Ecological Indicators 181 (2025) 114467

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original Articles

The times they are a changin': Temporal patterns in small cetacean abundance in the northeast Atlantic

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ARTICLE INFO

Keywords:
Seasonal abundance
Inter-annual variation
Aerial surveys
Line transect sampling
Phocoena phocoena
Tursiops truncatus
Delphinus delphis

ABSTRACT

Conserving marine species effectively requires spatially and temporally explicit knowledge of their abundance and distribution for assessing potential impacts (e.g., from fishery bycatch, anthropogenic sound, ship strikes) over different spatial and temporal scales. We used aerial surveys and distance sampling to estimate abundance of seven frequently encountered small cetacean species in the northeast Atlantic and examine seasonal and interannual variations. Five surveys were carried out over the Irish EEZ in summer 2016, 2021 and 2022 and winter 2016–2017 and 2022–2023. Seasonal and annual variations in abundance were seen across species. Harbour porpoise showed a steady decrease in abundance, with estimates ranging from 38,260 (CV: 23.7) individuals in summer 2016 to 6,604 (CV: 40.8) in winter 2022–2023. Similarly, bottlenose dolphin numbers ranged from an unprecedented 212,646 (CV: 15.5) individuals in the winter of 2016–2017 to 11,328 (CV: 42.4) in summer 2021. In contrast, common dolphins increased from 13,192 (CV: 75.2) individuals in winter 2016–2017 to 594,293 (CV: 28.2) in summer 2021. Other species, such as Risso's, Atlantic white-sided, white-beaked and striped dolphins were seen more sporadically and in lower numbers. The recent decline in harbour porpoise abundance is concerning and merits further investigation. Results provide the first measure of seasonal and interannual variability in abundance, possibly reflecting large scale movement patterns and population change and allow for the seasonal assessment of potential anthropogenic impacts.

1. Introduction

Marine ecosystems are experiencing rapid changes, leading to alterations in oceanographic and climate processes (Masson-Delmotte et al., 2021), shifts in species distributions (Pinsky et al., 2020; van Weelden et al., 2021; Melbourne-Thomas et al., 2022) and changes in communities (Gordó-Vilaseca et al., 2023). Differential impact on species can lead to spatial mismatches between predators and their prey (Pinsky et al., 2020), resulting in changes in ecosystem biomass, composition and dynamics to which top predators, such as cetaceans, must adapt (Poloczanska et al., 2013; Bryndum-Buchholz et al., 2019; Pinsky et al., 2020). In addition, marine species and ecosystems are affected by the cumulative effects of human activities (Rudd, 2014; Halpern et al., 2015). This is particularly relevant for cetaceans, whose conservation is threatened by the synergistic nature of human activities (Maxwell et al., 2013; Braulik et al., 2023), such as incidental capture in fishing gear or bycatch (Rogan & Mackey, 2007; Fernandez-Contreras et al., 2010; Peltier et al., 2021; Rogan et al., 2021), competition with fisheries (Piroddi et al., 2011; Giralt Paradell et al., 2021), collision with vessels (Laist et al., 2001; Redfern et al., 2013), disturbance by marine traffic (Pirotta et al., 2015, Pigeault et al., 2024), water pollution (Murphy et al., 2015; Jepson et al., 2016) and noise pollution derived

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https://doi.org/10.1016/j.ecolind.2025.114467
Received 4 November 2024; Received in revised form 19 November 2025; Accepted 20 November 2025
Available online 29 November 2025
1470-160X/

from *inter alia*, seismic surveys, naval sonar exercises, boat traffic and the development and construction of renewable energy sites (Tyack et al., 2011; Bailey et al., 2014; Williams et al., 2014; Kavanagh et al., 2019; Maxwell et al., 2022).

Recognising the impact caused by human activities, the European Union (EU) adopted the Marine Strategy Framework Directive (MSFD) in 2008, to ensure the preservation of the marine environment, by achieving a Good Environmental Status (GES) in all its seas by 2020 and by promoting an ecosystem-based approach to manage human activities and pressures (Directive 2008/56/EC). This Directive was integrated as the main framework defining environmental objectives within the EU's Integrated Maritime Policy, which, among other objectives, aims at maximising the sustainable use of oceans and seas (Paramana et al., 2023). The MSFD developed a series of descriptors to assess different components of marine ecosystems, which could be used to determine if GES was achieved and maintained. Among them, descriptor one pertains to the maintenance of biological diversity and descriptor four the correct functioning of marine food-webs. Both descriptors require that species abundance are in line with prevailing conditions and at levels capable of ensuring long-term species viability (Directive 2008/56/EC). In this regard, criterion D1C2 within descriptor one, considered a primary criterion, is used to assess if the abundance of a species is adversely affected by human activities. This is reported to MSFD with data collected in six-year cycles and relies on trends in abundance over a certain time period. Therefore, robust estimates of abundance and ideally a series of estimates, is one of the fundamental parameters needed for assessing conservation status of cetaceans and can provide further information on how combined impacts affect cetaceans over time (Hammond et al., 2021).

There are several methods to assess cetacean species abundance, but they all require standardised data collection techniques and compliance with the assumptions of each method to reduce potential biases and obtain robust estimates (Hammond, 2010). Furthermore, the intrinsic nature of cetaceans, which are cryptic and highly mobile, makes the assessment of their abundance at appropriate spatial and temporal scales challenging (Rogan et al., 2017; Authier et al., 2020). Visual line-transect surveys using airplanes cover large areas in short amounts of time (Bretagnolle et al., 2004; Buckland et al., 2012) and have been extensively used to assess cetacean abundance in different regions (Andriolo et al., 2006; Hedley et al., 2011; Hammond et al., 2013; ACCOBAMS, 2021; Gilles et al., 2023). Surveys are restricted to good weather conditions and depending on the geographical area, predominantly take place during summer months (e.g. Hammond et al., 2013; ACCOBAMS, 2021; Nachtsheim et al., 2021; Gilles et al., 2023), although some winter surveys have also been carried out (Panigada et al., 2011; Laran et al., 2017a,b; Peltier et al., 2024). Within the northeast Atlantic, abundance estimates for cetaceans at a European level have been derived from large-scale ship-based and aerial surveys conducted during the Small Cetaceans in European Atlantic Waters and the North Sea (SCANS) survey programmes on four occasions (SCANS-I in 1994, SCANS-II in 2005, SCANS III in 2016 and SCANS-IV in 2022) since 1994 (Hammond et al., 2002, 2013, 2017; Gilles et al., 2023). While the first two SCANS included waters within some or part of the Irish Economic Exclusive Zone (EEZ), no surveys were conducted in this area by SCANS-III and IV.

Here we present the results of two sets of aerial surveys carried out in Irish waters in 2016–2017 and 2021–2022 within the ObSERVE programme, which covered the Irish EEZ, not surveyed in SCANS III and IV. The primary aim of the study is to assess seasonal and interannual variability in the abundance of several species of small odontocetes, namely harbour porpoise (*Phocoena phocoena*), common bottlenose dolphin (*Tursiops truncatus*, hereafter bottlenose dolphin), common dolphin (*Delphinus delphis*), Risso's dolphin (*Grampus griseus*), white-sided dolphin (*Lagenorhynchus acutus*), white-beaked dolphin (*L. albirostris*) and striped dolphin (*Stenella coeruleoalba*) in part of the northeast Atlantic. Additionally, we aim to place these results within a wider geographical context by comparing them to SCANS-III and IV surveys conducted at the same time to provide a more comprehensive understanding of species dynamics in the northeast Atlantic. Understanding the seasonal fluctuations in abundance of these species is not only important for assessing conservation status, but can also inform management in relation to marine spatial planning, the designation of marine protected areas, to report under national and international legislation (e.g., EU MSFD and Habitats Directive), and can be incorporated into seasonally explicit risk assessments, which may be important for licensing or examining the impact of seasonal activities, e.g. fisheries.

2. Methods

2.1. Study area

The study was conducted in waters of the Irish EEZ (Fig. 1), which is a part of the Atlantic Margin of Northwest Europe (Shannon et al., 2005). The study area can be divided into three primary depth regimes: continental shelf waters less than 400 m deep, which includes the Irish Sea, the Celtic Sea and the Porcupine Bank; the continental margin, located along the western edge of the continental shelf and reaching depths of up to 2500 m; and the oceanic basins, located mainly in the Northwest (Rockall Trough) and the West (Porcupine Seabight and Porcupine Abyssal Plain), which are deeper than 2500 m (Dransfeld et al., 2014). Irish waters are characterised by oceanographic processes that influence the ocean circulation and primary production at different spatial scales. At a larger scale, the Atlantic Meridional Overturning Circulation (AMOC), carries warm shallow waters from the mid-Atlantic towards the northeast Atlantic (McCarthy et al., 2017). At regional scales, three currents, namely the Shelf Edge Current, the Slope Current and the Coastal Current, transport waters northwards and clockwise along the coast, acting as important transport for commercial fish eggs and larvae (White & Bowyer, 1997; Fernand et al., 2006; Dransfeld et al., 2014; Pingree & Garcia-Soto, 2014). In addition, underwater elevations and depressions lead to the presence of circular currents, such as the Irish Sea Gyre and the Taylor Column in the Porcupine Bank, which act as retention areas for nutrients and plankton (Mohn et al., 2002; Dransfeld et al., 2014). The combination of this varied topography and the different oceanographic processes results in a variety of habitats with different temperature, circulation and production regimes, that allow for the presence of at least 26 cetacean species (National Biodiversity Data Centre, 2022).

2.2. Data collection

Aerial surveys were carried out in an area covering 341000 km^2 in three summers (2016, 2021 and 2022) and two winter seasons (2016–2017 and 2022–2023) under the Irish government ObSERVE programme. Given the size of the area and limited weather windows for surveys, summer surveys were carried out between late May and early September, and winter surveys from November to March of the following year (Table 1). The survey area was divided into eight strata covering the different depth regimes (Fig. 1): stratum 1, 2 and 3 covering offshore waters in the outer edge of the continental shelf, the continental slope and beyond in the Northwest, along the Rockall Trough, the Porcupine Bank and the Porcupine Seabight; strata 4 and 5 covering continental shelf waters in the Celtic and the Irish Seas respectively; and three coastal strata (6A, 6B and 6C) covering waters of the north, west and south coasts, respectively (Fig. 1). Surveys followed a standard distance sampling line-transect methodology, designed to ensure equal coverage probability and representation of the survey area (Hiby & Hammond, 1989; Buckland et al., 2001). In 2016–2017 all strata were surveyed using two sets of randomly placed zig zag transects, while parallel transects, perpendicular to a baseline along the coast and the bathymetry gradient were used in strata 6A and 6C to improve coverage

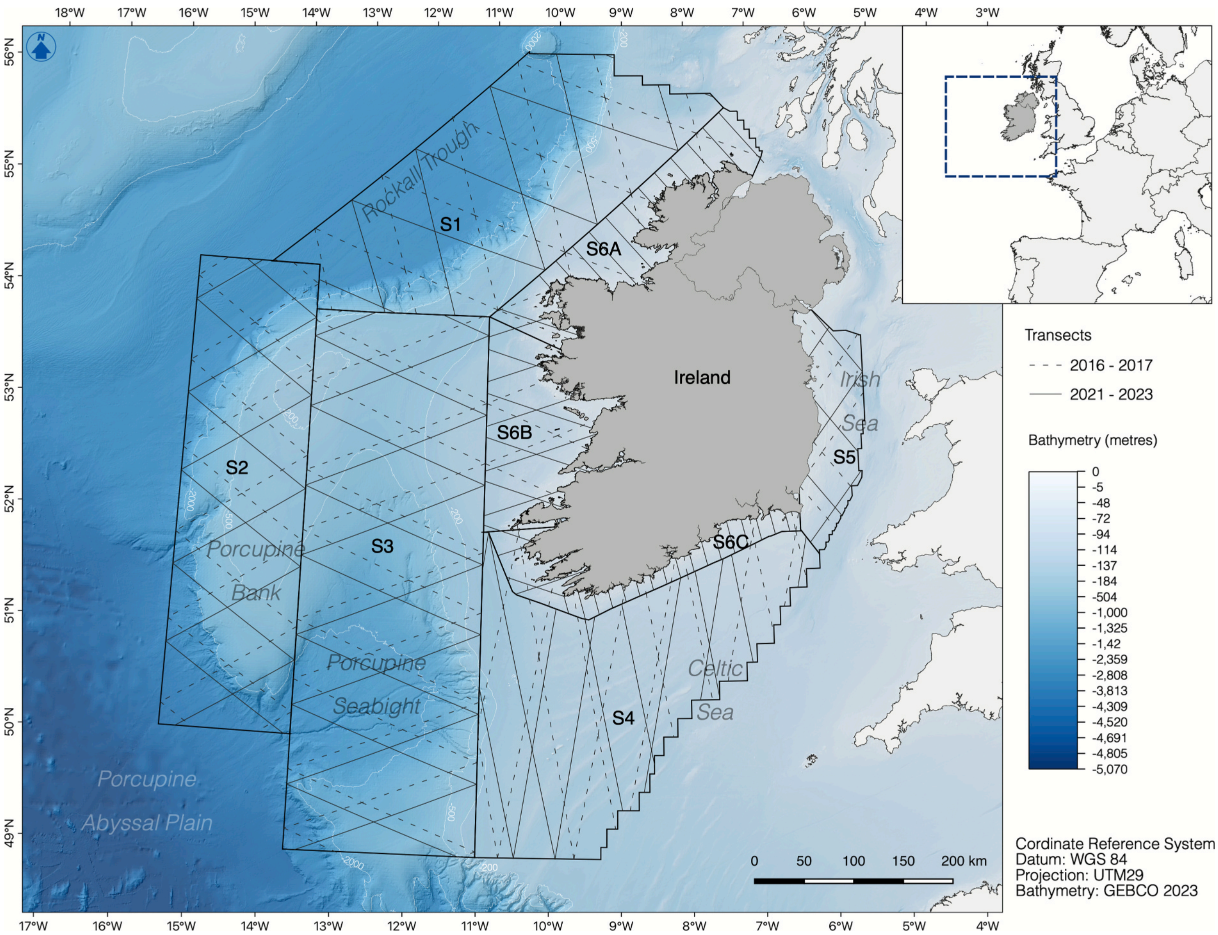

Fig. 1. Map of the study area showing the strata in which the area was divided to carry out the aerial surveys and the lines of the transects surveyed in 2016-2017 (dashed lines) and 2021-2023 (solid lines).

Table 1
Start and end date, total survey days and distance flown (km) on effort, in sea state ≤3 on the Beaufort scale for each survey season. Detailed information for each stratum and season is provided in Table SM3.

Season	Start date	End date	Survey days	Distance flown on effort (km)	Distance flown in sea state ≤3 (km)
Summer 2016	21 May 2016	7 July 2016	20	10,244	10,067
Winter 2016–2017	2 November 2016	15 March 2017	20	10,143	9,340
Summer 2021	7 July 2021	11 September 2021	22	9,994	9,356
Summer 2022	30 June 2022	14 August 2022	21	10,180	9,469
Winter 2022–2023	15 November 2022	06 March 2023	19	10,259	8996

in coastal areas in 2021–2023. The positions of the randomly placed zig zag transects in strata 1 to 5 and 6B differed between 2016–2017 and 2021–2023, by selecting a new randomised starting point while keeping the same coverage probability (Fig. 1).

Aerial surveys were carried out on board fixed-wing, high wing, twin engine aircraft (Britten-Norman BN-2 in 2016–2017 and Partenavia P-68 in 2021–2023) equipped with bubble windows to allow the observers an unobstructed view of the sea surface underneath the aircraft (Hammond et al., 2021). Flights were carried out at 183 m (600 feet) above the sea surface at a constant speed of 100 knots (185 km/h). Three or four scientific crew members were on board depending on the survey. Two observers, one at each side of the plane, reported the information related to environmental data and sightings to the data logger(s), who recorded all the information using the software VOR in 2016–2017 and 2021 and SAMMOA 2.1.0 (https://www.observatoire-pelagis.cnrs.fr/tools/sammoa/) in 2022–2023. Audio backup was recorded using a Zoom H1n Dictaphone in 2016–2017 and 2021 and an in-built audio recording system in SAMMOA in 2022–2023 and was used to validate the environmental conditions and sightings recorded. Team members were trained using the same protocol and new observers were always supervised by more experienced staff/members (Evans & Hammond, 2004).

Flights were carried out in target conditions of Beaufort sea state 3 or less, and visibility of 1 km or more. Observers were on effort for the duration of the transect, unless environmental conditions (presence of low clouds, fog or strong glare) obstructed the observers' view, in which case the observers were off effort. The position of the aircraft was automatically recorded every two seconds using a handheld GPS (Garmin GPS 60) connected to the computer. Environmental conditions were recorded at the beginning of every transect line and each time any of the conditions changed. These included sea state in the Beaufort scale, water turbidity, glare extent and severity, surface reflectance, known as "silvery shine", cloud cover and subjective conditions (good, moderate and poor), based on the probability of seeing a harbour porpoise (Hammond et al., 2002). Observers recorded all cetacean sightings within a perpendicular distance of 500 m from the aircraft. For each observation a time-stamped record containing the coordinates of the aircraft at that moment was created. Observers provided information on observation cue, species, group size, number of calves if present, declination angle measured with an inclinometer (to calculate the perpendicular distance of the group to the transect line), and main behaviour of the group. When identification was not possible to species level, sightings were recorded as unidentified dolphins and when distinction between common and striped dolphins was not possible, sightings were recorded as common/striped dolphins.

2.3. Data handling, validation and organisation

The collected data were stored in three different csv files in 2016–2017 and 2021 and in three different shapefiles in 2022–2023 (one with the locations of the aircraft every two seconds, one with all the locations at which the environmental conditions were changed and one with all the observations) and were collated in a single csv file using the packages *dplyr* (Wickham et al., 2023), *sf* (Pebesma, 2018) and *tidyverse* (Wickham et al., 2019) in the software R (R Core Team, 2023). The resulting files were validated using the audio recordings to ensure that there were no sightings missing and that the information on environmental data and sightings was correct.

2.4. Estimation of abundance and modelling framework

Mark-Recapture Distance Sampling (MRDS) was used to estimate the abundance of several small cetacean species. A central assumption in Conventional Distance Sampling (CDS) using line transect methodology is that the probability of detecting individuals or a group of cetaceans on the transect line g(0) = 1, and that the probability of detection decreases with distance from the transect line (Buckland et al., 2001). To calculate the probability of detection, CDS requires the perpendicular distance between the animal or group of animals and the transect line as a covariate to calculate the effective strip half width (esw). However, the amount of time cetaceans spend at the surface, combined with the speed of the aircraft, limits the time window that they are “available” to be detected (Buckland et al., 2001; Hammond et al., 2013). Therefore, the assumption g(0) = 1 can be violated due to two main aspects (Buckland et al., 2001): (1) cetaceans are unavailable, as they are underwater at the time the aircraft passes them (availability bias) and (2) cetaceans are available at the surface but missed by the observer (perception bias). Not correcting for these biases can lead to negatively biased abundance estimates (Hammond et al., 2021). We attempted a number of the “circle-backs” for harbour porpoise and common dolphins (Hiby, 1999) but were unable to conduct the necessary number to robustly estimate g(0). Therefore, we used g(0) derived from “circle-backs” conducted during SCANS-III and IV (Hammond et al., 2017; Gilles et al., 2023), for all species except bottlenose, Risso’s and unidentified dolphins, to improve the precision of the estimates. As most of the transects were surveyed in moderate subjective conditions, the g(0) derived for moderate conditions in SCANS-III and IV was selected for each species (Table SM1). Abundance for each species was then estimated using the design-based method for abundance estimation.

The design-based estimates were based on MRDS methods, which provide ways to estimate species detection probability at the track line (Burt et al., 2014) and to include covariates other than perpendicular distances to calculate the probability detection functions and esw for each species (Marques & Buckland, 2003). They also allow for the creation of a common detection function for species of similar detectability, but to calculate the abundance estimates separately (Marques & Buckland, 2003). Several covariates treated as factors, including sea state, subjective sighting conditions, glare, water turbidity, cloud cover (as this could create silvery shine on the water surface and make species identification more difficult), group size, individual observer, survey (2016–2017 or 2021–2023), year, a combination of season and year, airplane and species (when pooling two or more together) were incorporated in the models to estimate the best detection function for each species and season. The best detection function was selected with the Akaike’s Information Criterion AIC (Akaike, 1974), the performance of the qq-plot, and goodness of fit test (Cramer von Misses, Chi squared, Kolmogorov-Smirnov). Abundance was estimated with an adaptation of the Horvitz-Thompson estimator (Horvitz & Thompson, 1952), adapted to MRDS:

$$\widehat{N} = A\frac{n}{2L}\widehat{E}[s]\sum_{i=1}^{n}\frac{1}{\widehat{\mu}(Z_i)} \tag{1}$$

where, for each stratum, A is the total area, L is the distance travelled on effort in km, *n* is the number of primary sightings, $\widehat{\mu}$ is the estimated esw as a function of the covariate Z_i, and $\widehat{E}[s]$ is the estimate mean group size for a given species. To fit the detection functions to the species analysed, on-effort and off-effort on transect data were included in the analysis, and all data from all seasons and years were pooled. Abundance was estimated per season using MRDS distance sampling with the software Distance 7.0 (Thomas et al., 2010) in the 2016–2017 period and the *mrds* (Laake et al., 2023) and the *Distance* (Miller et al., 2019) packages of the statistical software R in the 2021–2023 period. Despite changes in software, Distance 7.0 calls the *mrds* package within R for the analysis (Thomas et al., 2010; Miller et al., 2019), and the analysis was the same in both survey periods, thus no effect in the outputs of the analysis was expected. The esw was obtained from the detection functions of each species, according to the variables included in it. The variation around the abundance estimates per species was calculated with the delta method combining the CV around the g(0) function with the CV from the detection function (Seber, 1982):

$$CV.total = \sqrt{\left(CV.df^2 + CV.g(0)^2\right)} \tag{2}$$

Final 95 % Coefficient Intervals (CIs) for the abundance estimates were obtained using the final CV and assuming that the estimates are lognormally distributed:

$$c = exp\left(1.96 \times \sqrt{log\left(1 + CV.total^2\right)}\right) \tag{3}$$

$$lower95CI = N.total/c$$

$$upper95CI = N.total \times c$$

For species with an adequate number of sightings (bottlenose dolphin, common dolphin and harbour porpoise), Density Surface Models (DSM, Miller et al., 2013) were used to generate abundance estimates and better understand the factors driving their abundance, as well as predict abundance spatially. The methodology and the covariates used are detailed in the Supplementary Material, and in Table SM2, respectively.

3. Results

3.1. Survey effort

Aerial surveys were carried out over 102 days across the five seasons, summer 2016, 2021 and 2022 and winter 2016–2017 and 2022–2023. Surveys were conducted mostly in July and August during summer, and in winter, mostly in November (Table 1 and Table SM3). Distance covered on effort varied between 9994 and 10259 km per season, totalling 50820 km (Table 1 and Table SM2). Of these, 47105 km (93 %) were flown in sea state 3 or less on the Beaufort scale (Figure SM1 and Table SM3). Subjective conditions were slightly better in the second survey period (Table SM4).

3.2. Small cetacean sightings

A total of 1796 small cetacean sightings were recorded across all surveys, comprising seven species (Table 2). Common dolphin (843 sightings) was the most frequently observed species, followed by bottlenose dolphin (573 sightings) and harbour porpoise (290 sightings). Species such as Risso's dolphin (51 sightings), white-sided dolphin (17 sightings), striped dolphin (11 sightings) and white-beaked dolphin (11 sightings) were less frequently seen. An additional 89 sightings were recorded as either common or striped dolphins (referred to as common/striped), and another 257 sightings were recorded as unidentified dolphins. A higher number of bottlenose dolphins was recorded in the 2016–2017 period, compared to the 2021–2023 period. In contrast, the number of sightings of common dolphins was higher in the 2021–2023 period, particularly in summer 2021, compared to the 2016–2017 period. Other species such as harbour porpoise showed markedly fewer sightings in summer and winter 2022–2023. Sightings of unidentified dolphins were higher in the 2016–2017 period, compared to the 2021–2023 period.

The distribution of sightings (Fig. 2, SM2, SM3) showed three different general patterns. For bottlenose and common dolphins, sightings were distributed ubiquitously throughout the survey area except for the Irish Sea. In contrast, harbour porpoise sightings were distributed in coastal and shallower areas, particularly in the Irish Sea, and to a lesser extent in the Celtic Sea and in coastal waters off the west and north coasts. Finally, Risso's, white-beaked, white-sided and striped dolphin sightings were scattered across the study area, primarily in continental shelf waters, and these species were sporadically observed.

3.3. Truncation distances

The final detection functions were obtained using Hazard Rate (HR) or Half Normal (HN) key functions depending on the species and survey period (Table SM5). The number of covariates included in the detection functions varied among species between one and three, with sea state, cloud cover or group size being the most commonly occurring

Table 2
Number of sightings, number of individuals, corrected density estimate (D in individuals/km^2), corrected abundance estimates (N), related CVs and 95 % CIs obtained for the different species for each season. In brackets, number of sightings and individuals for white-beaked and white-sided dolphins, respectively. A "–" is used for those seasons in which abundance could not be estimated.

	Season	Number of sightings	Number of individuals	D	D 95 % CIs	N	CV	95 % CIs
Harbour porpoise	Summer 2016	117	161	0.113	0.094–0.135	38,260	23.7	30,972–47,265
	Winter 2016	45	74	0.057	0.046–0.071	19,452	29.6	14,279–26,498
	Summer 2021	67	119	0.062	0.036–0.105	20,991	31.9	16,586–26,567
	Summer 2022	32	41	0.022	0.013–0.037	7,510	31.7	5,940–9,495
	Winter 2022	29	52	0.019	0.009–0.040	6,604	40.8	4,936–8,836
Bottlenose dolphin[a]	Summer 2016	172	1,132	0.257	0.219–0.302	87,330	20.7	58,029–131,426
	Winter 2016	289	2,381	0.626	0.554–0.708	212,646	15.5	157,026–287,967
	Summer 2021	22	83	0.014	0.006–0.029	4,690	40.0	2,202–9,989
	Summer 2022	55	236	0.046	0.029–0.071	15,521	23.0	9,950–24,210
	Winter 2022	35	153	0.029	0.018–0.047	9,844	25.3	6,040–16,044
Common dolphin	Summer 2016	19	79	0.040	0.024–0.068	13,633	86.0	5,214–35,646
	Winter 2016	12	98	0.039	0.024–0.063	13,192	75.2	5,558–31,308
	Summer 2021	451	3,083	1.748	1.231–2.481	594,293	26.2	487,570–724,375
	Summer 2022	193	1,528	0.853	0.599–1.214	289,893	26.3	237,650–353,621
	Winter 2022	168	1,114	0.753	0.490–1.159	256,142	29.2	205,917–318,618
Risso's dolphin[a]	Summer 2016	20	72	0.008	0.006–0.011	2,630	40.8	1,212–5,707
	Winter 2016	1	2	0.0003	0–0.001	97	97.6	19–491
	Summer 2021	8	20	0.002	0.001–0.006	806	49.3	353–2,013
	Summer 2022	13	31	0.004	0.002–0.008	1,372	38.2	665–2,833
	Winter 2022	9	40	0.006	0.002–0.016	2,080	53.9	772–5,608
White-beaked and white-sided dolphins*	WBD 2016–2017	3	6	0.006	0.004–0.008	2,019	41.1	1,213–3,362
	WSD 2016–2017	6	46	0.004	0.003–0.006	2,907	60.2	1,417–5,963
	Summer 2021	11 (0–11)	24 (0–24)	0.009	0.004–0.020	2,938	47.1	2,115–4,080
	Summer 2022	3 (3–0)	41 (41–0)	0.021	0.006–0.074	7,250	71.9	4,751–11,499
	Winter 2022	0	0	–	–	–	–	–
Striped dolphin	Summer 2016	0	0	–	–	–	–	–
	Winter 2016	0	0	–	–	–	–	–
	Summer 2021	9	96	–	–	–	–	–
	Summer 2022	2	13	–	–	–	–	–
	Winter 2022	0	0	–	–	–	–	–
Unidentified dolphins	Summer 2016	76	218	–	–	–	–	–
	Winter 2016	129	478	–	–	–	–	–
	Summer 2021	16	72	–	–	–	–	–
	Summer 2022	17	58	–	–	–	–	–
	Winter 2022	19	68	–	–	–	–	–

*Results for white-beaked (WBD) and white-sided dolphins (WSD) in 2016–2017 are presented as separate species, but for winter and summer combined. Tables SM6 to SM10 show results for each stratum.

[a] Densities, abundance estimates CVs and CIs for bottlenose and Risso's dolphin presented here are not corrected for g(0).

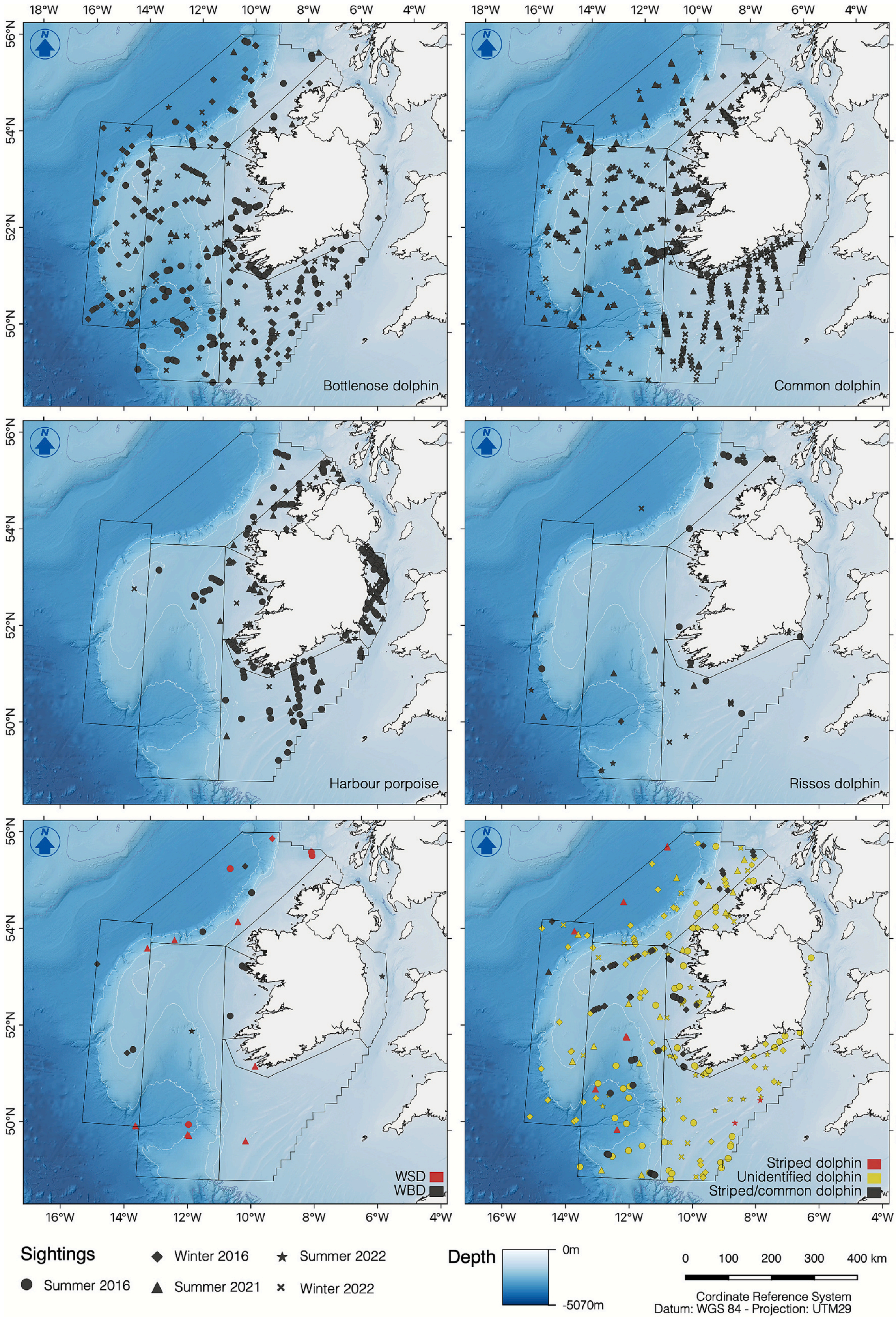

Fig. 2. The distribution of sightings of the different species during the aerial surveys. Sightings were classified by survey using different symbols. Where multiple species are shown within a map, they are denoted by colour. Some species are abbreviated as follows: WSD: White-sided dolphins and WBD: White-beaked dolphins.

(Table SM5).

3.4. Abundance estimates and density surface models

There were insufficient observations for model-based estimates to be generated for all species. Furthermore, abundance estimates generated using the design-based approach had overall lower Coefficients of Variation (CVs) than those obtained using the model-based approach. Therefore, for consistency, design-based estimates are shown in this paper. Abundance estimates using the design-based approach were generated for all species and seasons, except for white-sided, white-beaked, striped, common/striped and unidentified dolphins (Table 2). For the two *Lagenorhynchus* species, abundances were estimated for both species combined in summer 2021 and 2022 and separately for white-beaked and white-sided dolphins for summer and winter in 2016–2017. For striped, common/striped and unidentified dolphins there were insufficient sightings to generate separate abundance estimates. CVs around the abundance estimates were lower for the seasons with a higher number of sightings, whereas they increased in those seasons with fewer sightings (Table 2). Densities for harbour porpoise

Fig. 3. Density difference between each stratum in each season and the overall density (black horizontal line) found in summer 2016 (for the summer periods) and winter 2016-2017 (for the winter periods). Blue bars show higher stratum densities than the overall density of the 2016-2017 period, while yellow bars show lower stratum densities than overall density of the 2016-2017 period. Species names are abbreviated as follows: HP: Harbour porpoise, BND: Bottlenose dolphin, CD: Common dolphin, RD: Risso's dolphin.

showed the highest CVs, particularly for winter 2022–2023.

For species with sufficient sightings for model-based abundance estimates, significant factors determining species abundance and distribution varied between species and seasons and included an interaction between longitude and latitude for most species and seasons (Table SM6). Other significant factors included Sea Surface Height (SSH), Sea Surface Temperature (SST) and Average Chlorophyll-a concentration, and are detailed in each species section. Terms selected for each model are reported in Table SM7. Density distributions could be predicted for harbour porpoises (except for winter 2016–2017), bottlenose dolphin and common dolphin (except 2016–2017) and are described below and in Figs. SM9−SM12.

3.4.1. *Harbour porpoise*

Harbour porpoises were predominantly sighted in continental shelf waters (Fig. 2 and SM2). Data were truncated at 350 m perpendicular distance in 2016–2017 and at 334 m in 2021–2023 (Table SM5). The best model included subjective sighting conditions as covariates in 2016–2017 and sea state, cloud cover and plane in 2021–2023 (Figure SM4). Harbour porpoise showed seasonal and interannual variations in abundance. Summer had higher abundances than winter in both survey periods. Abundance estimates remained relatively stable between 2016 and 2021 but were notably lower in summer and winter 2022–2023, with highest abundances of 38260 individuals reported in summer 2016 and lowest abundances of 6604 individuals in winter 2022–2023. Spatiotemporal variation in density was also detected. Highest densities were generally found in the Irish Sea and to a lesser extent in the Celtic Sea and the northern and southern coastal strata (Fig. 3 and Table SM8). Predicted densities were lower in summer and winter 2022–2023 when compared with the 2016–2017 survey period (Fig. 3), particularly in the Celtic Sea and coastal waters off the north and south coasts (strata 6A and 6C).

Modelled harbour porpoise distribution was influenced by an interaction between latitude and longitude, with highest densities predicted in the Irish Sea; by water depth, with highest densities predicted around the 100 m depth contour; Sea Surface Height (SSH) with higher densities at higher SSH; the persistence of Chlorophyll-a fronts, with highest densities associated with weaker front persistence, and the standard deviation of Sea Surface Temperature (SST_SD, Table SM6 and Figure SM6). While Harbour porpoise showed higher densities primarily in the Irish Sea, Celtic Sea and parts of the north and west continental shelf waters in summer 2016, highest densities were mostly restricted to the Irish sea in 2021–2023 and were overall lower than in 2016–2017 (Figures SM9 and SM10).

3.4.2. *Bottlenose dolphin*

Bottlenose dolphins were sighted in all strata (Fig. 2 and SM2). Sightings data were truncated at 450 m perpendicular distance in 2016–2017 and at 469 m in 2021–2023 (Table SM5). The best models for detection functions included sea state as covariates in 2016–2017 and water turbidity and the logarithm of group size in 2021–2023 (Figure SM4). Abundance estimates for this species were not corrected for g(0) and therefore, likely underestimate total abundance. The abundance estimate was notably higher in 2016–2017 (87,330 individuals in summer and 212,646 individuals in winter) than in 2021–2023, when abundances ranged from 4,690 individuals in summer 2021 to 15,521 in summer 2022 (Table 2). Seasonal variations in abundance were recorded in 2016–2017, with highest abundances in winter. The abundance in the winter of 2023, however, was lower than the previous summer (Table 2). Predicted densities for bottlenose dolphin were high in all strata in the 2016–2017 period, but particularly in the coastal strata 6B and 6C, which showed higher density than the overall density (Fig. 3 and Table SM9). In the 2021–2023 period, although densities in all strata were considerably lower than the overall density in 2016–2017, highest densities occurred in the Celtic Sea (stratum 4) and lowest densities were found in the Irish Sea (stratum 5) in all seasons. Seasonal differences found in 2016–2017 were associated with increased densities in offshore areas (strata 1 and 2) and the Celtic Sea in winter and coastal areas in the north and west coast in summer (Fig. 3 and Table SM6).

Predicted bottlenose dolphin abundance and distribution were related to geographical factors such as an interaction between latitude and longitude, distance to the coast and slope with highest abundances found in less steep waters. In addition to these spatial variables, factors such as SST and CHL concentration played a role in predicting distribution. Higher dolphin densities were associated with elevated SST and CHL standard deviation (CHL_SD), and lower SST standard deviation (SST_SD) (Table SM6 and Figure SM7). Overall, bottlenose dolphin densities were noticeably higher in 2016–2017, with peak concentrations found in coastal waters off the southwest coast and in offshore areas south of the Porcupine Seabight during summer, along the 200 m depth contour and in the Celtic Sea during winter (Figures SM9 and SM11). In contrast, densities during 2021–2023 were significantly lower, with the highest concentrations observed in the Celtic Sea.

3.4.3. *Common dolphin*

Common dolphins were only observed in a few strata in 2016–2017, but were sighted in all strata in 2021–2023 (Fig. 2 and SM2). Data were truncated at 550 m perpendicular distance in 2016–2017 and at 409 m in 2021–2023 (Table SM5). The best models for detection functions included glare and group size as covariates in 2016–2017 and season, cloud cover and the logarithm of group size in 2021–2023 (Figure SM4). Common dolphins showed large interannual variation in abundance, with lower point estimates of around 13,000 individuals in summer and winter 2016–2017, and much higher estimates of up to 594,293 individuals in the 2021–2023 period (Table 2). If common/striped dolphins sightings are added, this results in an overall abundance estimate of 33,215 individuals in summer 2016 and 43,030 individuals in winter 2016–2017, notably lower than the common dolphin's abundance estimates in 2021–2023. Seasonal differences were also detected in 2021–2023 when abundance was twice as high in summer compared to winter. Densities were lower in all strata except for the Irish Sea in 2016–2017 compared to 2021–2023. In the latter period, higher densities occurred in continental shelf waters off the west coast of Ireland (stratum 6B) and in the Celtic Sea (stratum 4) in summer (Fig. 3 and Table SM10). In winter, the species showed highest densities in coastal areas of the south coast of Ireland and in the Celtic Sea (strata 6C and 4, respectively). Lowest overall densities were found in the Irish Sea (Fig. 3 and Table SM10).

Common dolphin distribution appeared to be primarily influenced by geographical factors, including interaction between latitude and longitude, distance to the 2000 m depth contour and overall depth, The highest densities were predicted to occur in shallower waters to the west of the survey area. Higher densities were also associated with lower CHL_SD, weaker SST fronts, and reduced SSH (Table SM6 and Figure SM8). Due to a limited number of sightings, density distributions could not be generated for common dolphins in the 1st survey period (2015–2016). In contrast, during 2021–2023, the highest densities were found in the Celtic Sea and along the west coast of Ireland in summer, and in waters shallower than 1000 m during winter (Figures SM9 and SM15).

3.4.4. *Risso's dolphin*

Risso's dolphins were sporadically seen in all strata (Fig. 2 and SM2). Data were truncated at 801 m in 2021–2023 (Table SM5). The best models included perpendicular distance as a covariate in 2016–2017 and season, and the logarithm of group size in 2021–2023 (Figure SM5 Abundance estimates for this species were not corrected for g(0) and therefore, likely underestimate the total abundance of the species. Abundance estimates showed an apparent interannual variation in winter, varying from 97 individuals in 2016–2017 to 2,080 individuals in 2022–2023, whereas summer abundances were slightly higher in

2016 than in 2021 and 2022 (Table 2). Compared with the previous three species, densities were low overall, although higher densities were recorded along the Rockall Trough (stratum 1) and coastal areas off the south and north coasts in summer 2016. Densities in summer 2021 and 2022 were lower in all strata except for strata 5 and 6C, than the overall density in summer 2016 (Fig. 3 and Table SM11). Densities in winter 2022–2023 were considerably higher than densities in winter 2016–2017, particularly in stratum 6A. These estimates must be considered with caution, as they were derived from a low number of sightings (ranging from 1 to 20 per season) and associated high CVs.

3.4.5. White-beaked and white-sided dolphins

These cold-water species were mostly seen in northern and western strata (Fig. 2 and SM3). Data were truncated at 600 m in 2016–2017 and at 469 m in 2021–2023 (Table SM5). The best models for detection functions included subjective conditions as covariates in 2016–2017 and species in 2021–2023 (Figure SM4). Seasonal abundance estimates could only be assessed for summer 2021 (2,938 individuals) and 2022 (7,250 individuals) and these were considerably higher in 2022 (Table 2). In 2016–2017 abundance was estimated at 2,019 individuals for white-beaked dolphins and 2,907 individuals for white-sided dolphin combining summer and winter sightings. Densities were low overall, with highest values in strata 1 and 2, particularly in summer 2022 (Table SM12).

3.4.6. Striped, common or striped and unidentified dolphins

Striped dolphins were recorded mostly in offshore areas of strata 1, 2 and 3 and in the Celtic Sea (Fig. 2 and SM3). Sightings of common/striped dolphins occurred mainly in continental shelf waters of the north and west coast, while dolphin sightings for which the species could not be determined occurred in all strata, being lowest in the Irish Sea (Fig. 2 and SM3).

4. Discussion

Aerial surveys have been extensively used to estimate cetacean abundance over large spatial scales as they can cover large areas in small amounts of time (Andriolo et al., 2006; Hedley et al., 2011; Laran et al., 2017a,b; ACCOBAMS, 2021; Peltier et al., 2024). The data collected during the ObSERVE aerial survey programmes allowed for the seasonal estimation of abundance for small cetaceans in two different temporal periods. The surveys carried out in summer complement other extensive aerial surveys in adjacent waters of the northeast Atlantic, such as SCANS-II, III and IV in summer 2005, 2016 and 2022, respectively (Hammond et al., 2013, 2017; Gilles et al., 2023) and contribute to a more comprehensive understanding of the dynamics of several species of cetaceans in the area. Both ObSERVE and SCANS III & IV surveys were carried out in the same summer periods, coinciding with EU mandated reporting time frames for reporting on MSFD and Habitats Directives. ObSERVE surveys were also carried out in winter (2016–2017 and 2022–2023), allowing for the assessment of seasonal as well as interannual variations in abundance over time. To our knowledge, this is the first estimate of small cetacean abundance during winter – a season when bycatch, a major anthropogenic threat, is known to occur in Irish waters.

4.1. Methodological considerations

The results presented here report the abundance of several species of small cetaceans in three summers and two winters in two survey periods that are six years apart. Given the limited number of yearly abundance estimates, results do not allow for the assessment of trends over time (Authier et al., 2020) and therefore should not be interpreted as such. However, results do show interannual and seasonal fluctuations for some species. The analysis using the Design-based approach resulted in fairly robust estimates (CVs < 45 % for most species and survey periods), similar to those obtained in other aerial surveys covering extensive areas (Laran et al., 2017a,b; Hammond et al., 2017; Gilles et al., 2023). However, a few estimates were derived from a relatively small number of sightings (e.g. Common dolphin in summer and winter 2016), which resulted in higher uncertainty. In these cases, abundance estimates must be considered with caution.

Some species like striped and common dolphins are difficult to distinguish from the air, especially in mixed groups. Identification to species level could not be inferred for a number of sightings, particularly in 2016–2017. Sea state, among other environmental variables, affects the ability of observers to detect and identify cetaceans to species level, with detection probabilities decreasing with increasing sea state (Hammond et al., 2002). However, the percentage of transects conducted at sea state ≤3 was similar for both survey periods, suggesting similar conditions to detect and identify species. In contrast, the percentage of transects surveyed under subjective good or moderate conditions was higher in the second survey period (Table SM3). To improve species identification, and consequently better assess abundance and risk to some of these species, the addition of digital photography as a complement to visual aerial surveys could vastly increase species identification precision in future surveys (Taylor et al., 2014).

4.2. Ecological and geographical context for species

Our results allowed for seasonal and interannual comparison of species abundances. While some species, such as common dolphins have apparently increased in abundance in this region between survey periods, the abundance estimates for harbour porpoises and bottlenose dolphins have decreased. As the surveys were conducted over a short time frame, these temporal patterns should not be considered as trends *per se*, but they do highlight the considerable fluctuations in abundance in a range of species. The reason(s) for these fluctuations are not known, but they could indicate a response to a combination of large scale oceanographic and climate processes that are occurring in the northeast Atlantic (Holliday et al., 2020; Ditlevsen & Ditlevsen, 2023), alongside the effects of human activities on ecosystems (Gascuel et al., 2016; Pedreschi et al., 2019; Baudron et al., 2020; Kempf et al., 2022) and on cetacean species (Pirotta et al., 2015; Jepson et al., 2016; Kavanagh et al., 2019; Giralt Paradell et al., 2021; Peltier et al., 2021; Rogan et al., 2021; Braulik et al., 2023). The factors behind these changes and the ultimate consequences at a population or species level likely vary between species.

4.2.1. Harbour porpoise

Harbour porpoises are widely distributed across the northern hemisphere, especially the shelf waters of the North Atlantic, where there is little evidence of genetically distinct populations (Lah et al., 2016; Ben Chehida et al., 2021; Autenrieth et al., 2024). In the northeast Atlantic, five management units (MUs) have been defined for conservation/management purposes under the MSFD, including the West Scotland & Northern Ireland MU, and the Celtic & Irish Seas MU (Figure SM5, Evans & Teilmann 2009; ICES WGMME, 2013). Declines have been noted in both MUs at both small and large scales. For instance, boat-based or acoustic surveys carried out every three to six years in three Special Areas of Conservation (SACs) within the Celtic & Irish Seas MU area, have reported declines in abundance in all three SACs in the last decade (O'Brien and Berrow 2020; Berrow et al., 2021; O'Brien et al., 2022; Todd, 2023). In a wider context SCANS-II and III revealed a substantial decrease in abundance in the West Scotland & Northern Ireland and the Celtic & Irish Seas MUs' areas between 2005 and 2016 (Hammond et al., 2013, 2017). A similar pattern is found for these two MUs when considering SCANS and ObSERVE surveys together, with lower abundance estimates in 2022 compared to 2016 (Figure SM5, Hammond et al., 2017; Rogan et al., 2018; Gilles et al., 2023). Harbour porpoise abundance estimates in adjacent MUs (North Sea and Iberian Peninsula) in the northeast Atlantic have remained stable or decreased slightly

(Hammond et al., 2017; Gilles et al., 2023).

During the ObSERVE programmes, harbour porpoise sightings were primarily distributed in the Irish Sea but also in the Celtic Sea and along the west coast in 2016, whereas they were more restricted to the Irish Sea in 2021–2023. Similarly, densities decreased substantially across all strata, particularly outside the Irish Sea in 2021–2023, suggesting a range contraction of the species in this region. This was confirmed by the density surface models, which predicted a contraction of areas of higher densities of harbour porpoise over time, highlighting the Irish Sea as an area of importance for the species. Potential explanations are a redistribution of the species within the northeast Atlantic, a marked decrease in abundance in Irish waters due to changes in their fitness in these waters, or a combination of both. Harbour porpoise have previously been shown to shift their distribution, progressively spreading into the English Channel from the North Sea (Hammond et al., 2017; Gilles et al., 2023), while a decrease in abundance has been detected in the Kattegat & Belt Sea MU, potentially as a consequence of the cumulative impact of bycatch, naval exercises, noise pollution and depletion of resources (Owen et al., 2024). Additionally, recent changes in abundance and distribution of several commercial fish species in the Irish and Celtic Seas (Baudron et al., 2020; Bentley et al., 2020a; Kempf et al., 2022), may have resulted in a shift in diet toward less energy-rich prey (Giralt Paradell et al., 2021; Rogan et al., 2025). This appears to be reflected in the increased number of strandings of harbour porpoise in poor body condition in Irish and UK waters (Coombs et al., 2019; Rogan et al., 2025).

There are several factors that suggest the interannual variation detected in Irish waters between 2016 and 2023 could reflect a population decline within this region: (1) decreases in abundance have also been detected at smaller scales in several Irish SACs over the same temporal scale (O'Brien and Berrow 2020; Berrow et al., 2021; O'Brien et al., 2022; Todd, 2023); (2) lower abundance in Irish waters does not coincide with an increase in abundance in the adjacent waters surveyed during SCANS-III and SCANS-IV (Hammond et al., 2017; Gilles et al., 2023); (3) the range contraction in Irish waters between the two survey periods (Goh et al., 2025); (4) bycatch. The latter is being highlighted as one of the major conservation concerns for harbour porpoises in Irish and UK waters (Carlén et al., 2021; Rogan et al., 2021), with bycatch estimates exceeding safe removal limits for the West Scotland & Northern Ireland, and the Celtic & Irish Seas MUs (Taylor et al., 2022). Although the abundance estimates presented here do neither confirm nor exclude specific causes for the observed decline, spatial modelling indicates a clear range contraction to the Irish Sea. This highlights growing concerns over increasing anthropogenic pressures and climate change in the region (Goh et al., 2025).

4.2.2. Bottlenose dolphin

In the northeast Atlantic, two genetically differentiated bottlenose dolphin ecotypes have been defined: (1) coastal: with several isolated coastal populations with high site fidelity, and transient individuals conducting long distance movements along the Irish coast and between Scotland and Ireland (O'Brien et al., 2009; Mirimin et al., 2011; Robinson et al., 2012; Louis et al., 2014; Oudejans et al., 2015; Nykänen et al., 2018, 2019); and (2) pelagic: less well known, genetically diverse and with lower potential for population genetic structure due to large scale movements across the north Atlantic (Quérouil et al., 2007; Louis, et al., 2014, Alexandre et al., 2024).

Differentiation among these two ecotypes from the aircraft is not possible. However, the large winter abundance estimate for 2016 comprised many sightings of large sized groups, mostly on the continental shelf and further offshore. This estimate likely represents the offshore or pelagic ecotype and contrasts with the estimates for "resident" populations in the area, which, based on photo-identification studies, number less than 200 individuals (Berrow et al., 2012; Nykänen et al., 2015; Rogan et al., 2015). The density distributions revealed areas of high concentration in coastal zones, likely representing resident coastal populations, as well as notable high-density regions offshore, suggesting the presence of the offshore ecotype. These offshore hotspots were particularly prominent during the summer of 2016–2017, coinciding with elevated abundance estimates for bottlenose dolphins, likely reflecting increased occurrence of this offshore ecotype. The larger group sizes, along with the pronounced seasonal and inter-annual variation in habitat use and the notably high abundance in 2016–2017, which showed an approximate 2.5-fold increase in density during winter (0.626 individuals/km^2) compared to summer (0.257 individuals/km^2), highlights how limited our understanding of this ecotype in the Atlantic remains.

Interannual variation in abundance in summer has also been found during the larger scale surveys SCANS-II, III and IV (2005, 2013 and 2022) and CODA (2007) in the northeast Atlantic (Hammond et al., 2009, 2013, 2017; Gilles et al., 2023). However, considering the areas surveyed in this study and the SCANS surveys together, which lie at the edge of the pelagic distribution, bottlenose dolphin summer abundance remained relatively stable between both survey periods. Movement patterns of pelagic bottlenose dolphins across the Atlantic have not been thoroughly studied. However, large-scale oceanographic changes are occurring in the area (Holliday et al., 2020; Ditlevsen & Ditlevsen, 2023) and these could result in seasonal and interannual fluctuations in prey distribution (Ottersen et al., 2013; Baudron et al., 2020; Bentley et al., 2020b; Báez et al., 2021), potentially triggering large-scale distribution shifts within the bottlenose dolphin pelagic population (Gilles et al., 2023). In this regard, interannual changes in bottlenose dolphin abundance found in this study, coupled with the low genetic differentiation shown by the pelagic population could reflect large-scale temporal redistribution of the species rather than a decrease in abundance.

4.2.3. Common dolphin

Results highlighted large interannual differences in common dolphin abundance between survey periods and between summer 2021 and summer 2022. The difference between summer 2021 and 2022 can be largely attributed to higher densities in strata 1 to 3. Even considering common/striped dolphins together with common dolphins, abundance estimates for 2016–2017 are still considerably lower than the estimates in 2021–2023. Variation in abundance between survey periods contrasts with wider SCANS-III and IV surveys, which reported low variation in common dolphin abundance between surveys. However, an increase in occurrence of common dolphins in areas of the Celtic Sea and Southwest of England was detected in 2022 (Gilles et al., 2023). Given that abundance estimates derived from SCANS-III and IV were consistent across surveys, and the co-occurrence of SCANS and ObSERVE surveys, the apparent increase in abundance detected in the Irish EEZ in 2021–2022 in this study is unlikely to reflect a redistribution of the species within European Atlantic waters, but changes occurring at larger scales.

In the northeast Atlantic common dolphins comprise a panmictic population with little genetic differentiation, and a low level of genetic differentiation from the northwest Atlantic population (Mirimin et al., 2009; Amaral et al., 2012; Moura et al., 2013, Becker et al., 2024), suggesting, *inter-alia*, long-distance dispersal or large-scale movements within the basin.

Recent studies have predicted a poleward expansion of the distribution range of the species following large scale oceanographic and climate changes (Lambert et al., 2011), and have reported increasing trends in abundance in the northeast Atlantic (Evans & Waggitt, 2020; Astarloa et al., 2021). Therefore, interannual changes in common dolphin abundance found in this study could reflect large-scale movements within the north Atlantic, coupled with a poleward expansion of the distribution range of the species. The drivers for these changes are not fully understood, but they could be linked to changes in prey availability or social structure (Murphy et al., 2013). Common dolphins feed on a wide variety of prey species in this region, likely with unpredictable availability, including mesopelagic prey in offshore waters (e.g. Brophy et al., 2009; Santos et al., 2013; Marçalo et al., 2018). In parts of the area surveyed, there have been large changes in prey availability, with species such as herring (*Clupea harangus*) seeing large scale fluctuations (Clarke & Egan, 2017), and wider ecosystem changes (Gascuel et al., 2016; Pedreschi et al., 2019; Kempf et al., 2022). Density distributions could not be generated for common dolphins for 2016–2017, hindering the comparison between survey periods. However, distribution models covering a larger scale could provide more detail on changes in the species distribution at a regional scale and highlight the potential drivers behind these changes. Results of this study suggest that the species has high mobility within the northeast Atlantic, and this should be considered when developing management and conservation strategies.

4.2.4. Risso's dolphin

Risso's dolphins typically inhabit deep (200–1,000 m), offshore waters, along the shelf edge (Jefferson et al., 2014), but in some areas of the northeast Atlantic, they also occur in shallower more coastal areas (e.g. Bloch, 2012; de Boer et al., 2013; Hodgins et al., 2024). Very little is known about population structure in this region, but they are currently managed as a single MU around the British Isles and the North Sea (IAMMWG, 2023). The depth range observed in this study (20–4,030 m) suggest that they can occupy a wide range of habitats, from shallow coastal environments to deeper canyon systems, including the Rockall Trough and Porcupine Basin.

Summer abundance estimates did not show interannual variability, which is consistent with SCANS-III and IV surveys in summer 2016 and 2022 (Hammond et al., 2017; Gilles et al., 2023) and contrasts with winter estimates, that showed evident interannual variations. The high estimate obtained in winter 2022–23, contrasts with seasonal patterns in Ireland and the UK, that reflect a higher number of sightings between May and September and lower predicted densities in winter (Berrow et al., 2010; Weir et al., 2019; Waggitt et al., 2020). Fine and large-scale movements in the region are poorly understood, although photo-identification studies suggest some degree of site fidelity in areas of Wales and Scotland (de Boer et al., 2013; Weir et al., 2019, Hodgins et al., 2024). Whether this highest estimate in winter 2022–23 is driven by species redistribution in the area, large scale movements or a pole-ward expansion of the distribution range of the species, as suggested by previous studies (Bloch, 2012), remains unclear.

4.2.5. White-beaked and white-sided dolphins

White-beaked dolphins show fine scale structure in the northeast Atlantic, with animals from Ireland and west Scotland forming a genetically distinct grouping from the animals occurring further north and west, i.e. Iceland and Barents Sea (Gose et al., 2024). Studies suggest that there has been a northward shift in its southern distribution range (Lambert et al., 2014; IJsseldijk et al., 2018; Waggitt et al., 2020). In contrast, broad-scale connectivity has been reported for Atlantic white-sided dolphins, with no evidence of population differentiation (Gose et al., 2023), suggesting that the species is wide ranging and probably numerous (Braulik, 2019). The species is restricted to cold-water habitats which may make it vulnerable to habitat contraction and climate change (MacLeod et al., 2005). Understanding the seasonal shifts in abundance and distribution of these two species may help inform management under different climate change scenarios.

Irish waters are likely located at the southernmost limit of both these species' range (MacLeod et al., 2005; Lambert et al., 2014). Thus, interannual changes in abundance and distribution might be expected. Abundance estimates obtained in this study are based on a small number of sightings and have associated high CVs and should therefore be considered with caution. As in SCANS-III and IV surveys, highest densities were found in the northern strata of the study area, although the majority of sightings in SCANS-III and IV occurred in waters at higher latitudes than the Irish EEZ (Hammond et al., 2017; Gilles et al., 2023). Given the low latitudes of the study area, an increase in sea water temperature might result in these species shifting range towards northern latitudes, making them particularly vulnerable to the impact of human activities or potential inter-species competition (Evans & Waggitt, 2020; Plint et al., 2023; Gose et al., 2024).

4.2.6. Striped dolphins

In contrast, striped dolphins are a warm-temperate species associated with and often found in deeper waters beyond the continental shelf (e.g. Archer, 2009, Laran et al., 2017a), with extra-limital distribution in the northeast Atlantic as far north as the Faroe Islands in some years (e.g. Bloch et al., 1996). To date, information on population structure within the Atlantic is lacking, with data showing some gene flow between adjacent areas. No formal abundance estimate exists for this species in the Irish EEZ but occasional sightings, strandings and bycatch records, suggest that individuals occur throughout the year (Berrow & Rogan, 1997; Rogan & Mackey, 2007; O'Connell & Berrow, 2014, 2015, 2017, 2018a,b, 2019, 2020, O'Connell et al., 2021a, b; Levesque & Berrow, 2022) and that the numbers of individuals stranding per year varies, showing an increasing trend since the 1980 s (Coombs et al., 2019; Williamson et al., 2021).

4.3. Implications for conservation

Estimates of cetacean abundance over time have been widely used in conservation biology, providing information on the status of a population or a species (Hammond et al., 2021). Abundance estimates derived from this work could be used to assess Criterion D1C2 of the MSFD in the marine subregion Celtic Seas, a primary criterion within Descriptor 1 *Marine Biodiversity* used to establish if a population is adversely affected by anthropogenic pressures. In addition, the abundances estimated here could be also used to report under Article 17 of the Habitats Directive, which requires the assessment of the conservation status of the species listed under Annexes II, IV and V (Council Directive 92/43/EEC). Among the different criteria used, the abundance estimates obtained here could contribute to assess population sizes within the Celtic Seas marine region.

Besides providing information necessary for reporting obligations at an EU level, the results presented here should be placed in a wider geographical and ecological context, aimed at inferring species dynamics. The factors causing the detected variations vary among species. The reasons behind the decrease in harbour porpoise abundance are not fully understood, but likely consist of a combination of factors exerting a synergistic impact including, but not limited to, bycatch, interspecific competition, overfishing and ecosystem mediated processes as suggested by a shift in diet (Deaville & Jepson, 2007; Deaville, 2020; Carlén et al., 2021; Taylor et al., 2022). Several studies have reported that the presence of harbour porpoise decreases with increasing SST (Wingfield et al., 2017; Rekdahl et al., 2023; Goh et al., 2025), likely mediated through shifts in prey distribution (Goh et al., 2025).

Although linking our results with any of these factors is beyond the scope of this work, we suggest that any management actions use an integrative approach to understand the cumulative impact of human pressures on this species (Likens & Lindenmayer, 2012). Given the decline in abundance of harbour porpoise highlighted in this study, any activity planned in this area should be planned carefully and mitigation measures such as the ones suggested by Virgili et al. (2024) considered to minimise any added impact on the species.

In contrast, changes in abundance of common and bottlenose dolphins suggest broad-scale movements within the northeast Atlantic and wider areas. Therefore, any conservation measures regarding these species must consider the wider ecological context. Ongoing environmental changes and human pressures are reshaping marine ecosystems worldwide (Maureaud et al., 2017, Gårdmark & Huss, 2020), and in the northeast Atlantic this is affecting all trophic levels and distribution shifts in marine top predators such as cetaceans have been documented (Maureaud et al., 2017; Evans & Waggitt, 2020; Hernvann & Gascuel, 2020; van Weelden et al., 2021; Williamson et al., 2021). These aspects make the assessment of the conservation needs of these species challenging, as change may occur over short periods of time (Notarbartolo di Sciara et al., 2016). For instance, bycatch has been the major concern for common dolphin conservation in the northeast Atlantic in recent decades as highlighted by several studies (Rogan & Mackey, 2007; Breen et al., 2017; Peltier et al., 2021; Rouby et al., 2022; Taylor et al., 2022). However, given the context of global change in the north Atlantic, the spatial overlap of common dolphins and fishing activities is likely to change in the future.

Density surface models provided a better understanding of how the changes in some species abundance translates into spatiotemporal changes in distribution, particularly for harbour porpoise and bottlenose dolphin. However, there are still some processes that remain unclear as they take place at a scale larger than the one covered in this study. Therefore, species distribution models at a larger scale, for instance, the northeast Atlantic region, could help elucidate the drivers behind these fluctuations in species abundance. Furthermore, these models could be used to evaluate the spatial and temporal overlap with human activities, to make predictions about the distribution of cetacean species in relation to topographical, physical and biogeochemical variables, to understand the current threats to the different species, highlight areas of importance and anticipate potential future impacts. The study also highlights the need for regular species abundance and distribution assessments, and the use of adaptive conservation and management measures that can respond to rapid changes in conservation needs.

CRediT authorship contribution statement

Oriol Giralt Paradell: Writing – original draft, Visualization, Investigation, Data curation, Conceptualization. **Ashley Bennison:** Writing – review & editing, Investigation, Data curation. **Meike Scheidat:** Writing – review & editing, Funding acquisition. **Mick Mackey:** Writing – review & editing, Investigation. **Helder Araújo:** Writing – review & editing, Investigation. **Steve C.V. Geelhoed:** Writing – review & editing, Investigation. **Dimitar Popov:** Writing – review & editing, Investigation. **Patricia Breen:** Writing – review & editing, Investigation, Data curation. **Mark Jessopp:** Writing – review & editing, Supervision, Investigation, Funding acquisition, Conceptualization. **Ana Cañadas:** Writing – review & editing, Funding acquisition, Formal analysis. **Emer Rogan:** Writing – review & editing, Supervision, Investigation, Funding acquisition, Conceptualization.

Funding

This study was funded by the Department of Environment, Climate and Communications (DECC) in partnership with the Department of Housing, Local Government and Heritage (DHLGH) and the Sustainable Energy Authority of Ireland (SEAI).

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.

Acknowledgements

The ObSERVE Programme is an Irish Government initiative developed by the Department of Environment, Climate and Communications (DECC) in partnership with the Department of Housing, Local Government and Heritage (DHLGH) and the Sustainable Energy Authority of Ireland (SEAI), and the authors are extremely grateful for the funding and guidance from the project Working Group. We are extremely grateful to the pilots and engineers from Aerosotravia and Action Air Environnement, and the dedicated team of observers that contributed throughout the programme. Finally, we would like to thank the three anonymous reviewers, the comments of which greatly improved this manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2025.114467.

Data availability

Data will be made available on request.

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