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## A synthetic analysis of post-construction displacement and attraction of marine birds at offshore wind energy installations

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### ABSTRACT

Changes in the distribution of marine birds in their at-sea foraging, resting, and migratory habitats are frequently observed following construction of offshore wind energy installations. However, the presence and strength of both displacement and attraction effects have been shown to vary widely among species and locations. An understanding of the underlying factors driving both occurrence and detection of distributional changes is required to inform wind farm design and develop best practices for environmental impact assessment, monitoring, and mitigation. Drawing from 39 publications and reports, we conducted a meta-analysis to assess the likelihood of detecting attraction or displacement of marine birds by wind energy infrastructure as well as the proportional change in use of the wind energy areas. We evaluated these outcomes as functions of wind farm characteristics, biological characteristics of birds, and the observation process. We found that the presence and strength of distributional change varied most strongly among taxa and seasons, followed by study design criteria and wind farm characteristics. Displacement and attraction effects were more frequently detected during the breeding season and in studies with a larger overall study area footprint relative to the size of the wind farm. Effects were also greater at wind farms further offshore and with lower turbine densities. Effects were significant and negative for loons, grebes, sea ducks, alcids, and gannets, and were mainly neutral or slightly positive for gulls, waterfowl, and cormorants. Less frequently-observed taxa such as fulmars and skuas had low frequency of significant displacement effects but large negative effect sizes, suggesting that displacement may be underestimated in these groups. Overall, we recommend that future monitoring studies clearly report underlying metrics (i.e., abundance and/or density) within defined study areas to allow for robust comparison among sites, species, and survey methodologies.

### 1. Introduction

Offshore wind energy development, considered by many nations to be a cornerstone strategy for reducing CO<sub>2</sub> emissions and meeting climate goals, is being developed at a rapid rate worldwide (Li et al., 2022). With this rapid pace of development comes a need to understand the effects of offshore wind energy installations on marine ecosystems. Offshore wind energy infrastructure can affect marine wildlife both directly, through interactions with structures and changes in habitat availability, and indirectly, through food web effects or shifts in hydrology (Allison et al., 2019). For marine birds, the principal pathways for direct effects are collision and distributional change. Collision effects

include mortality or injury caused by individuals being struck by wind farm structures and associated vessels (reviewed in Cook et al., 2018), while distributional change involves shifts in habitat use to avoid (i.e., displacement) or occupy (i.e., attraction) wind farm areas (Inger et al., 2009; Fox and Petersen, 2019; Degraer et al., 2021). Quantifying these effects—and, ultimately, their population-level consequences—is key to assessing the environmental impact of offshore wind energy development (Abramic et al., 2022) and mitigating its effects on vulnerable marine bird populations (Furness et al., 2013; Busch and Garthe, 2016; Pirota et al., 2022).

Large uncertainties remain around understanding and predicting the consequences of new offshore wind energy installations on marine bird

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distributions and habitat use (Masden et al., 2015). One of the most substantial bottlenecks to measuring changes in at-sea distribution of marine birds due to offshore wind energy development is the high magnitude of variability in ecosystem conditions (Searle et al., 2023). Marine systems are inherently dynamic, with considerable spatiotemporal heterogeneity in distribution of resources and species (Duck, 2012). Wide-ranging taxa like marine birds can respond to these dynamic ecosystem conditions by moving large distances to access resources, resulting in patchy and highly variable distributions across vast areas (Mueller and Fagan, 2008; Smith et al., 2019). In addition to their inherent variability, marine ecosystems worldwide are undergoing relatively rapid alterations due to other external changes (e.g., climate change, shipping activity, fishing activity), each of which further affects the distribution of marine resources (Tommasi et al., 2017; Gissi et al., 2019). Prior studies of the effects of offshore wind on marine bird distributions at the scale of individual wind farms often produce inconclusive results due to the difficulty of separating effects of offshore wind energy development from background variation at relatively small spatiotemporal scales with limited sampling effort (Maclean et al., 2013; Vanermen et al., 2015; Busch and Garthe, 2016). This highlights the need for intensive sampling to detect effects, particularly when effect sizes are small to moderate (Maclean et al., 2013).

To account for underlying variability, measuring distributional changes requires choosing the appropriate monitoring techniques to detect changes. Monitoring programs that compare marine bird distributions in offshore wind farms with surrounding habitats, either before and after construction or after construction only, are considered the standard method for separating effects of offshore wind energy development from underlying variation (Methratta, 2021), though more recently, individual tracking is also becoming increasingly used to study these effects using after-construction methods (e.g., Thaxter et al., 2018; Johnston et al., 2022; Peschko et al., 2021). Within these frameworks, there are many considerations that can affect a study design's ability to accurately detect and characterize distributional changes. Multiple methods are available for measuring bird abundance, each with specific strengths and weaknesses (Cook et al., 2018). Individual marine bird species vary in abundance, morphology, habitat needs, and behavioral responses, which in turn affect their detectability for distributional surveys (Camphuysen et al., 2004); therefore, a survey design or technique that is sufficient to detect changes in one species or season may not be appropriate for another. Variation in wind farm layouts and locations may also affect the scale and extent of distributional change (Drewitt and Langston, 2006). The effects of such site-, species-, and study-specific design considerations cannot be measured for any individual project, but rather require comparing results across multiple wind farm sites and locations. Such synthetic analyses can also help to address the spatiotemporal limitations of individual studies (Nakagawa et al., 2023) and can help to identify factors driving observed trends and to look for opportunities for improved assessment and mitigation across a range of conditions (Machado and de Andrés, 2023). Dierschke et al. (2016) previously conducted a qualitative synthesis to assess the comparative vulnerability of different seabird taxa to displacement; however, a comparative assessment of the potential drivers of observed differences in displacement and attraction effects is currently lacking.

To evaluate factors influencing the detection of displacement and attraction effects, we synthesized findings from multiple studies of marine birds at existing wind farms in a meta-analytical framework. We reviewed the literature to identify studies that tested for distributional changes in one or more marine bird species, extracted the observed effect detections and relative effect sizes, and used a generalized linear modeling framework to evaluate the relative influences of study design parameters, wind farm characteristics, species, and seasonal phenological variation on occurrence and strength of distributional changes. Our analysis seeks to provide guidance for designing future studies and reporting results, as well as to identify species- and site-specific factors that may contribute to the detection of displacement and attraction

responses of marine birds.

## 2. Methods

### 2.1. Literature review

We conducted a literature search focusing on empirical studies at operational wind farms examining displacement and attraction of marine birds building from two recent review papers (Dierschke et al., 2016; Cook et al., 2018) in addition to a literature search and expert-informed literature review. The published and grey literature cited in the above reviews were added to sources identified via Google Scholar and Tethys Knowledge Base (Copping et al., 2013), including relevant peer-reviewed papers, reports, and conference proceedings. Google Scholar search terms included: Avian/birds/seabirds + offshore wind/offshore wind farm/offshore wind energy/marine wind/marine wind farm + displacement/attraction/avoidance. The Tethys Knowledge Base only includes renewable energy studies and products and was filtered based on different filters: Wind energy/fixed offshore wind/floating offshore wind +attraction/avoidance/displacement + birds/seabirds. We also examined the literature cited in documents that met the search criteria to identify additional relevant publications and reports. Next, a group of subject matter experts (Avian Displacement Guidance Committee, New York State Environmental Technical Working Group; <https://www.nyetwg.com/avian-displacement-guidance>) identified additional potential sources for consideration. Since both the guidance committee and the Tethys Knowledge Base consortium include primarily North American and European offshore wind experts, we expect our literature search to be comprehensive for these regions; however, we may not have captured all relevant non-English-language literature from outside North America and Europe.

Following compilation, source documents were screened for relevance, and studies were included ( $n = 39$ ; Table 1) in the meta-analysis if they used empirical data from field studies to directly examine displacement and attraction and reported sufficient data to test for differences (i.e., mean values and measures of precision). Sources that were excluded from further review included those focused on method development, risk assessments, monitoring or mitigation plans, and publications on effects irrelevant to displacement (e.g., meso/micro-scale avoidance, collision risk). Sources were also excluded if their data were redundant with another study. In instances of duplicative data (e.g., multiple monitoring reports from the same site), we used the study with the most complete dataset.

### 2.2. Data extraction

As noted in previous reviews (Dierschke et al., 2016), study designs, field methods, analyses, and statistics differ considerably between studies. We extracted consistent information on biological parameters of the target bird populations across all studies, including species, taxon, season (e.g., breeding season, non-breeding season, migration, year-round), and stage (e.g., juvenile, adult, all) (Table 2). Since the only studies that targeted adults only (rather than all stages) occurred during the breeding season, we did not include stage in subsequent analyses. We also extracted study design parameters including study method (e.g., boat-based survey, digital aerial survey, visual aerial survey, multi-method survey, GPS telemetry, radar); temporal comparison (i.e., before and after construction or after only); spatial comparison (i.e., control-impact, gradient); distance to effect (i.e., the distance from the impact area at which displacement or attraction was measured, either the precise distance to a point or radius at which bird abundance was measured, or the mean distance to a fixed distance band, gradient width, and/or control area); buffer width (i.e., the distance around the outermost turbines assumed to be part of the impact area); minimum and maximum distances from the impact area (wind farm edge or buffer edge) to the reference area (gradient and/or control); total study area

**Table 1**

Abbreviated citation and study details for published and grey literature included in the literature review and subsequent meta-analysis of marine bird displacement and attraction at offshore wind energy sites. Wind farm id numbers (in brackets) correspond to numbered wind farm locations in Fig. 1.

Citation	Wind farm [id]	Region	Country	Taxa	Methods
Aumuller et al., 2013	Alpha Ventus [5]	North Sea	Germany	Gannets	Visual Observations
Canning et al., 2013	Robin Rigg [2]	Celtic Sea	U.K.	Alcids, Cormorants, Fulmars, Gannets, Gulls, Loons and grebes, Sea ducks	Boat-based surveys
Clewley et al., 2021	Walney and Burbo Bank Extensions [4]	North Sea	U.K.	Gulls	GPS telemetry
Degraer et al., 2021	Norther [16]	North Sea	Belgium	Gulls	GPS telemetry
Gill et al., 2008	Kentish Flats [13]	North Sea	U.K.	Alcids, Cormorants, Gulls, Loons and grebes, Sea ducks	Visual Aerial surveys
*Goddard et al., 2017	Westermost Rough [21]	North Sea	U.K.		Digital aerial surveys
Guillemette et al., 1998	Tuno Knob [27]	Baltic Sea	Denmark	Sea ducks	Visual Aerial surveys, Visual observations
Heinanen et al., 2020	Multiple – German Bight [23]	North Sea	Germany	Loons and grebes	Digital aerial survey, Satellite telemetry
Johnston et al., 2022	Multiple – South Walney cluster [3]	North Sea	U.K.	Gulls	GPS telemetry
*Kahlert et al., 2004	Nysted [26]	Baltic Sea	Denmark	Waterfowl	Radar
Leopold et al., 2013	Egmond aan Zee [8]	North Sea	Netherlands	Alcids, Cormorants, Fulmars, Gannets, Gulls, Loons and grebes, Sea ducks, Terns	Boat-based survey
Masden et al., 2009	Nysted [26]	Baltic Sea	Denmark	Sea ducks	Radar
Mendel, 2012	Alpha Ventus [5]	North Sea	Germany	Loons and grebes	Visual aerial survey
Mendel et al., 2019	Butendiek and Helgoland-cluster [22]	North Sea	Germany	Loons and grebes	Combined survey methods
Nilsson and Green, 2011	Lillgrund [25]	Baltic Sea	Sweden	Cormorants, Gulls, Sea Ducks, Waterfowl	Radar, Boat-based survey, Visual aerial survey
Project Management Support Services (PMSS), 2006	North Hoyle [1]	Celtic Sea	U.K.	Alcids, Cormorants, Gulls, Loons and grebes, Sea ducks	Boat-based survey, Visual aerial survey
Percival, 2013	Thanet [19]	North Sea	U.K.	Alcids, Gulls, Gannets	Boat-based survey
Percival, 2014	Kentish Flats [13]	North Sea	U.K.	Loons and grebes	Boat-based survey
Perrow et al., 2006	Scroby Sands [18]	North Sea	U.K.	Terns	Boat-based survey
Peschko et al., 2020a	Butendiek and Helgoland-cluster [22]	North Sea	Germany	Alcids	GPS telemetry
Peschko et al., 2020b	Helgoland-cluster [10]	North Sea	Germany	Alcids, Gulls	Combined survey methods
Peschko et al., 2021	Helgoland-cluster [10]	North Sea	Germany	Gannets	GPS telemetry
Petersen and Fox, 2007	Horns Rev. 1	North Sea	Denmark	Alcids, Gulls, Loons and grebes, Sea Ducks	Visual aerial survey
Petersen et al., 2006	Nysted [26] Horns Rev. 1 [11]	Baltic Sea North Sea	Denmark	Alcids, Gannets, Gulls, Loons and grebes, Sea ducks, Terns, Waterfowl	Visual aerial survey, Radar
Petersen et al., 2011	Nysted [26]	Baltic Sea	Denmark	Sea ducks	Visual aerial survey
Petersen et al., 2013	Nysted and Rødsand II [28]	Baltic Sea	Denmark	Sea ducks	Visual aerial survey
Petersen et al., 2014	Horns Rev. 2 [12]	North Sea	Denmark	Loons and grebes, Sea ducks	Visual aerial survey
Petersen et al., 2018	Nysted and Rødsand II [28]	Baltic Sea	Denmark	Sea ducks	Visual aerial survey
Pettersson, 2005	Utgrunden and Yttre Stengrund [29]	Baltic Sea	Sweden	Sea ducks	Visual observations
Plonczkier and Simms, 2012	Lynn and Inner Dowsing [15]	North Sea	U.K.	Waterfowl	Radar
Rehfishch et al., 2014	Greater Gabbard [9]	North Sea	U.K.	Gannets	Digital aerial survey
Rehfishch et al., 2016	London Array [14]	North Sea	U.K.	Alcids, Loons and grebes	Combined survey methods
Rexstad and Buckland, 2012	Kentish Flats [13]	North Sea	U.K.	Cormorants, Gulls, Loons and grebes	Boat-based survey
Trinder, 2019	Beatrice [6]	North Sea	U.K.	Alcids, Gannets, Gulls	Digital aerial survey
Vallejo et al., 2017	Robin Rigg [2]	Celtic Sea	U.K.	Alcids	Boat-based survey

(continued on next page)

**Table 1** (continued)

Citation	Wind farm [id]	Region	Country	Taxa	Methods
Vanermen et al., 2015	Bligh Bank [7]	North Sea	Belgium	Alcids, Fulmars, Gannets, Gulls, Skuas	Boat-based survey
Vanermen et al., 2016	Bligh Bank [7] Thornthorbank [20]	North Sea	Belgium	Alcids, Fulmars, Gannets, Gulls, Skuas	Boat-based survey
Vilela et al., 2021	Multiple – German EEZ [24]	North Sea	Germany	Loons and grebes	Combined survey methods
Welcker and Nehls, 2024	Alpha Ventus [5]	North Sea	Germany	Alcids, Gannets, Gulls, Loons and grebes, Terns	Boat-based survey

**Table 2**

Parameters extracted from studies of marine bird distributions before and after wind farm construction. In cases where several variants of a given parameter were extracted (listed in brackets), the variant that explained the most variance was included in the model (see methods). Several studies aggregated survey data across multiple wind farms.

Category	Parameter	Units/categories
Biological	Taxonomy [taxon, species]	listed in Table 3
	Stage	all, adults only
	Season	all, breeding, non-breeding
Study design	Survey method	boat-based, digital aerial, mixed survey, radar, telemetry, visual aerial
	Temporal comparison	after only, before-after
	Spatial comparison	control/impact, gradient
	Distance to effect	km
	Buffer width	km
	Distance from impact to reference area [minimum, maximum]	km
	Study area footprint (impact + reference areas)	km <sup>2</sup>
	Ratio of study area footprint to wind farm footprint	unitless
	Number of years of post-construction surveys	n
	Latitude [minimum, mean, maximum]	°
Wind farm	Turbine height [minimum, mean, maximum]	m
	Distance to shore [minimum, mean, maximum]	km
	Depth [minimum, mean, maximum]	m
	Footprint	km <sup>2</sup>
	Number of turbines	turbines
	Turbine density	turbines / km <sup>2</sup>
	Random terms	Source
	Country	country ID
	Year of completion	year

footprint including all impact and reference areas; ratio of total study area footprint to wind farm footprint; and number of years of post-construction data collected (Table 2). To reduce the number of categories for analysis, we combined all telemetry methods (GPS,  $n = 15$ ; satellite,  $n = 4$ ; and radio,  $n = 1$ ) into a single category, and classified observations collected during migration ( $n = 7$ ) as non-breeding.

For each wind farm included in the studies, we also extracted site information including region, latitude, distance to shore, wind farm footprint size, number of turbines, maximum water depth, and turbine height (Table 2). For studies that aggregated survey data across multiple wind farms, we summed wind farm footprints and number of turbines. Where values varied across the wind farm(s) (e.g., depth, distance from shore), we extracted average, minimum, and maximum values for the entire wind farm or study area and constructed preliminary models to select the summary statistic that explained the most variance in each response variable (see Section 2.3). Source documents did not consistently report wind farm characteristics; thus, these metrics were

extracted from Cook et al. (2018) and other sources where needed, including thewindpower.net, Wikipedia, individual wind farm websites, and direct communication with developers.

To estimate changes in marine bird distribution before and after wind farm construction, we extracted two response variables: effect detected (binary; response detected/no response detected) and effect size (i.e., the ratio of percent change in density in the wind farm area to percent change in background density in the control area(s), which we converted to log response ratio; Hedges et al., 1999). If a study reported density measurements and/or effect sizes at multiple distances from the wind farm edge (e.g., at 0, 2, and 4 km; Petersen et al., 2006), we extracted the presence/absence of a significant effect and the effect size at each identified distance band along with the distance at which the effect was measured (i.e., distance to effect parameter). Thus, a single study could have multiple distance-specific effects or effect sizes. Where studies did not test for significance, we used a chi square test comparing the value of the target species density within the impacted area before and after wind farm construction to the before/after values from the reference area and classified observed effects as statistically significant if the  $p$ -value of the  $\chi^2$  statistic was  $<0.05$ . We calculated log response ratios for all observations, regardless of whether the changes observed were determined to be significant, in order to account for the fact that individual studies with small sample sizes may have been unable to detect significance even if large changes were occurring (Maclean et al., 2013).

### 2.3. Analysis

To assess the impact of various wind farm, biological, and study design parameters on both the probability of detecting distributional change and the magnitude of observed changes, we used a generalized linear mixed modeling framework. All models were implemented in R 4.1.2 (R Core Team, 2021) using the lme4 package (Bates et al., 2015) for model fitting, and the MuMIn package (Bartoń, 2020) for model diagnostics and selection. Based on visual assessment of the data, we used a binomial distribution with a logit link function for the probability of detection and a Gaussian distribution with identity link for the effect size model, and tested model fit using quantile-quantile plots. Due to a lack of consistent reporting of variance among studies, we were not able to weight studies based on relative certainty; however, we included source document (i.e., the study in which the data were published) as a random factor to account for unmeasured differences among studies, along with country and year.

For the initial model selection, we compared the performance of highly correlated parameters (Pearson' correlation coefficient  $> 0.7$ ; Fig. 2) by constructing separate models for each of the correlated parameters and retaining the parameter from the model with the lowest Akaike's Information Criterion (AIC) value. For both the probability of detecting an effect and the effect size, we retained one measure each for distance to control (maximum), turbine height (mean), distance to shore (minimum), water depth (mean), and latitude (minimum). We also compared models containing either species or taxon effects, retaining taxon as a predictor (taxon AIC: 555.2; species AIC: 560.8). After the initial model selection, we ran a global model of each response variable for each of the three covariate categories (i.e., biological, study design,

and wind farm; Table 2), calculated the Variance Inflation Factor (VIF; Craney and Surlis, 2002) for each covariate, and sequentially removed covariates until  $VIF < 5$  for all remaining covariates.

To compare the relative suitability of wind farm, biological, and study design parameters for explaining variation in occurrence and magnitude of distributional change, we first constructed separate models for each of the three main categories of predictor variables. After removing correlated variables within each category as described above, we compared candidate models containing all possible permutations of all remaining predictors in an information theoretic framework, selecting the model with the lowest AIC value as the best-supported model (Tables S1-S4). Where multiple models received similar support (i.e., AIC values within two points of the lowest value; Burnham and Anderson, 2004), we selected the most parsimonious of the top models to simplify the final models and reduce opportunities for covariance among predictors. We generated pseudo- $R^2$  values (Nakagawa et al., 2017) to compare the variance explained by the best-performing model in each of the three categories. We tested interactions between parameters (species and season) in the biological models but did not include interaction terms in either the study design or wind farm models given the large numbers of predictors and categories relative to numbers of observations.

After selecting the best-fitting model for each response variable in each of the three categories (i.e., biological only, study design only, and wind farm only), we tested all combinations of these best-performing models to determine which combination of factors best explained detection and magnitude of change. Although we were again unable to test for all potential interaction effects due to the low ratio of observations to predictors, we included interactions between biological and study design parameters to test for potential variation in survey effectiveness among species and/or seasons. For each of the two response variables (probability of detecting distributional change and effect size), we selected the model with the lowest AIC value as the final model and extracted the overall  $R^2$  and coefficient values for each predictor.

### 3. Results

The literature review identified 48 studies presenting unique data on marine bird habitat use before and after wind farm construction. After removing studies that measured in-flight avoidance behavior rather than distributional change, we retained a total of 39 studies (Table 1), covering 29 unique wind farms or wind farm clusters, from which we extracted 445 unique records of displacement/attraction (i.e., study/species/distance combinations). Surveyed wind farms were primarily located in the North Sea, Baltic Sea, and Irish Sea between 51 and 58° latitude (Table 1; Fig. 1), with wind farm completion dates ranging from 1995 to 2018. Seabirds for which effects were evaluated included 29 species representing 10 different taxa (Table 3).

Among models predicting the likelihood of a significant change in distribution after construction of a wind farm, the best model containing biological parameters alone (taxon and season; Table S1) explained over half of the observed variance (pseudo  $R^2 = 0.54$ ; Table 4). The best models for only study design (which included survey area footprint, survey area to wind farm ratio, maximum distance to reference area, and distance to effect; Table S2; pseudo  $R^2 = 0.29$ ) or only wind farm characteristics (which included turbine density, minimum distance to shore, minimum latitude; Table S3; pseudo  $R^2 = 0.15$ ) had substantially poorer fit compared to biological parameters. The combined model with the lowest AIC value additionally combined the best-fitting biological, study design, and wind farm models with an interaction between taxon and distance to effect, and explained 77% of the variance in the likelihood of detecting distributional change (Table 4). The final model included a random effect of source (Fig. S1); random effects of year of wind farm establishment and country did not improve model fit.

Of the taxa included in this analysis, sea ducks had the greatest predicted probability of distributional change, followed by loons, alcids, and gannets (Fig. 3a). Changes were predicted less frequently for cormorants, gulls, terns, waterfowl, fulmars, and skuas (Fig. 3a). Likelihood of distributional change was higher during the breeding season than year-round, with non-breeding intermediate between the two (Fig. 3b). The probability of detecting distributional change decreased with

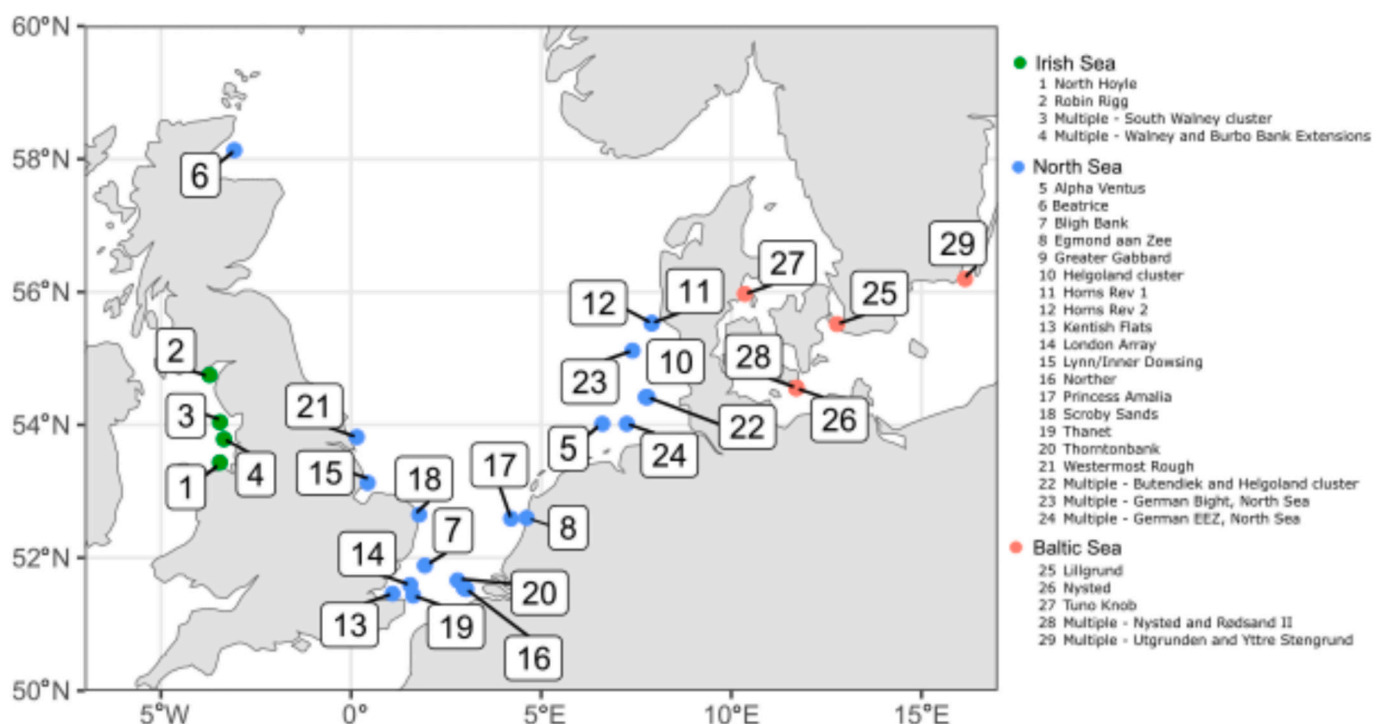


Fig. 1. Locations of wind farms included in this literature review reporting post-construction monitoring of marine bird distributions. Wind farm numbers are cross-referenced with study details in Table 1.

**Table 3**

Summary of sample sizes, species composition, latitude, and frequency and magnitude of distributional change of marine birds by taxon following construction of offshore wind farms. Where species were difficult to distinguish, some studies did not report species-level effects by species but instead provided a single multi-species value (e.g., for *Gavia arctica* and *Gavia stellata*).

Taxon	N <sub>studies</sub>	N <sub>sites</sub>	N <sub>observations</sub>	N <sub>species</sub>	Species names	Mean latitude	Significant effect detected (prop. studies)	Effect size	
								Log response ratio (lnRR)	% change (95% CI)
Alcids	16	15	69	3	<i>Alca torda</i> <i>Fratercula arctica</i> <i>Uria aalge</i> <i>Phalacrocorax carbo</i>	52.8	0.65	-0.93	[-69%, -49%]
Cormorants	6	6	9	2	<i>Phalacrocorax aristotelis</i>	53.6	0.56	-0.28	[-75%, +129%]
Fulmars	4	5	9	2	<i>Fulmaris glacialis</i> <i>Puffinus puffinus</i>	52.2	0.11	-3.05	[-99%, -80%]
Gannets	11	11	34	1	<i>Morus bassanus</i> <i>Chroicocephalus ridibundus</i> <i>Hydrocoloeus minutus</i>	52.8	0.56	-1.14	[-64%, -29%]
Gulls	17	17	174	7	<i>Larus argentatus</i> <i>Larus canus</i> <i>Larus fuscus</i> <i>Larus marinus</i> <i>Rissa tridactyla</i> <i>Gavia arctica</i>	52.8	0.37	-0.12	[-26%, +6%]
Loons/ Grebes	16	14	69	3	<i>Gavia stellata</i> <i>Podiceps cristatus</i> <i>Clangula hyaemalis</i>	53.7	0.8	-1.38	[-82%, -65%]
Sea ducks	14	12	45	3	<i>Melanitta nigra</i> <i>Somateria mollissima</i>	55	0.79	-1.12	[-80%, -46%]
Skuas	2	2	6	1	<i>Stercorarius skua</i> <i>Sterna hirundo</i>	51.6	0	-3.31	[-99%, -82%]
Terns	5	6	14	4	<i>Sterna paradisaea</i> <i>Sterna sanvicensis</i> <i>Sternula albifrons</i> <i>Anser brachyrhynchus</i>	53.5	0.5	-1.04	[-88%, +7%]
Waterfowl	4	3	14	4	<i>Bucephala clangula</i> <i>Cygnus olor</i> <i>Mergus serrator</i>	54.7	0.43	-0.35	[-72%, +75%]

distance from the wind farm edge for all species except gannets, alcids, and sea ducks, for which rates of detecting significant effects were higher at greater distances (Fig. 4a). Distributional changes were detected with greater probability at wind farms further from shore ( $\beta = 0.02$  [95% CI: 0.01–0.04]; Fig. 4b) and at lower turbine densities ( $\beta = -0.22$  [95% CI: -0.42 to -0.02]; Fig. 4c). Probability of detecting distributional change also tended to increase with total survey area ( $\beta = 0.0001$  [95% CI: 0–0.0003]; Fig. 4d), and survey area to wind farm ratio ( $\beta = 0.01$  [95% CI: 0–0.02]; Fig. 4e), although confidence intervals for the effects of these covariates included zero. Latitude and distance to reference area also appeared in the final model, but with non-significant coefficient values and no evident trends (latitude:  $\beta = 0.14$  [95% CI: -0.11–0.39]; distance to reference area:  $\beta = 0$  [95% CI: -0.002–0.003]).

The magnitude of distributional change was better explained by biological parameters (taxon, season, and their interaction; pseudo-R<sup>2</sup> = 0.24) than either study design (spatial comparison only; pseudo-R<sup>2</sup> = 0.09) or wind farm characteristics (intercept only; pseudo-R<sup>2</sup> = 0) (Table S4). The final model with the lowest AIC value combined the biological and study design models (i.e., taxon, season, spatial comparison, and their interactions) and explained 29% of the variance in the magnitude of distributional change (Table 4). The model also included random effects of source (Fig. S2) and country (Fig. S3). The magnitude of displacement was greatest for skuas and fulmars (log response ratios < -2.3, corresponding to ~90–100% decreases in use post-construction; Table 3), with smaller but significant displacement effects also occurring in loons/grebes, sea ducks, alcids, and gannets (log response ratios -1.38 to -0.93, corresponding to ~55–80% decreases in use; Table 3). Effect sizes for gulls, cormorants, terns, and waterfowl varied between positive and negative depending on the season but also overlapped

0 (Table 3, Fig. 5). Detected magnitude of displacement effects were stronger (i.e., more negative effect sizes) during non-breeding for most taxa and were generally strongest in studies that combined control-impact and gradient designs (Fig. 5).

#### 4. Discussion

By synthesizing estimates of distributional change across published and unpublished studies of existing offshore wind energy installations, we were able to identify several biological and methodological factors that contributed to the likelihood to statistically detect changes in seabird distributions following offshore wind energy development, as well as to the estimated magnitude of distributional change. Notably, displacement and attraction effects varied widely among seabird taxa, and were affected by spatial extent and configuration of studies, the distance from the wind farm at which effects were measured, and the density and distance to shore of wind turbines.

##### 4.1. Effects of biological parameters

We found that both probability of detecting distributional change and the strength of observed changes varied strongly among taxonomic groups, which explained more of the variation in these response metrics than any other covariate. Our analysis confirmed the descriptive assessments of Dierschke et al. (2016) suggesting systematic displacement of loons, grebes, sea ducks, alcids, and gannets, all of which had high probabilities of significant negative effects. We observed the largest negative effects (90–100% displacement) for fulmars and skuas, although with larger variation and lower probability of detecting changes in distribution compared to some of the other taxa. These

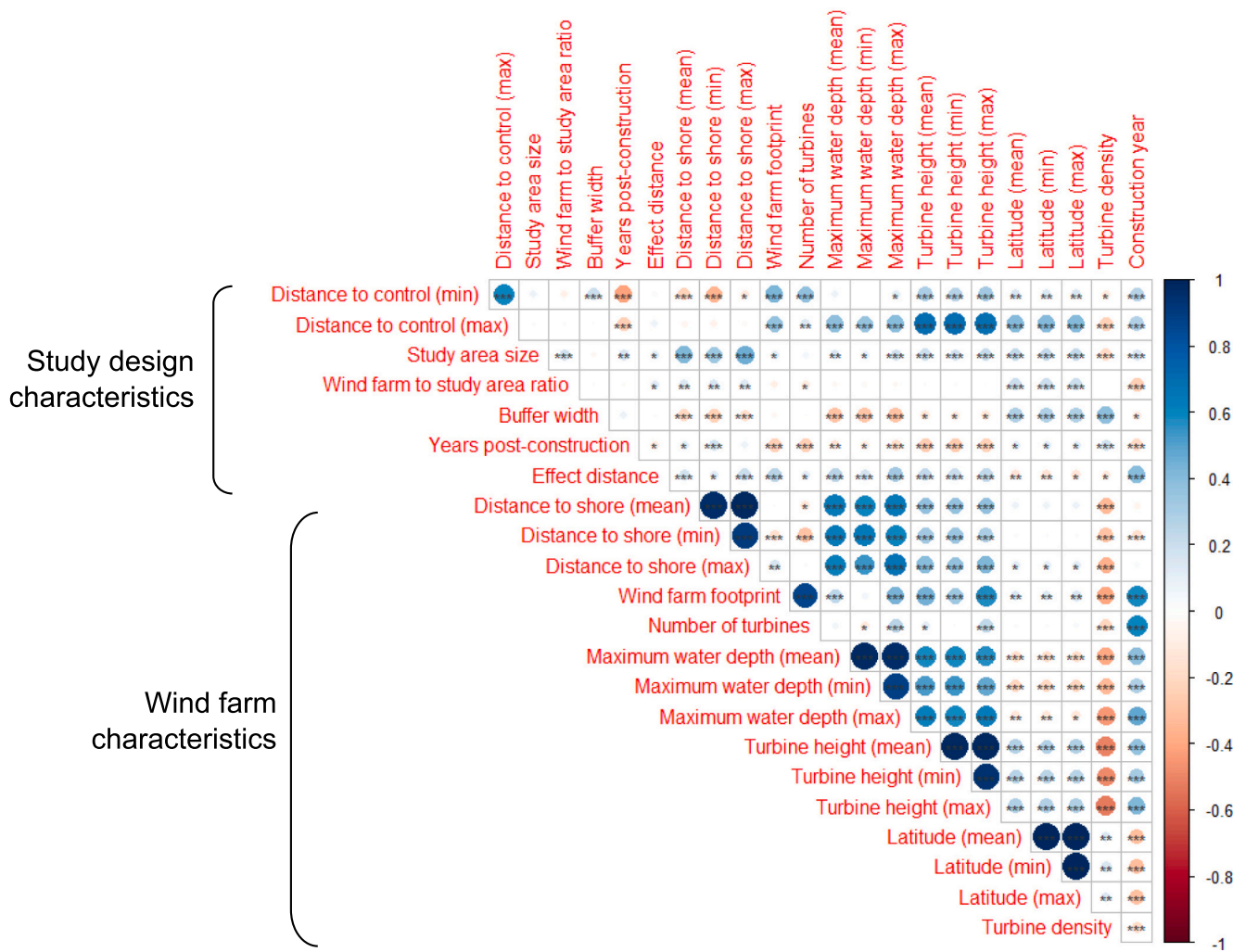


Fig. 2. Correlations among numeric wind farm and study design parameters used to predict distributional change of marine birds following offshore wind farm construction. Circle size increases with correlation strength (blue = positive; red = negative). Asterisks indicate correlation significance: \* $p < 0.05$ ; \*\* $p < 0.01$ ; \*\*\* $p < 0.001$ . (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

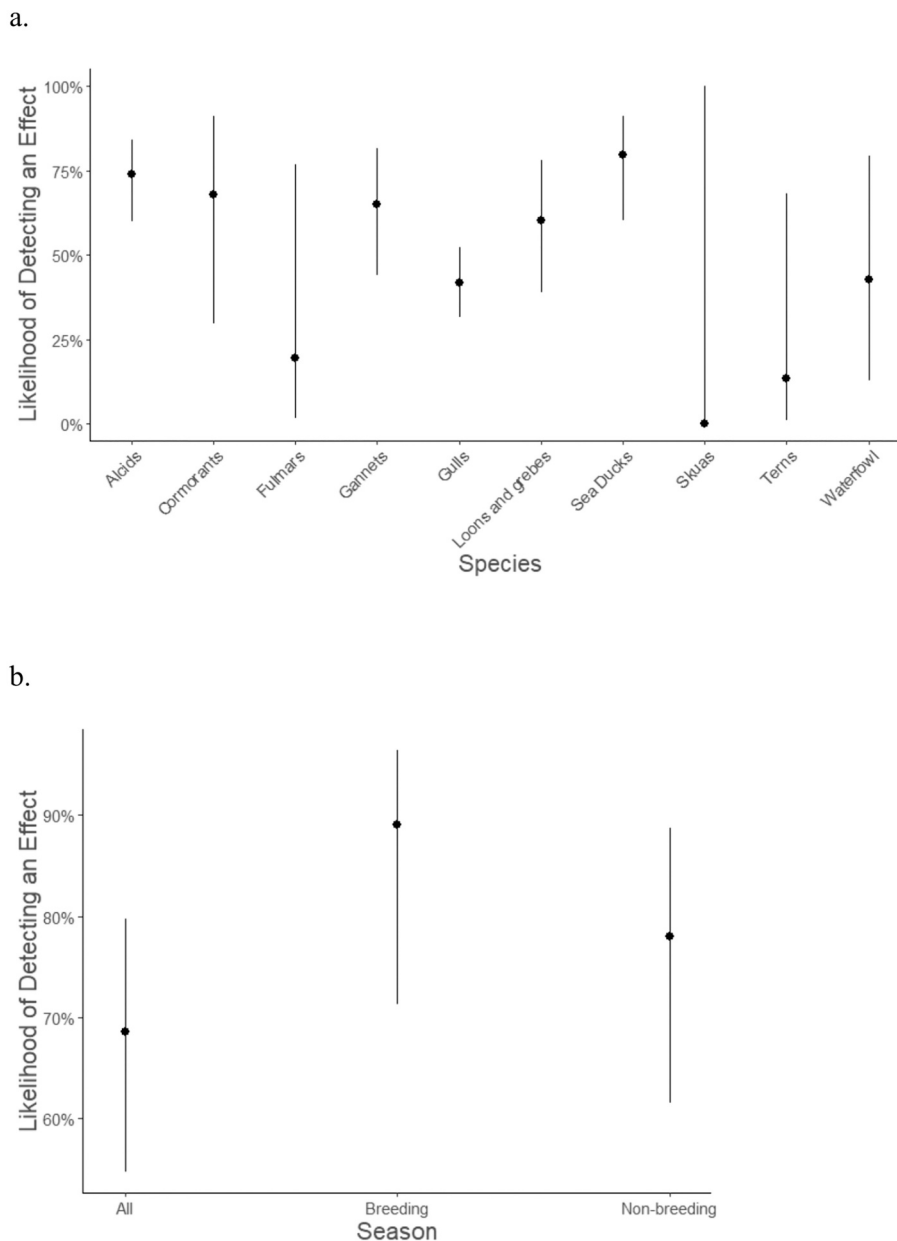
groups were sighted in few studies (Table 3) and tended to occur at low densities (e.g., Leopold et al., 2013; Vanermen et al., 2016), making detection of significant effects at the study level difficult. Since the studies in our analysis varied widely in how densities were calculated and reported, we were not able to directly incorporate absolute density estimates into our models. However, power analyses of boat-based surveys (e.g., Vanermen et al., 2015; Hall and Black, 2024) have confirmed that distributional characteristics have a greater influence than study design on power to detect changes after wind farm construction, with lower-density species and populations requiring longer monitoring periods and/or more intensive studies. It is also important to note that species observed in very small numbers are more likely to be absent due to random fluctuations in occurrence, which may exaggerate the magnitude of predicted decreases. However, such random fluctuations are just as likely to cause large increases as decreases in rarer species; thus, the fact that these taxa dramatically declined or disappeared from wind energy areas across multiple studies indicates that they may be more vulnerable than individual studies would suggest.

The remaining taxa showed a mixture of slight attraction (gulls, cormorants) and slight displacement (terns, waterfowl), but with effect size estimates varying among seasons and study methods and often overlapping zero. While cormorants, terns, and waterfowl were only detected in a small number of studies (4–6 studies per taxon), gulls were relatively common across the studies included in our analysis. The low

frequency of distributional change in this taxon, as well as the lack of significant displacement or attraction, suggests that effects for gulls may be limited or may vary among species and/or locations. Indeed, several recent studies (e.g., Thaxter et al., 2018; Vanermen et al., 2020; Johnston et al., 2022) suggest that gulls display complex responses to wind energy infrastructure including both attraction and displacement at various spatial scales.

Season also affected both the occurrence and magnitude of distributional change, with changes more likely during the breeding season than during non-breeding. Studies that pooled results across the annual cycle were less likely to detect changes than studies focusing on either breeding or non-breeding, possibly because varying responses among seasons could have diluted or negated one another when measured at annual scales. Indeed, for several taxa including cormorants and waterfowl, effect sizes estimates suggested a mixture of displacement during non-breeding with attraction or neutral responses during the breeding season (Fig. 5). While we were unable to test for differences within the non-breeding season, it is also likely that effects vary within this period, with elevated sensitivity during post-breeding (Busch and Garthe, 2016) and migration (\*Kahlert et al., 2004, Aumuller et al., 2013).

It is important to note that whether or not a significant effect was detected depends both on the presence of an effect and the statistical power to separate that effect from underlying variability. Marine birds



**Fig. 3.** Partial effects of a) taxon and b) season on the likelihood of significant distributional change of marine birds following wind farm construction. Note: significant effects were detected in 0/6 comparisons for skuas.

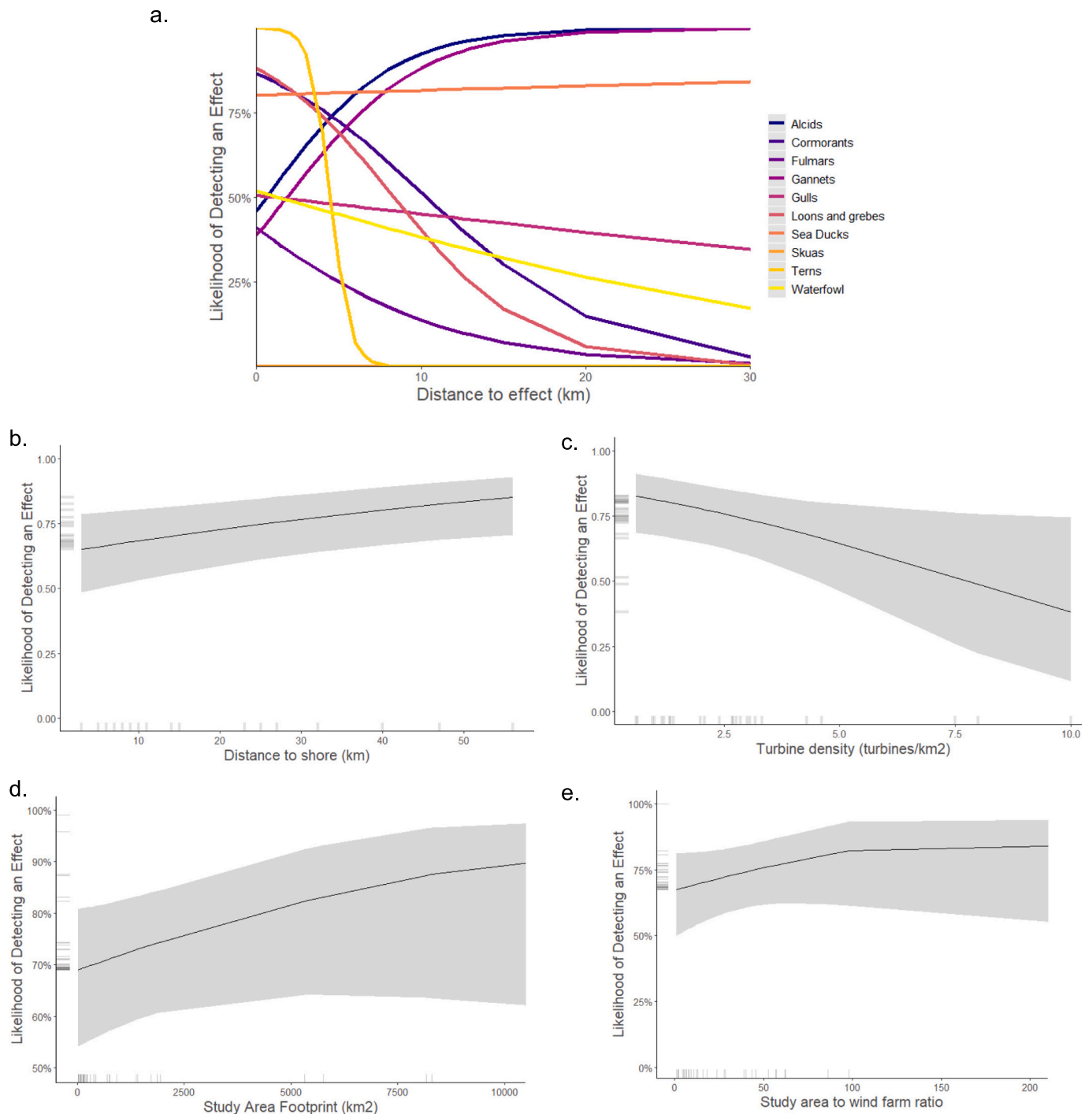
are central-place foragers during the breeding season, gathering in large aggregations at breeding colonies and foraging within a relatively limited radius of their nest sites (Orians and Pearson, 1979). Consequently, breeding birds are likely to be more densely aggregated in at-sea foraging areas than during non-breeding, and at-sea distributions may be less variable among years since breeders have limited flexibility to track shifting marine resources compared to non-breeders (Garthe, 1997; Huettmann and Diamond, 2001). Both higher densities and lower variability could positively influence power to detect changes in habitat use. Only one study in our analysis (Peschko et al., 2020b) directly compared displacement between non-breeding and breeding for two species, black-legged kittiwake *Rissa tridactyla* and common murre *Uria aalge*. They found that seasonal differences were species-specific: murre displayed greater decreases in use of wind farm areas and larger avoidance radii during non-breeding, while kittiwakes avoided wind farms only during the breeding season. Further longitudinal studies such as this one could help to disentangle detection effects from taxon- and

species-specific seasonal differences in sensitivity.

#### 4.2. Effects of study design parameters

In addition to biological factors, several covariates related to study design helped to explain both occurrence and magnitude of distributional effects. Probability of observing significant changes in distribution increased with the size of the study area footprint, as well as with the ratio of study area footprint to wind farm footprint. The influence of these covariates suggests that measuring changes across larger areas may improve power to separate changes in distribution from interannual variability and account for redistribution of seabirds across broad geographic regions. Our findings reinforce a recent analysis of offshore wind effects on marine mammals suggesting that the spatiotemporal scales of environmental impact assessments for many proposed offshore wind energy developments may be insufficient to encompass expected effects (Ryan et al., 2019). Changes were also more frequently detected

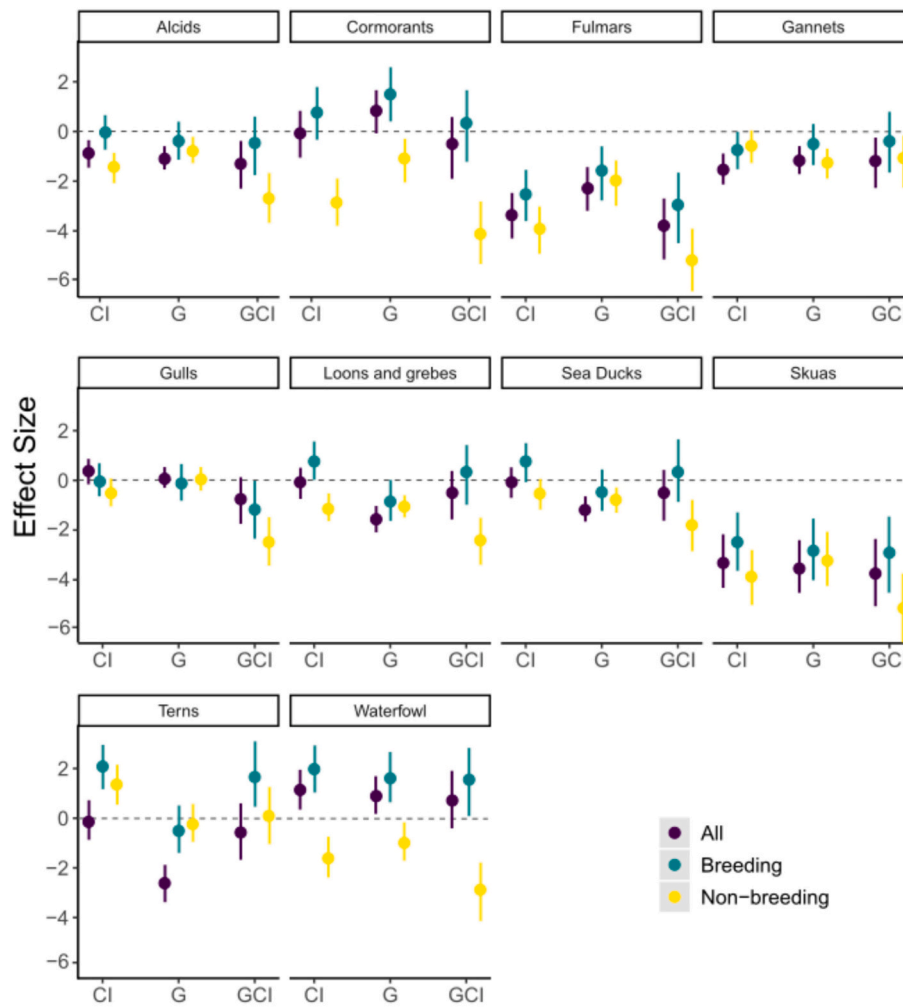




**Fig. 4.** Partial effects of study design and wind farm characteristics on the likelihood of detecting distributional change of marine birds following construction of offshore wind farms: a) distance to effect by taxon; b) distance to shore; c) turbine density; d) study area footprint; and e) ratio of total study area footprint to wind farm footprint. Shaded areas represent 95% confidence intervals. Hash marks on axes show densities of observations at different values, with darker lines indicating more observations.

closer to the wind farm edge in all taxa except gannets, alcids, and sea ducks (Fig. 4a), reflecting among-species differences in sensitivity (e.g., Busch and Garthe, 2016). This suggests that, for most taxa, displacement or attraction effects are strongest immediately adjacent to the wind farm and attenuate with distance. We observed the opposite pattern in gannets, alcids, and sea ducks; however, there was large uncertainty in these trends, and low rates of effect detection close to wind farms were driven by several studies that did not detect these taxa within wind farm footprints either before or after construction. These taxa often have

highly clumped distributions (e.g., Thiebault et al., 2014), which may make them more likely to be absent from specific grid squares due to random fluctuation compared to more evenly-distributed species. We were not able to incorporate group sizes for different species pre- and post-construction into our analysis; however, this could be a useful focus for future monitoring efforts, especially since group sizes may affect species-specific sensitivity to disturbance (Guillemette et al., 1998). Distance from the wind farm edge affected the probability of detecting displacement or attraction for 20–30 km from the wind farm edge,



**Fig. 5.** Partial effects of the three-way interaction between spatial comparison (CI = control-impact; G = gradient; GCI = gradient-control-impact), season, and taxon on the magnitude of marine bird distributional change following construction of offshore wind farms. Dashed lines indicate effect sizes of zero (i.e., no change), negative values indicate displacement, and positive values indicate attraction.

**Table 4**

Covariates and AIC values for best-fitting models of frequency (effect detected) and magnitude (effect size) of distributional change of marine birds following offshore wind farm construction using biological parameters only, study design parameters only, wind farm parameters only, and combinations of parameter categories. Other candidate models are arranged from lowest to highest AIC and compared with the top model.

Response	Model type	Covariates	AIC	Δ AIC	Likelihood	Weight	Pseudo R <sup>2</sup>
Effect detected	<b>Biological + Study design + Wind farm</b>	Taxon * Distance to effect + Season + Study area footprint + Ratio of study area to wind farm area + Distance to reference area + Turbine density + Latitude + Distance to shore	<b>530.30</b>	<b>0</b>	<b>1</b>	<b>0.89</b>	<b>0.77</b>
	Biological + Wind farm	Taxon + Season + Turbine density + Minimum latitude + Minimum distance to shore	534.40	4.10	0.13	0.11	0.71
	Biological + Study design	Taxon + Season + Study area footprint + Distance to effect + Ratio of study area to wind farm area + Distance to reference area (max)	547.15	16.85	<0.001	<0.001	0.67
	Biological	Taxon + Season	554.30	24.00	<0.001	<0.001	0.54
	Study design + Wind farm	Study area footprint + Distance to effect + Ratio of study area to wind farm area + Distance to reference area (max) + Turbine density + Latitude (min) + Distance to shore (min)	561.40	31.10	<0.001	<0.001	0.31
	Study design	Study area footprint + Distance to effect + Ratio of study area to wind farm area + Distance to reference area (max)	566.71	36.41	<0.001	<0.001	0.29
	Wind farm	Turbine density + Latitude (min) + Distance to shore (min)	569.01	38.71	<0.001	<0.001	0.15
Effect size	Intercept only		590.34	60.04	<0.001	<0.001	0
	<b>Biological + Study design</b>	<b>Taxon * Season * Spatial comparison</b>	<b>1528.13</b>	<b>0</b>	<b>1</b>	<b>1</b>	<b>0.29</b>
	Biological	Taxon * Season	1544.05	15.92	<0.001	<0.001	0.24
	Study design	Spatial comparison	1643.37	115.24	<0.001	<0.001	0.09
Wind farm/Intercept only		1645.76	117.63	<0.001	<0.001	0	

depending on taxon (Fig. 4a), suggesting that for some species the area impacted by a wind farm may extend beyond the limits of many gradient studies or overlap with control areas that are situated too close to wind farm sites. Overall, our work reinforces other recent evidence (Garthe et al., 2023) that standard monitoring radii of 2–4 km may be too small to detect all relevant changes in marine bird communities.

Besides the spatial extent of monitoring, the design of reference areas affected both the probability of detecting effects and the strength of observed effects. The maximum distance from impact to reference area was positively correlated with probability of detecting an effect, and studies that combined gradients with control-impact design generally detected larger displacement effects compared to control areas or gradients alone. This highlights the importance of accounting for displacement effects at large distances from the impact area, while also reinforcing recent evidence that gradients can improve ability to account for underlying variation (Bailey et al., 2014; Methratta, 2021). Other methodological factors, including the monitoring method used, buffer width included in the impact area, and whether monitoring was conducted both before and after construction or only after, did not substantially improve model fit for either response variable. However, the relatively small number of studies available for this analysis may have masked effects of key study design parameters, particularly for methods or temporal windows that were infrequently used. For example, only 9/39 studies (23%) used data collected only after construction, and five of the nine after-only studies were telemetry-based. This reflects the fact that before-after studies are already standard practice, particularly for surveys, and represent the best-available option for developing a pre-construction baseline against which to measure effects. In addition, our dataset was not large enough to include additional covariates for of method-specific factors that could have influenced results (e.g., frequency and duration of visual surveys, number and duration of transmitters used in tracking studies, etc.). Regardless of which temporal window and monitoring method is used, our study underscores the cross-cutting importance of surveying sufficiently large areas to detect effects, as well as ensuring that control or comparison areas reflect background conditions and are independent of any wind farm effects.

#### 4.3. Effects of wind farm parameters

We observed strong correlations between wind farm characteristics and study design parameters, suggesting a confounding effect of advances in wind farm technology with improvements in survey design. Newer wind farms tended to contain greater numbers of taller turbines at lower densities, occupy larger footprints, and be located in deeper waters further offshore. At the same time, effects of newer projects on seabird distributions were also assessed using larger survey areas and/or more distant control sites, which tended to contribute to a higher likelihood of detecting distributional changes. These survey characteristics likely reflect the evolution of standardized guidelines calling for larger survey areas to detect change (e.g., Camphuysen et al., 2004; Maclean et al., 2013), as well as the fact that survey activities around wind farms farther offshore are less naturally constrained by coastlines. Thus, while wind farm parameters alone explained less observed variation in seabird responses than either study design or biological factors, it was also difficult to fully disentangle their effects from changes in survey methodologies.

Although wind farm characteristics alone had relatively low explanatory power, increasing distance from shore, decreasing turbine density, and increasing latitude were associated with higher probabilities of detecting distributional change. The positive effect of distance from shore suggests that seabird responses to offshore wind energy installations may be weaker and/or less detectable in nearshore areas, where other human activities and high background variability may mask responses or favor species or taxa with lower sensitivity to disturbance. Seabirds in nearshore areas may also be more closely tied to

static habitat features, such as breeding colonies or river plumes (e.g., Scott et al., 2013; Warwick-Evans et al., 2016; Phillips et al., 2018), and therefore less likely to shift to different habitat areas post-construction than seabirds in offshore environments. We did not include distance to the nearest breeding site as a predictor, as the availability of breeding colony data varied among species and sites and would have only been relevant for a subset of species (local breeders) and seasons (breeding season). Moreover, since local competition and density-dependent effects can increase or decrease colony- and individual-specific foraging distances in complex ways (Bolton et al., 2019), using distance to colony alone to predict effects may be misleading. Using individual telemetry to disentangle effects of species, colony size, and distance to wind farm on displacement of breeding birds could be a valuable avenue for further study.

We also found a greater probability of detecting effects at lower turbine densities, which is somewhat unexpected and contrasts with recent evidence that in-flight avoidance of wind farms by Sandwich terns increases with turbine density (van Bemmelen et al., 2023). This effect could be confounded with sensitivity to larger turbines, since turbine height was negatively correlated with turbine density and positively correlated with distance from shore; however, turbine height itself was not supported as a predictor. Higher latitudes were strongly associated with increased probability of detecting distributional change when only wind farm characteristics were considered, but its effect was non-significant after accounting for taxa and seasonal occurrence. Further study could help to determine whether latitudinal differences are a result of differences in power (i.e., seabirds occurring at greater densities at higher latitudes, leading to greater probability of detecting effects), in seabird community sensitivity, or in other underlying drivers.

#### 4.4. Recommendations and conclusions

While our study represents an important step in quantitatively synthesizing offshore wind energy effects on seabirds, several areas for improvement remain. The number of surveys of existing wind farms for which data are publicly available in published literature and reports is still relatively small, which limits our ability to incorporate multiple covariates or account for complex interactions. As offshore wind farm buildout continues, ensuring the accessibility of survey results in a timely manner will be crucial to facilitate and update analyses like this one, as well as to assess the cumulative effects of offshore wind on seabirds and other marine species. Increases in the number and breadth of displacement studies available may reduce the confounding effects of covariance between survey methods and wind farm characteristics, as well as allowing for future analyses that incorporate additional method-specific (e.g., survey duration and frequency) and season-specific (e.g., spatial distribution of breeding sites, stage of non-breeding) covariates. However, the inclusion of these factors will only be possible with an increase in the quality and availability of published data across a range of sites and methods.

Moreover, the studies we used for this analysis varied widely in the amount and type of data included, making it difficult to derive standardized effect size estimates. We found that percent change (log response ratio) was a relatively straightforward metric for comparing among studies with varying metrics and levels of specificity, but deriving this value often required inferring and re-calculating data from figures, grey literature, and/or supplemental materials that were not readily available. Inconsistencies in reporting may have contributed to variability in effect size estimates, limiting our power to detect the influence of specific biological, methodological, and site-level characteristics on effects. We included a random effect of study to account for variation among studies; however, we were unable to weight studies based on the variance of their effect size estimates (e.g., Nakagawa et al., 2023). Most published studies did not report variance in survey-level abundance estimates, and those that reported variance often used different temporal and spatial scales (e.g., seasonal vs. annual) that

could not be compared directly. Re-analyzing raw survey data would allow for (re)calculating variance estimates and sample sizes using directly comparable methods, and could also allow for assessment of additional factors such as the effect of absolute density on detection of displacement or attraction. However, raw data are not readily available for most studies, especially for older projects, and deriving estimates from raw data would require accounting for differences in methodology using appropriate adjustment methods (Waggitt et al., 2020). Overall, we recommend that future studies provide straightforward and clear statistical tests comparing pre- and post-construction use estimates and percent change, along with survey-level raw and corrected abundance and/or density estimates.

Despite these limitations, our analysis provides some broad guidance that can help direct future environmental impact assessments and guide the design of monitoring studies for effects of offshore wind energy installations on marine bird distributions and habitat use. Notably, we found that several taxa (particularly skuas and fulmars) showed high levels of displacement but limited detection of significant effects. These taxa were present in relatively few studies, suggesting that displacement rates are likely underestimated in site-specific studies for rarer or more dispersed species such as these. In addition, although detection of species-specific effects was relatively robust regardless of study design, the size of the study area positively influenced detection of distributional change. Our analysis suggests that the incremental value of additional sampling effort increases with total survey area up to at least 2500 km<sup>2</sup> or 100 times the size of the wind farm footprint. These values, while potentially variable among species, indicate that monitoring programs seeking to achieve sufficient power to detect distributional changes should include regional as well as site-specific sampling, which would also contribute to development of large-scale and cumulative-impact monitoring frameworks (Masden et al., 2010). Our results also suggest that studies that sample both gradients around windfarms and separate control areas are likely to detect more substantial changes in distribution of focal species than either gradient or control-impact studies alone, and that the distance to the wind farm edge at which effects were measured influenced both the likelihood of detecting the effects and the magnitude of effects up to 30 km from the edge of the wind farm site. A recent multi-site analysis of displacement of loons in the North Sea (Garthe et al., 2023) also found displacement effects extending beyond 10 km from wind farm edges, with non-significant displacement up to 24 km away from wind farm sites. Taken together, our results suggest that effects of offshore wind energy development on marine birds extend well beyond the immediate surroundings of the wind farms. Thus, although fine-scale data remain crucial to measuring localized effects of wind farms on marine birds, collecting and/or integrating monitoring data at regional scales is key to detecting and quantifying displacement and attraction effects that extend beyond the immediate surroundings of wind farms.

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## Author statement

JSL conducted statistical analyses, wrote manuscript, and created figs. JG and KAW conducted the literature review. All authors contributed to the literature review and reviewed and edited the manuscript.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

Data will be made available on request.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.eiar.2024.107611>.

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