Assessing variability in marine traffic exposure between baleen whale species off the Galician Coast, Spain

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Abstract

Increases in marine traffic represent a growing issue for marine wildlife, posing threats through the impacts of ship strikes and noise pollution. Baleen whales are especially vulnerable to these impacts, yet regional and species-specific information on exposure to such threats is lacking. This study uses AIS and observational data to provide the first assessment of baleen whale exposure to vessel traffic on the NW coast of Spain. Overlap with vessel traffic was detected for all areas where whales were sighted, indicating that these species may be at risk of vessel exposure and its associated impacts. Level of exposure to vessel traffic experienced by whales was species-specific, with risk of exposure appearing highest for minke whales. Vessel exposure also displayed intra- and inter-annual variability and a significant influence of feeding behaviour highlighting the need for dynamic management tools to minimise interactions between baleen whales and marine traffic off the Galician Coast.

Keywords: Minke whale, Blue whale, Fin whale, Vessel exposure, Iberian Peninsula, Automatic identification systems

Introduction

As marine megafauna and top predators, whales have important roles in ecosystem engineering and food web stability (Bowen 1997; Smith 2007; Roman et al. 2014). Reductions in their populations may therefore cause changes to the structure and functioning of marine ecosystems (Worm et al. 2006; Ballance et al. 2007). However, marine environments are increasingly placed under pressures from both offshore and land-based activities, with humans now representing the largest driver of environmental change (Worm et al. 2006; Halpern et al. 2008; Halpern et al. 2015). Human activities have already caused whale populations to suffer detrimental impacts, namely the large-scale commercial whaling which overharvested many populations to the brink of extinction during the 20th century (Clapham et al. 1999; Thomas et al. 2016). These populations are still recovering, with one quarter of whale species listed on the IUCN Red List classified as Critically Endangered, Endangered or Vulnerable, including the blue (*Balaenoptera musculus*), fin (*B. physalus*) and North Atlantic right whale (*Eubalaena glacialis*) (Cooke 2018a, b, 2020). Many species have the potential to return to historical abundances (Kareiva et al. 2007): grey whales

(*Eschrichtius robustus*), for example, are thought to have returned to pre-whaling abundances within the Eastern Pacific (Alter et al. 2007). However, human exploitation of marine ecosystems for their goods and services through fisheries, commercial shipping, and the operation of polluting industries pose both lethal and sub-lethal threats to recovering populations (Clapham et al. 1999; Thomas et al. 2016). Vessel traffic in particular represents a growing threat due to the increases in seaborne trade to meet the demands of the growing global human population (Tournadre 2014; Pirotta et al. 2019).

Vessel traffic is one of the largest threats currently facing whales through the impacts of noise pollution and ship strikes (Laist et al. 2001; Thomas et al. 2016). As marine mammals that rely on sound as their primary sensory modality, whales use acoustic cues for critical life functions including navigation, communication with conspecifics, and for prey and predator detection (Clark et al. 2009; Williams et al. 2013; Cholewick et al. 2018). Much research effort has therefore focussed on the impacts of noise pollution. Whilst high amplitude sounds created by military sonar and seismic surveys have received considerable attention (Tyack et al. 2011; Goldbogen et al. 2013), commercial shipping represents the largest source of anthropogenic noise into the marine environment (Ellison et al. 2011; Williams et al. 2013). In addition, commercial shipping is the greatest contributor of anthropogenic noise of lower frequencies (20 - 200 Hz; Ross 1976), overlapping with those utilised by baleen whales (Mysticetes), low-frequency specialists, making this group particularly vulnerable (Clark et al. 2009).

This additional noise input into the marine soundscape has raised concerns of inducing physiological responses such as chronic stress, acoustic masking of important biological cues as well as the displacement of animals from critical habitat (Clark et al. 2009; Pirotta et al. 2012; Rolland et al. 2012; Williams et al. 2014; Holt et al. 2021). For example, reductions in glucocorticoid levels, a hormone linked to physiological stress, were observed in right whales within the Bay of Fundy, Canada, in association with a reduction of vessel traffic (Rolland et al. 2012). Clark et al. (2009) also discussed the potential impacts of acoustic masking in whales, with fin whale songs much less evident in the Mediterranean Sea, an area with higher shipping activity, compared to those singing in the Gulf of California, highlighting spatial differences in the distribution of vessel traffic impacts across the globe.

The largest anthropogenic source of whale mortality is attributed to ship strikes, with collisions causing direct mortality through blunt force trauma and propeller cuts, and indirectly by reducing fitness through severe injury (Laist et al. 2001; Soldevilla et al. 2017). Baleen whales, as some of the largest species, have been highlighted as those most frequently involved in these collisions (Laist et al. 2001; Schoeman et al. 2020). High mortality of blue, fin and humpback whales (*Megaptera novaeangliae*) due to vessel collisions, for example, have been reported for populations along the West Coast of the US and in the Mediterranean Sea (Berman-Kowalewski et al. 2010; Rockwood et al. 2017). The frequency and severity of vessel collisions have been found to be influenced by many factors including whale surface active behaviour, speed and size of vessels, density of vessel traffic and the extent of overlap in the spatiotemporal distribution of whale species and vessel traffic (Vanderlaan and Taggart 2007; Conn and Silber 2013; Soldevilla et al. 2017). Whales engaged in feeding activities, for example, have been reported to be less alert to surrounding noise and activities, including vessel traffic (Laist et al. 2001). Few assessments however of vessel exposure report on the influence of more than one of the above factors.

As demand for seaborne trade continues and a greater number of ships and those of greater size may become present in the oceans, there is a growing importance in evaluating the exposure of whale species to vessel traffic (Halpern et al. 2015; Pirotta et al. 2019). While spatiotemporal overlap does not confirm animals will be impacted, it is a necessary precursor for impact to occur and to identify species and areas of risk. Assessments of many cetacean species have been undertaken but are spatially biased towards the Mediterranean Sea and the coasts of North America, and in most cases are based only on a single year of shipping traffic data with cetacean presence averaged over time (Erbe et al. 2014; Pennino et al. 2017; Rockwood et al. 2017, for exceptions see Redfern et al. 2013; 2019; 2020 and Abrahms et al. 2019). This does not account for temporal variability in both vessel traffic and whale presence, factors which are fundamental to the development of effective management strategies for the protection of species and to minimise wildlife-user conflict (Redfern et al. 2020).

While the negative impacts of ship strike and noise pollution associated with marine traffic are well established, species-specific exposure to these threats remain unevaluated in many regions (Laist et al. 2001; Thomas et al. 2016; Erbe et al. 2019). The continental shelf in Galician waters (North-Western Spain), an area highly exposed to human activities and a historically significant whaling area, has been identified as a region frequently utilised by endangered rorqual whales for foraging (Sanpera and Aguilar 1992; Díaz López and Methion 2019), with confirmed year-round presence of minke whales as well as a high seasonality of blue and fin whale presence (Díaz López and Methion 2019). Some whales are known to migrate from the Azores however the specific migratory route of most individuals identified is unknown (Díaz López et al. 2022). Interactions between cetaceans and vessels have also been recorded in Galicia but research efforts have focussed on smaller cetaceans such as bottlenose dolphins (*Tursiops truncatus*) and their interactions with fisheries (Goetz et al. 2014; Díaz López and Methion 2018, Díaz López et al. 2019; Giralt Paradell et al. 2021). Here, we aim to address this gap by developing the first analysis of marine traffic exposure for baleen whales off the Galician Coast. Specifically, vessel types and density within the study region were identified and their spatiotemporal overlap compared with the presence of three key species: the blue whale, fin whale, and minke whale (*Balaenoptera acutorotrata*). We used 4 years of data to explore differences in exposure to marine traffic between different whale species.Generalised linear mixed models (GLMMs) were used to analyse whether whale species, group size, behaviour, season, or year were related to vessel density in the areas where whales were sighted. The results of this study will therefore provide useful insight for the identification of conservation priorities and may inform marine management strategies within the study area that can be designed with temporal differences of exposure risk in mind.

Materials and methods

Study area

This study was undertaken along the North-Western coast of the Iberian Peninsula, specifically the Galician coast, encompassing the continental shelf and inshore waters from Muros (42.79^{O} N, 9.25^{O} W) to the Cíes Islands (42.36º N, 8.94º W) (Fig. 1). Galicia lies along the northern edge of the eastern boundary of the North Atlantic upwelling system, one of the major upwelling systems of the world (Wooster et al. 1976). Seasonal upwelling of cold and nutrient-rich North Atlantic Central Water situated at depths of 70 – 500 m results in this oceanic region being one of the most biologically productive in the world (Blanton et al. 1984; Figueiras et al. 2002; Spyrakos et al. 2011). The productivity of the Galician coast is reflected in its high biodiversity, with at least 300 fish species, 80 cephalopod species and 20 cetacean species recorded in the area (Guerra 1992; Spyrakos et al. 2011). Whale presence has been noted in previous studies to peak at the end of summer and beginning of autumn due to higher concentrations of zooplankton produced by the seasonal upwelling events (Díaz López and Methion 2019; Methion and Díaz López 2019; Díaz López et al. 2021).

Data Collection

Presence data for baleen whales was collected by the Bottlenose Dolphin Research Institute BDRI (www.thebdri.com) as part of their long-term studies on the ecology and behaviour of cetacean species within Galician waters. Boat-based surveys were conducted year-round onboard a 12 m research vessel during daylight hours at a constant speed of 6 – 8 knots. Surveys were conducted systematically along transect lines designed to cover the study area equally, adapted to the specific conditions of the study area and the meteorological conditions of each day (Díaz López and Methion 2019). Spatial distribution of effort could vary according to weather conditions and time constraints throughout the study period, as surveys were conducted only when visibility was not reduced by rain or fog and when sea conditions were no greater than three on the Douglas Sea Force scale (Díaz López and Methion 2018).

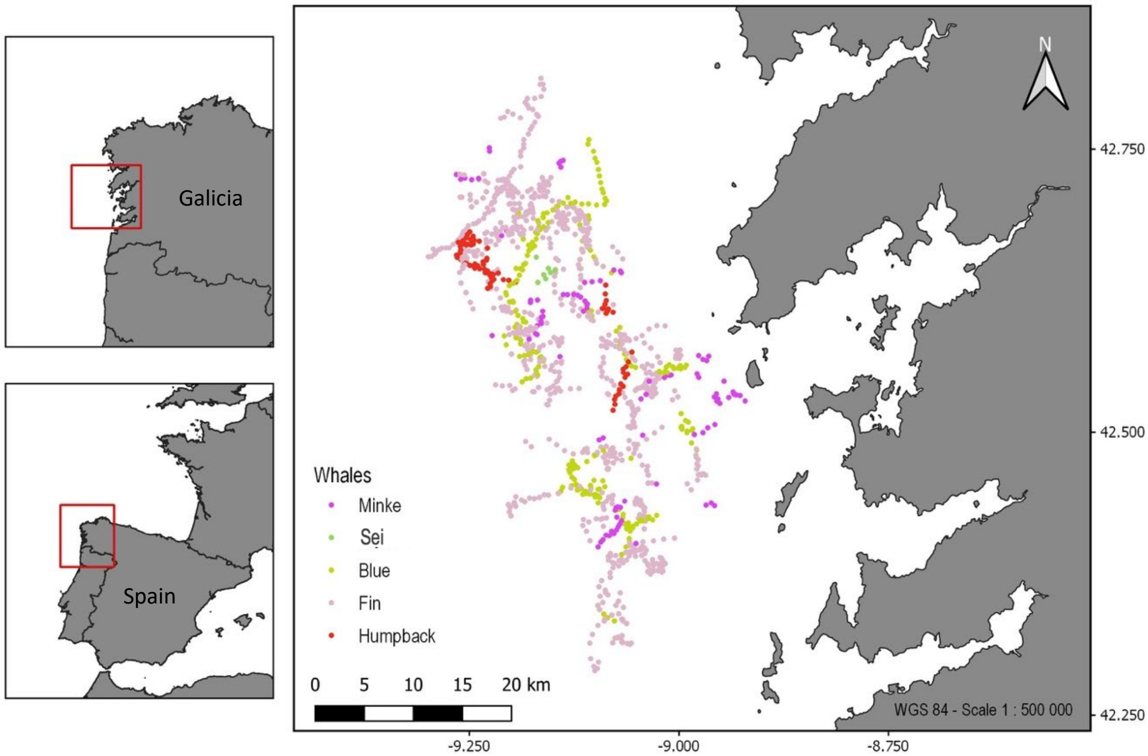

Fig. 1. Map of the study area (southern coast of Galicia, Spain) showing the distribution of the observed baleen whales between 2017 and 2020 (points indicate the position of instantaneous point samples recorded every 5 minutes of each sighting).

At least three experienced observers were stationed on the flying bridge of the vessel (4 m above sea level) at all times, scanning 360 degrees of the sea surface for presence of whales with the naked eye or with 10 x 50 binoculars (Díaz López and Methion 2019). Upon sighting a whale, the vessel slowly moved towards the individual or group to reduce disturbance during the approach (Díaz López and Methion 2018). A sighting was defined as one or more whales of the same species observed within 1 nautical mile radius engaged in the same behavioural activity (Díaz López and Methion 2019). Group size was estimated at the beginning of the observation and confirmed throughout the sighting. The species (identified by colour patterns, head shape, size and the shape of the dorsal fin), date, time, and geographical position (UTM longitude and UTM latitude: WGS 84 UTM Zone 29N) were recorded as an instantaneous point sample every five minutes. The behavioural state (feeding or not feeding) of the individual or group was determined at the end of each five-minute sample. Whales were considered to be feeding when swimming in different directions in the same area (lunge feeding: mainly observed in blue and fin whales; characterised by high-speed, vertical lunges in which the animal opened its mouth and distended the gular region a few metres from the water surface, turning on itself and showing the ventral region at the surface; or deep feeding: characterised by sequences of regular dives followed by long and steep dives (tail-stock or flukes-up dives) (Díaz

López et al. 2021). Samples were collected until the individual/group was lost or weather became unfavourable and the sighting was ended.

Presence data from 2017 to 2020 were selected to temporally match available vessel density data (see below), giving a total of 1800 samples from 187 sightings of 5 species (blue, fin, minke, humpback and sei whale: *Balaenoptera borealis*) over a total of 19 months. Presence data for sei and humpback whales were excluded from the analysis due to the low frequency of sightings (n = 2 and n = 4, respectively) and similarly, a small number of samples collected during the winter months were also excluded (n = 4).

Vessel density

Automatic identification systems (AIS) are tracking systems used on-board ships and by coastal authorities, capable of providing information on vessel identification, location and speed (IMO 2019). All passenger ships, ships ≥300 gross tonnage engaged in international voyages, and cargo ships ≥500 gross tonnage engaged in domestic voyages are required to be fitted with AIS under the SOLAS regulation V/19, and AIS transponders are carried voluntarily on many other vessel types (IMO 2019). AIS is therefore an appropriate data source for vessel density and has been utilised in previous studies assessing variability in ship traffic and its impacts on cetaceans (Moore et al. 2018; Redfern et al. 2020; Smith et al. 2020; Silber et al. 2021).

In the present study vessel data were obtained from the European Marine Observation and Data Network (EMODnet) (https://www.emodnet-humanactivities.eu/). Vessel density maps based on AIS data covering all EU waters and some neighbouring areas were available for the following vessel categories: fishing, service, dredging or underwater operations, sailing, pleasure craft, high speed craft, tug and towing, passenger, cargo, tanker, military and law enforcement, other, unknown, and all. Raster GIS files (GeoTIFF) of vessel density maps, available by month of year from 2017 to 2020, were downloaded and imported into a geographical information system (QGIS 3.16) and projected into the ETRS89/ETRS-LAEA coordinate reference system (CRS) (ESPG: 3035). Shipping density was shown in 1 x 1 km cells and expressed as hours per square kilometre per month.

To determine the spatiotemporal overlap between whale presence and vessel density, whale presence point samples were first imported into QGIS 3.16 as shapefiles, grouped by sightings per month of year of each species. Samples were then reprojected to the ESPG:3035 CRS to ensure all layers were in the same reference system for spatial analysis. Using the QGIS Python Point Sampling Tool Plugin, vessel density was extracted for each point sample, selecting the vessel density raster layer to probe values from that which corresponded with the month of year of the sighting. The extracted value therefore corresponded with the total number hours of vessel traffic within the 1 x 1 km grid cell in which the whale was sighted, in the month and year it was sighted in. While this value is not the real-time presence of boats during the sighting, this approach was taken to represent the general marine traffic activity within the areas that blue, fin and minke whales were sighted in.

Factors including the size, speed and noise produced by a vessel have been reported to influence whale presence (Campana et al. 2017; Blondin et al., 2020; Schoeman et al. 2020), but differences in these characteristics exist between vessel types, thus presenting different risks on exposure to whales. Due to the negligible contribution of many vessel types to marine traffic in the study area, only four vessel categories were used in further analysis: fishing boats, sailing boats, large vessels and all (representing the total number of hours of traffic from all vessel types combined). Vessel density values from 'dredging or underwater operations', 'cargo' and 'tanker' were aggregated to create the category of 'large' because of the risks to cetaceans associated specifically with larger vessels as reported in the literature (Laist et al. 2001; Thomas et al. 2016; Schoeman et al. 2020).

Statistical Analysis

All statistical analyses were conducted in R version 4.0.4 (R Core Team 2021). Differences in exposure of whales to vessel traffic were analysed using general linear mixed models (GLMMs) in the 'nlme' package (Pinheiro et al. 2021). Specifically, GLMMs were used to analyse whether whale species, group size, behaviour, season, or year were related to vessel density in the areas where whales were sighted. Vessel density of 'all' vessel types for point samples was fitted as the response variable. These data were slightly skewed and so were log transformed; this successfully produced a normal distribution. Four variables were initially considered to have potential effects on whale exposure to vessel traffic and so were fitted as fixed effects in the initial model: species of whale, group size, behaviour and season. Analysis was therefore restricted to point samples for which all of the above data were available ($n = 1294$). Season indicates upwelling and post-upwelling periods, with upwelling events occurring between May and August and post-upwelling occurring between September and November (Díaz López and Methion 2019; Díaz López and Methion 2021). Overlap between vessel traffic and cetacean feeding grounds has been reported in previous studies so to explore the effect of feeding behaviour on exposure to vessel traffic, behaviour was defined as "feeding" or "not feeding" (Goetz et al. 2014; Cruz et al. 2016; Díaz López et al. 2019; Ricci et al. 2021). To explore species-specific differences in vessel density in the areas that they were sighted in, species and its interactions with group size, behaviour and season were also fitted as fixed effects in the initial model. Year was fitted as a fixed effect to directly investigate whether the risk of exposure differed between years. To control for nonindependence of consecutive point samples within a sighting of the same individual or group, each sighting was given an identifier with sighting identity then fitted as a random effect.

Collinearity between explanatory variables was assessed by examining correlations and variance inflation factors (VIFs) prior to model selection, following Zuur et al. (2009). All variables were included in the initial model because correlations between them were weak ($r < 0.4$) with small associated VIFs (< 3). The model was then refined using backwards stepwise deletion. Explanatory variables were removed sequentially in order of increasing test statistic value if likelihood test ratios showed that they did not explain any significant variation. Each variable was assessed in turn until the minimal model was obtained and then reinstated into the model to confirm significant terms had not been inappropriately excluded and to determine the degree of nonsignificance. The final model included only significant variables and was validated by plotting the distribution of the residuals and the residuals against the fitted values, following Zuur et al. (2009). Estimates and standard error values for variables with multiple comparisons (species and year) were obtained in the 'multcomp' package (Hothorn et al. 2008).

Species-specific variability in vessel traffic exposure

To investigate variability in the type of vessel traffic (fishing, sailboats and/or large vessels) a species was exposed to and whether differences existed between years, species-specific two-way ANOVA tests were conducted. When a significant F value was identified, a Tukey's Honest Significant Difference test was run to find where the significant differences lay. Only the first sample of each sighting were selected to limit autocorrelation and pseudoreplication arising from consecutive samples (fin: n = 89; minke: n = 24). Analysis of minke whales were restricted to 2018 – 2020 due to the low representation of sightings of this species during 2017 (n = 4). Low sighting numbers of blue whales in 2018 and 2019 (n = 4 and n = 2, respectively) excluded this species from two-way ANOVA analysis, however a one-way ANOVA was conducted to investigate the variability in the type of vessel traffic this species was exposed to.

Results

Spatiotemporal overlap of whale presence and vessel traffic

Between 2017 and 2020, 132 sightings of blue, fin and minke whales were made along the study area (Table 1). The distribution of marine traffic varied both spatially and temporally within the study area, with coastal regions generally showing higher vessel densities than areas further offshore (see Fig. 2 for example).

Table 1. Number of sightings and individuals observed of each species during surveys between 2017 and 2020.

Species		***Year***				
		2017	**2018**	**2019**	**2020**	Total
Minke	Sightings	5	10	5	6	**26**
	Individuals*	7	12	5	7	**31**
Fin	Sightings	22	12	12	33	**79**
	Individuals*	36	22	15	136	**209**
Blue	Sightings	5	4	2	16	**27**
	Individuals*	5	4	2	24	**35**

*** The total number of individuals observed of the same species is the sum of the number of whales observed on each day without taking into account that some individuals may have been sighted on multiple different days.**

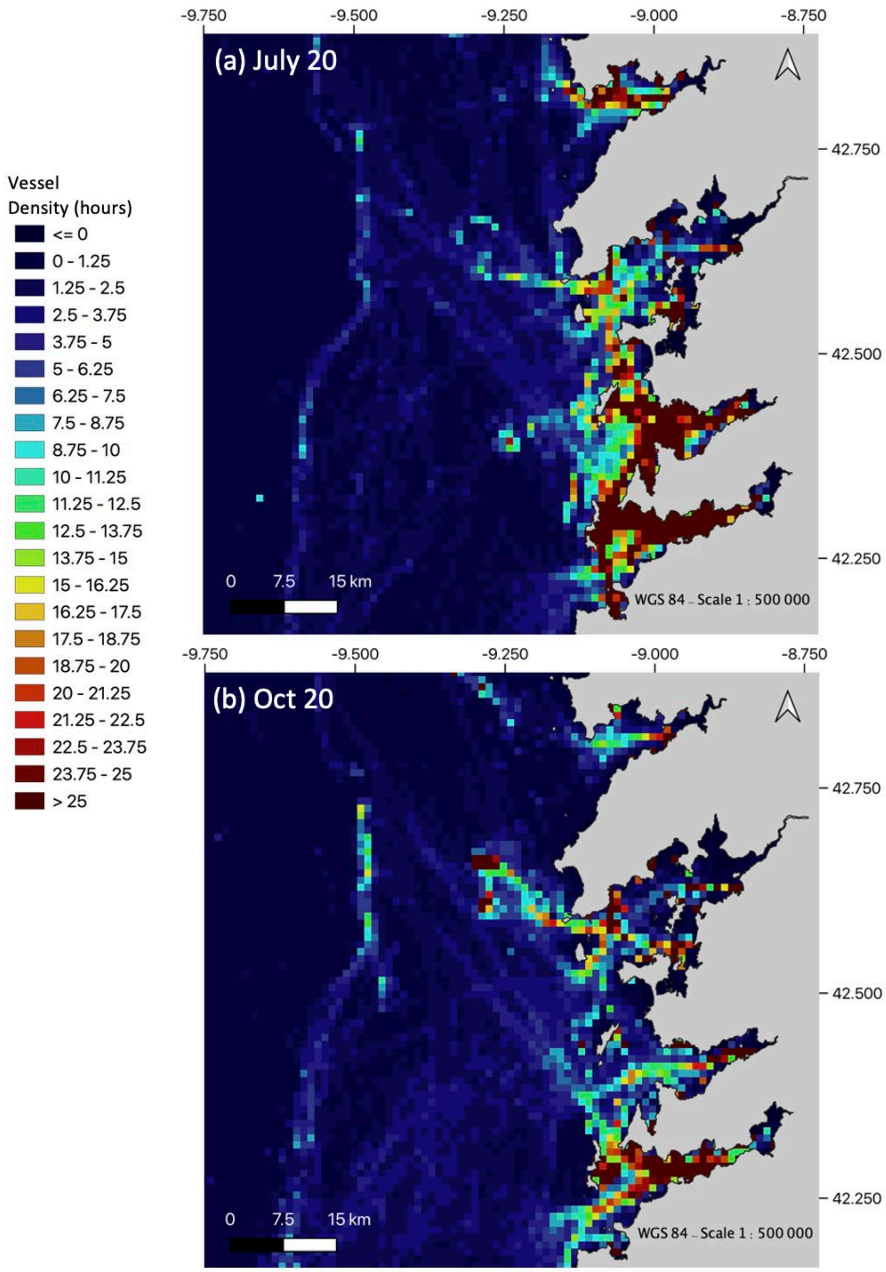

Fig. 2. Maps of vessel density within the study area during (a) summer and (b) autumn 2020. Vessel density of all vessel types of July and October 2020 were selected for illustrative purposes. Vessel density is shown in 1 x 1 km grid cells, expressed as hours per square kilometer per month.

All point samples (n = 1294) used in the analysis (representing the position of at least one individual every 5 minutes) were found in areas where the monthly density of marine traffic was greater than zero when considering all vessel types, confirming spatiotemporal overlap of whale presence and vessel traffic for minke, blue and fin whales within the study area in each year analysed (see Fig. 3 for example).

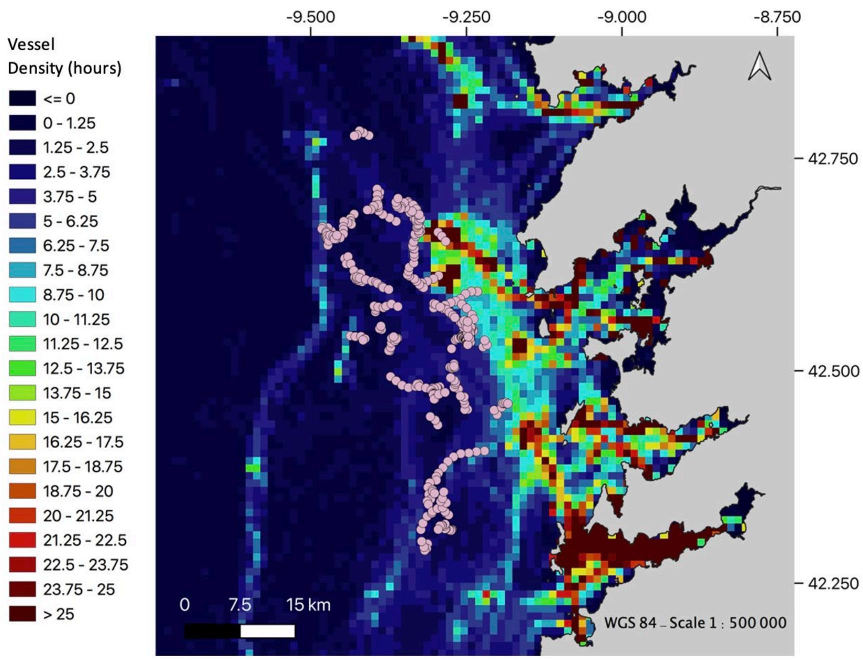

Fig. 3. Map of fin whale exposure to vessel traffic within the study region. Vessel density of all vessel types is shown for this species, the most frequently sighted in the area, during a high-risk month in the most recent year of available data (September 2020) to highlight spatiotemporal overlap of whale presence and vessel traffic. Pink circles indicate fin whale presence (instantaneous point samples) and vessel density is shown in 1 x 1 km grid cells, expressed as hours per square kilometer per month.

Factors influencing whale exposure to vessel traffic

Exposure to vessel traffic varied significantly between species (Table 2). For areas minke whales were sighted in, vessel density was significantly higher than for those where blue whales and fin whales were sighted (Table 1), with mean monthly vessel density for minke whale point samples 5.208 ± 0.333 hours compared to 3.400 ± 0.087 and 3.286 ± 0.062 hours for blue and fin whales respectively. The difference in vessel density between areas where fin whales and blue whales were sighted was found to be nonsignificant and no interactions between species and other variables were found to have a significant effect (Table 1). A significant relationship was, however, found between season and vessel traffic in areas where whales were sighted (Table 1). For each species, individuals were found in areas of higher vessel density in autumn than during summer. Feeding individuals were also sighted in areas of significantly higher vessel density than those not feeding (Table 1). Vessel density in the areas where whales were sighted also varied significantly with years (Table 1; Fig. 4). Specifically, whales were found in areas of significantly higher vessel density in 2017 compared to both 2019 and 2020, and in 2018 compared to 2019 (Table 3). All other pairwise comparisons between years were found to be nonsignificant (Table 3). Vessel density in the areas where whales were sighted did not vary significantly with group size (Table 3).

Table 2. Percentage of samples of presence of each whale species in areas with a density of marine traffic over 5 hours per month.

Species	*> 5 hours/month*
Minke whale	39.1 %
Fin whale	13.4 %
Blue whale	17.3 %

Table 3. The results of the minimal adequate general linear mixed model (GLMM) of the factors influencing whale exposure to vessel traffic. Analyses were restricted to sampling points for which complete data were available (n = 1294).

Random effect	*Variance*		
Sighting identity	0.521		
Residual	0.320		
Fixed effects	*Estimate ± SE*	*x^2 / z-value**	*P***
Intercept	2.074 ± 0.133	–	–
Species	–	11.172	**0.004**
Minke v Blue	- 0.485 ± 0.159	- 3.059	**0.006**
Minke v Fin	- 0.399 ± 0.132	- 3.019	**0.007**
Fin v Blue	0.086 ± 0.122	0.702	0.759
Season (Autumn v Summer)	- 0.601 ± 0.115	26.133	**< 0.001**
Behaviour (Feeding v Not Feeding)	- 0.190 ± 0.096	4.011	**0.045**
Group size	–	1.516	0.218
Year	–	77.116	**< 0.001**
2018 – 2017	- 0.149 ± 0.144	- 1.041	0.711
2019 – 2017	- 0.602 ± 0.778	- 7.740	**< 0.001**
2020 – 2017	- 0.434 ± 0.108	- 4.003	**< 0.001**
2019 – 2018	- 0.452 ± 0.128	- 3.540	**0.002**
2020 – 2018	- 0.284 ± 0.153	- 1.852	0.236
2020 – 2019	0.168 ± 0.121	1.395	0.485
Species: season	–	0.531	0.767
Species: behaviour	–	0.145	0.930
Species: group size	–	0.910	0.923

* Chi-squared values are given for the overall effects of species, season, behaviour and year. Z-values are given for the pairwise comparisons between species and across years.
** Significant values ($p < 0.05$) derived from likelihood ratio tests and Tukey's Honest Significance tests are highlighted in bold.

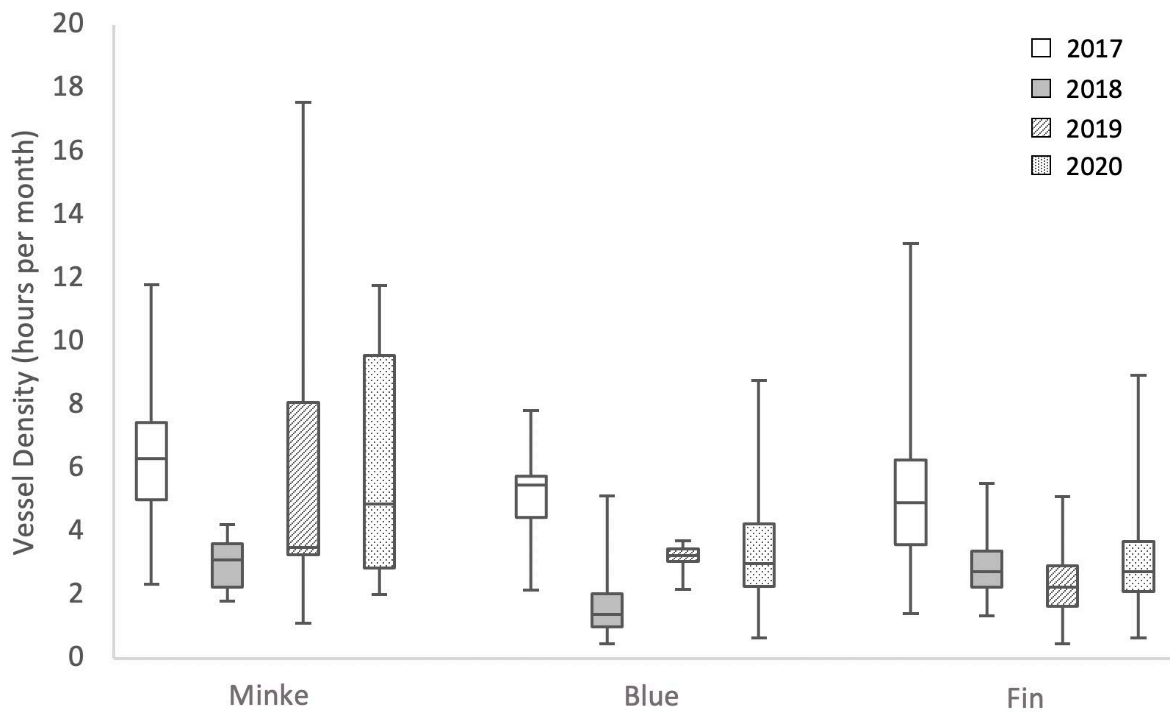

Fig. 4. The effect of year on species-specific exposure to vessel density. Box and whisker plots illustrate the median, interquartile range, lower and upper quartiles, and minimum and maximum values of vessel density in the areas where each species was sighted, each year from 2017 – 2020.

Variability in species exposure to different vessel types

Minke whales

No statistically significant interactive effect was detected between vessel type and year on vessel density in areas where minke whales were sighted (ANOVA, $F(4, 51) = 2.096$; $p = 0.095$). However, the effect of vessel type on mean vessel density in areas where minke whales were sighted was found to be significant (ANOVA, $F(4, 51) = 17.328$, $p < 0.001$; Fig. 5). Specifically, fishing vessels contributed a significantly greater proportion to the total vessel density in areas where minke whales were sighted compared to large vessels ($p < 0.001$) and sailboats ($p < 0.001$; Fig. 5). The difference between mean vessel density of sailboats and large vessels was nonsignificant ($p = 0.996$). The effect of year on mean vessel density was found to be significant (ANOVA, $F(4, 51) = 3.754$, $p =$

0.030; Fig. 5); vessel density in areas where minke whales were sighted in was significantly different in 2018 compared to 2020 (p = 0.029).

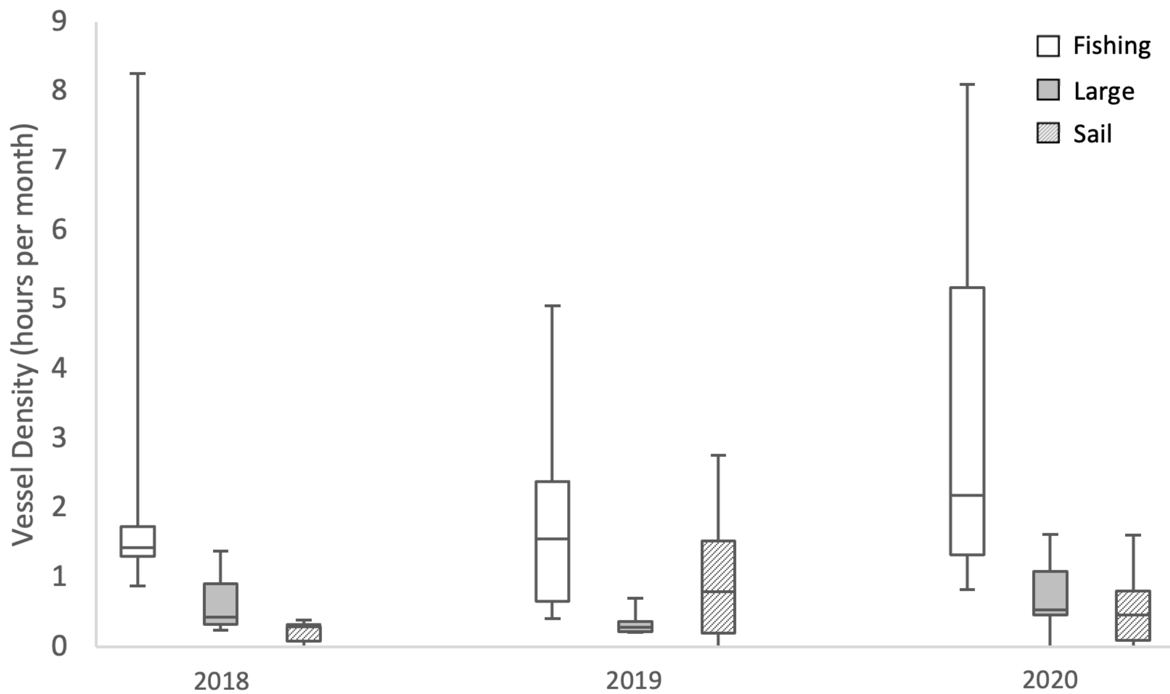

Fig. 5. Minke whale exposure to different vessel types. Box and whisker plots illustrate the median, interquartile range, lower and upper quartiles, and minimum and maximum values of vessel density of fishing vessels, large vessels and sailboats in the areas where minke whales were sighted each year between 2018 and 2020.

Fin whales

A statistically significant interaction between vessel type and year on mean vessel density in the areas that fin whales were sighted in was found (ANOVA, $F (6, 255) = 2.962$; $p = 0.008$; Fig. 6). Specifically, fishing vessel density in areas where fin whales were sighted during 2017 was significantly higher than in 2019 ($p = 0.015$). This vessel type also contributed a significantly greater proportion to the total vessel density in areas fin whales were sighted compared to both large vessels ($p < 0.001$) and sailboats ($p < 0.001$) in all years analysed (Fig. 6). There was no evidence that vessel density of large vessels or sailboats in areas where fin whales were sighted differed significantly between years, or that this species experienced greater exposure to large vessels compared to sailboats in any one year, and vice versa.

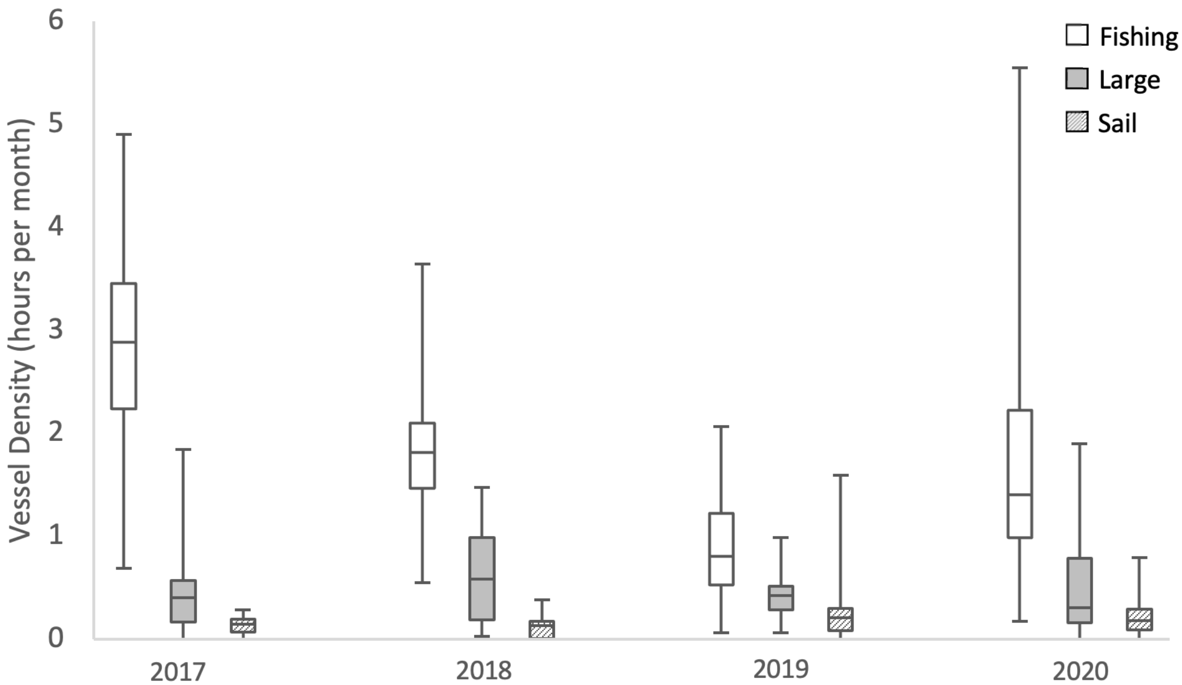

Fig. 6. Fin whale exposure to different vessel types. Box and whisker plots illustrate the median, interquartile range, lower and upper quartiles, and minimum and maximum values of vessel density of fishing vessels, large vessels and sailboats in the areas where fin whales were sighted each year from 2017 to 2020.

Blue whales

A statistically significant difference was detected between the mean vessel density of difference vessel types in the areas where blue whales were sighted in (ANOVA, $F (2, 84) = 53.64$; $p < 0.001$; Fig. 7); fishing vessels contributed a significantly greater proportion to the total vessel density than large vessels ($p < 0.001$) and sailboats ($p < 0.001$).

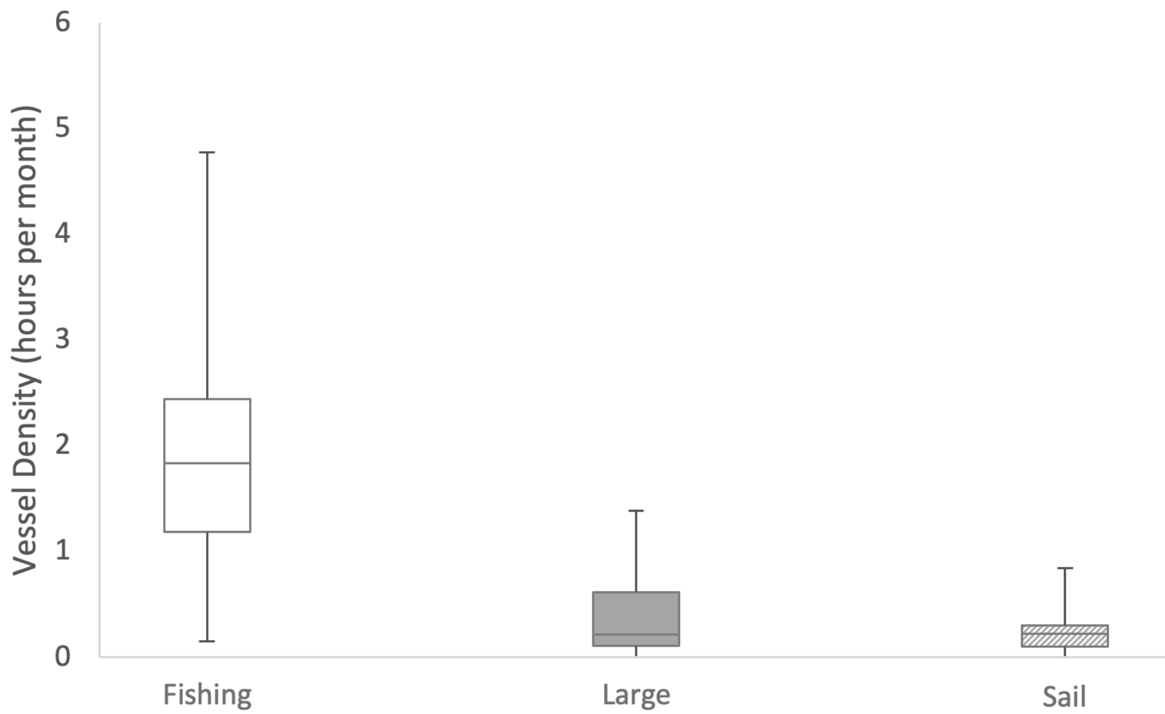

Fig. 7. Blue whale exposure to different vessel types. Box and whisker plots illustrate the median, interquartile range, lower and upper quartiles, and minimum and maximum values of vessel density of fishing vessels, large vessels and sailboats in the areas where blue whales were sighted between 2017 and 2020.

Discussion

The potential impacts of vessel traffic to marine megafauna have been extensively noted, including the effects of noise pollution (Clark et al. 2009; Erbe et al. 2019), ship strike (Laist et al. 2001; Schoeman et al. 2020) and vessel presence itself (Williams et al. 2006). Distribution of these threats is spatially and temporally heterogenous across the globe, making assessments of exposure critical for the identification of high-risk regions and species to inform conservation priorities (Clapham et al. 1999; Thomas et al. 2016). Yet, information on regional and species-specific threat exposure is lacking. This is likely due to the inherent challenges associated with the collection of distribution data on animals which spend the majority of their lives underwater, often relying on opportunistic cetacean sightings (Pompa et al. 2011; Silber et al. 2017). This study contributes to the growing literature on cetacean threat exposure by providing the first assessment of marine traffic exposure off the Galician coast to three baleen whale species.

Spatiotemporal overlap of whale presence and vessel traffic off the Galician coast confirms vessel traffic presents a real risk within the region, with blue, fin and minke whales all sighted in areas utilised by vessels on a monthly basis. The level of exposure to vessel traffic experienced by individuals in the study area displayed inter-annual variability and appears to be species-specific and significantly related to feeding behaviour and season.

Risk of exposure within the study region appears to be highest for minke whales, with this species sighted in areas with significantly higher monthly vessel density compared to both blue and fin whales. Sightings of this species during the study period included those in regions much closer to the coast than those of blue and fin whales, consistent with higher levels of vessel traffic within the study area. The use of inshore and coastal waters by minke whales has also been noted in previous studies (Northridge et al. 1995; Weir et al. 2007; Lee et al. 2017), indicating that the higher exposure of this species to vessel traffic in the study region may be the result of species-specific habitat preferences. This is also supported by the general preference of blue and fin whales for deep offshore waters (Azzellino et al. 2008; Panigada et al. 2008; Andriolo et al. 2010; Díaz López and Methion 2019), in which vessel density in the study area is typically lower.

Previous studies have suggested human-caused disturbances such as vessel presence or noise may be perceived as a form of predation risk (Frid and Dill 2002) and whales may therefore respond to vessel presence with avoidance and anti-predatory behaviours (Pirotta et al. 2015). Blue whales, for example, have been observed to alter diving behaviour in response to approaching large vessels (McKenna et al. 2015) and Campana et al. (2015; 2017) found significantly lower vessel traffic present in areas where cetaceans, including fin whales, were sighted in the Western Mediterranean Sea compared to areas where they were not sighted. Campana et al. (2015) discuss this may represent a negative response to traffic activity, where whales may avoid more highly trafficked regions with small or large scale displacements or increased dive activity. Other studies report a lack of response of cetaceans to approaching marine traffic, specifically fishing boats characterised by slow movements (Díaz López et al. 2008). Short-term behavioural changes have also been associated with vessel presence (Williams et al. 2006; Castellote et al. 2012; Pirotta et al. 2015; Dahlheim and Castellote 2016). Williams et al. (2006) noted for example that killer whales (*Orcinus orca*) within the Johnstone Straight of British Columbia were more likely to change their behavioural state in the presence of a vessel. This disruption raises concerns about population-level impacts of long-term vessel presence within Galicia if consistent interruption of foraging and feeding activities leads to reduced feeding opportunities and ultimately lower energetic acquisition (Williams et al. 2006; Blair et al. 2016). Previous studies have however reported that cetaceans engaged in feeding activities are less alert to their surroundings, including vessel traffic (Laist et al. 2001), with Campana et al. (2015; 2017) discussing co-existence of whales and vessels may be driven by ecological requirements, i.e. where favourable feeding grounds overlap with higher traffic pressure, indicating there may be a trade-off depending on the extent of risk perceived of vessel presence by the whale.

It should be noted that an inherent limitation of studies based on observational presence data is that the differences observed between species may be because individuals were simply not sighted during dedicated surveys, as opposed to species-specific differences in the areas that they utilise. This is a particular challenge for determining presence/absence of cetacean species, which spend most of their lives under the water surface where they are not visible. Future studies should aim to use spatial distribution models with presence/absence data for each whale species associated with multiple environmental and anthropogenic variables (including marine traffic) to determine habitat suitability.

Foraging often involves greater surface-active behaviour in whales, a factor noted to increase the likelihood of physical interaction with vessels, and thus injury or mortality resulting from collision (Parks et al. 2012; Crum et al. 2019). Previous studies have also found that when engaged in activities critical for survival, such as feeding, cetaceans are more likely to remain in their behavioural state, allowing vessels to approach more closely before they respond (Schuler et al. 2019; Bubac et al. 2020). This indicates that not only are feeding whales at risk of higher exposure to vessel traffic, but also at greater risk of collision. Baleen whales are often observed feeding at the surface in the study area (Díaz López et al. 2021). Surface feeding, a type of lunge feeding mainly observed in blue whales and fin whales (Díaz López et al. 2021), is a dynamic, unsteady, and unpredictable process, and so must be considered an important factor to consider as a potential risk for whales in the presence of vessels.

Seasonal changes in prey abundance are thought to drive blue and fin whale distribution specifically within Galician waters (Díaz López and Methion 2019). Baleen whales are opportunistic predators that feed on diverse prey including euphausiids (Friedlaender et al. 2006, 2015) such as northern krill (*Meganyctiphanes norvegica*), which aggregate in dense patches in Galician waters following the phytoplankton bloom, with a time lag of several weeks (Bode et al. 2009; Visser et al. 2011). Characteristics of the upwelling regime and the temporal synchrony of whales with their prey therefore leads to whale presence peaking at the beginning of autumn within the study region (Díaz López and Methion 2019, Díaz López et al 2021). As biological productivity off the Iberian Peninsula peaks after seasonal upwelling events (Blanton et al 1984), it is likely that fishing vessel presence also increases in autumn to temporally match increases in the abundance of target resources. This may explain why a significant effect of season on vessel exposure was detected: all whale species studied were sighted in areas with significantly higher monthly vessel traffic in autumn than during summer. The influence of season on exposure to vessel traffic was analysed only for summer and autumn due to low representation of sightings during other seasons, however the limited number of sightings during winter and spring appears to reflect the northward feeding migration pattern observed in rorqual whales from southerly grounds between July and October within the study region (Díaz López and Methion 2019).

It should be noted that in this study it is not possible to know whether the spatial distribution of whales is due to a different distribution of prey, with minke whales displaying greater preference for crustaceans and plankton compared to blue whales and fin whales which feed on krill (Friedlaender et al. 2006, 2015; Skaug et al. 1997), or whether it could be conditioned by vessel avoidance, a factor future studies should aim to consider.

Seasonal differences in vessel exposure to baleen whales have been noted in other regions of the world. In the Mediterranean Sea, for example, risk of exposure is typically reported to be highest during summer months (Panigada et al. 2006; Campana et al. 2017; Pennino et al. 2017). This increased risk during summer can generally be attributed to increases in passenger boats during warmer months (Schuler et al. 2019), although the ecotourism industry appears to be less evident along the Galician coast (Díaz López and Methion 2019). The importance of Galicia's fishing activity has, however, been commented on in previous studies (Cambiè et al. 2012; Surís-Regueiro and Santiago 2014), indicating that the significant seasonal variation in minke, fin and blue whale exposure to vessel traffic is likely due to changes in fishing vessel operations to correspond with peaks in commercially targeted species' abundance and distribution. Fishing vessels were the most common vessel type found in the areas where whales were sighted: a trend that applied across all species, years and seasons analysed. As Galicia is one of the most important fishing regions within Europe with the largest Spanish fishing fleet (Galician Institute for Statistics 2021), this finding was not unexpected but raises additional concerns of the anthropogenic threats posed to these species e.g., through fisheries bycatch (Surís-Regueiro and Santiago 2014).

Frequent cetacean-fishery interactions have been reported within Galicia and other regions of the Iberian Peninsula (Goetz et al. 2014), including bycaught long-finned pilot whales (*Globicephala melas*), blue whales and fin whales (López et al. 2003; Aguilar and Borrell 2022). Actual encounters between large cetacean species and fishing vessels may be low, with Goetz et al. (2014) reporting that baleen whales were sighted in <1% of all cetacean sightings by 283 Galician fishermen. However, interactions are known to occur, for example a fin whale was recently observed with a fishing net caught around its head within Galician waters (Díaz López and Methion 2019) and another fin whale observed with a fishing line around its body (Methion and Díaz López 2019). Interactions between vessels and cetaceans also frequently go unreported (Neilson et al. 2012; Peel et al. 2018; Schoeman et al. 2020) and it would thus be pre-emptive at this stage to underestimate the potential impact of fishing vessels within the region.

Significant inter-annual variability in whale exposure to vessel traffic was detected, with whales sighted in 2019 and 2020 generally found in areas with lower vessel traffic compared to those sighted in 2017 and 2018. Variability in vessel exposure each year may be the result of avoidance behaviours, for example if whales actively avoided areas of higher traffic in more instances during one year compared to another. However, as discussed earlier, limited literature on avoidance behaviours of whales exists and the significant inter-annual variability in vessel traffic exposure to minke, blue and fin whales in this study is most fully explained by the variation in the abundance of fishing boats, the most common vessel type in the region.

Almost all vessel types have been reported as involved in ship strikes around the world (Laist et al. 2001), but as risk of collision is proportional to the size of the vessel and its speed, the involvement of larger vessels (>80 m) is more frequently observed (Gende et al. 2019; Schoeman et al. 2020). Injuries sustained by larger vessels are also thought to carry a greater risk of fatality because of higher force on impact (Laist et al. 2001; Moore et al. 2013). Sailing vessels have also been highlighted as a growing threat to cetaceans (Ritter, 2012). While significantly lower presence of larger vessels and sailboats was found in areas where whales were sighted compared to fishing vessels, encounters may still occur, presenting ship-strike risk within the study area to the three study species as well as additional noise in their soundscape (Clark et al. 2009; Erbe et al. 2019).

Intrinsic sensitivity to vessel noise will differ with each species' unique ecology, however larger vessels typically generate sound at frequencies below 1000 Hz (Ross 1976), overlapping with those utilised and perceived by baleen whales (Clark et al. 2009). Minke, blue and fin whales may therefore be vulnerable to the impacts of acoustic masking off the Galician coast (Clark et al. 2009). Modelling noise pollution produced from vessel traffic in our study area was beyond the scope of our paper, however future studies should aim to characterise the acoustic environment of the study region as well as identify effective communication ranges of our three study species to investigate the extent to which their active space (the range over which the animals are able to communicate) is reduced in vessel presence (e.g. Merchant et al. 2014; Cholewiak et al. 2018).

Implications for conservation and management

Whales of the suborder Mysticeti are among some of the largest living animals in the world. These charismatic species, often deemed as ambassadors for marine conservation, hold important ecological roles in top-down regulation and the transference of nutrients and biomass (Bowen 1997; Smith 2007; Roman et al. 2014). Blue and fin whales are, however, classified as Endangered and Vulnerable respectively, in part owing to legacy effects of previous human persecution (Clapham et al. 1999; IUCN 2008) and are now faced with the threat of several other anthropogenic pressures (Halpern et al. 2008, 2015; Albouy et al. 2020).

The present study contributes to the growing body of literature detailing exposure to anthropogenic pressures, confirming a considerable spatiotemporal overlap of vessel traffic, particularly of fishing vessels, with endangered baleen whale species off the southern coast of Galicia (Díaz López and Methion 2019). This raises concerns not only of the exposure of minke, fin and blue whales within Galician waters to the impacts of ship strike and noise pollution (Laist et al. 2001; Clark et al. 2009; Cholewiak et al. 2018) but also to the risks associated with fisheries such as gear entanglement and bycatch (Van der Hoop et al. 2014; Giralt Paradell et al. 2021). It should also be noted that due to the nature of this study, results indicate the minimum spatiotemporal overlap between whales and marine traffic, as vessel density values represent the minimum monthly vessel traffic within the areas that whales were sighted. This is because while many vessels are legally required to carry and transmit AIS, not all carry this requirement and thus not all traffic will be represented within our dataset (IMO 2019). The potential threat of marine traffic off the Galician coast could therefore be much greater than presented.

There is a lack of information on ship collisions within the region, perhaps due do unreported interactions discussed earlier (Schoeman et al. 2020), however mortality due to fishery interactions have been recorded (López et al. 2003; Goetz et al. 2014; Aguilar and Borrell 2022). The accumulation of single mortality events arising from human-induced disturbance can result in population-level impacts in long-lived, k-selected species such as baleen whales (Rockwood et al. 2017; Díaz López and Methion 2019), making implementation of mitigation strategies a priority in regions where exposure to such disturbances has been noted.

The importance of the conservation of these species within Europe is recognised by their protection under Annex IV of the EU Habitats Directive (Council Directive 92/43/EEC). Special protection of minke, blue and fin whales (amongst other cetacean species) is given in Spanish waters under the Royal Decree 1727/2007, offering partial mitigation to the impacts of recreational vessels and whale-watching activities in Galician waters through the implementation of regulations such as reduced speed limits when approaching individuals, as well as exclusion zones which prohibit vessels from intentionally navigating within a 60 m radius of an observed cetacean. However, the protection measures in current Spanish legislation do not appear to help reduce the risk, as they do not focus on large vessels and fishing boats and appear to be static and non-specific. The results of the present study highlight that risk of vessel exposure to baleen whales off the Galician coast is dynamic, varying both temporally and between species, offering insight for management authorities to develop mitigation strategies which consider these differences, i.e., through the adoption of additional seasonal regulations when risk is greatest. Speed limits have been introduced in other highly trafficked regions of the world and proved effective at reducing ship-related mortality (Freedman et al. 2017; Joy et al. 2019), with several studies demonstrating that risk of mortality increases with vessel speed (Vanderlaan and Taggart 2007; Conn and Silber 2013). Introducing lower speed limits during summer months when coexistence of cetacean and vessel traffic is at its greatest should be considered.

Conclusions

Minke, blue and fin whales off the southern Galician coast may be at risk of vessel exposure and its associated impacts based on considerable spatial overlap between vessel traffic and whale distribution from AIS data and observations of whale presence over a four-year period. While these species are offered some protection within Spanish waters, avoidance behaviours of cetaceans and reductions in habitat quality due to vessel presence noted in other regions of the world (Nowacek et al. 2001; Williams et al. 2006, 2013; McKenna et al. 2015) indicates further protective legislation within Galicia may be required to ensure the continued utilisation of their waters as foraging ground for endangered whale species (Díaz López and Methion 2019). The results of the present study provide useful insight for the identification of conservation priorities within the region, highlighting minke whales are exposed to a higher level of marine traffic, likely due to the spatial overlap between the distribution of this species with fishing activities. However, despite lower levels of exposure of fin and blue whales to marine traffic, this should not be mistaken as low risk, because as larger species they are more vulnerable to vessel collisions and sighted in areas utilised by large vessels in the study area. Our findings can therefore inform marine management strategies to be designed with the consideration of temporal and species-specific variation in vessel exposure and risk.

Assessments of exposure to anthropogenic threats will continue to play an important role in identifying species and areas of conservation concern, however, vulnerability of individuals to anthropogenic stressors is influenced not only by their spatiotemporal overlap but also the intrinsic sensitivity of a species e.g., to vessel noise. Future studies should therefore aim to characterise species-specific sensitivity to such threats. In doing so, a greater understanding of the impacts of human activities on populations can be gained, which is critical to minimise wildlife-user conflict and prevent local and regional extinctions.

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Author Contributions

Rhian Bland: Conceptualization, Writing – Original Draft, Formal analysis, Review & Editing. Séverine Methion: Conceptualization, Investigation, Data curation, Writing – Review & Editing, Project administration, Funding acquisition. Stuart Sharp: Formal analysis, Review & Editing. Bruno Díaz López: Conceptualization, Investigation, Data curation, Methodology, Software, Formal analysis, Writing – Review & Editing, Supervision, Funding acquisition.

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Data Availability Data will be provided under request.

Code Availability R Script will be provided under request.

Declarations

Competing Interests The authors declare that they have no conflicts of interest.

Ethics approval Data collection complied with the current laws of Spain, the country in which it was performed.

Consent to participate All authors gave final approval to participate.

Consent to publish All authors gave final approval for publication.

References

Abrahms B, Welch H, Brodie S, Jacox MG, Becker EA, Bograd SJ, Irvine LM, Palacios DM, Mate BR, Hazen EL (2019) Dynamic ensemble models to predict distributions and anthropogenic risk exposure for highly mobile species. Divers Distrib 25:1182–1193. https://doi.org/10.1111/ddi.12940

Aguilar A, Borrell A (2022) Unreported catches, impact of whaling and current status of blue whales in the South European Atlantic Shelf. Sci Rep 12(5491). https://doi.org/10.1038/s41598-022-09570-6

Albouy C, Delattre V, Donati G, Frölicher TL, Albouy-Boyer S, Rufino M, Pellissier L, Mouillot D, Leprieur F (2020) Global vulnerability of marine mammals to global warming. Sci Rep 10(1):548. https://doi.org/10.1038/s41598-019-57280-3

Alter SE, Rynes E, Palumbi SR (2007) DNA Evidence for Historic Population Size and past Ecosystem Impacts of Gray Whales. Proc Natl Acad Sci 104(38):15162–15167. https://doi.org/10.1073/pnas.0706056104

Altmann J (1974) Observational Study of Behavior: Sampling Methods. Behaviour 49(3-4):227–266.

Andriolo A, da Rocha JM, Zerbini AN, Simões-Lopes PC, Moreno IB, Lucena A, Danilewicz D, Bassoi M (2010) Distribution and relative abundance of large whales in a former whaling ground off eastern South America. Zoologia (Curitiba) 27(5):741–750. https://doi.org/10.1590/S1984-46702010000500011

Azzellino A, Gaspari S, Airoldi S, Nani B (2008) Habitat use and preferences of cetaceans along the continental slope and the adjacent pelagic waters in the western Ligurian Sea. Deep Sea Res Part I Oceanogr Res 55(3):296–323. https://doi.org/10.1016/j.dsr.2007.11.006

Ballance LT, Pitman RL, Hewitt RP, Siniff DB, Trivelpiece WZ, Clapham PJ, Brownell RL (2007) The Removal Of Large Whales From The Southern Ocean Evidence For Long-Term Ecosystem Effects? In: Estes JA, Demaster DP, Doak DF, Williams TM, Brownell RL (eds) Whales, Whaling, and Ocean Ecosystems*,* 1st edn. University of California Press, Berkeley, pp 215-230

Berman-Kowalewski M, Gulland FMD, Wilkin S, Calambokidis J, Mate B, Cordaro J, Rotstein D, St Leger J, Collins P, Fahy K, Dover S (2010) Association Between Blue Whale (*Balaenoptera musculus*) Mortality and Ship Strikes Along the California Coast. Aquat Mamm 36(1):59–66. https://doi.org/10.1578/AM.36.1.2010.59

Blair H, Merchant N, Friedlaender A, Wiley D, Parks S (2016) Evidence for ship noise impacts on humpback whale foraging behaviour. Biol Lett 12(8):20160005. https://doi.org/10.1098/rsbl.2016.0005

Blanton JO, Atkinson LP, de Castillejo FF, Montero AL (1984) Coastal upwelling off the Rias Bajas, Galicia, northwest Spain I: Hydrographic studies. Rapports et procès-verbaux des réunions: The biological productivity of North Atlantic Shelf areas 183:79-90.

Blondin H, Abrahms B, Crowder LB, Hazen EL (2020). Combining high temporal resolution whale distribution and vessel tracking data improves estimates of ship strike risk. Biological Conservation, 250, 108757.

Bode A, Alvarez-Ossorio MT, Cabanas JM, Miranda A, Varela M (2009) Recent trends in plankton and upwelling intensity off Galicia (NW Spain). Prog Oceanogr 83(1-4):342–350. https://doi.org/10.1016/j.pocean.2009.07.025

Bowen WD (1997) Role of marine mammals in aquatic ecosystems. Mar Ecol Prog Ser 58:267-274

Bubac CM, Johnson AC, Otis R (2020) Surface behaviors correlate with prey abundance and vessels in an endangered killer whale (*Orcinus orca*) population. Marine ecology 42(1):e12626 https://doi.org/10.1111/maec.12626

Cambiè G, Ouréns R, Vidal D, Carabel S, Freire J (2012) Economic performance of coastal fisheries in Galicia (NW Spain): case study of the Cíes Islands. Aquat Living Resour 25(2):195-204. https://doi.org/10.1051/alr/2012010

Campana I, Angeletti D, Crosti R, Luperini C, Ruvolo A, Alessandrini A, Arcangeli A (2017) Seasonal characterisation of maritime traffic and the relationship with cetacean presence in the Western Mediterranean Sea. Mar Pollut Bull 115(1-2):282–291. https://doi.org/10.1016/j.marpolbul.2016.12.008

Campana I, Crosti R, Angeletti D, Carosso L, David L, Di-Méglio N, Moulins A, Rosso M, Tepsich P, Arcangeli A (2015) Cetacean response to summer maritime traffic in the western Mediterranean Sea. *Mar Environ Res* 109:1–8. https://doi.org/10.1016/j.marenvres.2015.05.009.

Carlucci R, Capezzuto F, Cipriano G, D'Onghia G, Fanizza C, Libralato S, Maglietta R, Maiorano P, Sion L, Tursi A, Ricci P (2021) Assessment of cetacean–fishery interactions in the marine food web of the Gulf of Taranto (Northern Ionian Sea, Central Mediterranean Sea). Rev Fish Biol Fisheries 31(1):135–156. https://doi.org/10.1007/s11160-020-09623-x

Castellote M, Clark CW, Lammers MO (2012) Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. Biol Conserv 147(1):115–122. https://doi.org/10.1016/j.biocon.2011.12.021

Cholewiak D, Clark CW, Ponirakis D, Frankel A, Hatch LT, Risch D, Stanistreet JE, Thompson M, Vu E, Van Parijs SM (2018) Communicating amidst the noise: modeling the aggregate influence of ambient and vessel noise on baleen whale communication space in a national marine sanctuary. Endanger Species Res 36:59–75. https://doi.org/10.3354/esr00875

Clapham PJ, Young SB, Brownell Jr RL (1999) Baleen whales: conservation issues and the status of the most endangered populations. Mamm Rev 29(1):37–62. https://doi.org/10.1046/j.1365-2907.1999.00035.x

Clark CW, Ellison WT, Southall BL, Hatch L, Van Parijs SM, Frankel A, Ponirakis D (2009) Acoustic masking in marine ecosystems. Mar Ecol Prog Ser 395:201–222. https://doi.org/10.3354/meps08402

Conn PB, Silber GK (2013) Vessel speed restrictions reduce risk of collision-related mortality for North Atlantic right whales. Ecosphere 4(4):43–16. https://doi.org/10.1890/ES13-00004.1

Cooke JG (2018a) *Balaenoptera musculus*. The IUCN Red List of Threatened Species 2018: e.T2477A156923585

Cooke JG (2018b) *Balaenoptera physalus*. The IUCN Red List of Threatened Species 2018: e.T2478A50349982

Cooke JG (2020) *Eubalaena glacialis*. The IUCN Red List of Threatened Species 2020: e.T41712A178589687

Council Directive 92/43/EEC of 21 May 1992 on the conservation of wild fauna and flora. https://eur-lex.europa.eu/eli/dir/1992/43/oj. Accessed 27 August 2021

Crum N, Gowan T, Krzystan A, Martin J (2019) Quantifying risk of whale–vessel collisions across space, time, and management policies. Ecosphere 10(4):e02713. https://doi.org/10.1002/ecs2.2713

Cruz M, Menezes G, Machete M, Silva M (2016) Predicting interactions between common dolphins and the pole-and-line tuna fishery in the Azores. PloS one 11(11):e0164107. https://doi.org/10.1371/journal.pone.0164107

Curé C, Antunes R, Alves A, Visser F, Kvadsheim P, Miller P (2013) Responses of male sperm whales (*Physeter macrocephalus*) to killer whale sounds: Implications for anti-predator strategies. Sci Rep 3(1):1579. https://doi.org/10.1038/srep01579

Dahlheim M, Castellote M (2016) Changes in the acoustic behavior of gray whales *Eschrichtius robustus* in response to noise. Endanger Species Res 31:227–242. https://doi.org/10.3354/esr00759

Díaz López B, Methion S (2018) Does interspecific competition drive patterns of habitat use and relative density in harbour porpoises? Mar Biol 165(5):1–11. https://doi.org/10.1007/s00227-018-3345-8

Díaz López B, Methion S (2019) Habitat drivers of endangered rorqual whales in a highly impacted upwelling region. Ecol Indic 103:610–616. https://doi.org/10.1016/j.ecolind.2019.04.038

Díaz López B, Methion S, Giralt Paradell O (2019) Living on the edge: Overlap between a marine predator's habitat use and fisheries in the Northeast Atlantic waters (NW Spain). Prog Oceanogr 175:115–123. https://doi.org/10.1016/j.pocean.2019.04.004

Díaz López B, Methion S, Giralt Paradell O, Mosca O (2021) Proyecto BALAENATUR: seguimiento del rorcual azul en la red Natura 2000. Bottlenose Dolphin Research Institute, O Grove. pp51.

Díaz López B, Methion S, Mosca O (2022) Proyecto BALAENATUR:ballenas azules y cambio climático dentro de la Red Natura 2000. Bottlenose Dolphin Research Institute, O Grove. pp36.

Díaz López B, Shirai J, Prieto A, Fernández P (2008) Diving activity of a solitary wild free ranging bottlenose dolphin (*Tursiops truncatus*). J Mar Biolog Assoc UK 88(6):1153-1157. https://doi.org/10.1017/S0025315408000921

Elgar M (1989) PREDATOR VIGILANCE AND GROUP SIZE IN MAMMALS AND BIRDS: A CRITICAL REVIEW OF THE EMPIRICAL EVIDENCE. Biol Rev 64(1):13-33. https://doi.org/10.1111/j.1469-185X.1989.tb00636.x

Ellison WT, Southall BL, Clark CW, Frankel, AS (2011) A New Context-Based Approach to Assess Marine Mammal Behavioral Responses to Anthropogenic Sounds. Conserv Biol 26(1):21–28. https://doi.org/10.1111/j.1523-1739.2011.01803.x

EMODnet (2021) *Vessel Density – EMODet Human Activities.* https://www.emodnet-humanactivities.eu/search-results.php?dataname=Vessel+Density. Accessed 7 July 2021

Erbe C, Marley S, Schoeman R, Smith J, Trigg L, Embling C (2019) The Effects of Ship Noise on Marine Mammals—A Review. Front Mar Sci 6:606. https://doi.org/10.3389/fmars.2019.00606

Erbe C, Williams R, Sandilands D, Ashe E (2014) Identifying modeled ship noise hotspots for marine mammals of Canada's Pacific region. PloS one 9(3):e89820. https://doi.org/10.1371/journal.pone.0089820

Figueiras FG, Labarta U, Fernández Reiriz MJ (2002) Coastal upwelling, primary production and mussel growth in the Rías Baixas of Galicia. Hydrobiologia 484(1):121–131. https://doi.org/10.1023/A:1021309222459

Freedman R, Herron S, Byrd M, Birney K, Morten J, Shafritz B, Caldow C, Hastings S (2017) The effectiveness of incentivized and non-incentivized vessel speed reduction programs: Case study in the Santa Barbara channel. Ocean Coast Manag 148:31–39. https://doi.org/10.1016/j.ocecoaman.2017.07.013

Frid A, Dill L (2002) Human-caused Disturbance Stimuli as a Form of Predation Risk. Conserv Ecol 6(1):11. https://doi.org/10.5751/ES-00404-060111

Friedlaender A, Goldbogen J, Hazen E, Calambokidis J, Southall B (2015) Feeding performance by sympatric blue and fin whales exploiting a common prey resource. Mar Mamm Sci 31(1):345–354. https://doi.org/10.1111/mms.12134

Friedlaender A, Halpin P, Qian S, Lawson G, Wiebe P, Thiele D, Read A (2006) Whale distribution in relation to prey abundance and oceanographic processes in shelf waters of the Western Antarctic Peninsula. Mar Ecol Prog Ser 317:297–310. https://doi.org/10.3354/meps317297

Galician Institute for Statistics (2021) *IGE - Pesca e acuicultura*. http://www.ige.eu/web/mostrar_actividade_estatistica.jsp?idioma=gl&codigo=0301004. Accessed 3 August 2021

Gende S, Vose L, Baken J, Gabriele C, Preston R, Hendrix A (2019) Active Whale Avoidance by Large Ships: Components and Constraints of a Complementary Approach to Reducing Ship Strike Risk. Front Mar Sci 6:592. https://doi.org/10.3389/fmars.2019.00592

Giralt Paradell O, Methion S, Rogan E, Díaz López B (2021) Modelling ecosystem dynamics to assess the effect of coastal fisheries on cetacean species. J Environ Manage 285:112175. https://doi.org/10.1016/j.jenvman.2021.112175

Goetz S, Read F, Santos M, Pita C, Pierce G (2014) Cetacean–fishery interactions in Galicia (NW Spain): results and management implications of a face-to-face interview survey of local fishers. ICES J Mar Sci 71(3):604–617. https://doi.org/10.1093/icesjms/fst149

Goldbogen JA, Southall BL, DeRuiter SL, Calambokidis J, Friedlaender AS, Hazen EL, Falcone EA, Schorr GS, Douglas A, Moretti DJ, Kyburg C, McKenna MF, Tyack PL (2013) Blue whales respond to simulated mid-frequency military sonar. Proc Royal Soc B 280(1765):20130657. https://doi.org/10.1098/rspb.2013.0657

Guerra A (1992) Mollusca, Cephalopoda. Fauna Ibérica, Vol. 1. Museo Nacional de Ciencias Naturales, Madrid

Halpern BS, Walbridge S, Fujita R, Heinemann D, Lenihan HS, Madin EMP, Perry MT, Selig ER, Spalding M, Steneck R, Watson R, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE (2008) A Global Map of Human Impact on Marine Ecosystems. Science 319(5865):948–952. https://doi.org/10.1126/science.1149345

Halpern BS, Frazier M, Potapenko J, Casey KS, Koenig K, Longo C, Lowndes J, Rockwood R, Cotton S, Elizabeth R, Selkoe KA, Walbridge S (2015) Spatial and temporal changes in cumulative human impacts on the world's ocean. Nat Commun 6(1):7615. https://doi.org/10.1038/ncomms8615

Hildebrand JA (2009) Anthropogenic and natural sources of ambient noise in the Ocean. Mar Ecol Prog Ser 395:5–20. https://doi.org/10.3354/meps08353

Holt M, Noren D, Veirs V, Emmons C, Veirs S (2008) Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. J Acoust Soc Am 125(1):EL27-32. https://doi.org/10.1121/1.3040028

Holt M, Tennessen J, Hanson M, Emmons C, Giles D, Hogan J, Ford M (2021) Vessels and their sounds reduce prey capture effort by endangered killer whales (Orcinus orca). Mar Environ Res 170:105429. https://doi.org/10.1016/j.marenvres.2021.105429

Hothorn T, Bretz F, Westfall P (2008) Simultaneous Inference in General Parametric Models. Biom J 50(3):346-363. https://doi.org/10.1002/bimj.200810425

IMO (2019) *AIS transponders* https://www.imo.org/en/OurWork/Safety/Pages/AIS.aspx. Accessed 22 July 2021

IUCN (2008) Status of the World's Marine Species. https://www.iucn.org/downloads/status_of_the_world_s_marine_species_factsheet_en.pdf. Accessed 7 August 2021

Janson CH, Goldsmith ML (1995) Predicting group size in primates: foraging costs and predation risks. Behav Ecol 6(3):326–336. https://doi.org/10.1093/beheco/6.3.326

Joy R, Tollit D, Wood J, MacGillivray A, Li Z, Trounce K, Robinson O (2019) Potential Benefits of Vessel Slowdowns on Endangered Southern Resident Killer Whales. Front Mar Sci 6:344 https://doi.org/10.3389/fmars.2019.00344

Kareiva P, Yuan-Farrell C, O'Connor C (2007) Whales Are Big and It Matters. In: Estes JA, Demaster DP, Doak DF, Williams TM, Brownell RL (eds) Whales, Whaling, and Ocean Ecosystems, 1st edn. University of California Press, Berkeley, pp 379-387

Skaug H, Gjosæter H, Haug T, Nilssen K, Lindstrøm U (1997) Do Minke Whales (*Balaenoptera acutorostrata*) Exhibit Particular Prey Preferences?. J Northwest Atl Fish Sci 22: 91-104. https://doi.org/10.2960/j.v22.a8

Laist D, Knowlton A, Mead J, Collet A, Podesta M (2001) Collisions Between Ships and Whales. Mar Mamm Sci 17(1):35–75. https://doi.org/10.1111/j.1748-7692.2001.tb00980.x

Lee D, An YR, Park KJ, Kim HW, Lee D, Joo HT, Oh YG, Kim SM, Kang CK, Lee SH (2017) Spatial distribution of common Minke whale (*Balaenoptera acutorostrata*) as an indication of a biological hotspot in the East Sea. Deep Sea Res Part II Top Stud Oceanogr 143:91–99. https://doi.org/10.1016/j.dsr2.2017.06.005

Lehtonen J, Jaatinen K (2016) Safety in numbers: the dilution effect and other drivers of group life in the face of danger. Behav Ecol Sociobiol 70(4):449–458. https://doi.org/10.1007/s00265-016-2075-5

López A, Pierce G, Santos M, Gracia J, Guerra A (2003) Fishery by-catches of marine mammals in Galician waters: results from on-board observations and an interview survey of fishermen. Biol Conserv 111(1):25–40. https://doi.org/10.1016/S0006-3207(02)00244-6

Mann J (1999) Behavioral sampling methods for cetaceans: a review and critique. Mar Mamm Sci 15(1):102–122. https://doi.org/10.1111/j.1748-7692.1999.tb00784.x

McKenna M, Calambokidis J, Oleson E, Laist D, Goldbogen J (2015) Simultaneous tracking of blue whales and large ships demonstrates limited behavioral responses for avoiding collision. Endanger Species Res 27(3):219–232. https://doi.org/10.3354/esr00666

Merchant ND, Pirotta E, Barton TR, Thompson PM (2014) Monitoring ship noise to assess the impact of coastal developments on marine mammals. Mar Pollut Bull 78(1-2):85–95. https://doi.org/10.1016/j.marpolbul.2013.10.058.

Methion S, Díaz López B (2019) Natural and anthropogenic drivers of foraging behaviour in bottlenose dolphins: Influence of shellfish aquaculture. Aquat Conserv 29(6):927–937. https://doi.org/10.1002/aqc.3116

Moore MJ, Van Der Hoop J, Barco SG, Costidis AM, Gulland FM, Jepson PD, Moore KT, Raverty S, McLellan WA (2013) Criteria and case definitions for serious injury and death of pinnipeds and cetaceans caused by anthropogenic trauma. Dis Aquat Organ 103(3):229–264. https://doi.org/10.3354/dao02566

Moore T, Redfern J, Carver M, Hastings S, Adams J, Silber G (2018) Exploring ship traffic variability off California. Ocean Coast Manag 163:515–527. https://doi.org/10.1016/j.ocecoaman.2018.03.010

Mul E, Blanchet M, McClintock B, Grecian W, Biuw M, Rikardsen A (2020) Killer whales are attracted to herring fishing vessels. Mar Ecol Prog Ser 652:1-13. https://doi.org/10.3354/meps13481

Neilson J, Gabriele C, Jensen A, Jackson K, Straley J (2012) Summary of Reported Whale-Vessel Collisions in Alaskan Waters. Journal of Marine Biology 2012:1–18.

Northridge S, Tasker M, Webb A, Williams J (1995) Distribution and relative abundance of harbour porpoises (*Phocoena phocoena* L.), white-beaked dolphins (*Lagenorhynchus albirostris* Gray), and minke whales (*Balaenoptera acutorostrata* Lacepède) around the British Isles. ICES J Mar Sci 52(1):55–66. https://doi.org/10.1016/1054-3139(95)80015-8

Nowacek DP, Johnson MP, Tyack PL (2004) North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. Proc Royal Soc B 271(1536):227–231. https://doi.org/10.1098/rspb.2003.2570

Nowacek SM, Wells RS, Solow AR (2001) Short-term effects of boat traffic on bottlenose dolphin, *Tursiops truncatus*, in Sarasota Bay, Florida. Mar Mamm Sci 17(4):673–688. https://doi.org/10.1111/j.1748-7692.2001.tb01292.x

O'Connell D (2005) Tuna, Dolphins, and Purse Seine Fishing in the Eastern Tropical Pacific: The Controversy Continues. UCLA Journal of Environmental Law and Policy 23(1).

Panigada S, Pesante G, Zanardelli M, Capoulade F, Gannier A, Weinrich M (2006) Mediterranean fin whales at risk from fatal ship strikes. Mar Pollut Bull 52(10):1287–1298. https://doi.org/10.1016/j.marpolbul.2006.03.014

Panigada S, Zanardelli M, MacKenzie M, Donovan C, Mélin F, Hammond P (2008) Modelling habitat preferences for fin whales and striped dolphins in the Pelagos Sanctuary (Western Mediterranean Sea) with physiographic and remote sensing variables. Remote Sens Environ 112(8):3400–3412. https://doi.org/10.1016/j.rse.2007.11.017

Parks S, Warren J, Stamieszkin K, Mayo C, Wiley D (2012) Dangerous dining: surface foraging of North Atlantic right whales increases risk of vessel collisions. Biol Lett 8(1):57–60. https://doi.org/10.1098/rsbl.2011.0578

Peel D, Smith JN, Childerhouse S (2018) Vessel strike of whales in Australia: The challenges of analysis of historical incident data. Front Mar Sci 5:69. https://doi.org/10.3389/fmars.2018.00069

Peng C, Zhao X. Liu G (2015) Noise in the sea and its impacts on marine organisms. IJERPH, 12(10):12304–12323. https://doi.org/10.3390/ijerph121012304.

Pennino M, Arcangeli A, Prado Fonseca V, Campana I, Pierce G, Rotta A, Bellido J (2017) A spatially explicit risk assessment approach: Cetaceans and marine traffic in the Pelagos Sanctuary (Mediterranean Sea). PloS one 12(6):e0179686. https://doi.org/10.1371/journal.pone.0179686

Pinheiro J, Bates D, DebRoy S, Sarkar D, R Core Team (2021) _nlme: Linear and Nonlinear Mixed Effects Models_. R package version 3.1-152. https://CRAN.R-project.org/package=nlme>

Pirotta E, Merchant N, Thompson P, Barton T, Lusseau D (2015) Quantifying the effect of boat disturbance on bottlenose dolphin foraging activity. Biol Conserv 181:82–89. https://doi.org/10.1016/j.biocon.2014.11.003

Pirotta E, Milor R, Quick N, Moretti D, Di Marzio N, Tyack P, Boyd I, Hastie G (2012) Vessel noise affects beaked whale behavior: results of a dedicated acoustic response study. PloS one 7(8):e42535. https://doi.org/10.1371/journal.pone.0042535

Pirotta V, Grech A, Jonsen ID, Laurance WF, Harcourt RG (2019) Consequences of global shipping traffic for marine giants. Front Ecol Environ 17(1):39–47. https://doi.org/10.1002/fee.1987

R Core Team (2021) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Redfern JV, McKenna MF, Moore TJ, Calambokidis J, DeAngelis ML, Becker EA, Barlow J, Forney KA, Fiedler PC, Chivers SJ (2013). Assessing the risk of ships striking large whales in marine spatial planning. Conserv Biol 27:292–302. https://doi.org/10.1111/cobi.12029

Redfern JV, Moore TJ, Becker EA, Calambokidis J, Hastings SP, Irvine LM, Mate BR, Palacios DM (2019) Evaluating stakeholder-derived strategies to reduce the risk of ships striking whales. Divers Distrib 25:1575–1585. https://doi.org/10.1111/ddi.12958

Redfern JV, Becker EA, Moore TJ (2020) Effects of Variability in Ship Traffic and Whale Distributions on the Risk of Ships Striking Whales. Front Mar Sci 6:793. https://doi.org/10.3389/fmars.2019.00793

Ricci P, Manea E, Cipriano G, Cascione D, D'Onghia G, Ingrosso M, Fanizza C, Maiorano P, Tursi A, Carlucci, R (2021) Addressing Cetacean–Fishery Interactions to Inform a Deep-Sea Ecosystem-Based Management in the Gulf of Taranto (Northern Ionian Sea, Central Mediterranean Sea). J Mar Sci Eng 9:872. https://doi.org/10.3390/jmse9080872

Ritter F (2012) Collisions of sailing vessels with cetaceans worldwide: first insights into a seemingly growing problem. J Cetacean Res Manag 12:119–127.

Roberts G (1996) Why individual vigilance declines as group size increases. Anim Behav 51(5):1077–1086. https://doi.org/10.1006/anbe.1996.0109

Rockwood RC, Calambokidis J, Jahncke J (2017) High mortality of blue, humpback and fin whales from modeling of vessel collisions on the U.S. West Coast suggests population impacts and insufficient protection. PloS one 12(8):e0183052. https://doi.org/10.1371/journal.pone.0183052

Rolland RM, Parks SE, Hunt KE, Castellote M, Corkeron PJ, Nowacek DP, Wasser SK, Kraus SD (2012) Evidence that ship noise increases stress in right whales. Proc Royal Soc B 279(1737):2363–2368. https://doi.org/10.1098/rspb.2011.2429

Roman J, Estes JA, Morissette J, Smith C, Costa D, McCarthy J, Nation JB, Nicol S, Pershing A, Smetacek V (2014) Whales as marine ecosystem engineers. Front Ecol Environ 12(7):377–385. https://doi.org/10.1890/130220

Ross D (1976) Mechanics of underwater noise. Pergamon Press, Oxford

Royal Decree 1727/2007. https://www.boe.es/boe/dias/2008/01/12/pdfs/A02292-02296.pdf. Accessed 27 August 2021

Sanpera C, Aguilar A (1992) Modern whaling off the Iberian Peninsula during the 20th century. Report of the International Whaling Commission 42:723-730.

Schoeman RP, Patterson-Abrolat C, Plön S (2020) A Global Review of Vessel Collisions With Marine Animals. Front Mar Sci 7:292. https://doi.org/10.3389/fmars.2020.00292

Schuler A, Piwetz S, Di Clemente J, Steckler D, Mueter F, Pearson H (2019) Humpback Whale Movements and Behavior in Response to Whale-Watching Vessels in Juneau, AK. Front Mar Sci 6:710. https://doi.org/10.3389/fmars.2019.00710

Silber G, Weller D, Reeves R, Adams J, Moore T (2021) Co-occurrence of gray whales and vessel traffic in the North Pacific Ocean. Endanger Species Res 44:177–201. https://doi.org/10.3354/esr01093

Silber GK, Lettrich MD, Thomas PO, Baker JD, Baumgartner M, Becker EA, Boveng P, Dick DM, Fiechter J, Forcada J, Forney KA, Griffis RB, Hare JA, Hobday AJ, Howell D, Laidre KL, Mantua N, Quakenbush L, Santora JA, Stafford KM, Spencer P, Stock C, Sydeman W, Van Houtan K, Waples RS (2017) Projecting marine mammal distribution in a changing climate. Front Mar Sci 4:413. https://doi.org/10.3389/fmars.2017.00413

Smith CR (2007) Bigger Is Better The Role Of Whales As Detritus In Marine Ecosystems. In: Estes JA, Demaster DP, Doak DF, Williams TM, Brownell RL (eds) Whales, Whaling, and Ocean Ecosystems, 1st edn. University of California Press, Berkeley, pp 286–300.

Smith J, Kelly N, Childerhouse S, Redfern J, Moore T, Peel D (2020) Quantifying Ship Strike Risk to Breeding Whales in a Multiple-Use Marine Park: The Great Barrier Reef. Front Mar Sci 7:67. https://doi.org/10.3389/fmars.2020.00067

Soldevilla MS, Hildebrand JA, Frasier KE, Aichinger Dias L, Martinez A, Mullin KD, Rosel PE, Garrison LP (2017) Spatial distribution and dive behavior of Gulf of Mexico Bryde's whales: potential risk of vessel strikes and fisheries interactions. Endanger Species Res 32:533–550. https://doi.org/10.3354/esr00834

Spyrakos E, Santos-Diniz T, Martinez-Iglesias G, Torres-Palenzuela J, Pierce G (2011) Spatiotemporal patterns of marine mammal distribution in coastal waters of Galicia, NW Spain. Hydrobiologia 670(1):87–109. https://doi.org/10.1007/s10750-011-0722-4

Surís-Regueiro JC, Santiago JL (2014) Characterization of fisheries dependence in Galicia (Spain). Mar Policy 47:99–109. https://doi.org/10.1016/j.marpol.2014.02.006

Thomas PO, Reeves RR, Brownell Jr RL (2016) Status of the world's baleen whales. Mar Mamm Sci 32(2):682–734. https://doi.org/10.1111/mms.12281

Tournadre J (2014) Anthropogenic pressure on the open ocean: The growth of ship traffic revealed by altimeter data analysis. Geophys Res Lett 41(22):7924–7932. https://doi.org/10.1002/2014GL061786

Tyack PL, Zimmer WMX, Moretti D, Southall BL, Claridge DE, Durban JW, Clark CW, D'Amico A, DiMarzio N, Jarvis S, McCarthy E, Morrissey R, Ward R, Boyd IL (2011) Beaked whales respond to simulated and actual navy sonar. PloS one 6(3):e17009. https://doi.org/10.1371/journal.pone.0017009

Van Der Hoop J, Moore M, Fahlman A, Bocconcelli A, George C, Jackson K, Miller C, Morin D, Pitchford T, Rowles T, Smith J, Zoodsma B (2014) Behavioral impacts of disentanglement of a right whale under sedation and the energetic cost of entanglement. Mar Mamm Sci 30(1):282–307. https://doi.org/10.1111/mms.12042

Vanderlaan ASM, Taggart CT (2007) Vessel collisions with whales: the probability of lethal injuty based on vessel speed. Mar Mamm Sci 23(1):144–156. https://doi.org/10.1111/j.1748-7692.2006.00098.x

Vanderlaan ASM, Corbett JJ, Green SL, Callahan JA, Wang C, Kenney RD, Taggart CT, Firestone J (2009) Probability and mitigation of vessel encounters with North Atlantic right whales. Endanger Species Res 6:273–285. https://doi.org/10.3354/esr00176

Visser F, Hartman K, Pierce G, Valavanis V, Huisman J (2011) Timing of migratory baleen whales at the Azores in relation to the North Atlantic spring bloom. Mar Ecol Prog Ser 440:267–279. https://doi.org/10.3354/meps09349

Weir CR, Stockin KA, Pierce GJ (2007) Spatial and temporal trends in the distribution of harbour porpoises, white-beaked dolphins and minke whales off Aberdeenshire (UK), north-western North Sea. J Mar Biolog Assoc UK 87(1):327–338. https://doi.org/10.1017/S0025315407052721

Williams R, Lusseau D, Hammond PS (2006) Estimating relative energetic costs of human disturbance to killer whales (*Orcinus orca*). Biol Conserv 133(3):301–311. https://doi.org/10.1016/j.biocon.2006.06.010

Williams R, Clark CW, Ponirakis D, Ashe E (2013) Acoustic quality of critical habitats for three threatened whale populations. Anim Conserv 17(2):174–185. https://doi.org/10.1111/acv.12076

Williams R, Erbe C, Ashe E, Beerman A, Smith J (2014) Severity of killer whale behavioral responses to ship noise: A dose–response study. Mar Pollut Bull 79(1-2):254– 260. https://doi.org/10.1016/j.marpolbul.2013.12.004

Wooster WS, Bakun A, McLain DR (1976) The seasonal upwelling cycle along the eastern boundary of the North Atlantic. J Mar Res 34:131–14

Worm B, Barbier EB, Sala E, Selkoe KA, Stachowicz JJ, Watson R, Beaumont N, Emmett Duffy J, Folke C, Halpern BS, Jackson JBC, Lotze HK, Micheli F, Palumbi SR (2006) Impacts of Biodiversity Loss on Ocean Ecosystem Services. Science 314(5800):787–790. https://doi.org/10.1126/science.1132294

Wrona FJ, Dixon RWJ (1991) Group Size and Predation Risk: A Field Analysis of Encounter and Dilution Effects. Am Nat 137(2):186–201.

Zuur AF, Ieno EN, Saveliev AA, Smith GM, Walker N (2009) Mixed Effects Models and Extensions in Ecology with R. Springer, New York