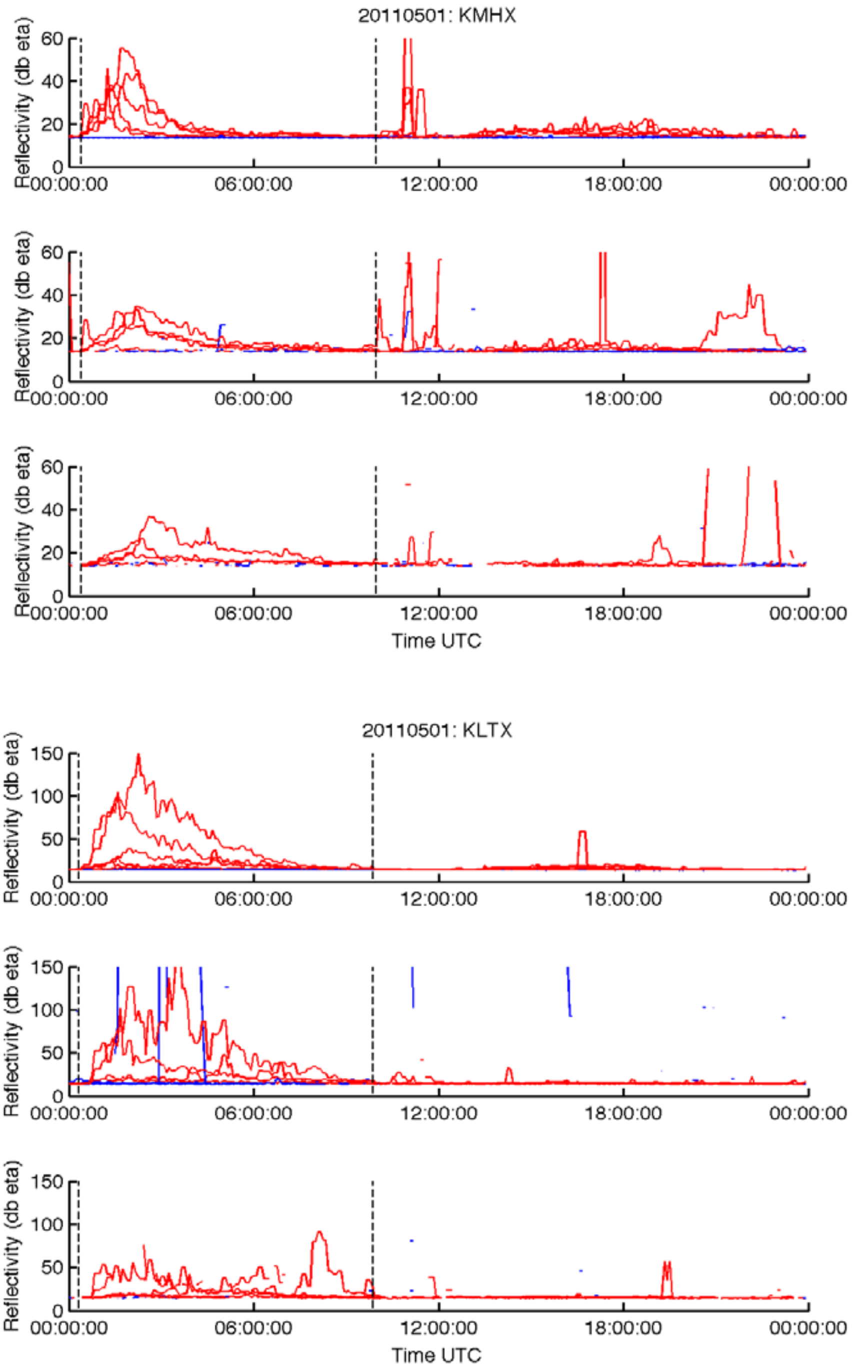

**Figure 27-10. Plots of the radar reflectivity in dBη for KMHX (upper 3 panels) and KLTX (lower 3 panels) for May 1, 2011.** The upper panels for a given radar show sampling locations 1-8 (30 km from radar), the middle panels show sites 8-16 (60 km), and the lower panels show sites 17-24 (90 km). Plots in red and blue are for locations over land and water, respectively. Sunset and rise times are indicated by vertical dashed lines (sunset is around 00 UTC, or 2000 hours in EDT; sunrise is around 10 UTC, 0r 0600 EDT). Sites over land (particularly in areas closer to the radar, since these values do not include a correction for this nuisance variable) show a pattern of "exodus" as migrants leave terrestrial stopover locations after sunset and begin their nocturnal flights. As expected, sites over water do not show this exodus pattern, and overall show lower dB η reflectivity values for this date.

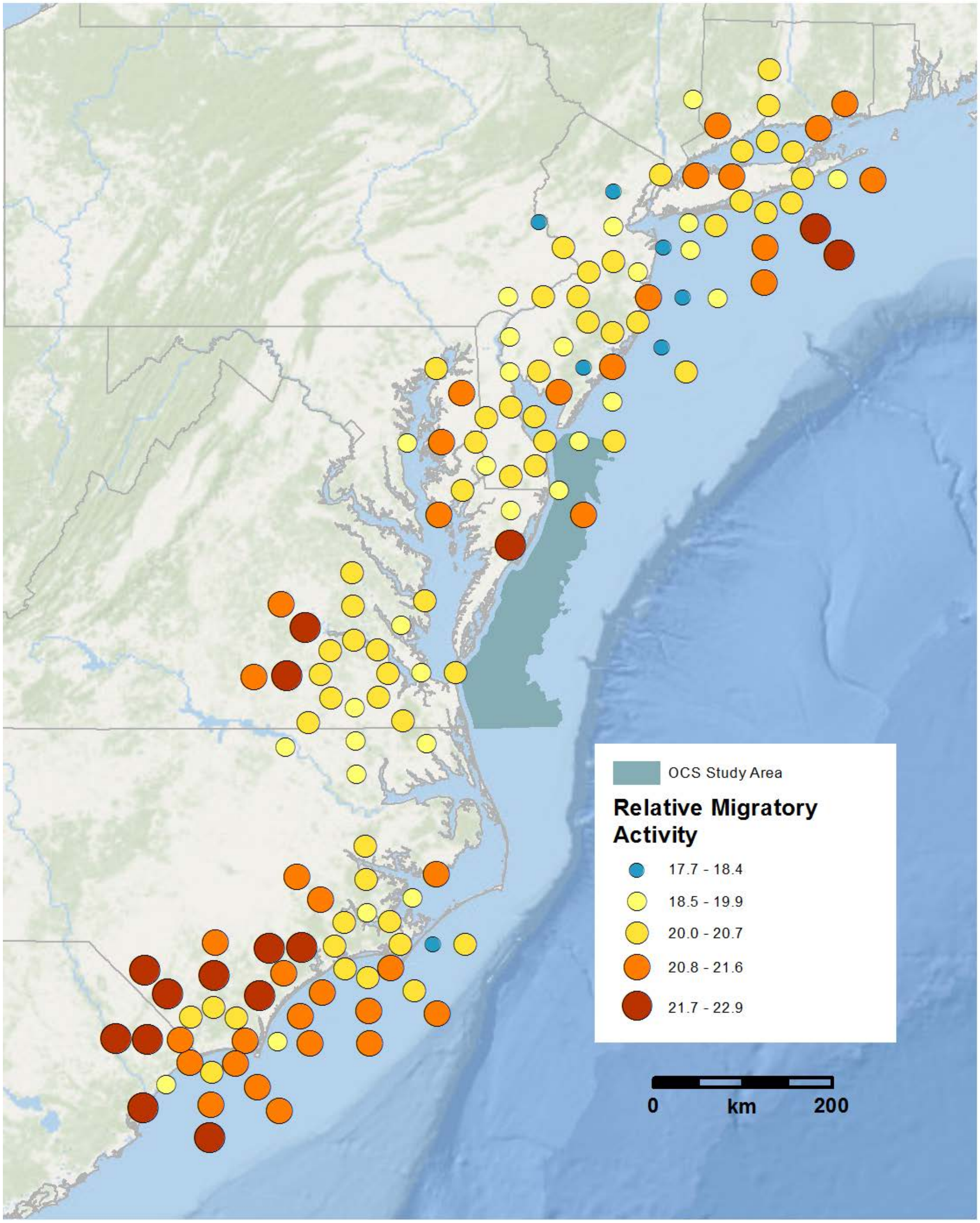

**Figure 27-11. Map of predicted levels of bioscatter across the radar study area for fall, averaged across all dates and years.** Elevation, radar unit, site type, and distance to the radar were standardized for each point so we could control for those nuisance variables and focus on the spatial factors that lead to differences in bioscatter. Relative migratory activity estimates are in values of dB η; the "OCS study area" is the mid-Atlantic study area referenced in other chapters in this report.

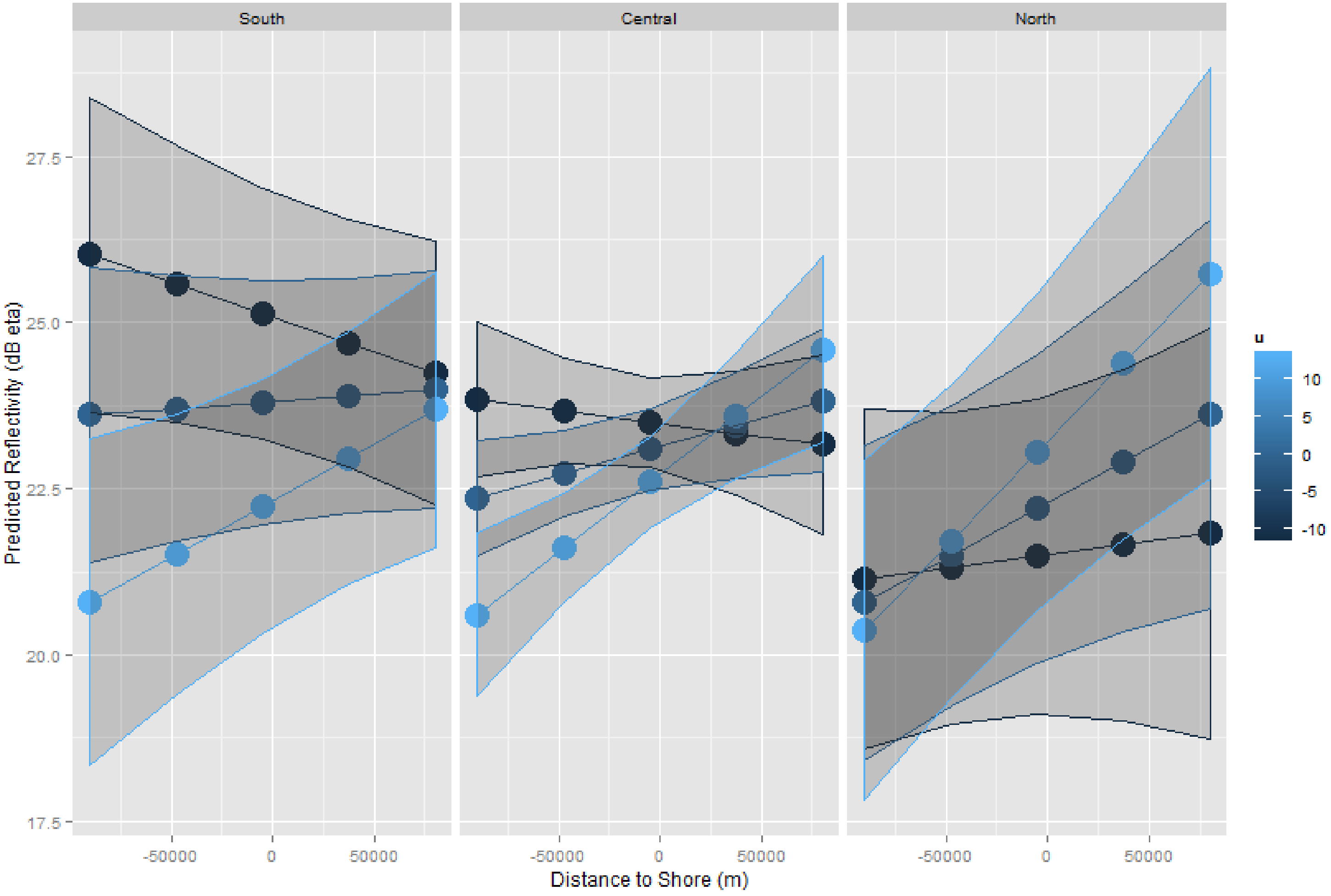

**Figure 27-12. The effects of fall East-West winds on bioscatter in relation to distance to shore.** Effects are shown separately for different latitudes within the radar study area, where South = the Carolinas, Central = Delaware, Maryland and Virginia, and North = New Jersey and New York. Wind direction is indicated by color, where easterlies are dark blue and westerlies and light blue. Zero is the shoreline, while positive values indicate locations further offshore and negative values are further inland. Shaded areas indicate the estimated 95% confidence interval of the prediction (when accounting only for variation in the model's fixed effects).

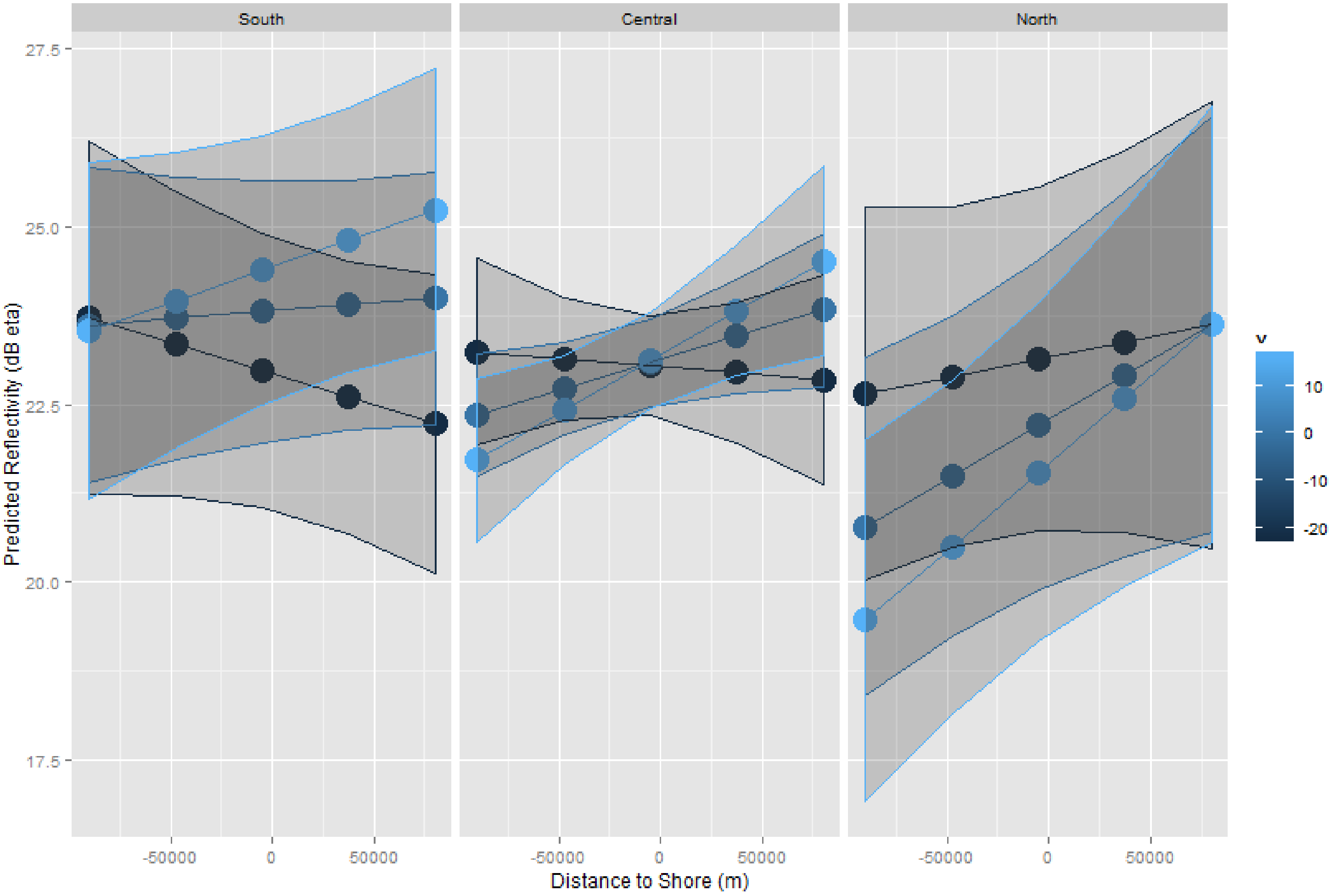

**Figure 27-13. The effects of fall North-South winds on bioscatter in relation to distance to shore.** Effects are shown separately for different latitudes within the radar study area, where South = the Carolinas, Central = Delaware, Maryland and Virginia, and North = New Jersey and New York. Wind direction is indicated by color, where northerlies are dark blue and southerlies and light blue. Zero is the shoreline, while positive values indicate locations further offshore and negative values are further inland. Shaded areas indicate the estimated 95% confidence interval of the prediction (when accounting only for variation in the model's fixed effects).

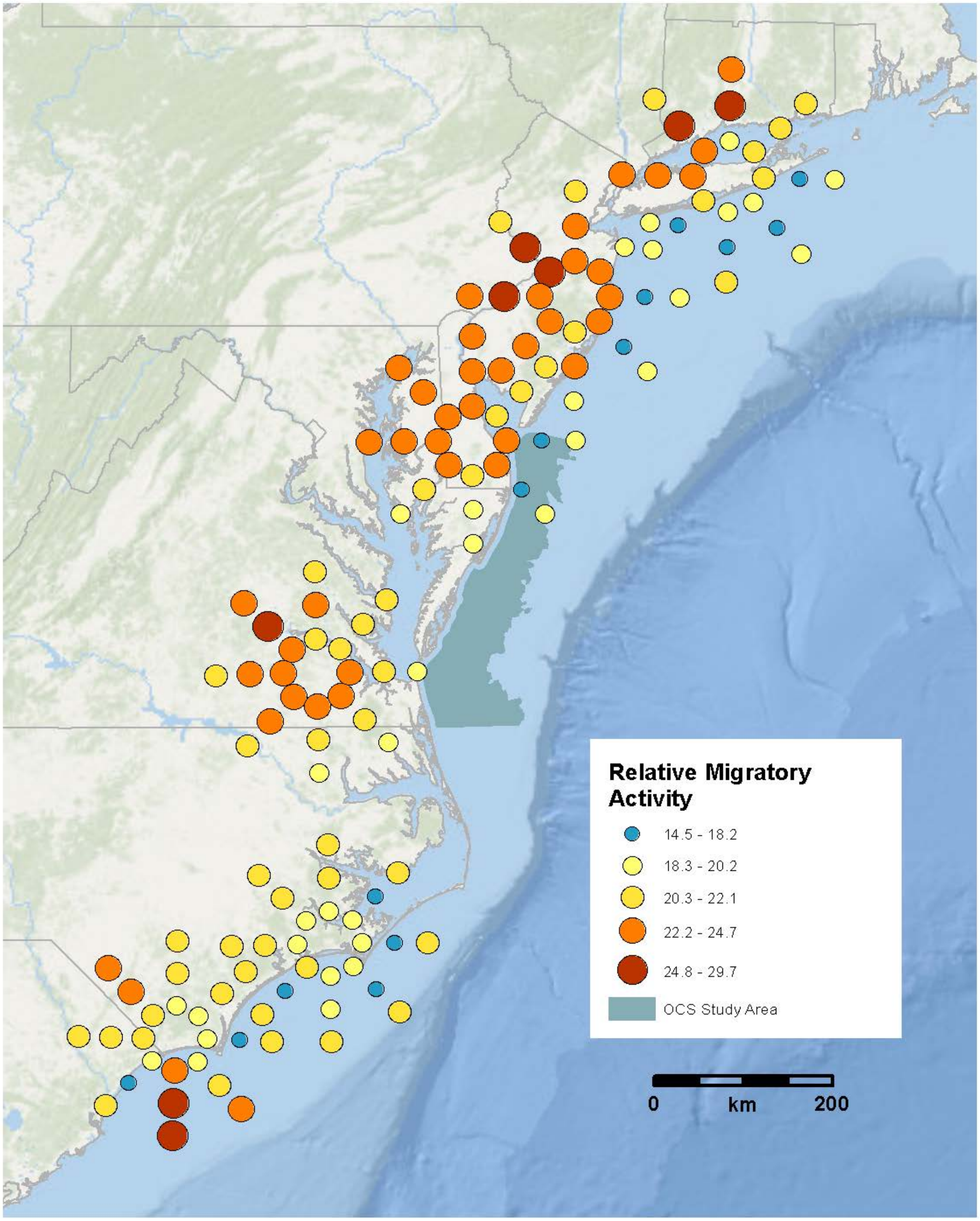

**Figure 27-14. Map of predicted levels of bioscatter across the radar study area for spring, averaged across all dates and years.** Elevation, radar unit, site type, and distance to the radar were standardized for each point so we could control for those nuisance variables and focus on the spatial factors that lead to differences in bioscatter. Relative migratory activity estimates are in values of dB η; the "OCS study area" is the mid-Atlantic study area referenced in other chapters in this report.

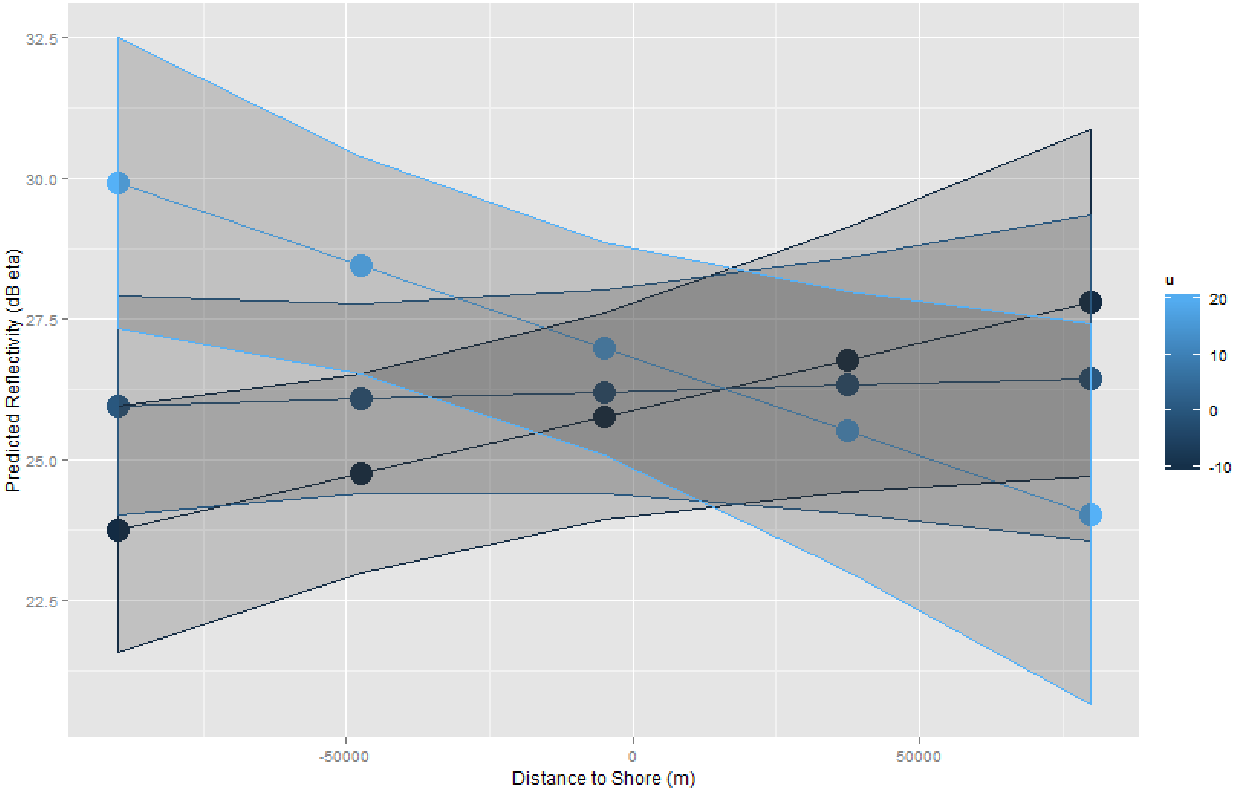

**Figure 27-15. The effects of spring East-West winds on bioscatter in relation to distance to shore.** Zero is the shoreline, while positive values indicate locations further offshore and negative values are further inland. Shaded areas indicate the estimated 95% confidence interval of the prediction (when accounting only for variation in the model's fixed effects).

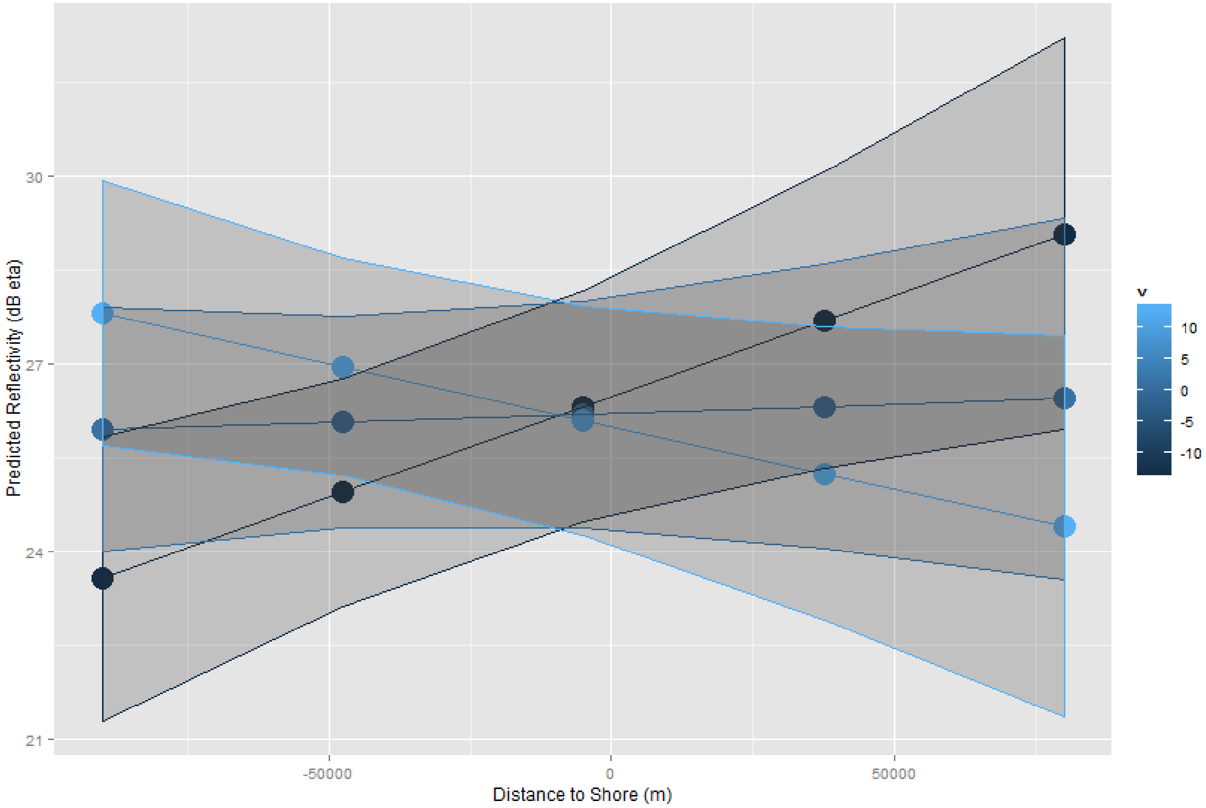

**Figure 27-16. The effects of spring North-South winds on bioscatter in relation to distance to shore.** Zero is the shoreline, while positive values indicate locations further offshore and negative values are further inland. Shaded areas indicate the estimated 95% confidence interval of the prediction (when accounting only for variation in the model's fixed effects).

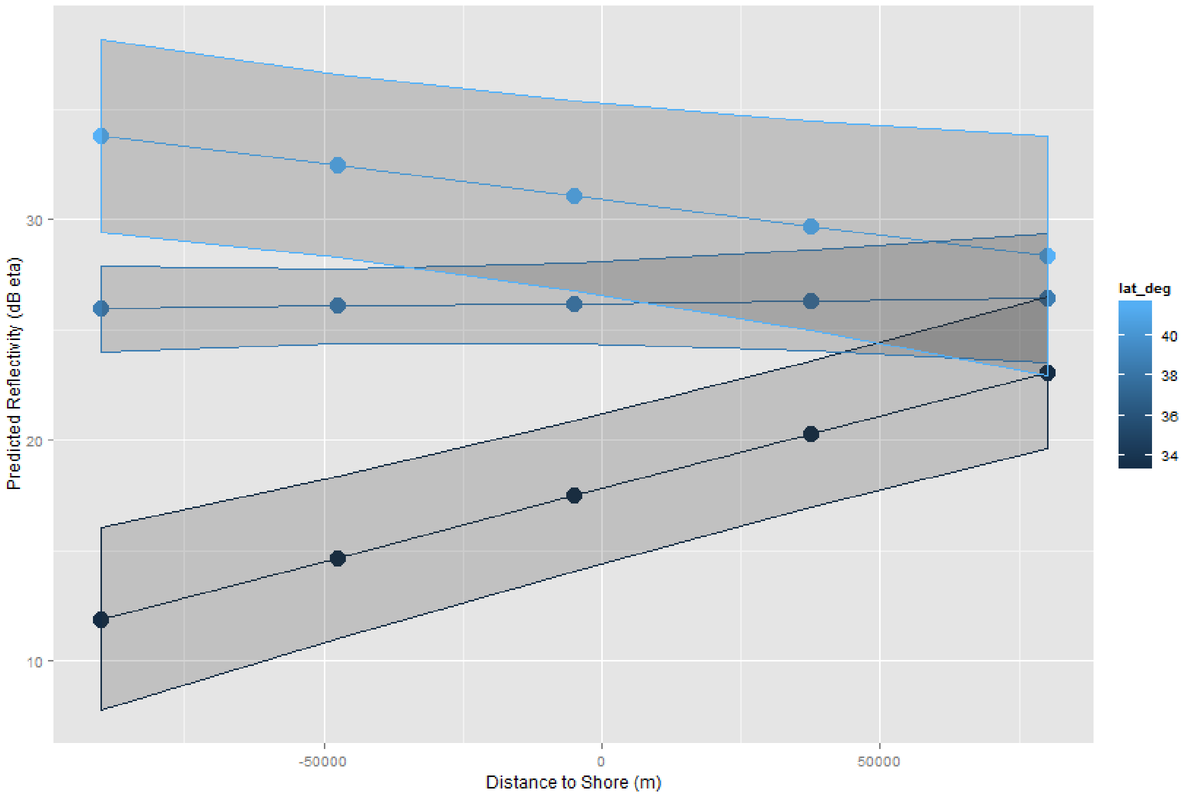

**Figure 27-17. The effects of spring latitude on bioscatter with distance to shore.** Zero is the shoreline while positive values indicate further away from the coast and negative values indicate further inland. Shaded areas indicate the estimated 95% confidence interval of the prediction when account only for variation in the model's fixed effects.

**Table 27-1. Analysis of Variance Table for the Fall Spatial Model.** Date, site and year were included as random variables. DF is degrees of freedom for the term, SSE is the sum squared error, MSE is the mean squared error and the F-value is a test to determine statistical significance on the F distribution.

| Model Term | DF | SSE | MSE | F value |
|---|---|---|---|---|
| Radar Unit | 5 | 358.6 | 71.7 | 6.6577* |
| Distance to Radar | 2 | 14809.1 | 7404.5 | 687.3636* |
| Distance to Shore | 1 | 28.8 | 28.8 | 2.6736 |
| Elevation | 1 | 30.5 | 30.5 | 2.8283* |
| Radar X Distance to Shore | 5 | 57.7 | 11.5 | 1.071 |

*Indicates the 95% confidence interval of at least one of the parameters in this term does not overlap zero

**Table 27-2. Analysis of Variance Table for the Fall Weather Model.** DF is degrees of freedom for the term, SSE is the sum squared error, MSE is the mean squared error and the F-value is a test to determine statistical significance on the F distribution.

| Model Term | DF | SSE | MSE | F value |
|---|---|---|---|---|
| Radar Unit | 5 | 654.6 | 130.9 | 10.3178* |
| Distance to Radar | 2 | 19413.5 | 9706.8 | 764.9804* |
| Land Cover | 1 | 85.6 | 85.6 | 6.7447* |
| U Wind | 1 | 71.7 | 71.7 | 5.6541* |
| V Wind | 1 | 13.9 | 13.9 | 1.0981 |
| Elevation | 1 | 7 | 7 | 0.5481 |
| Latitude | 1 | 55.9 | 55.9 | 4.4035 |
| Longitude | 1 | 46.7 | 46.7 | 3.683* |
| Distance to Shore | 1 | 23.7 | 23.7 | 1.8705 |
| U Wind X Distance to Shore | 1 | 150.3 | 150.3 | 11.8426* |
| U Wind X Latitude | 1 | 251.3 | 251.3 | 19.8031* |
| V Wind X Distance to Shore | 1 | 42.4 | 42.4 | 3.34 |
| V Wind X Latitude | 1 | 124.2 | 124.2 | 9.7847* |
| Distance to Shore X Latitude | 1 | 32.4 | 32.4 | 2.5502 |

*Indicates the 95% confidence interval of at least one of the parameters in this term does not overlap zero

**Table 27-3. Analysis of Variance Table for the Spring Spatial Model.** DF is degrees of freedom for the term, SSE is the sum squared error, MSE is the mean squared error and the F-value is a test to determine statistical significance on the F distribution. A star in the “significance” column indicates that the 95% confidence interval for β does not overlap zero.

| Model Term | DF | SSE | MSE | F value |
|---|---|---|---|---|
| Radar unit | 5 | 286.17 | 57.23 | 4.9257* |
| Distance to Radar | 2 | 2551.95 | 1275.98 | 109.8139* |
| Distance to Shore | 1 | 261.18 | 261.18 | 22.4776 |
| Elevation | 1 | 223.23 | 223.23 | 19.2116* |
| Radar X Distance to Shore | 5 | 274.44 | 54.89 | 4.7238* |

*Indicates the 95% confidence interval of at least one of the parameters in this term does not overlap zero

**Table 27-4. Analysis of Variance Table for the Spring Weather Model.** DF is degrees of freedom for the term, SSE is the sum squared error, MSE is the mean squared error and the F-value is a test to determine statistical significance on the F distribution. A star in the “significance” column indicates that the 95% confidence interval for β does not overlap zero.

| Model Term | DF | SSE | MSE | F value |
|---|---|---|---|---|
| Radar Unit | 5 | 261.9 | 52.38 | 3.3612 |
| Distance to Radar | 2 | 5068.4 | 2534.21 | 162.6332* |
| Land Cover | 1 | 1051.2 | 1051.16 | 67.4586* |
| U Wind | 1 | 214.4 | 214.42 | 13.7605 |
| V Wind | 1 | 0.6 | 0.63 | 0.0406 |
| Elevation | 1 | 214.6 | 214.58 | 13.7707 |
| Latitude | 1 | 16.7 | 16.67 | 1.07* |
| Longitude | 1 | 405.3 | 405.27 | 26.0084* |
| Distance to Shore | 1 | 214.5 | 214.45 | 13.7627* |
| U Wind X Distance to Shore | 1 | 325.1 | 325.07 | 20.8616* |
| U Wind X Latitude | 1 | 27.7 | 27.69 | 1.7767 |
| V Wind X Distance to Shore | 1 | 271.1 | 271.05 | 17.3947* |
| V Wind X Latitude | 1 | 13.3 | 13.29 | 0.853 |
| Distance to Shore X Latitude | 1 | 481.5 | 481.54 | 30.9026* |

*Indicates the 95% confidence interval of at least one of the parameters in this term does not overlap zero