



RESEARCH ARTICLE

Resource selection and wintering phenology of White-winged Scoters in southern New England: Implications for offshore wind energy development

Dustin E. Meattey,^{1,4*} Scott R. McWilliams,¹ Peter W. C. Paton,¹ Christine Lepage,² Scott G. Gilliland,³ Lucas Savoy,⁴ Glenn H. Olsen,⁵ and Jason E. Osenkowski⁶

¹ Department of Natural Resources Science, University of Rhode Island, Kingston, Rhode Island, USA

² Canadian Wildlife Service, Environment and Climate Change Canada, Quebec, Quebec, Canada

³ Canadian Wildlife Service, Environment and Climate Change Canada, Sackville, New Brunswick, Canada

⁴ Biodiversity Research Institute, Portland, Maine, USA

⁵ USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA

⁶ Rhode Island Department of Environmental Management, West Kingston, Rhode Island, USA

* Corresponding author: dustin_meattey@uri.edu

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ABSTRACT

Southern New England provides key wintering habitat for White-winged Scoters (*Melanitta fusca*). This area has also pioneered the development of offshore wind energy in North America. The U.S. Bureau of Ocean Energy Management (BOEM) has established 9 Wind Energy Area (WEA) lease blocks along the Atlantic Outer Continental Shelf in areas that may provide important staging and wintering habitat for scoters and other species of sea ducks. Concern over the potential impact of offshore wind energy on sea duck populations has led to efforts to develop models to understand their distribution, habitat use, and site fidelity. We used satellite telemetry to document winter phenology and site fidelity, as well as fine-scale resource selection and habitat use, of 40 White-winged Scoters along the southern New England continental shelf. Scoters spent over half of the annual cycle on the wintering grounds and demonstrated a high degree of interannual site fidelity to composite core-use areas. Sizes of individual 50% core-use home ranges were variable ($\bar{X}$ = 868 km²; range: 32–4,220 km²) and individual 95% utilization distributions ranged widely ($\bar{X}$ = 4,388 km²; range: 272–18,235 km²). More than half of all tagged birds occupied 2 or more discrete core-use areas that were up to 400 km apart. Throughout the study area, scoters selected areas with lower salinity, lower sea surface temperature, higher chlorophyll-*a* concentrations, and higher hard-bottom substrate probability. Resource selection function models classified 18,649 km² (23%) of the study area as high probability of use, which included or immediately bordered ~420 km² of proposed WEA lease blocks. Future offshore wind energy developments in the region should avoid key habitats highlighted by this study and carefully consider the environmental characteristics selected by sea ducks when planning and siting future WEAs.

Keywords: habitat use, *Melanitta fusca*, New England, offshore wind energy, phenology, resource selection function, satellite telemetry, White-winged Scoter

Sélection des ressources et phénologie de l'hivernage chez *Melanitta fusca* dans le sud de la Nouvelle-Angleterre: implications pour le développement de l'énergie éolienne en mer

RÉSUMÉ

Le sud de la Nouvelle-Angleterre fournit un habitat d'hivernage essentiel pour *Melanitta fusca*. Cette région est également un pionnier du développement de l'énergie éolienne en mer en Amérique du Nord. Le Bureau of Ocean Energy Management (BOEM) des États-Unis a établi neuf blocs de concession de zone d'énergie éolienne (WEA) le long de la zone externe du plateau de l'Atlantique dans des secteurs susceptibles de fournir un important habitat de halte migratoire et d'hivernage pour les macreuses et d'autres espèces de canards de mer. L'inquiétude suscitée par l'impact potentiel de l'énergie éolienne en mer sur les populations de canards de mer a mené à des efforts pour élaborer des modèles permettant de comprendre leur répartition, leur utilisation de l'habitat et leur fidélité au site. Nous avons utilisé la télémétrie satellitaire afin de documenter la phénologie hivernale et la fidélité au site, de même que la sélection des ressources et l'utilisation de l'habitat à fine échelle de 40 individus de *M. fusca* le long de la partie sud du plateau continental de Nouvelle-Angleterre. Les macreuses ont passé plus de la moitié du cycle annuel sur les quartiers d'hivernage et ont démontré un degré élevé de fidélité au site interannuelle envers les zones composites d'utilisation principale. La taille des domaines vitaux individuels à 50% d'utilisation principale était variable ($\bar{X}$ = 868 km²; étendue: 32–4220 km²) et les distributions individuelles de 95 % d'utilisation variaient considérablement ($\bar{X}$ = 4388 km²; étendue: 272–18,235 km²).

Plus de la moitié de tous les oiseaux marqués occupaient au moins deux zones discrètes d'utilisation principale qui étaient distantes de 400 km au maximum. Dans l'ensemble de la zone d'étude, les macreuses ont sélectionné des zones avec une plus faible salinité, une plus faible température de la mer en surface, une concentration plus élevée en chlorophylle-*a* et une probabilité plus élevée de substrat de fond dur. Les modèles de fonction de sélection des ressources ont classé 18,649 km² (23 %) de la zone d'étude comme ayant une probabilité élevée d'utilisation, qui incluaient ou juxtaposaient ~420 km² de blocs projetés de concession WEA. Les projets éoliens futurs dans la région devraient éviter les habitats clés mis en évidence par la présente étude et considérer attentivement les caractéristiques environnementales sélectionnées par les canards de mer lors de la planification et la localisation des futures WEA.

Mots-clés: énergie éolienne en mer, fonction de sélection des ressources, *Melanitta fusca*, Nouvelle-Angleterre, phénologie, télémétrie satellitaire, utilisation de l'habitat

INTRODUCTION

Effective management and conservation of any migratory species relies on a thorough understanding of seasonal distribution and resource use, as well as threats from anthropogenic and other sources. In North America, there is increasing concern over declines in populations of several sea duck species (Sea Duck Joint Venture Management Board 2014, Bowman et al. 2015). Reasons for these apparent declines are uncertain, although poor habitat conditions and foraging availability on wintering grounds have been linked to significant mortality events (Camphuysen et al. 2002), reduced annual survival (Petersen and Douglas 2004), and decreased productivity in subsequent breeding seasons (Oosterhuis and van Dijk 2002). Because sea ducks spend much of their annual cycle utilizing habitats in nonbreeding areas where direct anthropogenic threats are often greater than during the breeding season, understanding winter habitat use dynamics is important for conservation.

In North America, the first offshore wind energy development (OWED), a 5-turbine, 30-megawatt facility off Block Island, Rhode Island, became operational in December 2016. Thus, the potential impact of OWEDs on sea duck populations is a recent conservation concern in the United States, particularly on their wintering grounds; numerous other multi-turbine wind energy leases have been issued for offshore areas in New England and mid-Atlantic states (Manwell et al. 2002, Breton and Moe 2009, Musial and Ram 2010). Potential negative interactions between sea ducks and OWED include collision risk, disturbance, and exclusion from key habitats and prey resources (Drewitt and Langston 2006, Fox et al. 2006, Furness et al. 2013, Dierschke et al. 2016). In Europe, where biologists have been investigating the potential impacts of OWEDs on marine birds for over 20 yr (Guillemette and Larsen 2002, Desholm and Kahlert 2005, Langston 2013, Bailey et al. 2014, Vallejo et al. 2017), collision risk is likely minimal for sea ducks (Desholm and Kahlert 2005, Bradbury et al. 2014), but avoidance behaviors including displacement from key foraging sites likely have more significant population-level impacts (Hüppop et al. 2006, Furness et al. 2013, Dierschke et al. 2016).

Sea ducks are particularly vulnerable because they usually forage in shallow, subtidal areas in substrates that are often favored for OWED (Fox 2003, Kaiser et al. 2006, Loring et al. 2014, Meattey et al. 2015). A review of post-construction studies at 20 offshore wind farms in Europe classified Common Scoters (*Melanitta nigra*) and Long-tailed Ducks (*Clangula hyemalis*) as weakly avoiding offshore wind farms (Dierschke et al. 2016). Petersen and Fox (2007) documented short-term displacement of Common Scoters from an OWED in Denmark for 3 yr, although follow-up studies suggest that this displacement may be more long-lasting (Petersen et al. 2014). This loss of potential foraging habitat, as a result of avoidance and displacement, in areas with large concentrations of wintering sea ducks could have detrimental population-level effects. Habitat conditions and availability during the wintering period may have strong carryover effects on reproductive success and productivity during the subsequent breeding season (Camphuysen et al. 2002, Oosterhuis and van Dijk 2002, Gurney et al. 2014). Also in Denmark, Common Eiders (*Somateria mollissima*) avoided flying close to wind turbines (Larsen and Guillemette 2007), suggesting that habitat use within and around wind farms may be greatly reduced. The cost of avoidance behaviors may be trivial relative to the energetic costs of long-distance migration, but the cumulative impact of avoiding multiple developments along a migration route could be significant (Masden et al. 2009). Thus, identification of important habitats used by sea ducks prior to offshore wind energy development informs the planning process and helps avoid displacement of sea ducks from preferred habitats.

Satellite telemetry provides an increasingly effective tool for assessing population delineation, movement ecology, and habitat selection of marine birds including sea ducks (Oppel et al. 2008, Loring et al. 2014, Meattey et al. 2015, Berlin et al. 2017, Meattey et al. 2018). In New England, recent telemetry studies of Black Scoter (*Melanitta americana*; Loring et al. 2014) and Common Eider (Beuth et al. 2017) provided essential information on their movement ecology and habitat use patterns. However, there remains a lack of information on the seasonal changes in abundance and distribution, as well as environmental drivers of habitat use, for other priority sea duck species in southern New England, such as White-winged Scoters (*Melanitta*

fusca; Zipkin et al. 2010, Silverman et al. 2013, Sea Duck Joint Venture 2015).

The White-winged Scoter is a long-lived species that winters along both the Atlantic and Pacific coasts of North America and breeds throughout the interior boreal forest from Alaska to central Canada (Brown and Fredrickson 1997). On their wintering grounds, White-winged Scoters generally feed on benthic mollusks and crustaceans in waters ≤ 20 m deep (Stott and Olson 1973, Lewis et al. 2007). The continental population of White-winged Scoters has experienced a long-term decline throughout the last half-century (Alisauskas et al. 2004, USFWS 2011), with steady rates of decreased annual harvest being recognized on the wintering areas, particularly on the Atlantic Coast (Rothe et al. 2015). White-winged Scoters remain one of the least-studied waterfowl species, thus management and conservation efforts, particularly on the wintering grounds, have been impeded by a lack of vital life history information.

The objectives of our study were to use satellite telemetry to document (1) timing of movements of White-winged Scoters in southern New England, and (2) fine-scale resource selection and habitat use patterns. This combined information identifies key locations and times when White-winged Scoters may be vulnerable to ecological impacts from offshore wind energy development.

METHODS

Study Area

Our 82,572 km² study area included coastal and offshore habitat in the southwestern Gulf of Maine and the southern New England continental shelf ranging from Gloucester, Massachusetts, to ~80 km south of Long Island, New York, and extending ~90 km east of Cape Cod, Massachusetts (39.93°N–42.52°N, 68.86°W–74.10°W; Figure 1). This region includes a geologically diverse array of coastline and offshore islands including Block Island, Martha's Vineyard, and Nantucket Island, as well as several moderately shallow bays and sounds including Cape Cod Bay, Nantucket Sound, Narragansett Bay, Buzzard's Bay, Block Island Sound, Rhode Island Sound, and Long Island Sound. This area also includes the Nantucket Shoals (41.03°N, 69.94°W), a 2,000 km² expanse of shallow (4–35 m deep), sandy-bottom habitat identified through extensive aerial surveys as having a high density of White-winged Scoters and other sea birds (Veit et al. 2016). The shoals support high concentrations of benthic amphipods (Avery et al. 1996), which are important in the winter diet of some sea duck species including Long-tailed Duck (White et al. 2009) and the breeding season diet of White-winged Scoters (Brown and Fredrickson 1986, Benoit et al. 1996, Haszard and Clark 2007). The highest Atlantic densities of White-winged Scoters occur between Cape Cod and Long Island Sound (Silverman et al. 2013), accounting

for approximately 94% of the entire western Atlantic Coast wintering population (Silverman et al. 2012). Our study area included ~4,000 km² of commercial offshore wind energy leases and planning areas off the coasts of Massachusetts, Rhode Island, and New York (Bureau of Ocean Energy Management 2012; Figure 1).

Capture and Marking

We used floating mist nets (36 m long, 100 or 127 mm mesh; Brodeur et al. 2008) to capture White-winged Scoters in Cape Cod Bay, Massachusetts (41.75°N, 70.31°W), and Long Island Sound, New York (40.99°N, 72.83°W), during November 2015 and March 2016 ($n = 104$). We also captured birds with a submerged gill net (adapted from Breault and Cheng 1990) at a prominent molting location in the St. Lawrence River Estuary, Quebec, Canada (48.69°N, 69.06°W; $n = 15$ from 2010 to 2012 and $n = 262$ in August 2016). We determined the age and sex of all captured birds based on plumage characteristics (Carney 1992), cloacal examination, and bursal depth (Mather and Esler 1999).

Licensed veterinarians implanted platform terminal transmitters (PTTs) in 52 female White-winged Scoters (Cape Cod Bay $n = 22$; Long Island Sound $n = 4$; Quebec $n = 26$) from 2015 to 2016. This study was part of a larger project assessing population-level linkages between wintering, breeding, and molting areas (Meattey et al. 2018). Because females of most sea ducks exhibit a higher degree of natal and breeding philopatry than males (Eadie and Savard 2015, Mallory 2015), we implanted PTTs in second-year and after-second-year females. We used 35–38 g coelomic-implant PTT with an external antenna manufactured by either Microwave Telemetry (Columbia, Maryland, USA; $n = 31$) or Telonics (Model IMPTAV-2635; Mesa, Arizona, USA; $n = 21$). All PTTs were wrapped in nylon mesh with a felt cuff at the antenna base to provide additional anchor points to stabilize the PTT within the body cavity and provide additional surface area for adhesion to the body wall (D. M. Mulcahy personal communication). Transmitters were then sterilized with ethylene oxide and allowed to de-gas before implanting. All transmitters were implanted using sterile surgical procedures (Korschgen et al. 1996). All birds were administered subcutaneous boluses of lactated Ringer's solution (30 mL kg⁻¹). Isoflurane given by mask, followed by intubation, was used for the general anesthesia. All birds received a line-block of bupivacaine (2 mg kg⁻¹) or bupivacaine and lidocaine (2 mg kg⁻¹) at the site of the skin incision. During holding, transport, and recovery, we held birds separately in small pet carriers equipped with padded sides to avoid bill damage and a raised mesh floor above a bed of pine shavings to allow them to remain clean and dry. We allowed birds to recover in their crates for 1–2 hr after surgery and then released them at or near their original capture location within 11 hr of initial capture ($\bar{X} = 7.5$ hr, range: 3.0–11.0).

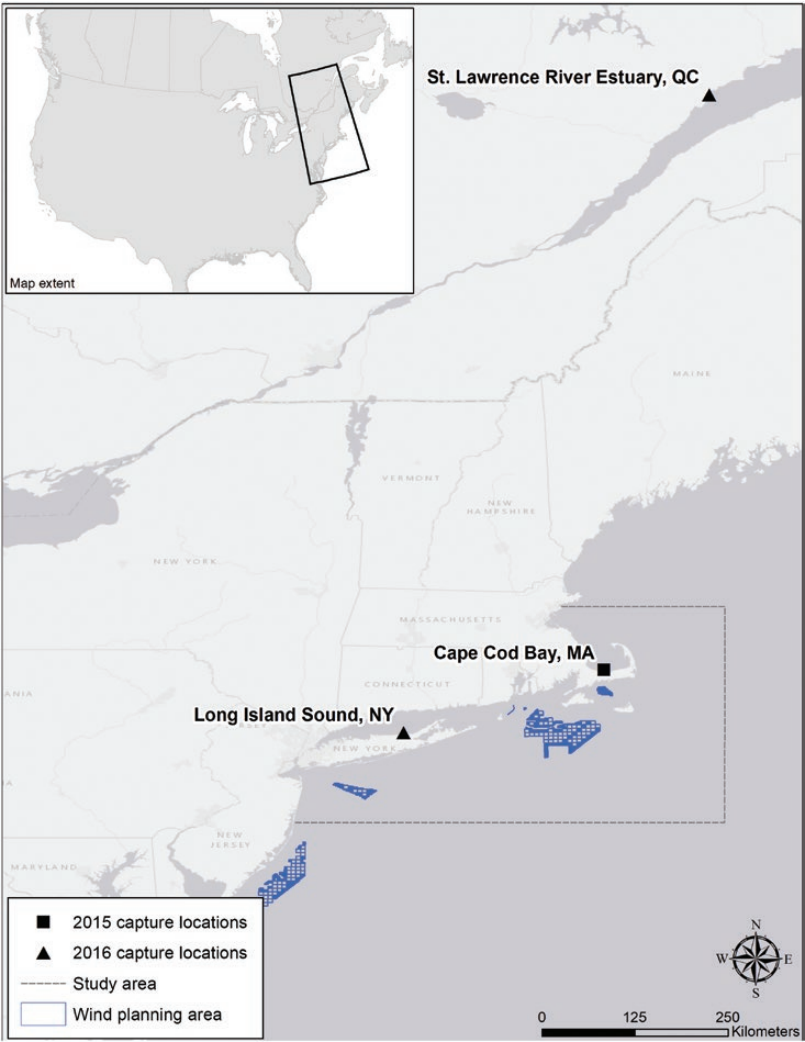


FIGURE 1. Study area (dashed line) and capture locations (solid symbols) of White-winged Scoters ($n = 40$) implanted with satellite transmitters in southern New England and Quebec during 2015 and 2016.

Location Data

We used the Argos satellite-based location and collection system (Collecte Localisation Satellites 2017) to receive transmission signals and PTT diagnostic data from all deployed birds. We downloaded and archived transmission data nightly and subsequently filtered data through the Douglas Argos Filter (DAF; Douglas et al. 2012) to remove redundant data and unlikely point locations. Using the DAF, we employed a hybrid filter to retain the single location with the highest accuracy from each duty cycle to reduce redundant daily positional information in our analyses. Argos processing centers report calculated accuracy estimates for each of the 4 highest-quality location classes (Table 1). These accuracy estimates may be overestimated; O'Connor (2008) calculated estimates of approximately

TABLE 1. Location class and frequency of randomly selected locations ($n = 40$ per bird) used to generate individual and composite 95% utilization distributions and 50% core-use areas for 40 White-winged Scoters wintering in the southern New England study area.

Location class ^a	Frequency	Percent
LC 3	738	45
LC 2	482	30
LC 1	201	12
LC 0	64	4
LC A	95	6
LC B	55	3

^a Locations classified by accuracy intervals (m): LC 3 (<250), LC 2 (250 to <500), LC 1 (500 to <1,500), LC 0 (>1,500) (Douglas et al. 2012, Collecte Localisation Satellites 2017). Accuracy estimates are not assigned for location classes LC A or LC B.

660 m, 1,000 m, and 1,700 m for class 3, 2, and 1 locations, respectively.

We programmed PTTs on 2 separate duty cycles that would alternate throughout the year (Table 2). In short, duty cycles increased the frequency of locations during the winter period and prolonged battery life through multiple seasons to allow for full annual cycle analysis of migration and connectivity (Meattey et al. 2018). Due to changes in capture timing, PTTs deployed during August 2016 began on a more conservative duty cycle that lasted through the first winter. While this was counter to our earlier fall deployments, we projected that this duty cycle would still provide an acceptable number of winter locations (~40) for habitat analysis.

To increase battery life, we programmed transmitters deployed in 2010–2012 with a duty cycle of 2 hr on and 72 hr off. To minimize potential bias in habitat use and movement behavior associated with capture and surgery trauma, we excluded the first 14 days of data collected after release (Esler et al. 2000; Sexson et al. 2014). We only included birds that transmitted >60 days after release in our analyses.

We used only data collected over a single winter (Nov–Apr) for each bird when calculating winter resource selection functions to standardize for mortality and PTT longevity and avoid biasing the analysis toward individuals that provided data over more than one winter. Some researchers have suggested that PTT implantation can affect movement patterns following capture and deployment (Barron et al. 2010, White et al. 2013), therefore we preferentially used data for an individual in their second winter if available ($n = 5$). When calculating movement phenology and interannual site fidelity, multiple years of data were used when possible. We managed and analyzed all telemetry data, as well as produced all maps, using ArcGIS 10.3.1 (Environmental Systems Research Institute, Redlands, California, USA). We performed all statistical analyses using the software R 3.3.1 (R Core Team 2016).

Winter Phenology and Length of Stay

We calculated fall arrival dates, spring departure dates, and overall length of stay following criteria described by De La Cruz et al. (2009). For these calculations, we only used

scoters that spent the majority of at least one winter within our study area and survived to undergo at least one migration to and from the study area ($n = 36$). We defined the fall arrival date into the study area as the median date between the last location outside the study area and the first location immigrating into the study area during fall migration. Similarly, we calculated the spring departure dates as the median date between the last location within the study area and the first location emigrating out of the study area during spring migration. We calculated spring departure dates for White-winged Scoter females deployed across all capture locations. We defined the first winter length of stay as the period between transmitter deployment and the spring departure date. We estimated the length of stay during the second winter period as the difference between the fall arrival date and the spring departure date plus one additional day, to account for the possibility that birds could have been present within the study area on either or both the arrival date and departure date. We report the overall winter length of stay as mean $\pm$ SE, whereas we report only the median (range) arrival and departure dates. Fall arrival dates were calculated for scoters captured during the molting period in the St. Lawrence River Estuary in 2016, as well as those captured in Cape Cod Bay that returned during the second winter period.

Wintering Area Distribution

Following Loring et al. (2014), we calculated utilization distributions for birds wintering within our study area by first randomly selecting 40 location points from within the study area for each individual. We then calculated individual kernel-based utilization distributions using the Gaussian kernel and likelihood cross-validation bandwidth estimator within Geospatial Modeling Environment 0.7.4.0 (Beyer 2015). The likelihood cross-validation bandwidth estimator provides a better fit and less variability than least squares cross-validation when estimating utilization distributions with sample sizes <50 (Horne and Garton 2006). We then pooled the 40 locations for each bird to estimate composite 95% kernel utilization distributions and 50% kernel core-use areas. In ArcGIS, we clipped each utilization distribution and core-use area to the National Oceanic and Atmospheric Administration's (NOAA) Medium

TABLE 2. PTT duty cycle schedule based on capture period and location for adult female White-winged Scoters ($n = 52$) deployed in 2015 and 2016.

Deployment period	Location	<i>n</i>	First duty cycle	1st switch	Second duty cycle	2nd switch	Third duty cycle
Oct/Nov 2015	Cape Cod Bay	22	Intensive ^a	Mid-Apr 2016	Conservative ^c	–	–
Mar 2016	Long Island Sound	4	Conservative ^b	Mid-Oct 2016	Intensive	Mid-Apr 2017	Conservative ^c
Aug 2016	St. Lawrence River Estuary	26	Conservative	Mid-Mar 2017	Intensive	Mid-Sep 2017	Conservative ^c

^a Intensive = 4 hr on, 72 hr off.

^b Conservative = 4 hr on, 96 hr off.

^c Remained on this duty cycle until battery died.

Resolution Digital Vector Shoreline data (1:70,000; NOAA 2017a), because White-winged Scoters are not typically found on inland freshwater areas during the wintering period in our study area. To calculate winter home range size and resource selection, we only used birds ($n = 25$) that had >40 high-quality locations (i.e. location classes = 3, 2, or 1) within the study area. We supplemented the winter home range and resource selection analyses with an additional 15 scoters (9 males, 6 females) captured in the St. Lawrence River Estuary, Quebec, between 2010 and 2012, for a total sample size of 40 individuals. We then calculated the total area (km²) of the individual and composite utilization distributions and core-use areas. We reported total area for individual and composite utilization distributions and core-use areas as $\bar{X} \pm \text{SE}$. We used Wilcoxon rank-sum tests to compare total area of utilization distributions and core-use areas by sex and capture location. For birds with 2 or more distinct 50% core-use areas, we calculated the Euclidian distance (km) between centroids of each area. For birds that spent 2 consecutive winters within the study area, we compared total area of utilization distributions and core-use areas between winters using paired t -tests.

Site Fidelity

We assessed winter site fidelity between consecutive winter periods by determining the number of second winter (2016–2017) locations within the study area that fell within an individual's first winter (2015–2016) 50% core-use area and 95% utilization distribution as well as those that occurred within the composite 2015–2016 core-use areas and utilization distributions. We measured mean ($\pm \text{SE}$) geodesic distances between first and second winter core-use areas for each individual. We also calculated the percentage of second-winter points for each individual that occurred within the first winter 50% core-use areas of all other individuals for which we had 2 winters of data to assess whether birds preferentially occurred within their own core-use area compared to the core-use areas of other birds in the population.

Resource Selection During Winter

We used the composite 95% utilization distributions and 50% core-use areas to assess habitat use and resource selection within the study area. We made no distinction between diurnal and nocturnal locations when calculating individual and composite home ranges, so resource selection estimates were based on a full 24-hr diel period. Following Loring et al. (2014) and Beuth et al. (2017), we investigated third-order resource selection (Johnson 1980) by quantifying and comparing habitat covariates within the composite 95% utilization distributions (available) and 50% core-use areas (used; Manly et al. 2002; Sampling Protocol-A). We used Spatial Analyst in ArcGIS to generate the maximum number of random points within the

95% utilization distribution and 50% core-use area, with a minimum separation distance between points of 500 m to reduce spatial autocorrelation. We did not assess overlap between used and available samples as resource selection functions (RSF) are robust to such contamination (Johnson et al. 2006).

Distribution patterns of wintering sea ducks are driven in large part by available food resources (Žydelis et al. 2006, Kirk et al. 2008) and ocean bottom geography (Loring et al. 2014, Beuth et al. 2017, Heinänen et al. 2017). We chose a set of 8 geophysical and oceanographic habitat variables that we hypothesized could serve as proxies for benthic invertebrates. To quantify distance from shore, we calculated the Euclidian distance (km) from each resource unit to the nearest segment of the NOAA Medium Resolution Digital Vector Shoreline (1:70,000; NOAA 2017a). We measured water depth (m) and slope (degrees) within each resource unit using the NOAA National Geophysical Data Center Coastal Relief Model (3 arc-second) for the United States (NOAA 2017b). To estimate likely areas of hard bottom occurrence, we used a kernel-based probabilistic model (Loring 2012). We obtained sediment grain size data from The Nature Conservancy's Northwest Atlantic Marine Ecoregional Assessment data portal (Greene et al. 2010). These data were interpolated from point-based sampling and classified based on grain size (Wentworth 1922). To convert this to a continuous dataset, we assigned the median grain size value from each ordinal class to pixels within each category. Following Meattey et al. (2015), we used Marine Geospatial Ecology Tools in ArcGIS to create long-term mean raster sets for oceanographic habitat variables including sea surface temperature, sea surface salinity, and chlorophyll-*a* concentrations.

We obtained smoothed daily sea surface temperature (SST) estimates derived from interpolated data from high-resolution satellite imagery and floating buoys (Stark et al. 2007). These data are collected at a spatial resolution of 0.05 degrees latitude and longitude. To estimate sea surface salinity (practical salinity units [psu]), we used the Hybrid Coordinate Ocean Model, which produces daily estimates at a spatial scale of 1.5 degrees latitude and longitude by the National Ocean Partnership Program (Chassignet et al. 2009). As an estimated proxy for biological productivity, we used monthly estimates of chlorophyll-*a* concentrations (mg/m³) produced by the NASA Goddard Space Flight Center's Ocean Data Processing System. These data were derived from the Aqua sensor aboard the MODIS satellite system, which produces radiometric measurements of chlorophyll fluorescence at a 4-km scale (Mueller et al. 2003). To account for the ~6 yr of sampling data included in this study, we calculated 6-yr mean datasets for each of the oceanographic variables by averaging winter-month (Oct 1 to May 1) raster values. We randomly sampled habitat variables at 25% of resource units from both the

95% utilization distributions and 50% core-use areas to reduce spatial autocorrelation between variables. All habitat data were in raster format and resampled to a standardized 250 m × 250 m cell size (hereafter: resource units) prior to extraction and analysis.

We calculated Pearson product-moment correlations to assess correlations between all possible pairs of habitat covariates and checked for multicollinearity of variables using variance inflation factors (VIF). Within samples throughout the study period, pairwise correlation among variables did not exceed 0.6 and VIF values were ≤2.0. Therefore, we retained all variables through the modeling step. We used logistic regression to model habitat covariate effects and estimate the parameters for exponential resource selection models (Manly et al. 2002). All environmental variables, including quadratic terms, were included in an initial global model. Nonlinear terms for some variables (e.g., water depth, distance to shore) suggested significance in the global model, but parameter estimates were exceedingly low and not ecologically meaningful, thus only linear terms for each variable were included in further modeling. We performed backwards stepwise model selection, excluding uninformative parameters in order of least significance. We compared each model iteration, as well as an intercept-only and individual-parameter models using Akaike's Information Criterion adjusted for small sample size (AIC_c). We ranked models using AIC_c differences (ΔAIC_c) and AIC_c weights (w_i) to estimate the relative likelihood of each candidate model (Burnham and Anderson 2002). Competitive models were considered at ≤2.0 ΔAIC_c from the best-performing model if they contained no uninformative parameters, and we selected the parameter coefficients from the most parsimonious model to calculate the RSF. Model residuals were checked for spatial autocorrelation by using a Moran's *I* test in the R package SPDEP (Bivand 2009).

We predicted relative probability of use for 77,390 km² of our study area using the RSF derived from our highest-ranked logistic regression model. We were unable to predict probability of use for 5,182 km² of our study area due to incomplete spatial coverage of the sea surface temperature, salinity, and chlorophyll-*a* datasets. We calculated the RSF model using Equation (5.11) in Manly et al. (2002):

$$W(x) = \exp(\beta_1 x_1 + \dots + \beta_n x_n)$$

where W is relative probability of use, β_n are the model coefficients estimated from the logistic regression for each habitat parameter, and x_n are the predictor variables. We used RASTER CALCULATOR in ArcGIS to complete the above equation and then reclassified the distribution into 4 quantile bins to characterize relative probability of use from low to high.

We evaluated the predictive ability of the top-ranked RSF using k-fold cross-validation methods described by

Johnson et al. (2006). We used Huberty's (1994) rule of thumb to partition resource units into 3 k-folds with approximately 37% of used resource units being used for model testing against 63% of remaining model training data. We partitioned resource selection functions predicted from the model training data into 4 equal-sized quantile bins following Morris et al. (2016), who suggested that RSFs should be validated using the same classification scheme as presented visually. We determined strong predictive ability of the RSF model by a high R^2 value and a nonsignificant chi-square (χ^2) goodness-of-fit value between observed and expected proportions of use across quantile bins (Johnson et al. 2006). We assumed that areas classified with a high probability of selection in the RSF model were high quality habitat that should warrant conservation from developers when planning and siting future wind energy areas.

RESULTS

Survival and Transmitter Performance

Of 22 female White-winged Scoters implanted in Cape Cod Bay during fall 2015, 2 died within 2 weeks of PTT deployment, and 8 PTTs went offline in presumed live birds either during the first winter or outside the study area during migration or breeding. Thus, 16 female birds transmitted data throughout the entire 2015–2016 winter period, of which 14 spent the majority of at least one winter within the study area; 6 of these birds also transmitted throughout the winter of 2016–2017. Of the 4 females implanted in Long Island Sound in March 2016, 2 died before returning to the study area the following winter and 2 transmitters went offline in presumed live birds, thus none of these birds provided a full winter of data for resource selection analyses. Thirteen of the female White-winged Scoters deployed in the St. Lawrence River Estuary in August 2016 spent most of the 2016–2017 winter period within the study area. Two birds died within 2 weeks of transmitter deployment, 1 bird died ~3 mo after deployment while still on the molting grounds, and 2 transmitters went offline in presumed live birds.

Locational accuracy classes of the best-per-duty-cycle locations used to calculate winter movement phenology and generate winter home ranges ranged from location class (LC) 3 to LC B, with 75% of randomly selected locations classified as either LC 2 or LC 3 (Table 1).

Winter Phenology

We found no significant difference in spring departure date based on initial capture location ($F = 2.36$, $df = 3$ and 29, $P = 0.09$; Table 3). Tagged females spent an average of ~53% (193 days) of their annual cycle within the southern New England study area.

TABLE 3. Fall arrival, spring departure, and total winter length of stay (LOS) within the southern New England study area.

Winter	Capture location	<i>n</i>	Fall arrival	Spring departure	LOS (days)
2015–2016	Cape Cod Bay	12	–	May 20 (Apr 22 to May 27)	198 (±3)
2016–2017	Cape Cod Bay	7	Nov 14 (Oct 17–Jan 12)	May 19 (May 8–27)	179 (±16)
	Long Island Sound	4	–	May 12 (May 12–13)	–
	St. Lawrence River	13	Oct 31 (Oct 13–Nov 12)	May 17 (Apr 27 to May 26)	193 (±6)

Wintering Area Distribution

White-winged Scoter 50% core-use areas ranged widely from 32 to 4,220 km² ($\bar{X}$ = 868 ± 174 km²). Individual 95% utilization distributions ranged from 272 to 18,235 km² ($\bar{X}$ = 4,387 ± 761 km²). For the 40 White-winged Scoters (31 females, 9 males) that spent an entire winter within the study area (including the additional 2010–2012 Quebec-caught birds), the composite core-use area was 2,054 km² and the composite utilization distribution was 9,790 km² (Figure 2). Core-use areas were located in Cape Cod Bay, the outer edge of Nantucket Sound between Monomoy Island and Nantucket Island, Buzzards Bay, Long Island Sound and Montauk Point, as well as the Nantucket Shoals south of Nantucket Island. We found no significant difference in individual core-use areas (Wilcoxon rank-sum test: z = −0.6, P = 0.6) or utilization distribution size (z = −0.5, P = 0.6) based on initial capture location, and core-use areas were similar between sexes (z = −0.2, P = 0.9). For birds that spent consecutive winters in the study area, total area of utilization distributions and core-use areas decreased by ~30% and ~20%, respectively, although this was not significant (paired t -test: t_4 = 1.6, P = 0.2 in both cases). Twelve birds spent the majority of the 2015–2016 or 2016–2017 winter outside the study area, including Lake Ontario, mid-coast Maine, and coastal Nova Scotia.

Individual birds occupied 1–5 distinct 50% core-use areas, with 29 of 40 birds occupying 2 or more. Mean distance between multiple core-use areas was 101 km (±16), ranging from 37 to 404 km. Composite 95% utilization distributions and 50% core-use areas overlapped with or immediately bordered 484 km² and 69 km² of current wind energy area lease blocks, respectively.

Site Fidelity

Six females tagged in Cape Cod Bay in 2015 survived and continued to provide data into the winter of 2016–2017. All 6 birds returned to the study area, but only 5 provided enough winter locations in their second winter to calculate home ranges. Females that returned exhibited moderate to high degrees of winter site fidelity. Most locations during the 2016–2017 winter period were within the 2015–2016 composite core-use area ($\bar{X}$ = 66% ± 14, range: 33–100%) and utilization distribution ($\bar{X}$ = 97% ± 2, range: 91–100%; Figure 3). However, individuals that returned to the study area during the 2016–2017 winter period exhibited varying degrees of site fidelity to their individual core-use areas ($\bar{X}$

= 44% ± 19, range: 0–90%) and utilization distributions ($\bar{X}$ = 59% ± 21, range: 0–100%) from the first winter. In comparison, scoters returning to the study area during the second winter were equally philopatric to first winter utilization distributions ($\bar{X}$ = 57% ± 8, range: 0–100%), but had notably fewer locations within the first winter core-use areas ($\bar{X}$ = 27% ± 6, range: 0–79%) of other birds in the population. The mean distance between first- and second-year core-use areas across individuals was 106 ± 15 km, with a maximum distance of 188 km.

Resource Selection During Winter

Core-use areas within our study area were generally shallower and closer to shore relative to utilization distributions, while bottom slope and sediment grain size were similar throughout (Table 4). The best-performing logistic regression model estimating relative probability of use (n = 40; 31 females, 9 males) included 4 significant parameters (sea surface temperature, hard bottom probability, sea surface salinity, and chlorophyll-*a* concentration) and accounted for 49% of Akaike weight (Table 5). The second-ranked model was within 2 Δ AIC_c but contained an uninformative parameter and thus was not considered competitive. Based on this best model, scoter core-use areas were negatively associated with sea surface temperature and sea surface salinity and positively associated with probability of hard bottom substrate and mean chlorophyll-*a* concentrations, relative to utilization distributions (Table 6). Results from the k -fold cross-validation showed strong positive correlation (R^2 = 0.83) between area-adjusted proportions of observed and expected habitat use and χ^2 goodness-of-fit tests were nonsignificant across all 3 iterations of model validation (K_1 : χ^2 = 1.53, P = 0.67; K_2 : χ^2 = 1.34, P = 0.72; K_3 : χ^2 = 1.65, P = 0.65), indicating that the top-ranked RSF model was capable of reliably predicting cross-validated use locations. The model slightly under-predicted use in the highest quantile bin. We determined that a small degree of positive spatial autocorrelation was present in the residuals of our best-performing model (Moran's I = 0.2, P = 0.001).

The top-ranked RSF model was able to predict relative probability of use by White-winged Scoters for 77,390 km² of the 82,572 km² study area. Throughout the study area, 18,654 km² (24.1%) were classified as low probability of use, 19,122 km² (24.7%) were classified as medium-low, 20,965 km² (27.1%) were classified as medium-high, and 18,649 km² (24.1%) were classified as high probability of

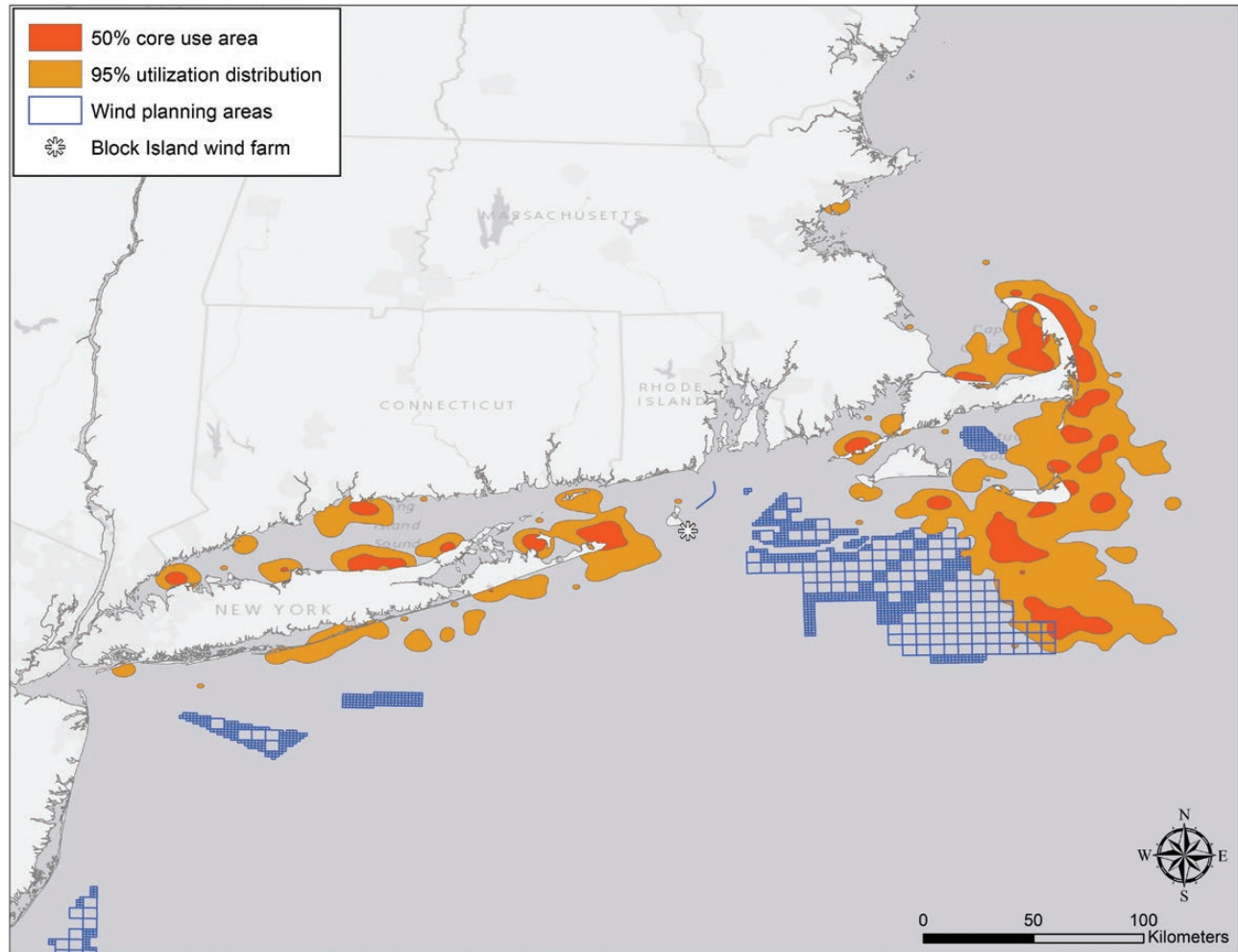


FIGURE 2. Composite kernel-based winter 95% utilization distributions and 50% core-use areas of White-winged Scoters ($n = 40$; 31 females, 9 males) implanted with satellite transmitters between 2010 and 2016 in relation to current and proposed offshore wind energy areas.

use (Figure 4). Approximately 420 km² of current wind energy area lease blocks fell within or immediately bordered areas classified as high probability of use.

DISCUSSION

Winter Phenology

The results from this study confirm past survey data and other telemetry studies highlighting the importance of this region for White-winged Scoters during winter (Silverman et al. 2013, Baldassarre 2014, Sea Duck Joint Venture 2015, Veit et al. 2016, C. Lepage personal communication). Our satellite-tagged birds spent ~53% (193 days) of their annual cycle within the southern New England study area. This is longer than estimates for Black Scoters (Loring et al. 2014) and Common Eiders (Beuth et al. 2017), which spent an average of 147 days and 135 days, respectively, within the same New England wintering area. Surf Scoters

(*Melanitta perspicillata*) wintering along the mid-Atlantic coast of the U.S. spent an average of 133 days on the wintering grounds (Meattey et al. 2015), while King Eiders (*Somateria spectabilis*) in the Bering Sea spent an average of 160 days on their wintering grounds (Oppel et al. 2008).

White-winged Scoters typically departed the study area by the third week of May. This is consistent with Black Scoters in the same area (range: Mar 4 to May 24; Loring et al. 2014) but later than Common Eiders (range: Mar 18 to Apr 20; Beuth et al. 2017). Also, aerial surveys from 2011 to 2015 documented White-winged Scoters were most abundant along the western edge of the Nantucket Shoals during the spring period (Veit et al. 2016), which was at the eastern edge of the Massachusetts offshore WEA lease blocks. While our home range analyses did not consider date of locations within the study area, many White-winged Scoters staged on the shoals during mid-May for 1–2 weeks prior to spring departure. This further suggests that Nantucket Shoals are a seasonally

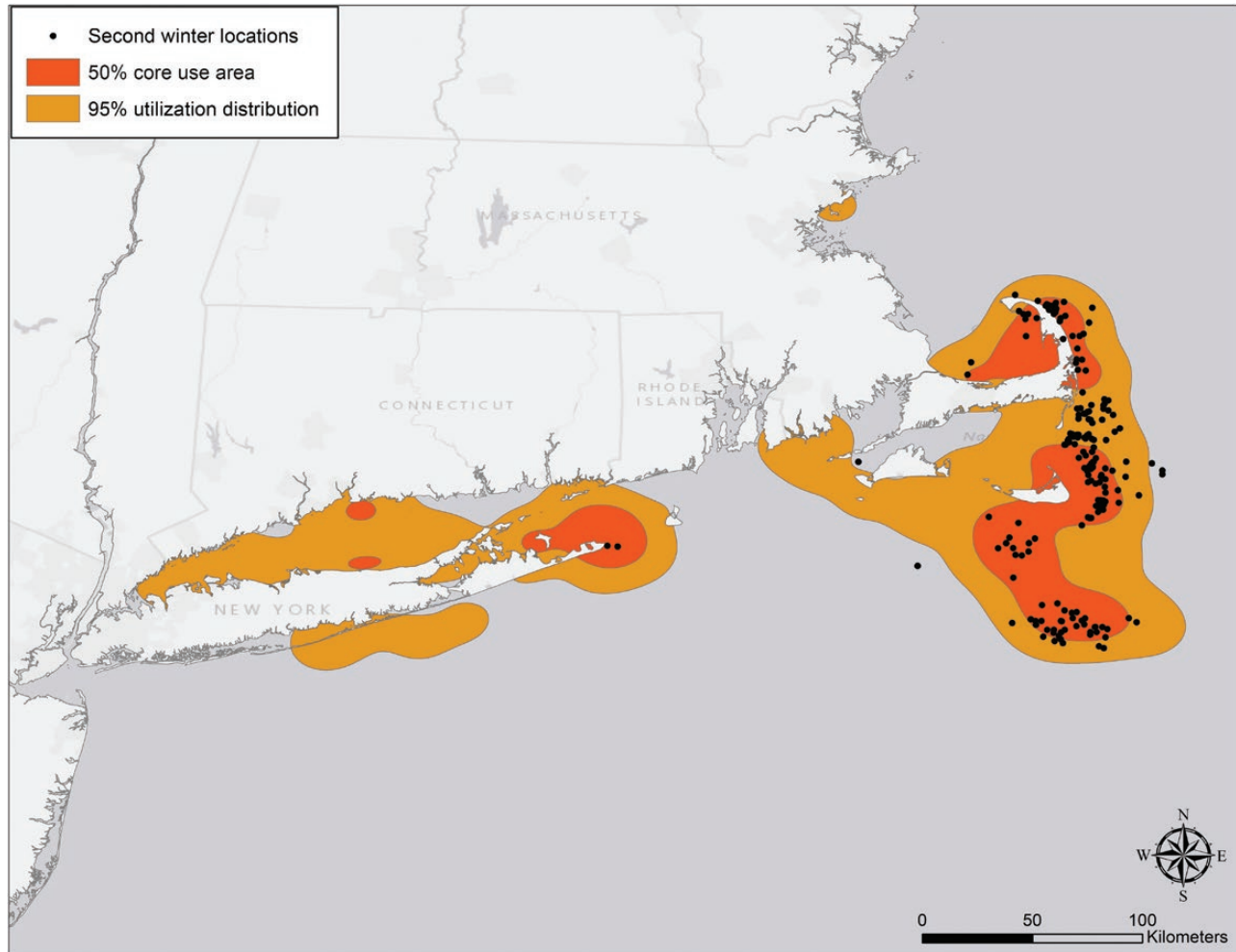


FIGURE 3. Second-winter (2016–2017) locations of 5 female White-winged Scoters in relation to composite ($n = 30$ individuals) kernel-based first-winter (2015–2016) 95% utilization distributions and 50% core-use areas.

important area for sea ducks, likely due to high densities of high-quality prey (e.g., the pelagic amphipod *Gammarus annulatus*) that sea ducks may rely on for reserve-building prior to long-distance migration (White et al. 2009). Most studies of winter diet composition of White-winged Scoters report very small percentages of non-bivalve prey (e.g., Polychaeta; Anderson et al. 2008), although the importance of soft-bodied prey such as amphipods has been well documented during the breeding period (Brown and Fredrickson 1986, Benoit et al. 1996, Haszard and Clark 2007), suggesting the possibility that White-winged Scoters may take advantage of similar food sources during the winter months in pelagic habitats if available. Any disturbance to this area could have detrimental effects on White-winged Scoters during a crucial part of the annual cycle.

Wintering Area Distribution

Individual White-winged Scoters wintering in the study area varied widely in the size of their core-use and utilization

distributions (32–4,220 km² and 272–18,235 km², respectively), with generally larger winter home ranges than other species of sea ducks. As kernel home range estimates can vary depending on kernel method, sample size, and bandwidth estimators, direct comparisons among studies with nonidentical methods should be interpreted with caution. Schamber et al. (2010) documented 50% core-use areas of 448 km² and 160 km², and 95% utilization distributions of ~3,670 km² and ~1,298 km², for King Eider and Black Scoter in Bristol Bay, Alaska. Winter home range sizes of White-winged Scoters in this study were higher than those reported for Harlequin Ducks (*Histrionicus histrionicus*) in Prince William Sound, Alaska (11.5 km²; Iverson and Esler 2006). Communal foraging behavior in sea ducks and resulting depletion of local food sources is well documented (Guillemette et al. 1996, Kirk et al. 2007, Kirk et al. 2008, Loring et al. 2013). Large and variable winter home range sizes in our study may have been a result of changing prey densities throughout the winter, requiring birds to expand

TABLE 4. Mean ($\bar{X}$), standard error (SE), and range of values for habitat variables sampled within composite 95% utilization distributions (available) and 50% core-use areas (used) of White-winged Scoters ($n = 40$).

Habitat covariate	Used		Available	
	$\bar{X} \pm \text{SE}$	Range	$\bar{X} \pm \text{SE}$	Range
Distance from shore (km)	17.3 $\pm$ 1.1	0.3–63.4	19.5 $\pm$ 0.5	0.0–72.4
Water depth (m)	24.0 $\pm$ 0.6	3.0–47.0	25.7 $\pm$ 0.3	0.0–87.0
Bottom slope (°)	0.3 $\pm$ 0.1	0.0–3.3	0.2 $\pm$ 0.1	0.0–2.5
Hard bottom probability (0–1)	0.04 $\pm$ 0.01	0.0–0.8	0.02 $\pm$ 0.01	0.0–0.8
Sediment grain size (mm)	0.3 $\pm$ 0.1	0.02–0.5	0.3 $\pm$ 0.0	0.02–0.5
Sea surface temperature (°C)	8.6 $\pm$ 0.1	7.8–11.4	8.7 $\pm$ 0.1	7.6–13.3
Sea surface salinity (psu)	32.7 $\pm$ 0.1	31.9–32.9	32.7 $\pm$ 0.1	31.5–32.9
Chlorophyll- <i>a</i> (mg m ⁻³)	5.3 $\pm$ 0.1	2.3–8.3	5.1 $\pm$ 0.04	2.3–21.6

TABLE 5. Number of model parameters (K), Akaike Information Criterion differences (ΔAIC_c), maximized log-likelihood [$\log(L)$], and Akaike weights (w_i) for logistic regression models of winter habitat use vs. availability used to estimate coefficients for exponential resource selection functions. Model parameters include distance from shore (DIST), water depth (WD), bottom slope (SL), hard bottom probability (HB), sediment grain size (SED), sea surface temperature (SST), sea surface salinity (SAL), and chlorophyll-*a* concentrations (CHL).

Model parameters	K	ΔAIC_c	$\log(L)$	w_i
CHL, HB, SAL, SST	5	0.0 ^a	−731.2	0.49
CHL, HB, SAL, SL, SST	6	1.0	−730.7	0.29
CHL, HB, SAL, SED, SL, SST	7	2.5	−730.5	0.14
CHL, HB, SAL, SED, SL, SST, WD	8	4.4	−730.4	0.06
CHL, DIST, HB, SAL, SED, SL, SST, WD	9	6.4	−730.4	0.03
HB	2	9.7	−739.1	0.00
SAL	2	14.4	−741.5	0.00
WD	2	17.9	−743.2	0.00
CHL	2	18.3	−743.4	0.00
SST	2	18.9	−743.7	0.00
DIST	2	18.9	−743.7	0.00
SL	2	19.0	−743.8	0.00
Intercept only	1	20.2	−745.4	0.00
SED	2	22.2	−745.4	0.00

^a Lowest AIC_c value: 1472.5.

beyond core-use areas to locate food. More than half the birds in our study occupied 2 or more disjunct core-use areas throughout a single winter. Several of these individuals occupied core-use areas in both the eastern (Cape Cod) and western (Long Island) portions of the study area, with up to 404 km between core-use areas. Several cases of White-winged Scoters using secondary wintering sites have also been documented with the 2010–2012 Quebec-caught birds, with several birds moving from Long Island to the Nantucket area in late winter, prior to spring migration (C. Lepage personal communication).

These instances of long-distance within-winter movements highlight the potential vulnerability of White-winged Scoters to offshore wind energy developments in the area. Additionally, a recent study on Black Scoters throughout the migratory and wintering period in

southern New England highlighted a tendency to venture outside nearshore core-use areas to locations farther offshore, increasing the likelihood of encountering offshore wind energy facilities (Loring et al. 2014). While the locations of current wind energy lease blocks in the study area have minimal overlap with scoter core-use areas, the development of offshore structures such as wind turbines could act as an impediment to White-winged Scoters moving between important areas in Cape Cod Bay and Long Island Sound during the winter period.

Site Fidelity

White-winged Scoters in our study were highly philopatric to the broad southern New England wintering area and to the location of composite home ranges but exhibited varying degrees of interannual site fidelity to their individual home ranges. While the sample size of birds with location data spanning consecutive winters was small ($n = 5$ females), our results are generally consistent with other studies of winter philopatry in sea ducks. In comparison, 82% of White-winged Scoters studied in Quebec from 2010 to 2012 returned to the same wintering area ($n = 17$; C. Lepage personal communication). A study of Common Eiders wintering in southern New England found that about half of second-winter locations fell within an individual's first-year core-use areas (~51%) and nearly all locations were within the composite core-use area (96%; Beuth et al. 2017). In the same study area, wintering Black Scoters exhibited only 24% and 32% spatial overlap between first- and second-year core-use areas and utilization distributions, respectively (Loring et al. 2014). The site fidelity we observed was also generally lower than that reported for Surf Scoters in the mid-Atlantic (91%; Meattey et al. 2015), Common Eiders on the Pacific coast (95%; Petersen et al. 2012), and Harlequin Ducks (62%; Robertson et al. 2000).

Knowledge of local prey distributions is one of several advantages that could result from a high rate of site fidelity among sea ducks (Robertson and Cooke 1999). The high rate of population-level site fidelity we observed supports this hypothesis, as many core-use areas we identified were

located near high-productivity areas known to be of seasonal importance to sea ducks (i.e. Nantucket Shoals; [White et al. 2009](#), [Veit et al. 2016](#)). The variability in individual-level site fidelity reported in our study suggests that White-winged Scoters are also able to adjust to changes in local environmental conditions between years to respond to shifting prey distributions and habitat quality.

TABLE 6. Coefficients (β) and upper and lower 95% confidence intervals of environmental parameters from the best-fit resource selection function model for White-winged Scoters ($n = 40$).

Variable	β	Lower 95% CL	Upper 95% CL
Sea surface salinity (psu)	-0.8384	-1.573	-0.079
Hard bottom probability (0–1)	2.6098	1.210	4.004
Sea surface temperature (°C)	-0.2519	-0.475	-0.043
Chlorophyll- <i>a</i> (mg m ⁻³)	0.0867	0.002	0.171

Resource Selection During Winter

This is the first study to document spatially explicit resource selection and habitat use of White-winged Scoters wintering on the Atlantic Coast of North America. Scoter core-use areas within our study area were associated with areas of lower sea surface temperatures, lower salinity, higher probability of hard bottom substrate, and higher mean chlorophyll-*a* concentrations relative to utilization distributions. [Loring et al. \(2014\)](#) found a similar significant positive effect of hard bottom probability on Black Scoters in Rhode Island Sound, presumably foraging on sessile prey such as blue mussels (*Mytilus edulis*) frequently abundant in harder substrates ([Goudie and Ankney 1986](#)). We found the importance of hard bottom probability in our models surprising, as White-winged Scoters prefer prey in predominantly soft-sediment habitats ([Stott and Olson 1973](#), [Anderson et al. 2008](#)). However, the significance of this habitat characteristic may be tied to interactions with

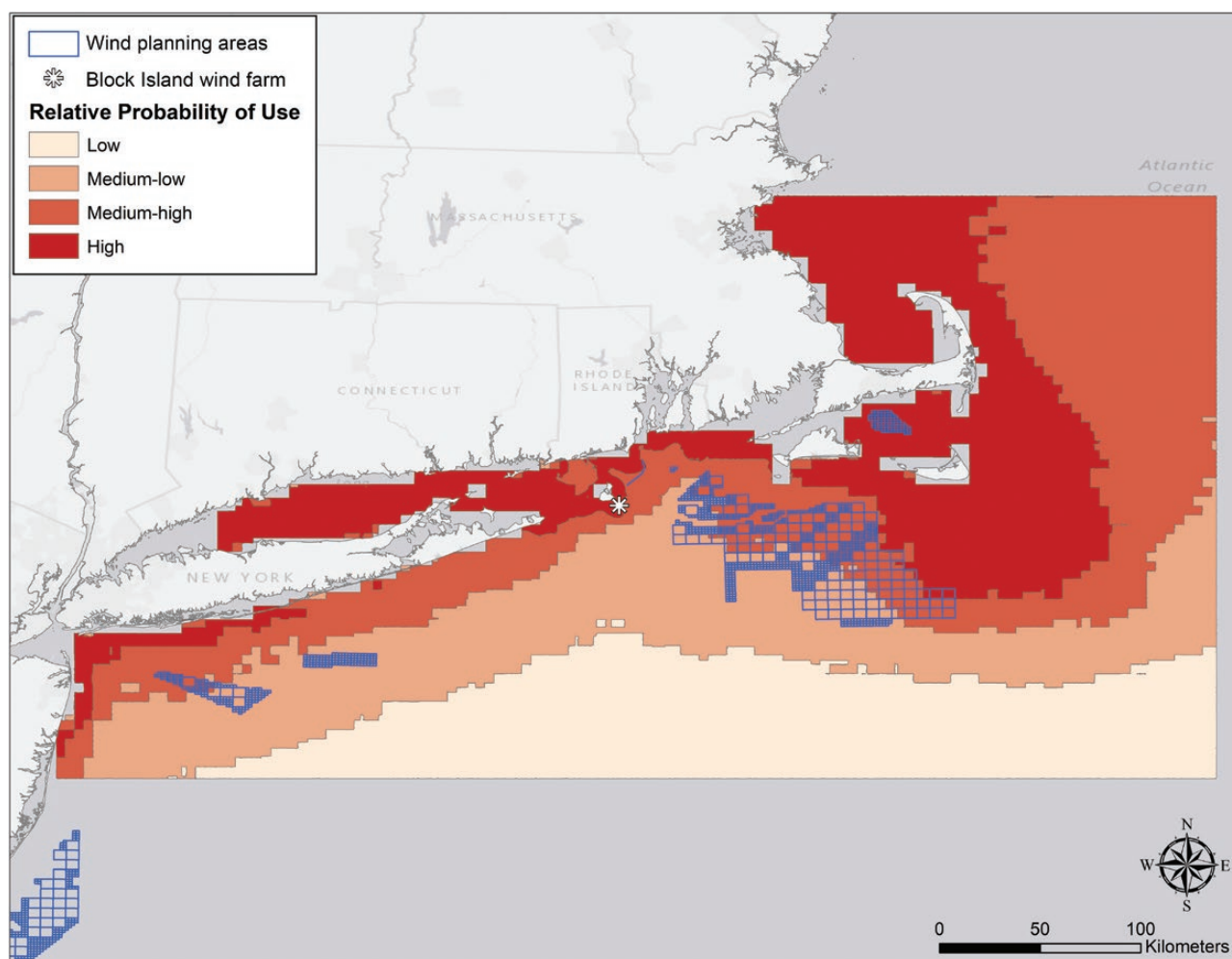


FIGURE 4. Quartile-based relative probability of use (<25% to >75%) predicted for White-winged Scoters across the southern New England study area by the top-ranked resource selection function model in relation to current and proposed offshore wind energy areas.

other parameters not explored in our analyses rather than foraging.

While studies directly associating seasonally dynamic climate variables to sea duck habitat use and distribution are limited, our study does corroborate findings by Zipkin et al. (2010) who found sea surface temperature to have a significant negative effect on long-term White-winged Scoter count data along the Atlantic Coast. More frequently, oceanographic climate variables affect benthic invertebrate physiology and distribution (Lesser et al. 2010, Waldeck and Larsson 2013, Sorte et al. 2016), which in turn can directly influence sea duck abundance and distribution (Perry et al. 2007, Kirk et al. 2008, Loring et al. 2013). Increases in sea surface temperature by only a few degrees, corresponding to a mild vs. cold winter period, were associated with 15–19% body mass loss in blue mussels in the Baltic Sea (Waldeck and Larsson 2013). Similarly, Lesser et al. (2010) documented blue mussels in the Gulf of Maine exhibiting increased expression of heat shock proteins and antioxidant enzyme activity when exposed to higher seawater temperatures. Such environmental stress has been associated with slower growth and impaired reproductive capacity (Petes et al. 2007). During the winter period when White-winged Scoters must build energy reserves for migration and breeding, selection for areas of lower sea surface temperature may be indicative of higher-quality prey.

Chlorophyll-*a*, as a proxy for overall levels of primary productivity, and salinity can be important parameters for predicting both seabird and benthic invertebrate distributions (Chester et al. 1983, Ballance et al. 1997, Suryan et al. 2012). We assumed that higher chlorophyll concentrations corresponded to increased primary productivity, and thus higher benthic biomass or food availability to foraging birds (Grebmeier 1993, Phillips et al. 2006). King Eiders in the Bering Sea during the molting period and winter were associated with areas of lower salinity (Phillips et al. 2006); preference for foraging in low-temperature and high-chlorophyll areas has also been documented in other upper trophic-level seabirds, such as the Cape Gannet (*Morus capensis*) in the Benguela upwelling zone off South Africa (Grémillet et al. 2008). It is worth considering the narrow range of sea surface salinity values in both the used and available areas of this study. While our RSF model identified salinity as a significant, informative parameter, this may be a result of the large sample size and not indicative of ecological significance. Future studies should prioritize identifying the correlations between these oceanographic habitat variables and shellfish beds in southern New England, as these are likely a primary driver of sea duck distributions in the area.

We found that White-winged Scoters inhabited relatively shallow (<25 m) areas that averaged ~17 km from shore. This corroborates aerial surveys conducted south of Nantucket Island in Massachusetts between 2011 and

2015, which found that White-winged Scoters were the most pelagic of the sea duck species recorded (Silverman et al. 2013, Veit et al. 2016). Much of this bias toward areas farther from shore comes from the high abundance of White-winged Scoters utilizing the Nantucket Shoals. This area also sits adjacent to a large expanse of wind energy lease blocks that skirt the shoals along their western edge. Future development in this area could pose a high risk of displacement, or act as a barrier to White-winged Scoters moving into or within this important habitat. Core-use areas for Black Scoters in Rhode Island averaged ~15 m in water depth but were much closer to shore (~4 km; Loring et al. 2014). Similarly, Common Eiders in the same study area were found primarily in <20 m of water within 2 km of shore (Beuth et al. 2017). King Eiders in the Bering Sea during winter were in slightly deeper water (~38 m) but were within 12 km of shore (Phillips et al. 2006). In Europe, Common Scoters primarily forage in waters shallower than 20 m (Fox 2003). Foraging scoters are well documented to tend to congregate in areas with high prey density (Kirk et al. 2008, Loring et al. 2013), which occurs along the southern New England shelf at depths shallower than 26 m (Theroux and Wigley 1998). It is assumed that offshore development within this depth range would have the highest potential for displacement of wintering sea ducks in the study area. While core-use areas were found in areas close to shore, the results of our RSF model likely underestimate or incompletely predict probability of use in nearshore areas due to a lack of spatial coverage of habitat variables used in the model. Finally, we acknowledge that the presence of positive spatial autocorrelation in the model residuals from our top-ranked logistic regression model is a potential limitation of our study that we do not directly address in our analyses. However, we remain confident that our model results are not strongly impacted by these limitations, as the degree of autocorrelation was relatively low and our cross-validation results show very good predictive ability of our top model.

Management and Conservation Implications

In the United States, several sites along the mid-Atlantic Outer Continental Shelf have been proposed for offshore wind energy facilities, and commercial wind energy leases have been issued for offshore areas in Massachusetts, Rhode Island, Delaware, Maryland, and Virginia (Manwell et al. 2002, Breton and Moe 2009, Musial and Ram 2010). Large-scale surveys suggest these areas provide important staging and wintering habitat for several sea duck species (Silverman et al. 2013, Veit et al. 2016), and detailed studies of fine-scale habitat selection have confirmed this importance for multiple species that utilize these offshore waters (Loring et al. 2014, Meattey et al. 2015, Berlin et al. 2017, Beuth et al. 2017). Throughout southern New England from Long Island to Cape Cod, several state agencies have

invested millions of dollars toward site-planning of wind energy facilities. For example, Rhode Island recently committed funding toward baseline monitoring of natural resources, including sea ducks (Winiarski et al. 2014), for the Rhode Island Ocean Special Area Management Plan (RI Ocean SAMP 2010). Using the most current modeling frameworks, animal movement information gathered through these monitoring efforts can help integrate ecological data into marine spatial planning and policy (Masden et al. 2012, Lascelles et al. 2016).

The results from our study demonstrate that current lease areas for offshore wind energy development show minimal overlap with White-winged Scoter winter home ranges and areas predicted by our RSF model as having a high probability of use. However, the large proportion of birds utilizing multiple disjoint core-use areas, often on opposite sides of the study area, suggests caution when planning future offshore wind energy developments. While direct collision risk is of minimal concern for sea ducks, the effects of displacement and obstruction could have compounding effects on birds' ability to utilize the entirety of the wintering area and respond to seasonally dynamic habitat quality. Numerous White-winged Scoters spent more than half of the annual cycle on wintering grounds in southern New England, exhibited a high degree of interannual site fidelity to composite core-use areas, and demonstrated a tendency to range widely within the study area, often traveling across areas where current wind energy lease areas exist. Thus, important habitats and key environmental characteristics identified by this study should be carefully considered when siting and developing future offshore wind energy areas.

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Ethics statement: All required permits were secured prior to field work, and the project and methodology were approved by the University of Rhode Island Animal Care and Use Committee (IACUC #AN1516-002). Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Author contributions: DEM, SRM, PWCP, LS, and JEO conceived the ideas and designed methodology; GHG oversaw the surgical team and advised on protocols; all authors contributed to field work and data collection; CL and SGG provided supplemental telemetry data; DEM analyzed the data; DEM led the writing of the manuscript. All authors provided critical input and edits to the drafts and gave final approval for publication.

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