IMPACTS OF SHIPPING ON MARINE FAUNA

EDITED BY: Christine Erbe, David Peel, Jessica Redfern and Joshua Nathan Smith

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frontiers Research Topics

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IMPACTS OF SHIPPING ON MARINE FAUNA

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EDITORIAL
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Editorial: Impacts of Shipping on Marine Fauna

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Editorial on the Research Topic

Impacts of Shipping on Marine Fauna

SHIP TRAFFIC IN THE OCEAN KEEPS INCREASING

About 80% of international trade goods are transported by ships (UNCTAD, 2019). Over the 49 years from 1970 to 2018, the volume of global seaborne trade increased by a factor 4.4[1] (**Figure 1**). The number of ships and the size of ships have also been increasing (**Figure 1**).

AT THE SAME TIME, CONCERN OVER ENVIRONMENTAL IMPACTS OF SHIPPING IS INCREASING

These include chemical pollution of water and air (from fuel spills, waste dumping, and exhaust; Lachmuth et al., 2011; Endres et al., 2018; Arzaghi et al., 2020; Czermański et al., 2020), biofouling on hulls and invasive species (from discharge of ballast water; Jones, 2009), noise pollution in water and air (Wysocki et al., 2006; Badino et al., 2016; Erbe, Marley, et al.), and collision with marine fauna (Jägerbrand et al., 2019; Pirotta et al., 2019; UNCTAD, 2019). Shipping potentially impacts (directly or indirectly) a great range of marine taxa, not just locally but globally.

RESEARCH INTO THE EFFECTS OF SHIP TRAFFIC AND MANAGEMENT OF THE ASSOCIATED RISKS ARE TYPICALLY COMPARTMENTALIZED BY TYPE OF IMPACT

Yet the impacts may be cumulative and they may be linked. For example, quieter ships might be harder for animals to detect (i.e., detected over shorter ranges) and thus may represent a higher risk of collision. Faster ships may traverse animal habitat in less time, but at a cost of increased fuel consumption (hence exhaust), higher noise emission levels, and an increased risk of fatal collision. As Leaper highlights, reduced vessel speed could be financially prudent while also reducing greenhouse gas emission, ship noise, and ship strike risk.

THERE IS POTENTIAL BENEFIT FROM A MORE HOLISTIC APPROACH TO STUDYING AND MANAGING THE IMPACTS OF SHIPPING

In this special issue, we've therefore brought together research on the diverse impacts of shipping, on a variety of marine fauna, with examples from the equatorial regions to both poles (**Figure 2**).

[1] https://unctadstat.unctad.org/EN/BulkDownload.html (accessed June 7, 2020).

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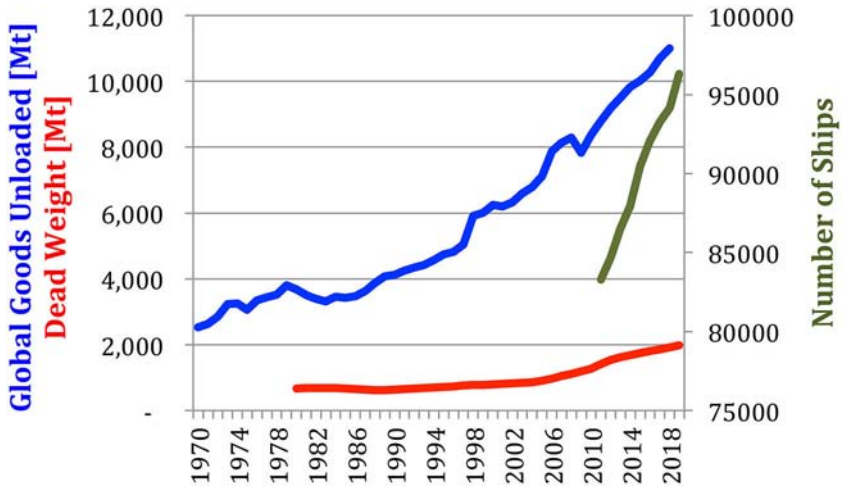

FIGURE 1 | Volume of global goods unloaded from ships (in million tons), dead weight of ships (in million tons), and number of ships[1].

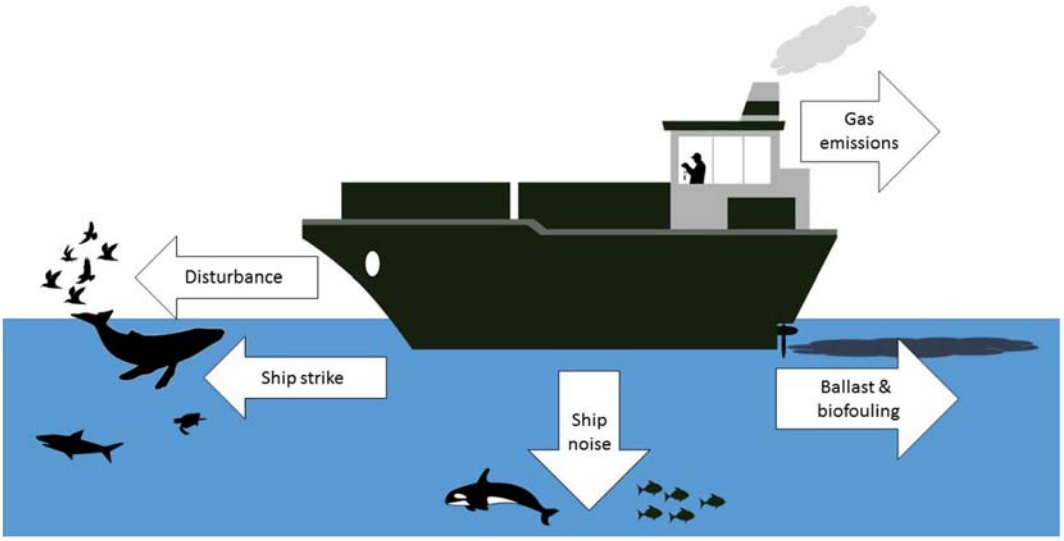

FIGURE 2 | Infographic on the potential impacts of shipping addressed in this special issue: ship strike (Arregui et al.; Calambokidis et al.; Aschettino et al.; Cloyed et al.; Flynn and Calambokidis; Gende et al.; Greig et al.; Keen et al.; Leaper; Peltier et al.; Redfern et al.; Schoeman et al.; Szesciorka et al.; Silber and Adams; Smith et al.), ship noise (Chion et al.; Erbe, Dähne et al.; Erbe, Marley et al.; Heenehan et al.; Joy et al.; Leaper; Silber and Adams), gas emission (Leaper; Silber and Adams), chemical spill (Silber and Adams), introduced pests (Hayes et al.), and induced flushing (i.e., seabird disturbance, Fliessbach et al.).

SEVERAL ARTICLES IN THIS SPECIAL ISSUE PROVIDE AN INTRODUCTION TO AND OVERVIEW OF THE DIFFERENT TYPES OF IMPACT

Hayes et al. review the risks of biofouling and ballast water discharge and associated management strategies. While their historical overview focuses on Australia and New Zealand, international guidelines and conventions are presented, and the efficacy, practicality and costs of treatment options are discussed. Erbe, Marley, et al. provide an overview of ship noise generation and propagation, and the impacts on marine mammals: change of behavior, auditory masking, and stress. Study challenges, knowledge gaps, and research needs are discussed. Schoeman et al. present a global review of vessel collisions with marine fauna and found that 75 marine species are affected by ship collisions, including whales, dolphins, porpoises, dugongs, manatees, whale sharks, sharks, seals, sea otters, sea turtles, penguins, and fish. Information about collisions with species other than large whales is scarce and Schoeman et al. recommend establishing species-specific necropsy protocols and creating a ship-strike database for smaller species to fill this data gap.

THE BEHAVIOR OF ANIMALS AFFECTS VULNERABILITY TO POTENTIAL IMPACTS

Calambokidis et al. show baleen whales can exhibit differential vulnerability to ship strike based on their movement patterns, swimming, and diving behavior during the day (**Figure 3**), although diurnal patterns showed a commonality of greater vulnerability at night. The same diurnal risk of collision was found for fin whales by Keen et al. who identify seasonal implications of risk with higher risk during winter where nighttime duration is longer, particularly at high latitudes. Cloyed et al. show that manatees are exposed to interactions with a range of vessels, from large commercial to small recreational vessels, and understanding manateee migration patterns and use of shipping channels is integral to linking and mitigating risk between the offshore and nearshore environment. Similarly, Aschettino et al. document an affinity to high-traffic areas by humpback whales. Despite the overlap of shipping routes with animal habitat and the associated risks of collision and noise exposure, Szesciorka et al. demonstrate for Eastern North Pacific blue whales the importance of the animals' behavioral response in their ability to avoid serious impacts, such as mortality from vessel strike. Fliessbach et al. developed a disturbance vulnerability index (DVI) for 26 seabird species in Northwestern Europe accounting for shyness, escape costs, and compensatory potential. The DVI can be used with distribution data to identify areas vulnerable to disturbance.

DETERMINING THE RISKS POSED BY SHIPPING IS THE FIRST STEP IN ENVIRONMENTAL IMPACT ASSESSMENTS

Vessel operations need to be documented and monitored, as Silber and Adams did in the Arctic, which is a marine ecosystem experiencing increased opportunities for maritime activities in historically inaccessible areas. Greig et al. examined some of the issues with speed calculated from Automatic Identification System (AIS) data, which are used in ship-strike risk analysis. While understanding the magnitude of ship strike rates globally is notoriously difficult because they often occur offshore and go unnoticed, Peltier et al. demonstrate the importance of monitoring marine mammal strandings and undertaking necropsies to allow an assessment of the risk of collision. However, when carcasses are in advanced stages of decomposition, it is challenging to distinguish whether trauma occurred ante- or post-mortem. Arregui et al. demonstrate that fat emboli detection can be a feasible, reliable, and accurate forensic tool to determine ante-mortem ship strikes in stranded whales, even in decomposed tissues kept in formaldehyde for long periods of time. Heenehan et al. suggest that soundscape monitoring is useful to assess noise impacts and determine co-occurrence of marine fauna and ships.

THE RISKS OF COLLISION AND NOISE IMPACTS CAN BE MITIGATED

Redfern et al. explored the consequences of interannual variability on ship-strike risk. They found that areas containing the highest predicted number of whales were generally the same across years. Consequently, either nearshore or offshore ship traffic consistently had the highest risk for each whale species. The consistency in risk suggests that static spatial management measures (e.g., changing shipping lanes, creating areas to be avoided, and seasonal speed reductions) can provide an effective means of mitigating risk in their study area. Smith et al. identified differences in ship strike risk based on the reproductive status of female humpback whales. They found that temporal dynamics in whale movement within a breeding season could affect risk, which can be countered by changes in whale density, and that common mitigation measures (e.g., re-routing shipping lanes) are not always possible. Studies have also assessed the possibility of reducing risk through "active whale avoidance" defined as a mariner making operational decisions to reduce the chance of collision with a sighted whale. Gende et al. generated a conceptual model of active whale avoidance and explored the model using observations of humpback whales surfacing in the proximity of large cruise ships and simulations run in a full-mission bridge simulator and commonly used pilotage software. They identified several options for enhancing whale avoidance and conclude that active whale avoidance is a feasible yet underdeveloped tool for reducing collision. The practicality and effectiveness of placing marine mammal observers on commercial vessels were examined by Flynn and Calambokidis.

Chion et al. compiled a large dataset of published ship noise levels from the literature and demonstrate that reducing speed is a means of achieving lower noise emission levels for any specific ship class and size category. Voluntary commercial vessel slowdown reduced underwater noise and associated "lost foraging time" despite increased transit duration in Joy et al. 's study on endangered southern resident killer whales (*Orcinus orca*), which used field measurements and modeling.

While this special issue collates articles on various types of shipping impact, on a diversity of taxa, in equatorial to polar regions, commonalities in identified *knowledge gaps and research needs* have been identified. Risk assessments utilize both ship distribution data (i.e., AIS) and species distribution information. Developments are needed in AIS coverage and accuracy. A better understanding of species distributions is also needed. Uncertainty needs to be considered in risk assessments. This includes uncertainty due to temporal variability (as examined in Redfern et al.), spatial uncertainty (e.g., AIS positional uncertainty), sampling and measurement error, and model uncertainty. Cumulative risk models are needed to assess risk over successive exposures in space and time, and to combine the potentially synergistic impacts of shipping. Understanding how impacts interact and accumulate will improve mitigation solutions by increasing our understanding of when reducing one impact may reduce or increase other impacts.

FIGURE 3 | Photo of a blue whale in front of a large vessel off California on 4 September 2014. Photo taken by John Calambokidis of Cascadia Research, as part of the diurnal ship strike risk assessment (Calambokidis et al.).

MANAGING THE IMPACTS OF SHIPPING REQUIRES COLLABORATION AMONGST STAKEHOLDERS AND ACROSS POLITICAL BORDERS

The shipping industry, marine scientists, environmental groups, government regulators, socio-economists, etc., need to work together to improve outcomes for all stakeholders and the environment. Political borders never line up with habitats or environmental regions, necessitating collaboration among countries on guidelines and regulations. Examples include the European Marine Strategy Framework Directive (van der Graaf et al., 2012) and the Protocol on Environmental Protection of the Antarctic Treaty, which was the framework for the impact assessment of underwater noise in Antarctica by Erbe, Dähne, et al.. As our understanding of the potential impacts of shipping matures, international standards can ensure consistency in research, mitigation, and management.

At a time when shipping is continuing to increase, we believe the articles in this special issue highlight the important issues facing our global society and serve as a starting point for managing the potential impacts of shipping.

AUTHOR CONTRIBUTIONS

CE prepared the first draft and created **Figure 1**. JS created **Figure 2**. All authors summarized 5, 6 articles each and approved the final version of this Editorial.

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A Ship Traffic Disturbance Vulnerability Index for Northwest European Seabirds as a Tool for Marine Spatial Planning

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Ship traffic in Northwestern European seas is intense and continuing to increase, posing a threat to vulnerable seabird species as a result of disturbance. However, information on species-specific effects of ship traffic on seabirds at sea is limited, and tools are needed to prioritize species and areas to support the integration of conservation needs in Marine Spatial Planning. In this study, we investigated the responses of 26 characteristic seabird species in the German North and Baltic Seas to experimental ship disturbance using large datasets collected as part of the Seabirds at Sea counts. We developed a Disturbance Vulnerability Index (DVI) for ship traffic combining indicators for species' shyness, escape costs, and compensatory potential, and analyzed the relationships among shyness, escape costs, and vulnerability. The DVI was calculated using the following eight indicators: escape distance, proportion of escaping birds, proportion of birds swimming prior to disturbance, wing loading, habitat use flexibility, biogeographic population size, adult survival rate, European threat and conservation status. Species-specific disturbance responses differed considerably, with common scoters (*Melanitta nigra*) and red-throated loons (*Gavia stellata*) showing the longest escape distances and highest proportions of escaping individuals. Red-throated loon, black guillemot (*Cepphus grylle*), Arctic loon (*Gavia arctica*), velvet scoter (*Melanitta fusca*), and red-breasted merganser (*Mergus serrator*) had the highest DVI values, and gulls and terns had the lowest. Contrary to theoretical considerations, shyness correlated positively with escape costs, with the shyest species also being the most vulnerable among the species studied. The strong reactions of several species to disturbance by ships suggest the need for areas with little or no disturbance in some marine protected areas, to act as a refuge for vulnerable species. This DVI can be used in combination with distribution data to identify the areas most vulnerable to disturbance.

Keywords: seabirds, ship traffic, disturbance, behavior, escape distance, vulnerability index, risk-disturbance hypothesis, marine spatial planning

INTRODUCTION

The German North Sea and Baltic Sea are heavily impacted by ship traffic (OSPAR, 2010; Bahlke, 2017; HELCOM, 2018). A growing maritime economy in general and the construction and maintenance of offshore wind farms in particular will lead to further increases in ship traffic, including outside designated shipping lanes (Ecorys et al., 2012; Fridell et al., 2015; Bahlke, 2017; Matczak, 2018). Ship traffic is known to be associated with various negative environmental impacts as a result of emissions into the water and air (OSPAR, 2010; HELCOM, 2018). In addition, approaching vessels may present a threatening stimulus to marine birds, with subsequent risk-avoidance behavior reducing the time available for other activities such as feeding, resting, or mating (Gill et al., 1996; Frid and Dill, 2002; Beale and Monaghan, 2004b). Observable responses by seabirds include flying off, escape diving, and increased alertness, which can result in loss of energy and opportunities, displacement, and net habitat loss (Bélanger and Bédard, 1990; Madsen and Fox, 1995; Béchet et al., 2004). Disturbance by ships may thus reduce survival and reproductive success and affect population dynamics (Goss-Custard et al., 1995a; Madsen, 1995; Carney and Sydeman, 1999; Sutherland, 1998).

The German North and Baltic Seas are important wintering sites for a large number of seabirds (Mendel et al., 2008; Markones et al., 2015). Several species are listed in Annex 1 of the European Union Birds Directive, which obliges member states to conserve their "most suitable territories" as Special Protection Areas. Winter and spring are considered to be the most critical times for accumulating body fat and establishing pair bonds in most waterbirds (Madsen and Fox, 1995; Knapton et al., 2000). However, despite the importance of the area and the high frequency of vessel traffic, disturbance of seabirds as a result of ship traffic is often neglected in (cumulative) impact assessments and planning processes. This can be partly attributed to the lack of detailed information and tools to identify and prioritize vulnerable species and areas.

Vulnerability indices have been established as tools to estimate levels of concern for species and areas, and have been developed for several human activities in the marine environment (surface pollutants: Williams et al., 1995; oil pollution: Camphuysen, 1998; traffic disturbance: Camphuysen et al., 1999; set-net fishery: Sonntag et al., 2012; wind energy: e.g., Kelsey et al., 2018). When combined with distribution data, they can be used to identify the most vulnerable areas (e.g., Garthe and Hüppop, 2004; Sonntag et al., 2012; Bradbury et al., 2014). An extensive literature has documented the effects of disturbance on breeding and non-breeding waterbirds in coastal and freshwater habitats (Carney and Sydeman, 1999; Rodgers and Schwikert, 2002; Steven et al., 2011; Glover et al., 2015; Krüger, 2016; McFadden et al., 2017), but information on disturbance responses of seabirds at sea to vessel traffic is limited to a few species (Bellebaum et al., 2006; Kaiser et al., 2006; Schwemmer et al., 2011).

Disturbance responses differ among species, with some species being more sensitive than others. Responses are measurable as escape distances, the proportions of escaping birds, and as physiological responses such as heart rate and corticosterone levels. Given that the physiological responses of free-ranging seabirds at sea are extremely difficult to measure, most studies of disturbance effects have reported escape distances as a measure of effect (Blumstein et al., 2005). However, a species' vulnerability to disturbance cannot be assessed based on escape distance alone, given that the decision of when to take flight represents a trade-off between safety and fitness-enhancing activities (Ydenberg and Dill, 1986; Lima and Dill, 1990; Gill et al., 2001; Frid and Dill, 2002; Gill, 2007). A bird in good body condition and with sufficient feeding alternatives might flush earlier than a bird short of resources, as demonstrated in an experimental study with waders (Beale and Monaghan, 2004a). Visible disturbance responses alone are thus generally not considered to be a good indicator of vulnerability (Gill et al., 2001; Frid and Dill, 2002; Beale and Monaghan, 2004a; Beale, 2007). Vulnerability analysis should therefore consider the total costs of disturbance events including the ability to compensate for losses at the individual and population levels.

This study aimed to further our knowledge of species-specific behavioral disturbance responses at sea for all common and characteristic seabirds in German waters using experimental disturbance. We also aimed to develop a Disturbance Vulnerability Index (DVI) for ship traffic, combining indicators for species' shyness, escape costs, and compensatory potential, which can be used as a management tool to assess different vulnerabilities of a given sea area with respect to disturbance by ships. Finally, we aimed to investigate the general relationships among shyness, escape costs, and vulnerability in seabirds by cross-species comparisons of disturbance-related factors.

MATERIALS AND METHODS

Behavioral Observations

All behavioral observations were carried out during ship-based Seabirds at Sea counts in the coastal and offshore zones of the German North Sea (**Figure 1**) and Baltic Sea (**Figure 2**), following internationally standardized methods (Tasker et al., 1984; Webb and Durinck, 1992; Garthe et al., 2002; Camphuysen and Garthe, 2004). All birds were classified as either swimming or flying. Swimming birds were counted continuously within a 300 m wide transect parallel to the ship's keel line. Flying birds were only counted at full minutes and within a distance of 300 m to the side and to the front of the vessel, to avoid overestimation. We analyzed three different datasets collected within this framework from all seasons combined, described in the following paragraphs.

Proportion of Swimming Birds

We calculated the proportion of swimming birds for each species from data collected on more than 1,200 survey days in the years 2000 to 2017. Distance-correction factors (Garthe et al., 2007, 2009; Markones et al., 2013) were applied to the number of swimming birds to account for overlooked birds, resulting in a dataset of over 1.1 million birds. In our context 'swimming' encompassed all activities performed on the water surface, including resting, preening, active swimming, or others,

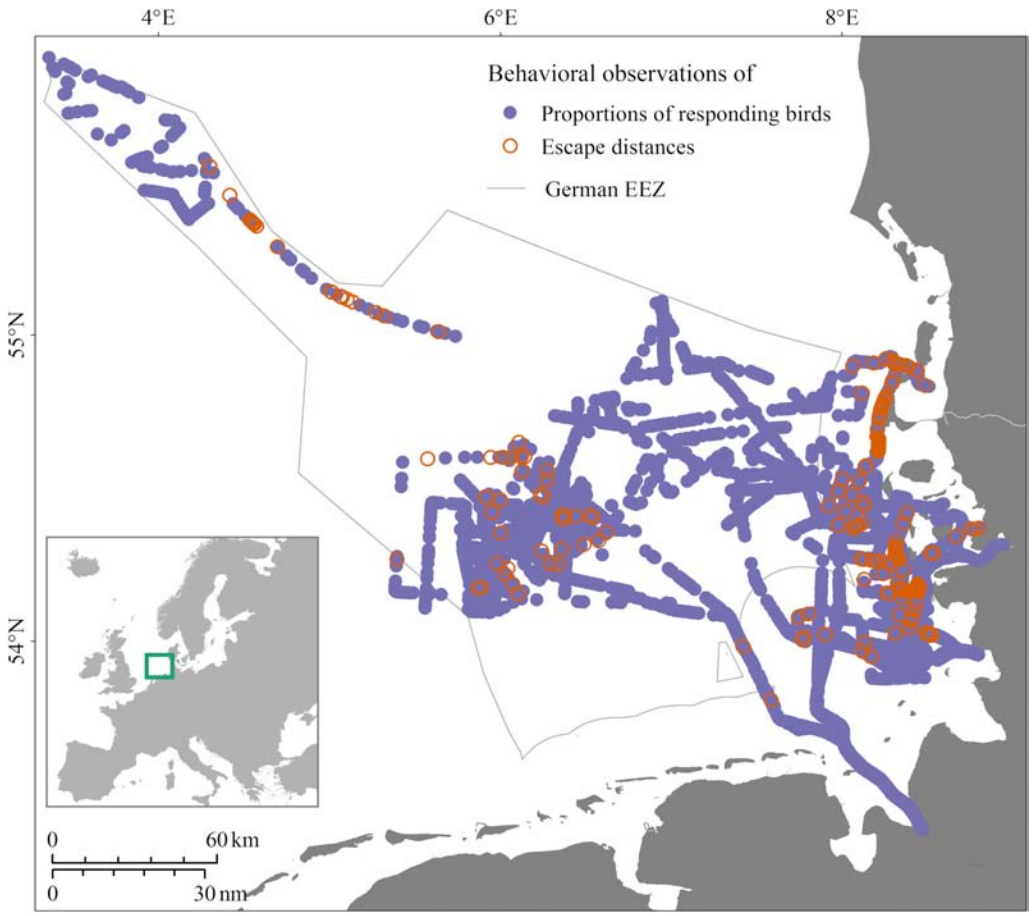

FIGURE 1 | Study area and locations of data collection in the years 2016/2017 in the German North Sea. Purple locations, data collected on proportions of responding birds; orange locations, additionally measured escape distances.

each of which implies slightly different, but low energetic costs compared with flying (Norberg, 1996). Because birds reacted to the approaching vessels, we looked far ahead to detect swimming birds before flushing, and always recorded flushed birds as swimming birds prior to disturbance.

Proportion of Escaping Birds

Species-specific disturbance responses were recorded on 139 survey days in 2016 and 2017 (**Figures 1**, **2**). We distinguished between two types of disturbance responses: flying off and escape diving. Common murres (*Uria aalge*) with young were excluded from the analysis, because we assumed that adult birds escaped less often to stay in the vicinity of their offspring. We further excluded species with fewer than 15 observations from the analysis of the proportion of escaping birds. The total dataset comprised 221,071 individuals from 25 species and species groups (loons and auks).

Escape Distance

We additionally measured the escape distances of disturbed birds, also called flight initiation distance (e.g., Bonenfant and Kramer, 1996; Blumstein, 2006) or flush distance (e.g., Rodgers and Smith, 1995; Schwemmer et al., 2011), on 51 of the 139 survey days in the years 2016 and 2017. Our method was based on the distance estimation using geometrical functions, as described by Heinemann (1981). We recorded escape distances following the same principle as Schwemmer et al. (2011), but using individualized rulers instead of calipers or binoculars with reticles. To keep a consistent method, we refrained from measuring distances with radar or rangefinder, which are not consistently feasible under conditions at sea (Schwemmer et al., 2011; Borkenhagen et al., 2017). We randomly selected flocks of different sizes and measured the distance between the observer vessel and the first escaping bird in a flock at the moment of flushing or escape diving. Measurements were taken in directions between 90° and 0° of the course of the ship. Five research vessels were used: MS Haithabu (39 m long, n = 465 measurements); FS Heincke (55 m long, n = 42 measurements); MS Odin (32 m long, n = 161 measurements); MS Prandtl (31 m long, n = 1575 measurements); and MS Skoven (42 m long, n = 17 measurements). The measurements were taken at an average speed of 18.5 ± 1.8 km/h. We scanned the water surface constantly using binoculars to ensure that birds further away were not missed. Because escape distance

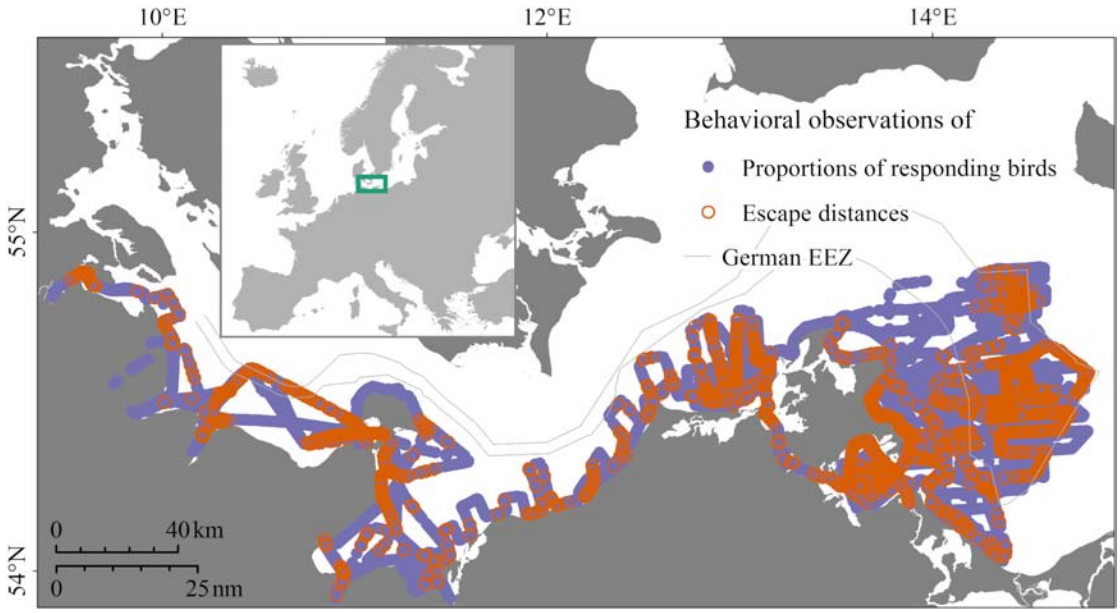

FIGURE 2 | Study area and locations of data collection in the years 2016/2017 in the German Baltic Sea. Purple locations, data collected on proportions of responding birds; orange locations, additionally measured escape distances.

generally increases with flock size (Burger and Gochfeld, 1991; Mori et al., 2001; Kaiser et al., 2006; Schwemmer et al., 2011), we calculated the mean escape distance per individual for later use in the DVI. Escape distance per flock was also presented to allow comparisons with other studies. A considerable proportion of auks and loons cannot be identified to species level, especially at greater distances. If differences between species (e.g., red-throated loon) and their respective species group (loons) occurred, we presented both values separately to ensure that all behavioral observations were included. Only species with at least five observations were included in the analysis of escape distances. We calculated mean escape distances for a total of 22 species and species groups (loons and auks) based on 2,260 measurements. Statistical analysis was performed in R 3.2.4 using simple summary statistics (R Core Team, 2016; RStudio Team, 2016).

Disturbance Vulnerability Index

We constructed the DVI for ship traffic to reflect the total costs of disturbance, defined by three components: (1) the probability of a disturbance event based on species' shyness; (2) the energetic costs of escape of each disturbance event; and (3) the costs on the population level based on status factors. We chose eight factors as indicators of the described components:

(1) Shyness:

- (a) Proportion of escaping birds: species with a high proportion of escaping birds flush or dive more often to escape from ships.
- (b) Escape distance: the affected area is larger for species with a long escape distance.

(2) Escape costs:

- (c) Proportion of swimming birds: flying is energetically much more costly than swimming (floating), and a flushing event is thus proportionally more costly for birds that seldom fly than for frequently flying birds.
- (d) Wing loading: species differ in the energy expenditure required for reactions such as flushing or diving because of morphological and physiological differences. Birds with a higher wing loading have higher energetic costs for flushing (Norberg, 1996).
- (e) Habitat use flexibility: disturbance may displace birds from suitable habitat. Species relying on specific habitat features thus have higher costs than species with less specific habitat preferences.

(3) Population status:

- (f) Biogeographic population size: energetic losses, displacement, and habitat loss may increase mortality and reduce reproduction. Species with small biogeographic populations are considered more vulnerable to additional losses.
- (g) Adult survival rate: species with high adult survival rates are more affected by additional adult mortality than species with low adult survival rates (Sæther and Bakke, 2000).
- (h) European threat and conservation status: species with a high conservation status are considered more vulnerable to any additional pressures.

Each factor was scored on a 5-point scale from 1 (low) to 5 (high). Factors (a–c) above were based on data collected in the

present study. We used the weighted mean value of species and species group for auks and loons in factor (b), because birds could often not be identified to species level at longer distances. Factor (e) was assessed by subjective considerations based on at-sea experience by Garthe and Hüppop (2004). If data for one species was missing, we used scores for closely related species. The sources of the values for each factor and their scores are given in **Table 1**. An average score was calculated for each component and subsequently multiplied by each other to produce the DVI for each species, following the methodology of the wind farm sensitivity index developed by Garthe and Hüppop (2004):

$$DVI = \frac{(a+b)}{2} \times \frac{(c+d+e)}{3} \times \frac{(f+g+h)}{3}$$

The relationships among the three components of the DVI were investigated by Spearman's rank-order correlations in R 3.2.4 (R Core Team, 2016; RStudio Team, 2016; Wickham, 2016).

TABLE 1 | Data sources and scoring of factors used in the DVI.

Factor	Source	Scoring
(a) % Escaping birds	Present study: experimental disturbance, German Seabirds at Sea database (years 2016 and 2017); total proportion of birds that either flushed or dived to escape within the transect	0–20%: 1 21–40%: 2 41–60%: 3 61–80%: 4 81–100%: 5
(b) Escape distance	Present study: experimental disturbance, German Seabirds at Sea database (years 2016 and 2017); distance from the vessel in m in the moment of flushing or diving	0–200 m: 1 201–400 m: 2 401–600 m: 3 601–800 m: 4 >800 m: 5
(c) % Swimming	Present study: German Seabirds at Sea database (data from 2000 to 2017); proportion of birds swimming within the transect	0–20%: 1 21–40%: 2 41–60%: 3 61–80%: 4 81–100%: 5
(d) Wing loading	Greenewalt, 1962; Guillemette, 1994; Alerstam et al., 2007; FLIGHT 1.22 for Windows (Pennycuick, 2008)	<50 N/m^2: 1 50–79 N/m^2: 2 80–109 N/m^2: 3 110–150 N/m^2: 4 >150 N/m^2: 5
(e) Habitat use flexibility	Garthe and Hüppop, 2004	1–5
(f) Biogeographic population size	Wetlands International, 2018 Northern gannet, Northern fulmar and Auks: BirdLife International (2017); excluding Greenland, individuals were multiplied by 1.5 and breeding pairs by 3 for consistency with Wetlands International	>3,000,000: 1 1,000,001–3,000,000: 2 500,001–1,000,000: 3 100,001–500,000: 4 ≤100,000: 5
(g) Adult survival rates	Horswill and Robinson, 2015	≤0.75: 1 0.751–0.80: 2 0.801–0.85: 3 0.851–0.90: 4 >0.90: 5
(h) European threat and conservation status[1]	BirdLife International, 2017	Non-SPEC: 1 Non-SPEC^E: 2 SPEC3: 3 SPEC2: 4 SPEC1: 5

[1] *Categories after BirdLife International (2017):*
Non-SPEC: species whose global population is not concentrated in Europe, and whose European population status is currently considered to be 'Secure.' Non-SPEC^E: species whose global population is concentrated in Europe, but whose European population status is currently considered to be 'Secure.' SPEC3: species whose global population is not concentrated in Europe, but which is classified as 'Regionally Extinct,' 'Critically Endangered,' 'Endangered,' 'Vulnerable,' 'Near Threatened,' 'Declining,' 'Depleted,' or 'Rare at European level.' SPEC2: species whose global population is concentrated in Europe, and which is classified as 'Regionally Extinct,' 'Critically Endangered,' 'Endangered,' 'Vulnerable,' 'Near Threatened,' 'Declining,' 'Depleted,' or 'Rare at European level.' SPEC1: European species of global conservation concern, i.e., classified as 'Critically Endangered,' 'Endangered,' 'Vulnerable,' or 'Near Threatened' at global level.

RESULTS

Behavioral Observations

Proportion of Birds Swimming

The proportion of time the birds spent swimming differed strongly among species (**Figure 3**). Grebes, seaducks, auks, and loons were detected swimming most often, with proportions ranging from 91% (red-throated loon) to 100% [red-necked grebe (*Podiceps grisegena*) and horned grebe (*Podiceps auritus*)]. Moderate proportions of great cormorants (*Phalacrocorax carbo*), gull species, northern fulmars (*Fulmarus glacialis*), and northern gannets (*Morus bassanus*) were seen swimming, while terns had the lowest proportions of swimming birds (16–23%).

Proportion of Escaping Birds

The proportion of individuals showing disturbance responses such as flushing or escape diving differed greatly between species (**Figure 4**). Overall, flushing was the most common disturbance response, with 73% of all recorded birds flushing in front of the vessel, compared with 1% that escaped by diving. Even among species capable of diving, only a small proportion of birds dived, except for Arctic loons, common murres, and red-necked grebes.

The highest total proportions of birds with observed disturbance responses were calculated for unidentified loons (96%), red-throated loons (95%), unidentified auks (94%), and black guillemots (92%), followed by red-breasted mergansers (*Mergus serrator*; 86%), common scoters (83%), velvet scoters (82%), horned grebes (80%), razorbills (*Alca torda*; 78%), and long-tailed ducks (*Clangula hyemalis*; 81%). The lowest proportions of disturbance responses were found in gull species and northern fulmars, among which black-legged kittiwakes (*Rissa tridactyla*) had the highest (32%) and black-headed gulls (*Croicocephalus ridibundus*) the lowest (10%) proportions.

There were sometimes large differences in the proportions of individuals of closely related species displaying certain behaviors; 92% of red-throated loons took flight in front of the vessel and only 3% dived to escape, while only 30% of Arctic loons took flight, and 32% dived to escape. The proportion of flushed individuals was very high among black guillemots (90%), compared with razorbills (65%) and common murres (17%). In contrast, the proportion of individuals that dived was considerably higher among common murres (20%) compared with razorbills (13%) and black guillemots (2%). Horned grebes (75%) flushed more often than red-necked grebes and great crested grebes (*Podiceps cristatus*; each 46%), but red-necked

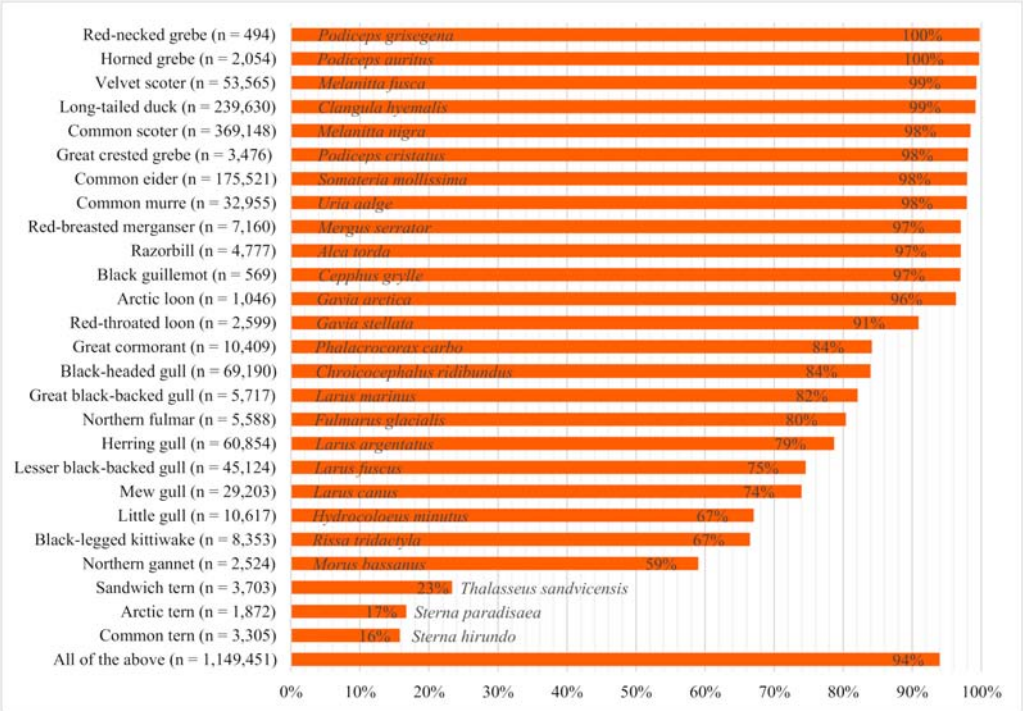

FIGURE 3 | Proportion of swimming birds observed during ship-based surveys in the German North and Baltic Seas in the years 2000 to 2017 (*n* = total number of individuals considered).

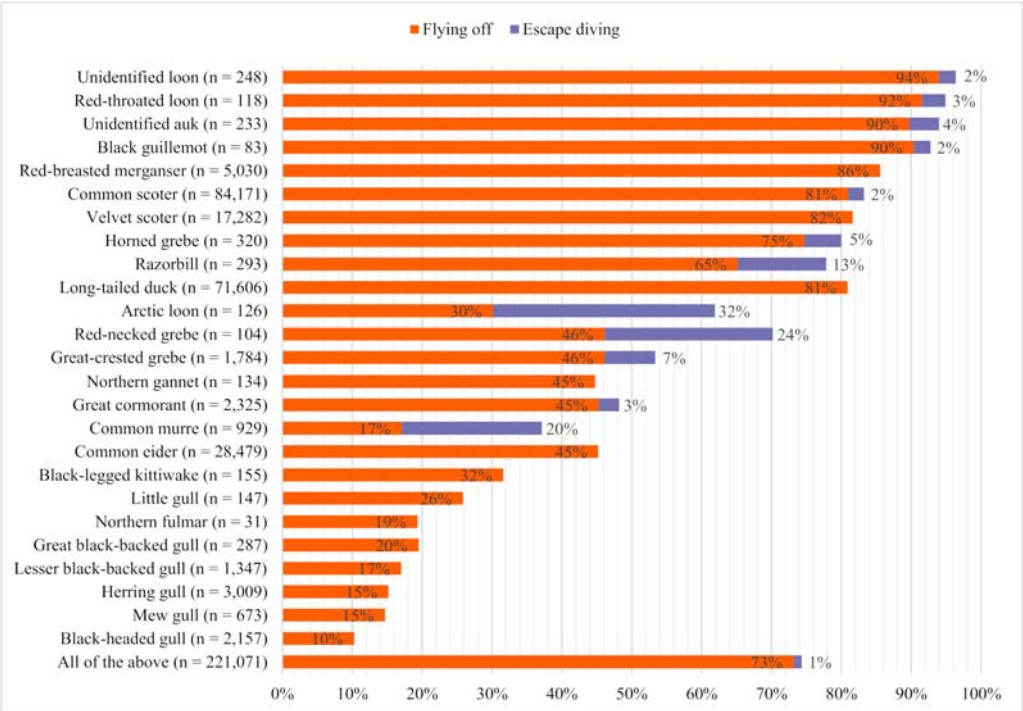

FIGURE 4 | Species-specific proportions of birds showing different disturbance responses in front of approaching research vessels in 2016 and 2017 (*n* = total number of individuals considered).

grebes dived more often (24%). Velvet scoters, common scoters, and long-tailed ducks showed similar proportions of flushed individuals (82%, 81%, and 81%, respectively), while the proportion of common eiders that flushed was only about half (*Somateria mollissima*; 45%).

Escape Distance

Escape distances differed widely among species. The mean escape distance per individual was higher than the mean escape distance per flock in most species (**Table 2**). This effect was most pronounced in red-breasted mergansers (1,178 m per individual vs. 681 m per flock) and common scoters (1,600 m per individual vs. 1,015 m per flock). Of all species, common scoters had the highest mean escape distance per individual (1,600 ± 777 m), followed by unidentified loons (1,374 ± 416 m), red-breasted mergansers (1,178 ± 617 m), red-throated loons (750 ± 437 m), unidentified auks (750 ± 379 m), and Arctic loons (721 ± 616 m). The escape distances of the remaining seaduck species, razorbills, black guillemots, grebes, and great cormorants were considerably lower, with mean values between 474 ± 304 m (velvet scoter) and 221 ± 171 m (red-necked grebe). The lowest mean escape distances were calculated for gull species (lesser black-backed gull (*Larus fuscus*): 157 ± 105 m, herring gull (*Larus argentatus*): 133 ± 83 m, mew gull (*Larus canus*): 118 ± 113 m, black-headed gull: 84 ± 70 m, great black-backed gull (*Larus marinus*): 79 ± 81 m), northern gannets (127 ± 82 m), and common murres (127 ± 110 m) (**Table 2**). The maximum escape distance was observed in common scoters (3,200 m). Other seaduck species and loons also had high maximum escape distances between 1,500 m and 2,000 m. Seaducks sometimes flushed at a distance of around 3,000 m, but could not be identified to species level and were not included in the analysis.

Disturbance Vulnerability Index

The species differed strongly in their DVI values (**Table 3**). The highest values were calculated for red-throated loon, black guillemot, and Arctic loon, followed by velvet scoter, red-breasted merganser, razorbill and horned grebe. Mew gull, black-headed gull, common tern (*Sterna hirundo*) and Arctic tern (*Sterna paradisaea*) were the least vulnerable species. Rankings based on behavioral sensitivity (shyness × escape costs) and population sensitivity diverged in some cases, and we therefore presented both values separately. Behavioral sensitivity was highest in red-throated loon, common scoter, red-breasted merganser and Arctic loon and lowest in Sandwich tern (*Thalasseus sandvicensis*), lesser black-backed gull, common tern and Arctic tern. Common eider and black guillemot ranked highest in terms of population sensitivity, and common scoter, great-crested grebe, black-headed gull and Arctic tern lowest. In the DVI, common scoter thus ranked lower and common eider ranked higher than suggested based on behavioral sensitivity alone.

The scores for the three components of the index (shyness, escape costs, population status) correlated positively with each

TABLE 2 | Escape distances of seabirds in the German North and Baltic Seas.

	Individual				Flock		
Species	**Mean ± SD**	***N***	**Min**	**Max**	**Mean ± SD**	**Median**	***N***
Common scoter	1,600 ± 777	9,417	40	3,200	1,015 ± 727	800	591
Unidentified loon	1,374 ± 416	64	340	2,000	1,281 ± 424	1,200	40
Red-breasted merganser	1,178 ± 617	193	120	2,000	681 ± 485	500	41
Red-throated loon	750 ± 437	31	250	1,700	702 ± 348	600	21
Unidentified auk	750 ± 379	4	200	1,000	667 ± 416	800	3
Arctic loon	721 ± 616	31	80	2,000	562 ± 450	450	18
Velvet scoter	474 ± 304	1,062	30	2,000	444 ± 307	350	241
Black guillemot	417 ± 186	6	180	700	417 ± 186	410	6
Razorbill	395 ± 216	53	30	900	330 ± 219	280	23
Long-tailed duck	389 ± 227	8,274	10	1,500	325 ± 235	250	604
Horned grebe	343 ± 255	33	30	1,100	325 ± 268	265	24
Great-crested grebe	308 ± 248	58	50	900	288 ± 245	165	36
Common eider	277 ± 218	1,496	20	1,600	255 ± 195	200	290
Great cormorant	258 ± 215	187	30	1,500	287 ± 241	225	124
Red-necked grebe	221 ± 171	7	80	600	230 ± 186	175	6
Lesser black-backed gull	157 ± 105	51	30	500	159 ± 106	130	50
Herring gull	133 ± 83	115	15	300	110 ± 85	65	60
Northern gannet	127 ± 82	7	20	250	127 ± 82	120	7
Common murre	127 ± 110	86	15	500	137 ± 123	100	48
Mew gull	118 ± 113	12	20	400	118 ± 113	70	12
Black-headed gull	84 ± 70	9	20	250	86 ± 75	60	8
Great black-backed gull	79 ± 81	7	25	250	79 ± 81	40	7

Values presented for individuals and flocks for comparability reasons. Values for individuals calculated from value for flock, weighted by the number of individuals. Distances given in meters.

TABLE 3 | Factor scores and resulting Disturbance Vulnerability Index (DVI) values for 26 common European seabird species.

	a	*b*	*c*	*d*	*e*	*f*	*g*	*h*			
Bird species	**% Escaping**	**Mean escape distance**	**% Swimming**	**Wing loading**	**Habitat use flexibility**	**Biogeographic population size**	**Adult survival rate**	**European threat and conservation status**	**Behavioral sensitivity ((a + b)/2) × ((c + d + e)/3)**	**Population sensitivity (f + g + h)/3**	**DVI**
Red-throated loon	5	5	5	5	4	4	3	3	23.3	3.3	77.8
Black guillemot	5	3	5	4	5	4	4	4	18.7	4.0	74.7
Arctic loon	4	5	5	5	4	4	3	3	21.0	3.3	70.0
Velvet scoter	5	3	5	5	4	4	2	5	18.7	3.7	68.4
Red-breasted merganser	5	5	5	4	4	4	2	3	21.7	3.0	65.0
Razorbill	4	3	5	4	3	2	4	5	16.0	3.7	51.3
Horned grebe	4	2	5	4	5	5	1	5	14.0	3.7	51.3
Common eider	3	2	5	5	4	3	4	5	11.7	4.0	46.7
Common scoter	5	5	5	4	4	3	2	1	21.7	2.0	43.3
Long-tailed duck	5	2	5	4	4	2	1	5	15.2	2.7	40.4
Red-necked grebe	4	2	5	3	5	5	1	1	13.0	2.3	30.3
Great cormorant	3	2	5	3	3	3	4	1	9.2	2.7	24.4
Great crested grebe	3	2	5	4	4	4	1	1	10.8	2.0	21.7
Common murre	2	1	5	5	3	1	5	3	6.5	3.0	19.5
Northern gannet	3	1	3	3	1	3	5	2	4.7	3.3	15.6
Little gull	2	1	4	1	3	4	2	3	4.0	3.0	12.0
Great black-backed gull	1	1	5	2	2	4	5	2	3.0	3.7	11.0
Black-legged kittiwake	2	1	4	1	2	1	4	3	3.5	2.7	9.3
Northern fulmar	1	1	5	2	1	1	5	3	2.7	3.0	8.0
Sandwich tern	1	1	2	1	3	4	4	2	2.0	3.3	6.7
Lesser black-backed gull	1	1	4	1	1	4	4	2	2.0	3.3	6.7
Herring gull	1	1	4	2	1	1	3	4	2.3	2.7	6.2
Mew gull	1	1	4	1	2	2	3	2	2.3	2.3	5.4
Black-headed gull	1	1	5	1	2	1	3	2	2.7	2.0	5.3
Common tern	1	1	1	1	3	2	4	1	1.7	2.3	3.9
Arctic tern	1	1	1	1	3	2	3	1	1.7	2.0	3.3

other (**Figure 5**). There was a strong significant correlation between shyness (mean of factors a and b) and escape costs (mean of factors c–e) ($r = 0.81$, $p \leq 0.001$, $n = 26$). Escape costs also correlated significantly with population status (mean of factors f–h) ($r = 0.43$, $p \leq 0.05$, $n = 26$), but the positive correlation coefficient between shyness and population status was not significant ($r = 0.27$, $p = 0.189$, $n = 26$).

DISCUSSION

Behavioral Observations

The current study detected large interspecific differences in the proportions of swimming birds prior to disturbance, associated with species' ecology. Species adapted to diving for (relatively) stationary prey exhibited the highest percentages of swimming individuals. Flying is the energetically most demanding form of locomotion per unit time (Norberg, 1996), and can be up to 31 times more costly than the basal metabolic rate (see Elliott et al., 2013 for an example in thick-billed murres, *Uria lomvia*). Short flights are especially costly per unit time, because take-off, ascent, and descent form a large part of the total flight time (Nudds and Bryant, 2000). Birds that rarely fly under normal circumstances thus suffer proportionally higher flight costs due to disturbance than birds that fly more frequently.

Disturbance responses to ships differed strongly among species. Species with long escape distances were also among the species with the highest proportions of escaping birds, reflecting the fact that these parameters are related and measure the same trait (shyness). However, differences between the shyest species were much more pronounced in terms of escape distances than in the proportions of escaping birds. Investigating both factors thus gave a more detailed picture of interspecies differences in disturbance behavior, and both were included as indicators for species' shyness in the vulnerability index.

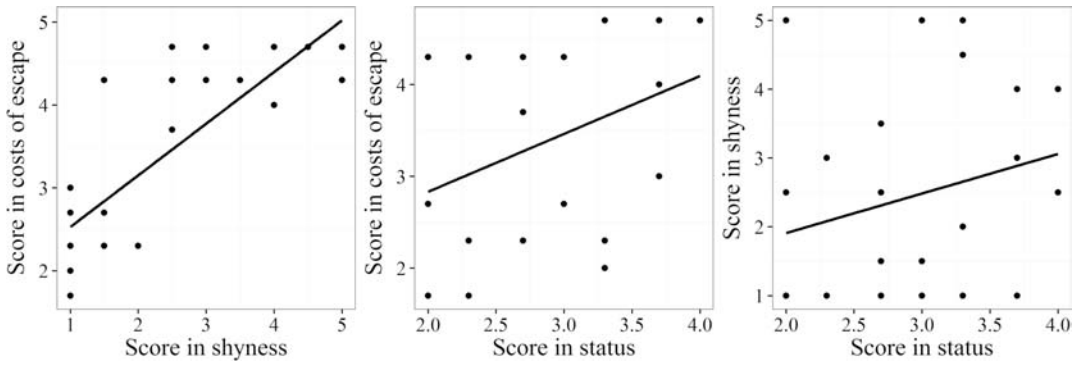

FIGURE 5 | Associations of the three components, shyness, escape costs, and population status, used to compute the DVI. Trend lines were calculated using linear regression.

The detected differences in disturbance responses among species could largely be explained by different perceptions of predation risk, and less by the cost of escape. The decision on when to take flight is the result of a trade-off between safety and feeding or other fitness-enhancing activities (Ydenberg and Dill, 1986; Lima and Dill, 1990; Frid and Dill, 2002). Birds will thus flush when the costs of remaining and escape are equal (Ydenberg and Dill, 1986). In theory, escape distance should increase with higher flight costs (high wing loadings) and higher costs due to lost opportunities (low habitat flexibility). In our study, however, species with higher escape costs were overall also those with stronger disturbance responses (except for common eider and common murre), suggesting that these species must have highly different predation risks, i.e., costs of remaining (Ydenberg and Dill, 1986), which dominated the influence of escape costs on escape distances. Birds with high wing loadings are generally less maneuverable in flight, need a longer time to take off, and thus have more difficulty escaping from predators. Many of those species are also subject to hunting by humans (Hirschfeld and Heyd, 2005; Hirschfeld and Attard, 2017), which might have increased their shyness toward human activities. In contrast, purely pelagic seabirds such as northern gannets, northern fulmars, and terns have a lower predation risk, and like gulls, often benefit from increased feeding opportunities in the presence of ships by using discards (Garthe and Hüppop, 1994; Garthe et al., 2016).

Wing loading, as an escape cost, might partially explain differences in disturbance behavior between closely related species. Common murres and Arctic loons each have approximately 30% higher wing loading than the closely related razorbills and red-throated loons (Thaxter et al., 2010; Alerstam et al., 2007; Pennycuick, 2008). Common murres had a much lower escape distance than razorbills, while Arctic loons seemed to escape slightly later than red-throated loons. We also observed fewer common murres and Arctic loons escaping compared with razorbills and red-throated loons, and a much higher percentage dived to escape instead of flying off. Similarly, the wing loading of common eiders is higher than in other seaduck species (Guillemette, 1994; Alerstam et al., 2007), and the mean escape distance and proportion of birds taking flight was considerably lower (see also Schwemmer et al., 2011). However, differences among the other seaduck species cannot be explained by wing loading. The positive relationship between body mass and escape distance reported in several other studies (Blumstein, 2006; Fernández-Juricic et al., 2006; Weston et al., 2012) did also not seem to exist in the seabird species studied here. A detailed knowledge of the costs, habitat and resource uses, and predation risks for each species would be required to understand the interspecific differences fully.

We also detected high intraspecific variability in escape distance. Escape distances can be influenced by various environmental (season, weather condition, location, habitat quality, size, speed, and noise of approaching vessels, angle of approach) and individual parameters (body condition, body size, flock size, previous experiences, molting stage, state, personality; e.g., Madsen, 1985; Burger and Gochfeld, 1991; Schwemmer et al., 2011). Notably, habitat quality and the state of the individual are likely to have strong effects on escape distances, leading to differences between populations and between different life cycle stages for the same individual. For example, high habitat quality and few habitat alternatives may lead to lower escape distances because birds should avoid leaving profitable areas (Frid and Dill, 2002); however, birds feeding in a high-quality habitat might be in better body condition and thus be able to maximize their safety by flushing earlier (Beale and Monaghan, 2004a). Some species undergo a flightless period during molting, leading to reduced escape distances (own observations); however, the energy demand during this period is high and individuals are thought to be more vulnerable to disturbance (Thiel et al., 1992). During courtship, males are less prepared to leave females and breaking of pair bonds due to disturbance might have a direct negative effect on reproduction. These examples illustrate the facts that escape distances are context-dependent and thus highly variable within species, and that escape distances alone do not translate into vulnerability (Beale, 2007). Although the above parameters generated variation in our study, a recapitulation of their effects was beyond the scope of the current study. Our data were collected over a large study area and at different seasons to represent birds in different states and different habitats to allow the calculation of a mean escape distance per species, which could

serve as an indicator of the probability of disturbance responses for that species.

Common scoters and loons are known to escape at long distances in front of ships or low flying planes and helicopters (Camphuysen et al., 1999; Garthe and Hüppop, 2004; Kaiser et al., 2006; Thaxter and Burton, 2009; Schwemmer et al., 2011), but documented information on the disturbance behaviors of other species at sea is limited. The flush distances of seaduck flocks presented by Schwemmer et al. (2011) were similar to the current results, in terms of both absolute values and proportions between species, suggesting consistency of our method and a negligible observer effect (see also Guay et al., 2013). Kaiser et al. (2006) measured flush distances of common scoter flocks of 1,000–2,000 m using radar, which also falls within the range of our observations. The values for velvet scoters, long-tailed ducks, and loons given by Bellebaum et al. (2006) were not directly comparable because of the different methods used, but the ratios between species were similar to those in the present study. Flush distances at sea appear to be considerably higher than in estuarine areas (Ronconi and Clair, 2002; McFadden et al., 2017). This could be explained by habituation to vessel traffic, which is usually high in estuarine areas. Another explanation might be longer sighting distances in the open seascape. Differences in escape distances between habitats highlight the importance of studying the responses within the area of interest to define setback distances (Rodgers and Schwikert, 2002; Blumstein et al., 2003; McFadden et al., 2017) for a specific habitat.

Notably, use of the mean escape distances calculated here must take account of the fact that they were based on ships of a specific size and speed, and bigger and/or faster ships might induce longer escape distances and higher proportions of flushed birds. Furthermore, the mean escape distances were still likely to be underestimated, because birds flushing at longer distances were less likely to be detected or the species was less likely to be identified. Finally, we only measured visible disturbance responses, which are energetically the most costly reactions to disturbance. However, physiological stress responses, measurable as corticosterone levels (Cockrem, 2007; Fowler, 1999) and increased heart rate (e.g., Nimon et al., 1995), which is linked to an elevated metabolic rate (Green, 2011), commence well before behavioral changes become visible (Weimerskirch et al., 2017). Similarly, individuals might still experience stress despite behavioral habituation, as shown in penguins habituating to human disturbance (Walker et al., 2006). Thus a reduced escape distance in certain areas, which could be interpreted as indicating habituation or lowered vulnerability, might also be a consequence of limited alternative habitat. Setback distances should thus be considerably higher than mean escape distances to minimize the physiological effects of disturbance.

Disturbance Vulnerability Index

Ship traffic disturbance has been the subject of a few vulnerability indices in the past. Camphuysen et al. (1999) evaluated the behavioral sensitivity of seabird species to traffic disturbance, but did not include conservation status. In Garthe and Hüppop (2004), sensitivity to disturbance by ship and helicopter traffic was ranked by expert judgment as one factor for assessing seabird vulnerability to offshore wind energy developments. Both these indices ordered species similarly to the current index with respect to behavioral sensitivity, highlighting the consistency and reliability of expert-based evaluations, even when different methodologies are used.

Similar to other vulnerability indices, the current DVI depends on the selected factors, the scoring system, and the relative grouping and weighting of the factors. Another limitation of our index is that it only covers species occurring in German waters, and therefore does not include some especially sensitive species such as common loon (*Gavia immer*). We therefore suggest including measurements of escape distances and records of disturbance responses from other European monitoring programs to broaden the set of species. This might also provide more information on intra-specific behavioral differences and allow adjustment of population-specific vulnerability indices to account for variations in the selected factors between populations.

For the DVI, we only chose those factors that we considered to be most relevant for estimating the total costs of disturbance, and thus gave all the chosen factors the same weight. We included behavioral factors as well as population status factors, and combined them in a systematic way to estimate the total costs of disturbance. All but one factor, habitat use flexibility (evaluated by Garthe and Hüppop, 2004), were based on real data. Habitat use flexibility was used to indicate how well a bird can compensate for the cost of lost opportunities. Feeding time needed to meet energetic requirements would be a meaningful additional indicator for the ability to compensate for losses (Mayhew, 1988; Madsen and Fox, 1995; Wisniewska et al., 2016). However, integration of this factor in the index would require a detailed knowledge of the species' activity budgets, including activity at night, and time needed for digestion and recuperation from diving. Although such information can be obtained from telemetry studies, it is not yet available for most species. The application of telemetry data in behavior-based and individual-based models also helps to improve predictions regarding the impacts of disturbance on habitat use, survival, and reproduction (Goss-Custard et al., 2006a). However, even sophisticated modeling techniques may not be able to describe the whole complexity of the interactions between wildlife and humans (May et al., 2019). We therefore consider that vulnerability indices like the one presented here provide the best available solution for assessing the potential vulnerabilities of a large set of species.

Relationship Between Shyness and Vulnerability in Seabirds

Cross-species comparisons and the systematic combination of disturbance-related factors within the DVI also allowed us to draw conclusions about the general relationships among shyness, escape costs, and disturbance vulnerability in seabirds. Theoretical considerations and experimental evidence suggest that visible disturbance responses alone are generally not a good indicator of vulnerability, because birds in good body condition and with sufficient feeding alternatives should flush

earlier than birds short of resources (Gill et al., 2001; Frid and Dill, 2002; Beale and Monaghan, 2004a). In contrast to the expected situation however, shyness and escape costs were positively correlated in the present study. Escape costs also correlated positively with population status, such that several species ranked high in all three factor groups of our index. Among the species studied here, the shyest thus also seemed to be the most vulnerable. We propose that, while escape distance is not a good indicator of vulnerability of individuals within the same species, it may serve as an indicator of vulnerability within a range of species. This is in line with Møller (2008) and Møller et al. (2014), who found that long escape distances were associated with population declines among European and Australian birds. A better understanding of the relationships among shyness, escape costs, and population status might have important implications for conservation management. The next step should thus be to carry out further investigations based on the emerging patterns for seabirds using an extended set of species and possibly populations.

The Dimension of the Problem

Escape costs do not comprise only direct energetic costs and reduced energy uptake through lost time for feeding (Owens, 1977; Bélanger and Bédard, 1989); flushed birds might also be displaced from the best feeding resources (Madsen, 1998; Madsen and Fox, 1995). Altered distribution patterns within shipping lanes (Schwemmer et al., 2011) and in relation to vessel traffic to and from offshore wind farms (Mendel et al., 2019) have already been demonstrated in loons. We observed many common scoter flocks flying so far away after flushing that they could not be seen resettling before moving out of sight. Schwemmer et al. (2011) found that most common scoters did not return within 3 h after disturbance by a vessel, while common eiders and long-tailed ducks returned to pre-disturbance numbers within one to 3 h after disturbance. This suggests that very shy species may abandon an area completely, while others may suffer temporary habitat loss.

If birds cannot compensate for energetic losses, disturbance will affect body condition, reproduction, and survival (Madsen, 1985; Goss-Custard et al., 1995a,b, 2006a,b; McHuron et al., 2018). Ducks and geese have been observed feeding at night to compensate for being disturbed during the day (Bélanger and Bédard, 1990; Knapton et al., 2000; Merkel et al., 2009) and shorebirds were shown to increase feeding rates to compensate for lost feeding time (Swennen et al., 1989; Urfi et al., 1996). However, feeding rates and times cannot be extended limitlessly. The time needed to meet energetic requirements determines by how much feeding rates can be increased.

Seabirds might be able to habituate and even adapt to disturbance by ship traffic, if they were able to identify vessels as non-threatening objects. Habituation of birds to particular types of disturbance and within certain areas has been documented before (Smit and Visser, 1993; Samia et al., 2015). For example, among waterbirds, snow geese became accustomed to gunfire (Bélanger and Bédard, 1989) and common eiders and long-tailed ducks showed reduced flush distances within shipping lanes (Schwemmer et al., 2011). However, ships differ greatly in size, shape, speed, and engine noise, making recognizing them as non-threatening objects difficult. Furthermore, waterbirds are hunted using motorboats in some parts of Europe (Laursen and Frikke, 2008). In an environment where predation risk exists, either from natural predators or human activity, birds are thus likely to regard big moving objects as potential threats, and the potential for habituation among sensitive species seems very limited under the current conditions. Notably, even after decades of intense ship traffic in European waters, most species still reacted strongly to our experimental disturbance.

CONCLUSIONS AND RECOMMENDATIONS

This study provides further evidence for the disturbance effects of ship traffic on seabirds, and additional information on the species-specific responses. We present the first comprehensive set of data on escape distances and relative numbers of escaping birds, covering most seabird species found in Northwest European waters. Our findings are based on extensive field experience and data sets from up to 17 years of behavioral observations of seabirds at sea. We show that species differ strongly in their disturbance responses. Our DVI is based on a comprehensive set of variables and addresses a wide range of disturbance-related aspects of bird ecology. It allows for objective quantification of species' vulnerability to ship traffic disturbance. It identified red-throated loon, black guillemot, Arctic loon and velvet scoter as the most vulnerable species to ship disturbance, and common and Arctic tern as the least vulnerable. The shyest seabird species also had the highest escape costs and seemed to be the most vulnerable.

The strong reactions of several species to disturbance by ships have important management implications. Because most human activity at sea involves vessel movements, their effects should be considered more comprehensively in environmental impact assessments and conservation planning. Our DVI can be used to inform marine spatial planning, conservation management, and impact assessments to identify and prioritize the most vulnerable areas in a practical way: The values given here can be multiplied by rasterized species abundances and then summed over all species in a raster cell to produce vulnerability maps (see Garthe and Hüppop, 2004; Sonntag et al., 2012 for examples). Following this, a possible management solution would be to implement low-disturbance or disturbance-free zones in some marine protected areas (von Nordheim et al., 2006). An alternative option in other vulnerable areas could involve the spatial and temporal coordination of ship traffic, which might be especially relevant in areas where ship traffic has increased dramatically as a result of the construction and maintenance of offshore wind turbines. The displacement of sensitive seabirds from wind farm areas has been shown to be a combined effect of the wind farm itself and the associated ship traffic, but these effects are difficult to separate (Mendel et al., 2019). Our study

demonstrated strong reactions to ships alone, emphasizing the importance of minimizing wind farm-associated ship traffic in order to achieve an environmentally friendly transition to the use of renewable energies. Vulnerability maps produced using our DVI can be applied in this context to identify the most suitable corridors.

Ship traffic might also have to be included in single-species action plans as a relevant threat to declining seabird species identified as most vulnerable to ship traffic disturbance (see Wildfowl and Wetlands Trust, 2015 for an example of a species action plan under AEWA). The mean flush distances for such species reported in this study can be used to develop species-specific conservation measures, such as setback distances from important feeding and resting sites. In this context, the setback distances should be longer than the mean escape distances in order to minimize the behavioral and physiological effects of disturbance. Setback distances might also be necessary with respect to human activities in the vicinity of marine protected areas to conserve their proposed function as refuges for vulnerable species.

Clearly, the effects of disturbance events by ships are cumulative and equate to net habitat loss (see also Madsen and Fox, 1995). If regulation of ship traffic is applied as a management tool, threshold levels are needed above which species' abundance will be significantly reduced. The results of this study highlight the fact that these thresholds will be species-specific, and need to be investigated further.

ETHICS STATEMENT

Data collection was carried out within the German Marine Biodiversity Monitoring and did thus not require approval of an ethics committee.

AUTHOR CONTRIBUTIONS

KF and SG contributed to conception and design of the study. PS developed the methodology for measuring escape distances. NM and SG organized the database. KB, KF, NG, and NM conducted field work and recorded data. KF performed the analysis and wrote the first draft of the manuscript. KB, NG, NM, PS, and SG provided critical revision and discussions, and corrected the manuscript. All authors contributed to manuscript revision, and read and approved the submitted version.

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Linking Use of Ship Channels by West Indian Manatees (*Trichechus manatus*) to Seasonal Migration and Habitat Use

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Research on marine mammal occurrence in ship channels often focuses on large cetaceans in offshore shipping routes, while nearshore research largely addresses small vessel strikes. Marine mammals, such as the West Indian manatee, that reside in or migrate through nearshore areas, have potential to travel through a wide range of channel types, encountering a greater diversity of vessels than previously recognized. We tested the extent and conditions of ship channel use by manatees along the north-central Gulf of Mexico (nGoM) coast by combining data from telemetry-tracked individuals, opportunistic citizen-sourced sightings, and environmental attributes linked to manatee movements. Manatees used both nearshore boat channels (130 and 300 m wide) and open water fairways but used nearshore channels much more frequently, consistent with habitat requirements. Satellite-tracked individuals swam faster and moved more directly in all channel types, indicating use of these channels as migratory and travel corridors. Accordingly, generalized additive models revealed that manatees used channels most often during spring/early summer and fall and at temperatures coincidental with entry to and exit from the nGoM during migration. Manatees also occurred in ship channels when freshwater discharges were low, likely because timing of peak manatee occurrence in the nGoM coincides with seasonally low discharge periods. Expanding shipping activity world-wide is likely to increase interactions between marine mammals and a variety of vessel types, and these effects may be particularly impactful to migratory animals like manatees that use nearshore habitats at the interface of recreational boating and commercial shipping. Linking near- and offshore ship channel use to migration and habitat use will better aid risk-assessment for vessel collision and other shipping related activities for migratory marine species globally.

Keywords: movement ecology, citizen science, generalized additive models, satellite telemetry, northern Gulf of Mexico, fairway, vessel strike

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INTRODUCTION

Boating and shipping activity are ubiquitous to global oceans and coasts. The intensity of shipping has increased drastically during the last several decades to accommodate the expanding global economy (Tournadre, 2014). More than 41,000 large merchant vessels were in operation globally in 2016, and nearly 16 million recreational vessels were in use in the United States alone as of 2017

(USDOT, 2016; National Marine Manufacturers Association, 2017). Vessel traffic can negatively impact marine mammals and other marine fauna by generating chemical and noise pollution (Weilgart, 2007; Liubartseva et al., 2015; Pirotta et al., 2019) and causing direct injury and mortality via collisions (Carrillo and Ritter, 2010; van der Hoop et al., 2015; Pirotta et al., 2019). Most research on vessel interactions with marine megafauna focuses on large cetaceans in offshore shipping routes or recreational boat collisions in nearshore waters (Laist et al., 2001; Laist and Shaw, 2006; van der Hoop et al., 2015; Edwards et al., 2016; Crum et al., 2019). Pressure from shipping, however, is increasing in nearshore as well as offshore channels (Tournadre, 2014; Zhang et al., 2015) and will likely affect smaller cetaceans, pinnipeds, and sirenians that live in nearshore areas at the interface of recreational and commercial channels. Little research has focused on the use of nearshore channel types by marine mammals.

The West Indian manatee (*Trichechus manatus*) is a sirenian vulnerable to multiple types of vessel interactions in nearshore areas. The Florida (*T. m. latirostris*) subspecies (Domning and Hayek, 1986) inhabits estuarine and nearshore coastal waters of southeastern North America (Gannon et al., 2007; Hieb et al., 2017), where they frequently interact with humans (**Figure 1**). Vessel collisions are the primary anthropogenic cause of injury and mortality in Florida manatees (Ackerman et al., 1995; Wright et al., 1995; Aipanjiguly et al., 2003). Manatees are frequently exposed to potential vessel collisions while foraging in shallow water, using thermal refuges, and moving among essential habitats (Flamm et al., 2005; Bauduin et al., 2013). The risk of vessel collision is additionally complicated during warm seasons when some manatees leave thermal refuge sites and migrate to widely distributed foraging areas (Deutsch et al., 2003; Cummings et al., 2014; Hieb et al., 2017). Thus, understanding manatee movement and habitat use in areas of high boat traffic is essential to conservation and management efforts (Flamm et al., 2005; Calleson and Frohlich, 2007; Bauduin et al., 2013; Rycyk et al., 2018).

While well-documented historically in the north-central Gulf of Mexico (nGoM; Powell and Rathbun, 1984; Fertl et al., 2005; Pabody et al., 2009), manatee sightings have drastically increased along the Alabama, Mississippi, and Louisiana coasts in warm months during the last several decades (Fertl et al., 2005; Pabody et al., 2009; Hieb et al., 2017). Increasing manatee population numbers in western and northwestern Florida may contribute to the increasing number of individuals migrating to the nGoM during recent years (Runge et al., 2004), but cold winter temperatures in the region limit year-round occupancy (Irvine, 1983; Bossart et al., 2003; Hieb et al., 2017). Cold-induced physiological problems can result in serious disease or death (Irvine, 1983; Bossart et al., 2003), forcing manatees in the United States to return to Florida to overwinter (Deutsch et al., 2003; Reep and Bonde, 2006). Thus, similar to many terrestrial, mammalian herbivores that migrate seasonally among foraging habitats, some manatee individuals move between winter ranges in peninsular Florida and summer ranges farther north (Deutsch et al., 2003; Ferguson and Elkie, 2004; White et al., 2010; Cummings et al., 2014), and other sirenian species follow similar migration patterns (Sheppard et al., 2006; Arraut et al., 2010; Castelblanco-Martínez et al., 2013).

Manatees, like other migratory species, face energetic costs during migrations and risk exposure to extreme conditions and dangerous situations (Sumich, 1983; Hein et al., 2012; Hopcraft et al., 2014; Le Corre et al., 2017). To reduce these costs, animals minimize the duration of migration by moving relatively fast and using direct routes (Sheppard et al., 2006; Tudorache et al., 2007; Åkesson et al., 2012). Animal trajectories during migration are often relatively straight, containing long step lengths (i.e., the distance between GPS pings) and small angles between steps that allow them to take the most direct migratory routes (Edelhoff et al., 2016; Michelot et al., 2017). These distinct movement patterns can be used to define areas and time periods of migratory and travel corridor use among tagged animals when direct observation is not possible.

The most direct migration routes for manatees in the nGoM overlap with an extensive network of ship channels (Gray et al., 2003; Zhang et al., 2015; U.S. Army Corps Of Engineers, 2018). This network of channels includes smaller, nearshore boat channels (**Figures 1D,E**) used by recreational boaters, commercial fisherman and shrimpers, barges, and other shipping vessels, as well as open water fairways (**Figure 1F**) that are mostly offshore and used by larger shipping freighters, seismic vessels, and cruise liners (Gray et al., 2003; Zhang et al., 2015). The Intracoastal Waterway (ICW), a heavily used route between Texas and Florida, and shipping fairways connect to inshore ports and rivers via overlapping boat channels and fairways and can convey larger vessels miles inshore. All of these channels pass through areas frequented by manatees (Hieb et al., 2017), potentially introducing them to various risks. The nearshore channels are typically used by smaller, faster vessels that allow less time for manatees to effectively react and avoid collisions (Nowacek et al., 2004; Laist and Shaw, 2006; Calleson and Frohlich, 2007; Rycyk et al., 2018). Larger vessels found in channels and fairways often move at slower speeds, but their large size may make collisions more lethal (Rommel et al., 2007; Rycyk et al., 2018). The collective use of these different ship channels is unknown but may put migratory species such as manatees at greater risk as species distributions and habitat change through time. Hence, defining the use of ship channels by manatees traveling in nearshore waters is necessary to assess risk from associated vessel interactions and support conservation for these and other migratory species.

We combined data from telemetry-tracked manatees and from opportunistic sightings to determine when and under what conditions manatees use the combination of ship channels in the nGoM. To determine if manatee movements were consistent with the use of channels as travel corridors and for migration, we compared swim speed, step lengths between GPS pings, and angles between steps for manatees in and outside channels (Michelot et al., 2017). We used generalized additive models (GAMs) to determine if nGoM water temperature, local air temperature, and freshwater discharge (as a proxy for salinity) affected manatee use of boat and ship channels at 2-week (fortnight) intervals year-round. This study has applications

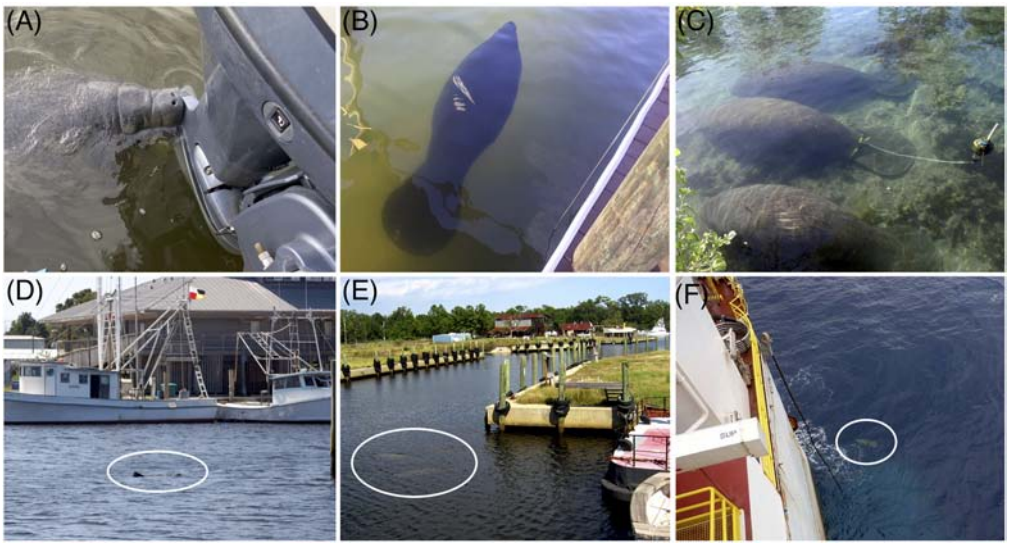

FIGURE 1 | Typical examples of manatee interactions with vessels and channels. Manatee nudging a boat motor in Horseshoe Beach, FL 2013 **(A)**. Fresh propeller scars on a manatee in Bayou St. John, AL 2018 **(B)** and healed propeller scars (bottom animal below tagged manatee) at Florida wintering ground 2012 **(C)**. Manatees traveling in local commercial and recreational harbors in Apalachicola, FL in 2010 and Bayou La Batre, AL 2008, respectively **(D,E)**. Manatee next to a seismic vessel in a fairway in the Gulf of Mexico 2013 **(F)**. Photo credits: DISL/MSN staff and contributors.

for ongoing management and recovery efforts for manatees and other species that occupy similar nearshore habitats, and it introduces a robust approach to help delineate the factors affecting channel use by many megafauna species.

MATERIALS AND METHODS

Study Area

We examined manatee occurrence in ship channels along the nGoM, defined as eastern Louisiana, just west of Lake Pontchartrain (91°W) to western Florida at Pensacola Bay (87°W). Latitudinal boundaries for the study area were defined as the northern Mobile-Tensaw Delta (31°N) to the southeastern Louisiana coast (29.5°N). Longitudinal and latitudinal boundaries were based on regional manatee occurrence data and habitat requirements (Fertl et al., 2005; Pabody et al., 2009; Hieb et al., 2017). Channel and fairway locations in the nGoM were based on established GIS data layers from the US Department of Transportation National Transportation Atlas Databases (NTAD), U.S. Army Corps of Engineers Navigable Waterway Network, and the Bureau of Ocean Energy Management (BOEM) Data Center. We measured nearshore boat channels using ArcMap Version 10.3 (ESRI, 2014) following the NTAD "Navigable Waterways" GIS polyline layer with polygons at nGoM regional channel widths of 130 and 300 m (Gray et al., 2003; U.S. Army Corps Of Engineers, 2012; BIST, 2013). After initial analyses, we combined manatee occurrence data for the 130 and 300 m channels as representative of manatee occurrence in the nearshore boat channels throughout the region.

Data Collection

Telemetry Data

Manatee movement data were collected via GPS satellite telemetry, using established methods for capturing and tagging manatees (Bonde et al., 2012). We captured a total of 10 manatees during September 2009, August 2010, August 2012, and September 2014 (**Table 1**). Manatees were located via an aerial observer, and individual manatees were captured in a net deployed from a specialized capture boat (Bonde et al., 2012). Each manatee was then hauled aboard the boat and underwent a full health assessment onboard prior to fitting with a floating, tow-behind tagging platform (Telonics Inc., Mesa, AZ, United States; Bonde et al., 2012). Tags were attached to the peduncle with a belt and tether following standard tagging procedures for manatees (Deutsch et al., 1998; Weigle et al., 2001). Tags were programmed to record a GPS location every 15 or 30 min and locations were downloaded following tag recovery. Capture and release occurred at the same location, and manatees were typically released within 1 h of capture. Following initial tagging, some individuals lost tags, but retained the peduncle belts, and a new tag and tether combo was attached opportunistically at a later date by snorkeling or during a subsequent capture event. Tags were recovered by removing them at the end of their battery life and replacing with a new tag or when accidental loss from the animal occurred due to breaking at weak points in the belt or tether, which are designed to breakaway to protect against entanglement. Tag recovery was 98% because manatees use shallow, coastal systems and lost tags are easily located.

Continuous tagging duration was from 2 weeks to 13 months and non-continuous tagging (i.e., an individual tagged more

TABLE 1 | Manatee capture and retagging events.

Manatee ID	Sex	Tagging period	Tagging location
TMA001	F	September 2009 – July 2010	Delvan Bay, Mobile-Tensaw Delta, Mobile, AL, United States*
		January 2011 – June 2011	Three Sisters Spring, Crystal River, FL, United States
TMA002	M	September 2009 – September 2009	Delvan Bay, Mobile-Tensaw Delta, Mobile, AL, United States *
		February 2010 – January 2011	Three Sisters Spring, Crystal River, FL, United States
TMA003	M	August 2010 – August 2011	Dog River, Mobile, AL, United States*
TMA004	M	August 2010 – September 2010	Dog River, Mobile, AL, United States*
		August 2012 – September 2012	Polecat Bay, Mobile-Tensaw Delta, Mobile, AL, United States*
		October 2012 – October 2013	D'Olive Creek, Mobile-Tensaw Delta, Mobile, AL, United States
		December 2013 – March 2014	Idiot's Delight, Crystal River, FL, United States
TMA005	M	August 2010 – September 2010	Spanish River, Mobile, AL, United States*
		August 2011 – August 2011	Wakulla River, Wakulla, FL, United States
		August 2012 – December 2012	Magnolia Springs, Crystal River, FL, United States*
		January 2013 – August 2013	Kings Bay, Crystal River, FL, United States
TMA006	M	August 2012 – December 2012	Rattlesnake Bayou, Dog River, Mobile, AL, United States*
		February 2013 – August 2013	Teco Power Plant, Tampa Bay, FL, United States
TMA008	F	August 2012 – September 2012	Raft River, Mobile-Tensaw Delta, Mobile, AL, United States*
TMA010	M	August 2012 – January 2013	Delvan Bay, Mobile-Tensaw Delta, Mobile, AL, United States*
		March 2013 – April 2013	Magnolia Springs, Crystal River, FL, United States
		May 2013 – August 2013	Horseshoe Key, FL, United States
		December 2013 – April 2014	Three Sisters Spring, Crystal River, FL, United States
		November 2014 – September 2015	Magnolia Springs, Crystal River, FL, United States
		December 2015 – February 2016	Magnolia Springs, Crystal River, FL, United States
TMA011	M	September 2014 – October 2014	Dog River, Mobile, AL, United States*
TMA012	M	September 2014 – March 2015	Dog River, Mobile, AL, United States*

*Gaps in time are due to researcher removal of tag or accidental loss. Manatees were either tagged at health assessments (denoted with *) or a new tag was attached to a belt, which had been fitted during a prior capture event, following removal or loss.*

than once) duration ranged from 4 to 22 months. We monitored animal locations using ARGOS service and with focal visual observations taken periodically while animals were tagged (Deutsch et al., 1998; Weigle et al., 2001). Data were downloaded directly from the tagging platforms and included standard GPS locations or quick-fix pseudoranging (QFP) positions accurate within 10 or 75 m, respectively. All locations with a successful GPS fix or a resolved QFP were included in the dataset, which were plotted in ArcMap 10.3 to verify accuracy of locations and all locations on land were removed (0.00006%). All work on manatees was performed under US Fish and Wildlife Service Permits MA107933-1 and MA37808A-0, Alabama Department of Conservation and Natural Resources Wildlife and Freshwater Fisheries Division annual permits, and University of South Alabama IACUC protocols 581568 and 1038636.

Sighting Data

We used manatee sighting data reported to the Dauphin Island Sea Lab's Manatee Sighting Network (DISL/MSN), a citizen science program in the nGoM (Pabody et al., 2009). Sightings were compiled from DISL/MSN records to include live animal sightings obtained from public reports and targeted research efforts. Opportunistic, publicly reported sighting data were collected during 2007–2017 as described by Pabody et al. (2009) and Hieb et al. (2017). At a minimum, publicly reported sightings included the date, location, and number of manatees per sighting. Publicly reported sightings that did not provide sufficient location descriptions to derive GPS coordinates were excluded from the dataset. Duplicate sightings, which were defined as sightings reported by multiple observers but occurring at the same location, date, and time (within 20 min), were removed from the dataset. We augmented opportunistic sighting data with sightings during research efforts from 2009 to 2017 in Alabama, Mississippi, and Florida, including aerial observations during health assessments and surveys, boat-based monitoring by trained observers, and sightings of live animals during stranding response and rescue efforts.

Environmental Conditions

To examine relationships among manatee ship channel use and local environmental conditions, we divided the study area into five sub-regions defined longitudinally by major waterbodies: Lake Pontchartrain, Bay St. Louis, Pascagoula River, Mobile Bay, and Pensacola Bay (**Supplementary Figure S1** and **Table 2**). For each sub-region, data for nGoM water temperature, local air temperature, and discharge were averaged for the week prior to each GPS ping or sighting date to represent conditions leading up to and during channel use. In cases when water temperature data were unavailable for the nGoM (14%), data were averaged for that fortnight period from other years the data were available. We obtained nGoM water temperature from 42040-Luke buoy located 63 nautical miles south of Dauphin Island, AL (29.208 N 88.226 W). We used data from the National Oceanic and Atmospheric Administration's (NOAA) National Data Buoy

TABLE 2 | Weather and gauging stations for longitudinally defined sub-regions within the study area.

Sub-region	Western boundary (°W)	Eastern boundary (°W)	Weather station	River and gauging station
Lake Pontchartrain	91	>89.5	Station NWCL1 New Canal Station	Mississippi River USGS 07374000
Bay St. Louis	89.5	>89	Station WYCM6 Bay Waveland Yacht Club	Pearl River USGS 02489500
Pascagoula River	89	>88.25	Station DKCM6 Dock C, Port of Pascagoula	Pascagoula River USGS 02479000
Mobile Bay	88.25	>87.75	Station MBLA1 Middle Bay Lighthouse	Alabama River USGS 02428400
Pensacola Bay	87.75	>87	Station PCLF1 Pensacola	Escambia River USGS 02376033

We obtained air temperature from weather stations and discharge (cf^{-1}) from gauging stations.

Center to calculate average local air temperature from centrally located buoys within each sub-region (**Table 2**). Discharge data were obtained from the US Geological Survey (USGS) National Water Information System[1] for major discharge sources in each sub-region (**Table 2**).

Statistical Analyses and Modeling

To determine differences in movement of tagged manatees inside *versus* outside of ship channels, we estimated manatee speeds, step lengths (i.e., the distance traveled between two GPS pings), and angles between step lengths (Edelhoff et al., 2016). To reduce error in our estimate of step length caused by large differences in the durations between GPS fixes, we only fit estimated step lengths for path segments where durations between GPS fixes were 25 – 35 min, and for tags that transmitted every 15 min, we only used fixes with 30-min durations (i.e., every other GPS fix). We estimated speed in kmh^{-1} as the step length between two successive GPS ping locations divided by the time between pings. We used Welch's *t*-tests to determine differences in manatee speeds and natural-log transformed step lengths inside *versus* outside different channel types. Angles between step lengths were calculated using the '*moveHMM*' package in R (Michelot et al., 2016), and we used a circular ANOVA, with the R package '*circular*' (Agostinelli and Lund, 2013) to determine differences in angles inside *versus* outside of ship channels.

We used GAMs to determine how environmental variables—time-of-year (fortnight), nGoM water temperature, local air temperature, and freshwater discharge—affected manatee use of ship channels. We used fortnight because a 2-week period balanced a finer temporal resolution with the ability to compare detections and sightings at a seasonal scale. We constructed four sets of GAMs with the package *mgcv* in R (Wood, 2012); one set for 130 and 300 m boat channels and a second set for fairways for both tagged manatees and manatee sightings. To determine if collinearity existed among environmental variables, we used paired scatterplots and Pearson's correlation statistics (Zuur, 2009) and found that nGoM water temperature, local air temperature, and discharge were all correlated with fortnight, but these correlations were too weak for collinearity to affect the results of the model (Dormann et al., 2013). Therefore, each GAM set included 20 models containing a combination of individual variables and their interactions (**Supplementary Tables S1–S4**). We used penalized thin plate smoothing splines to determine the main effects because they have been demonstrated to be the best smoother for these types of models (Wood, 2003). We used tensor smoothing to determine the interactions because this method is scale and unit invariant and can generalize between one- and multiple-dimensions (Wood et al., 2013).

For tagged manatees, we divided the number of GPS pings each day detected within each channel type (130 and 300 m channels and fairways) by the total number of pings from that day, resulting in the proportion of pings in each channel type per day. We used binomial models for tagged manatees with the *logit* link, as is standard with proportional data, and, because the total number of detected GPS pings varied each day, we used the total number of pings per day as an offset in the models (Zuur, 2009). For manatee sightings, we used binomial models with the *logit* link, as is standard with binomial data. To simplify models for sighting data, we excluded year if it was an insignificant predictor of ship channel use. We validated models graphically and tested the fit of the residuals.

To determine the best-fitting models for tagged manatees and manatee sightings in the 130 and 300 m channels and fairways, we used an information criterion approach (Johnson and Omland, 2004; Burnham et al., 2011). In this approach, we calculated Akaike information criterion (AIC) values as an indicator of model fit, where lower AIC values indicate better fit (Johnson and Omland, 2004). To measure the relative strength of each model, we calculated the normalized Akaike weights, *w,* for each model *i,* where $w_i = \frac{e^{-0.5^* \Delta AIC_i}}{\sum_{r=1}^{R} e^{-0.5^* \Delta AIC_i}}$, which is the probability that the given model is the best approximating model (Burnham et al., 2011; Symonds and Moussalli, 2011). Values of $w > 0.9$

[1]https://waterdata.usgs.gov

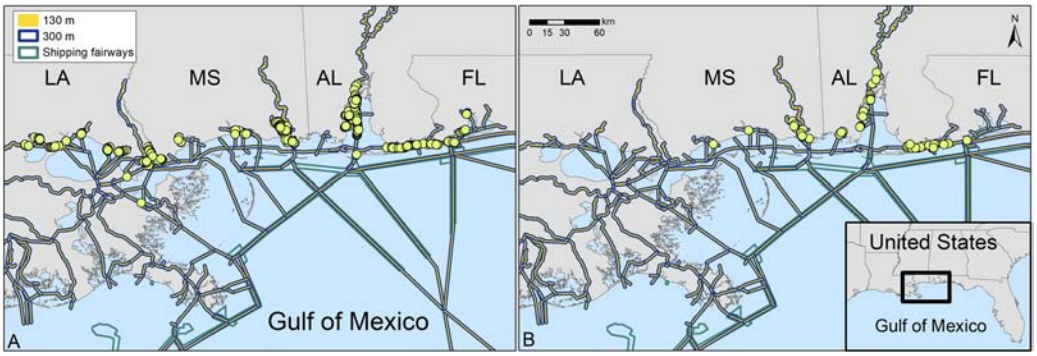

FIGURE 2 | Maps of the nGoM with boat channels and fairways and locations of tagged **(A)** and sighted **(B)** manatees in ship channels.

TABLE 3 | Number of tagged and sighted manatees in 130 and 300 m boat channels and shipping fairways (FW) and percentage of detected or sighted individuals in each type of channel.

				Channel type				
		n_{in} (%)	n_{out} (%)	130 m	300 m	130 m+FW	300 m+FW	FW
Sighted	# (%)	237 (11)	2, 000 (89)	166 (70)	47 (20)	3 (1)	3 (1)	18 (8)
Tagged	# (%)	1, 525 (5)	29, 646 (95)	676 (44)	633 (41)	39 (3)	61 (4)	116 (8)

The 130 m wide channel is most often the inner core of boat channels while the 300 m wide channel is the periphery; therefore, manatees can co-occur in the 130 and 300 m wide channels, but manatees in the 300 m channel were not in the 130 m. The 130 m + FW and 300 m + FW represent channels where the 130 and 300 m boat channels overlap with fairways, respectively.

indicate strong model support, while $w < 0.9$ indicate that other models may also fit the data well and should be considered in the analysis (Burnham et al., 2011; Symonds and Moussalli, 2011). To determine the relative importance of each variable, we calculated the normalized Akaike weight for each parameter w_{ip}, which is the sum of the w_{im} values for all the models in which that variable was present (Symonds and Moussalli, 2011).

RESULTS

Manatee Use of Ship Channels

Manatees broadly used ship channels across the nGoM, primarily occupying nearshore channels in rivers, canals, estuaries, and the ICW, with fewer manatees detected and sighted in offshore waters (**Figure 2** and **Table 3**). Tagged and sighted manatees followed similar spatial patterns across the study area (**Figure 2**). For the 10 tagged manatees, GPS data included a total of 723 manatee days in the study area: 544 days in Alabama, 80 in Mississippi, 61 in Louisiana, and 37 in Florida. A total of 2,237 manatee sightings occurred from 2007 to 2017, including 1,945 sightings in Alabama, 205 in Mississippi, 79 in Florida, and 8 in Louisiana. Opportunistic, publicly reported sightings accounted for the majority of sighting data (93%) with an additional 150 sightings recorded during research efforts, including 137 sightings in Alabama, 12 in Mississippi, and 1 in Florida.

Tagged manatees were detected in ship channels March – October (**Figure 3A**), and manatees were sighted in ship channels March – December (**Figure 3E**). Both tagged and sighted manatees occurred much more frequently in the 130 and 300 m boat channels than in fairways, including areas where fairways overlapped with boat channels (**Figures 3A,E** and **Table 3**). Tagged manatees occurred in ship channels when nGoM water temperature ranged from 22 to 32°C and local air temperature ranged from 12 to 31°C (**Figures 3B,C**), while manatees were sighted in ship channels when nGoM water temperatures ranged from 20 to 32°C and local air temperature ranged from 7 to 31°C (**Figures 3F,G**). Tagged and sighted manatees used ship channels when freshwater discharges ranged from 9 to 100,733 $\cdot$cfs^{-1} (**Figure 3D**) and 164 to 78,329 $\cdot$cfs^{-1} (**Figure 3H**), respectively. An example of these patterns can be seen in movements of a tagged manatee (TMA001, "Bama") during fall 2009, when she used channels, including the Mobile Bay ship channel to access foraging sites in late October (nGoM water temperature = 26.8°C, local air temperature = 16.8°C, and discharge = 71,598 $\cdot$cfs^{-1}), before exiting the region via the ICW between Bon Secour, AL and Pensacola, FL in early November (**Supplementary Video S1**).

Movements of tagged manatees were different inside compared to outside ship channels. We estimated that manatees swam faster inside the 130 and 300 m boat channels and fairways (130 and 300 m channels: $t_{625.81} = 4.19$, $p < 0.001$; Fairways: $t_{667.56} = 4.72$, $p < 0.001$; **Figures 4A,C** and **Table 4**), and step lengths were significantly longer inside the 130 and 300 m channels and fairways (130 and 300 m channels: $t_{633.32} = 9.75$, $p < 0.001$; Fairways: $t_{683.53} = 11.20$, $p < 0.001$; **Figures 4B,D** and **Table 4**). Step angles were not significantly different in or out of the 130 and 300 m channels or fairways (130 and 300 m channels: $F_{1,26507} = 0.05$, $p = 0.83$; Fairways: $F_{1,26507} = 0.20$, $p = 0.65$; **Supplementary Figure S2**). Tagged

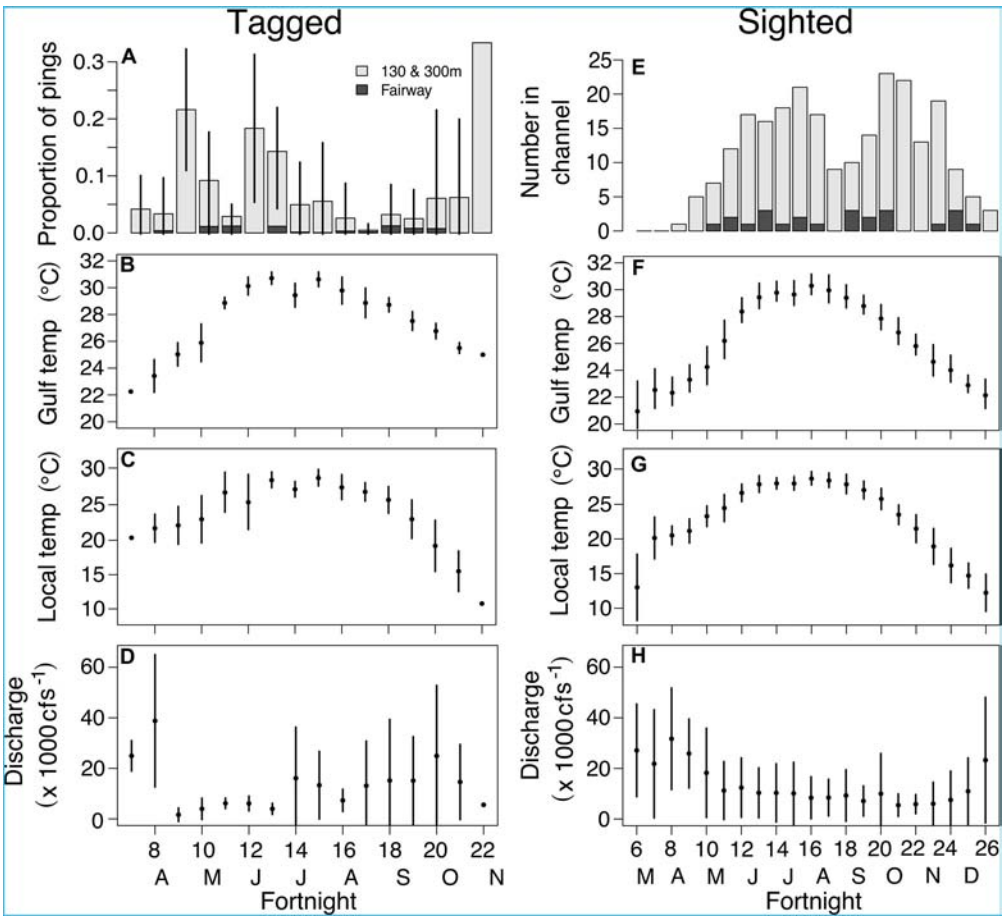

FIGURE 3 | The average proportion of GPS pings (±SD) from tagged manatees (left column) in ship channels per fortnight period **(A)** and the sum of sighted manatees (right column) per fortnight period **(E)**. The average (±SD) nGoM water temperature **(B,F)**, local air temperature **(C,G)**, and freshwater discharge of the nearest river **(D,H)** per fortnight for the periods of time that manatees were detected or sighted in ship channels.

manatees also typically used ship channels for < 5 h (median: 2.36 and 1.60 h for 130 and 300 m channels and fairways, respectively); however, durations in ship channels up to 24 h were not uncommon, and, in several instances, manatees spent > 30 h in the channels (**Figures 5A,C**). Likewise, the distances that manatees traveled continuously in the ship channels were generally < 5 km (median: 3.08 and 1.62 km for 130 and 300 m channels and fairways, respectively), but distances up to 15 km were not uncommon and, in a few instances, manatees traveled up to 35 km in the channels (**Figures 5B,D**).

Modeling of Tagged Manatees

The best-fitting model for telemetry-tracked manatees in the 130 and 300 m boat channels included fortnight, nGoM water temperature, and local air temperature and the interactions between fortnight and nGoM water temperature, local air temperature, and discharge (% Deviance Explained [R^2] = 77%; **Supplementary Table S1**), which were all statistically significant predictors of ship channel use (**Figures 6A–D**, **7A–C** and **Table 5**). The probability of detecting tagged manatees in the 130 and 300 m channels peaked May–June, decreased through the summer, and increased again September–October (**Figure 6A**). The probabilities of detections were highest when nGoM water temperatures were ∼24°C, corresponding to spring (May, June) and fall (October), and ∼31°C during the summer (July, August; **Figures 6B**, **7A**). Results were similar when compared to local air temperatures, with peak use at 10–20°C during spring and fall with an additional peak at ∼31°C during the summer (**Figures 6C**, **7B**). Tagged manatees were most likely to be detected in 130 and 300 m channels at low freshwater discharges during the spring and fall (**Figures 6D**, **7C**) and at very high discharges during the spring (**Figure 7C**). All parameters had high weights except discharge (**Supplementary Table S5**), indicating that discharge had little effect on the fit of the model compared to other variables.

The best-fitting model for tagged manatees in open water fairways included the three interactions (R^2 = 65.3%; **Supplementary Table S2**), all of which were significant (**Table 5**). Manatees were most likely to be detected in fairways when nGoM water temperatures ranged from 24 to 31°C, generally in the late spring and early summer (**Figure 8A**), and local air temperatures

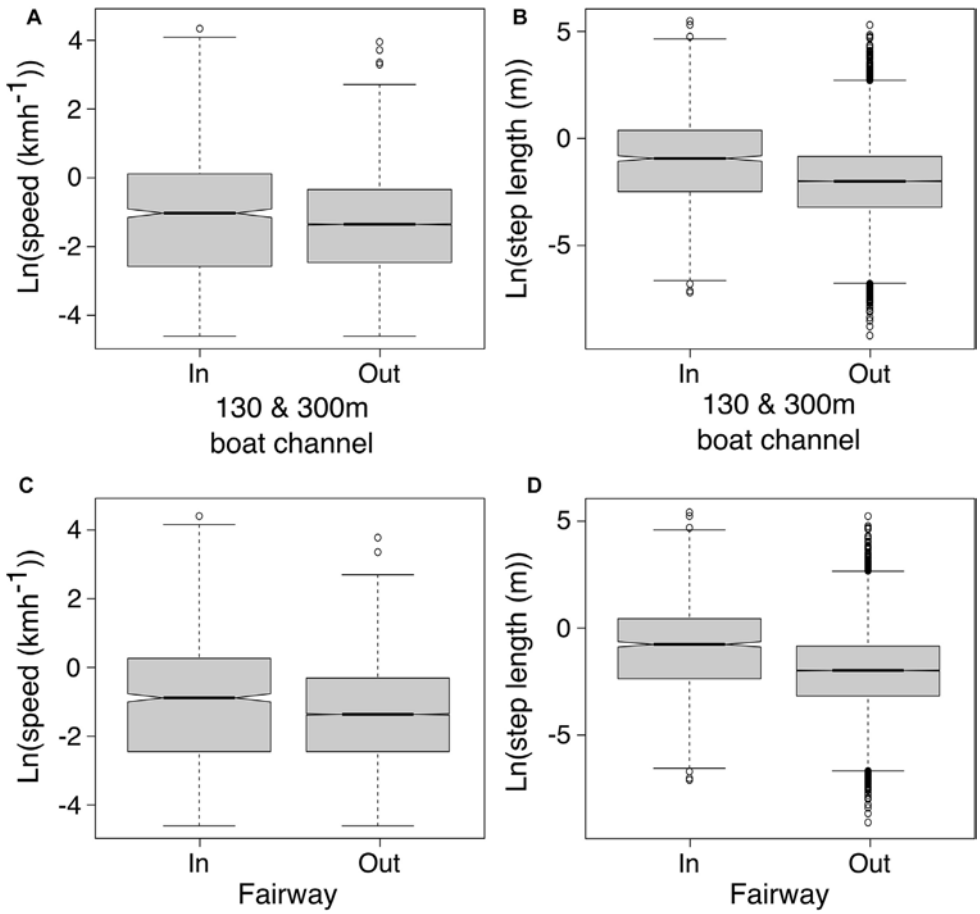

FIGURE 4 | Estimated averages of natural log transformed manatee swimming speeds in and out of 130 and 300 m channels **(A)** and shipping fairways **(C)** and natural log transformed step lengths in and out of 130 and 300 m channels **(B)** and shipping fairways **(D)** for tagged manatees.

TABLE 4 | Estimated average speed and step lengths of tagged manatees inside and outside of 130 and 300 m boat channels and shipping fairways.

Trait	Ship channel type			
	130 and 300 m		Fairways	
	In	Out	In	Out
Speed (kmh^{-1})	1.15	0.59	1.32	0.58
Step length (km)	0.32	0.12	0.35	0.12

ranged from 18 to 30°C, with peaks at ∼22°C in the spring, 26 and 30°C in the early and late summer, respectively, and at ∼19 and 14°C in the fall (**Figure 8B**). The probability of manatee detections in the fairways occurred at relatively low freshwater discharges in the late spring, early summer, and fall (**Figure 8C**).

Modeling of Sighted Manatees

For sighted manatees, year was not a significant predicator of 130 and 300 m channel use (edf = 4.38, χ^2 = 7.279, p = 0.225) and was not included in the model analysis. The best-fitting model included all four environmental variables (fortnight, nGoM water temperature, local air temperature, and freshwater discharge) and all three interactions (R^2 = 12.5%; **Supplementary Table S3**), of which nGoM water temperature, discharge, and each interaction were statistically significant predictors of channel use (**Figures 6F,H**, **7D–F** and **Table 5**). The probability of sighting manatees in the 130 and 300 m channels was relatively high April–June, decreased through the summer, and then increased September – December (**Figure 6E**). Manatees were most often sighted in any ship channels at nGoM water temperatures of 23°C (April–June) and 26°C (October–December) (**Figures 6F**, **7D**) and when local air temperatures ranged between 10 and 25°C in the spring and fall (**Figures 6G**, **7E**). Manatees were sighted most frequently in ship channels when freshwater discharge was low in the spring and fall (**Figures 6H**, **7F**). Two other models, one that did not contain discharge and one that did not contain fortnight, but each contained all other factors and interactions, had relatively high model weights (**Supplementary Table S3**). The high weights of these models, as well as the lack of significance for fortnight and local air temperature and the low parameter weight for discharge (**Supplementary Table S5**), emphasize the relative importance of nGoM water temperature to manatee occurrence in the 130 and 300 m channels.

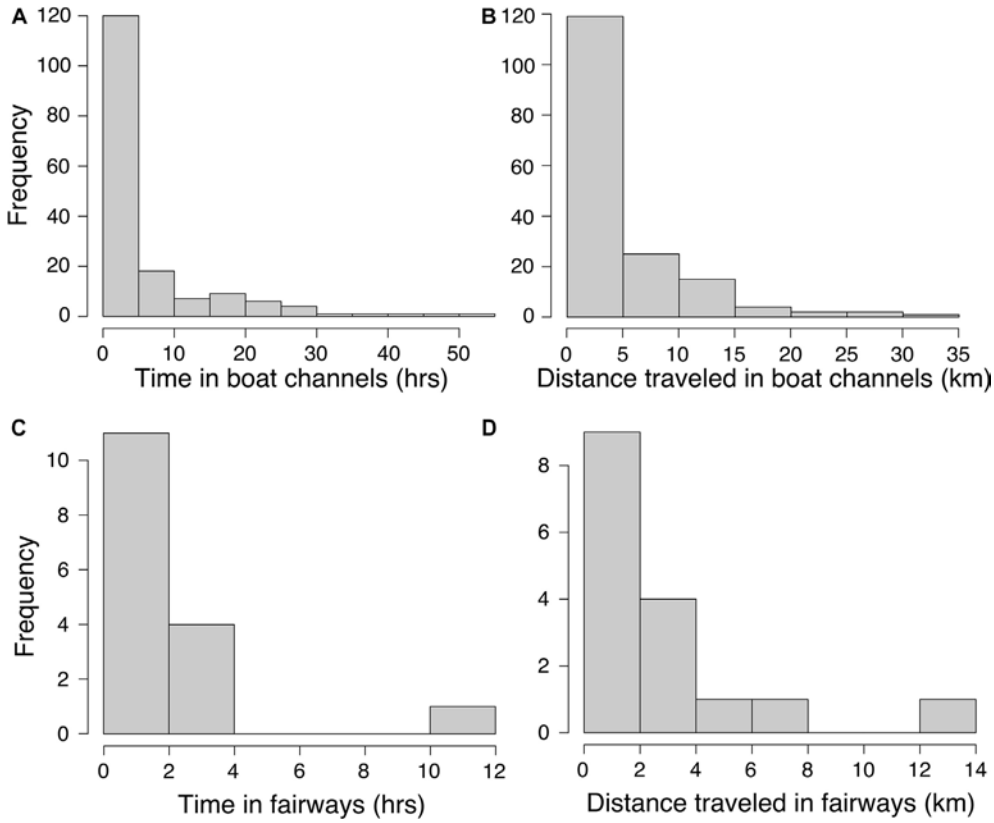

FIGURE 5 | Histograms of continuous time durations that manatees spent in 130 and 300 m boat channels **(A)** and fairways **(C)** and associated distances traveled in the 130 and 300 m boat channels **(B)** and fairways **(D)**.

For manatees sighted in shipping fairways, year was not a significant predictor (edf = 1.003, χ^2 = 2.83, p = 0.09) and was not included in the remaining model selection analysis. The best-fitting model included only nGoM water temperature, local air temperature, and fortnight (R^2 = 8.94%; **Supplementary Table S4**); however, none of these factors were significant predictors of fairway use (**Table 5**). Furthermore, the model containing only local air temperature and another containing only the interaction between fortnight and local air temperature had high weights.

DISCUSSION

Manatee Use of Ship Channels

Manatee movements in ship channels along the nGoM coast suggest they use these channels as migration routes and travel corridors among habitats. Manatees mostly used nearshore boat channels, moving east–west in the ICW, a heavily trafficked shipping route, and north-south in estuaries and rivers, consistent with known foraging habitats and other essential resources (Flamm et al., 2005; Cummings et al., 2014; Hieb et al., 2017). Manatees used the ICW most frequently between Gulf Shores, AL and Pensacola, FL during spring and fall, suggesting they use this channel to enter and exit the nGoM during seasonal migration. Accordingly, movements of tagged manatees in channels were typical of animals during migration or when moving among habitats, where manatees swam faster and had longer step lengths between GPS pings (Tudorache et al., 2007; Åkesson et al., 2012; Michelot et al., 2017). Speeds of manatees inside ship channels in this study were nearly identical to speeds of dugongs performing large-scale movements (Sheppard et al., 2006) but were faster than manatees migrating on the eastern coast of the United States (Deutsch et al., 2003). The similarity of angles between steps inside and outside of shipping channels may be explained by the temporal resolution of GPS pings (Wilson et al., 2013). The duration between GPS detections (∼30 min) could allow manatees moving in a discontinuous or meandering line outside of channels, but following a general heading, to appear to have slower speeds and shorter step lengths, but no difference in turn angle, as was seen in our analysis (Wilson et al., 2013). Additionally, manatee behavior can affect the quality of GPS fixes (e.g., swimming at depths and speeds that submerge the tag), which in turn can affect the step angle calculations. Despite these caveats, manatee movements in ship channels are consistent with use as travel corridors, particularly during seasonal migration, but finer temporal resolution of tagged individuals may reveal more detailed movements to explain differences in movements inside and outside channels.

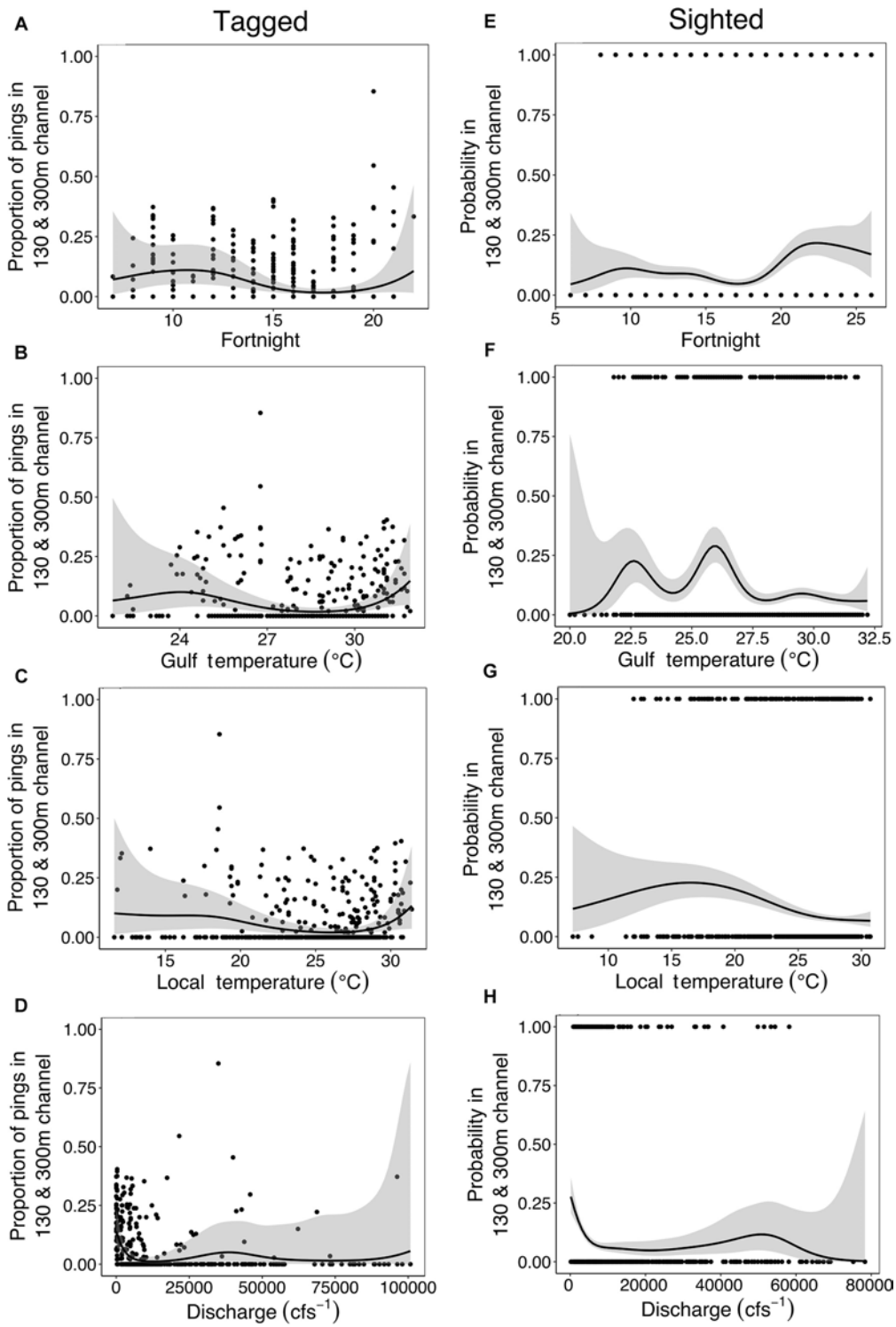

FIGURE 6 | Smoothing curves for fortnight **(A,E)**, nGoM water temperature **(B,F)**, local air temperature **(C,G)**, and freshwater discharge **(D,H)** from the generalized additive models for tagged (left column) and sighted (right column) manatees in the 130 and 300 m boat channels.

Temperature is a migratory cue for manatees and dugongs (Deutsch et al., 2003; Sheppard et al., 2006; Cummings et al., 2014), and manatees used ship channels in our study at temperatures and times of the year associated with migration into or out of the region. Manatees and other sirenians that are susceptible to potentially fatal cold-stress typically migrate

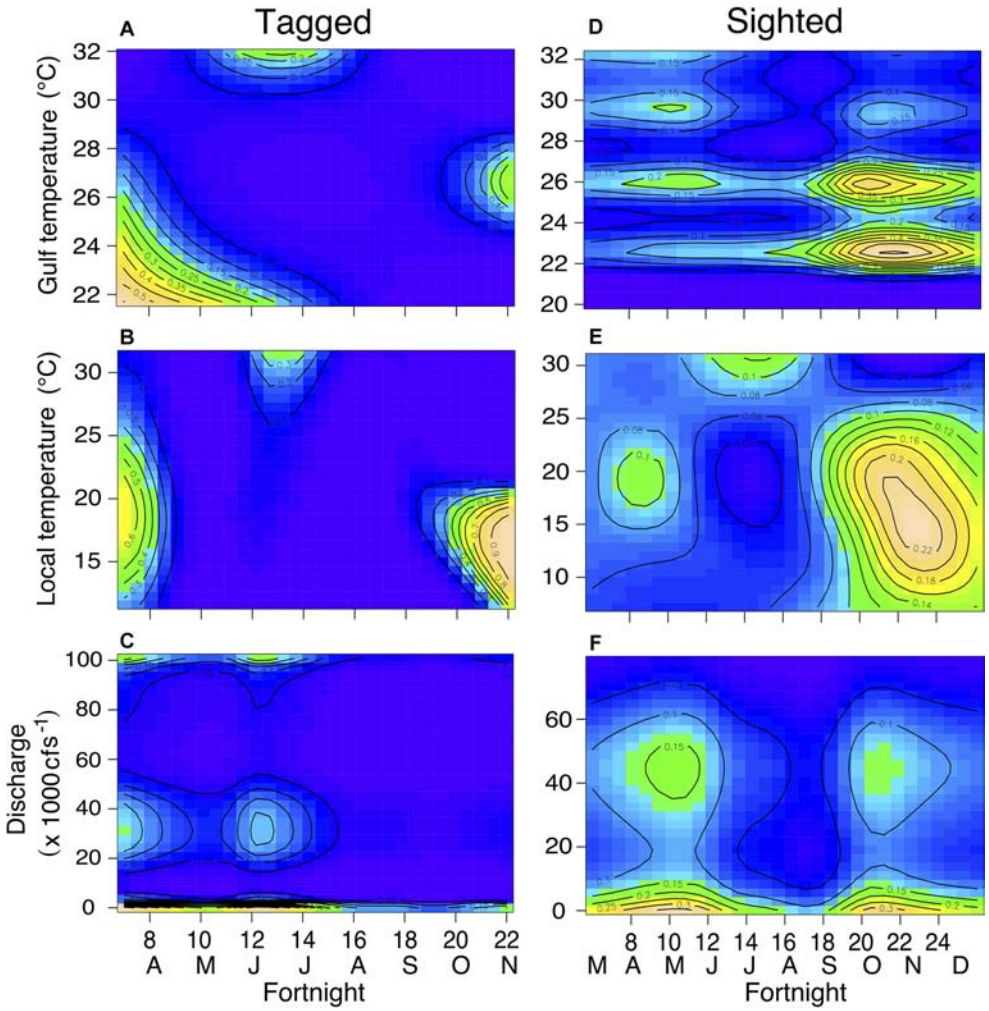

FIGURE 7 | Contour plots displaying interactions between fortnight period and nGoM water temperature **(A,D)**, local air temperature **(B,E)**, and freshwater discharge **(C,F)** for tagged (left column) and sighted (right column) individuals in the 130 and 300 m ship channels. Cooler colors (e.g., blue, green) indicate lower probabilities while warmer colors (yellow, tan) indicate higher probabilities. Contour lines represent probabilities.

from summer ranges to wintering grounds (in peninsular Florida for manatees in the southern United States) when temperatures drop below ∼20°C, a threshold for healthy thermoregulation (Irvine, 1983; Bossart et al., 2003). In our study, manatees most frequently used ship channels during the spring/early summer as temperatures rose above ∼22°C, corresponding to entering the region, and the fall as temperatures dropped below ∼26°C, corresponding to exiting the region (Deutsch et al., 2003; Pabody et al., 2009; Hieb et al., 2017). Tagged manatees used open water fairways at similar times and temperatures as both tagged and sighted manatees in 130 and 300 m boat channels, but, overall, manatees were rarely detected in fairways, which are mostly located in offshore waters and less consistent with manatee habitat requirements (Reep and Bonde, 2006; Gannon et al., 2007; Marsh et al., 2011). These patterns of ship channel use by manatees in the nGoM are very similar to migratory patterns reported to-date along the southern United States Atlantic coast, including use of the ICW (Cummings et al., 2014), suggesting broad use of ship channels as migratory corridors by manatees throughout their United States range. Accordingly, temperature was the most significant predictor of manatee ship channel use in our models. While the interaction between fortnight and nGoM water and local air temperature were similar, local air temperature was consistently a few degrees lower than water temperature and a slightly weaker predictor of channel use. Because air temperature drives local water temperature in this study area (Park et al., 2007; Dzwonkowski et al., 2011), changes in air temperature can indirectly initiate manatee migration. Manatees have been documented to delay their north- or southward migrations during periods of unseasonably cold or warm air temperatures (Deutsch et al., 2003), which may explain some variability in relationships between temperature and manatee ship channel use detected in our study. Thus, while water temperature more directly influences manatee migrations, our data also indicate that air temperature has potential for use as

TABLE 5 | Best-fitting models for tagged and sighted manatees in the 130 and 300 m boat channels and fairways.

Parameter	edf	χ^2	*p*
Tagged: 130 and 300 m Channel			
Fortnight * Gulf Temp	26.914	186.21	<0.001
Fortnight * Local Temp	26.971	234.48	<0.001
Fortnight * Discharge	27.783	396.28	<0.001
Gulf Temp	8.913	71.17	<0.001
Local Temp	8.840	57.14	<0.001
Fortnight	8.996	117.16	<0.001
Tagged: Open Water Fairway			
Fortnight * Gulf Temp	28.000	61.22	<0.001
Fortnight * Local Temp	25.380	45.66	0.010
Fortnight * Discharge	27.580	46.93	0.014
Sighted: 130 and 300 m Channel			
Gulf Temp	8.451	26.84	0.001
Local Temp	8.912	8.90	0.458
Fortnight * Gulf Temp	6.161	10.26	0.004
Fortnight * Local Temp	5.588	12.48	<0.001
Fortnight * Discharge	0.002	0.00	0.051
Fortnight	1.008	0.33	0.569
Discharge	3.655	31.99	<0.001
Sighted: Open Water Fairway			
Local Temp	8.733	6.85	0.651
Gulf Temp	1.000	2.04	0.153
Fortnight	1.001	0.01	0.920

The effective degrees of freedom (edf) is an index of the non-linearity of the parameter.

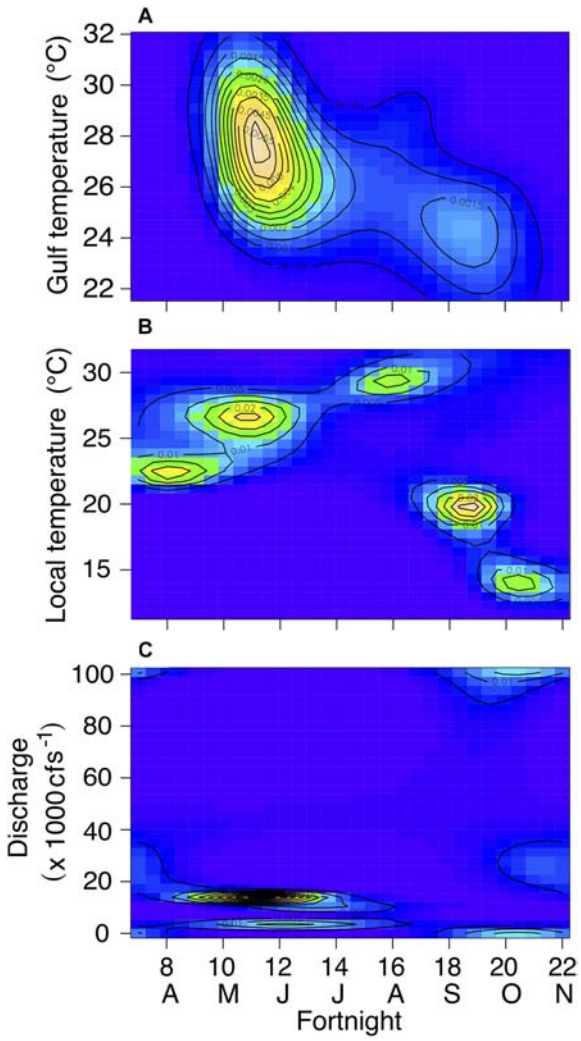

FIGURE 8 | Contour plots displaying interactions between fortnight period and nGoM water temperature **(A)**, local air temperature **(B)**, and freshwater discharge **(C)** for tagged individuals in fairways. Cooler colors (e.g., blue, green) indicate lower probabilities while warmer colors (yellow, tan) indicate higher probabilities. Contour lines represent probabilities.

a proxy in manatee movement studies when water temperature data are unavailable and broader relationships between water and air temperature are understood.

Manatees also used ship channels most frequently during periods of low freshwater discharge (<10,000 cfs^{-1}). Low discharges are most common during the summer when manatees are present at higher numbers in the nGoM region; therefore, correlations between ship channel use and discharge may be coincidental to timing of manatee occurrence in the study area (Dzwonkowski et al., 2011; Hieb et al., 2017). Low discharge periods also correspond to higher summer temperatures, which could prompt manatees to seek refuge in cooler, deeper waters found in ship channels (Stith et al., 2011). Conversely, during periods of high discharge, salinity stratification in deeper channels such as the Mobile Bay ship channel (Hummell, 1990) results in more saline conditions that may drive manatees out of ship channels in favor of lower salinity waters. Manatees may also avoid ship channels during high discharge periods, particularly in large riverine-driven systems such as Mobile Bay, because movement may be more difficult and energetically costly when the direction of flow opposes the direction of travel (Fish, 1994). While discharge may not be as significant a factor as nGoM water or local air temperature in determining manatee use of ship channels, patterns of freshwater flow may act as a secondary factor mediating day-to-day use at local scales.

Combining datasets that used different methodologies helped to fill spatial and temporal data gaps and added confidence that the movement patterns we detected are meaningful for manatees in our study region. While sources of bias must be considered when using citizen-sourced data (e.g., uneven sampling effort or detection probability), these types of data are increasingly recognized as valuable contributions to long-term monitoring efforts and answering broad ecological questions (Dickinson et al., 2010; McKinley et al., 2017). In this study, the seasonal and spatial distribution of opportunistic manatee sightings both in and out of ship channels was highly consistent with patterns of manatee occurrence documented in the region during the last four decades (Fertl et al., 2005; Pabody et al., 2009; Hieb et al., 2017). Furthermore, we found very similar patterns in ship channel use between tagged and opportunistically sighted manatees. Tagged individuals provided a smaller set of shorter-term data but with high

spatial resolution, while opportunistic, citizen-sourced sightings provided a large database that captured decadal-scale patterns of variation in manatee location and habitat use. Hence, our two datasets accentuate a common tradeoff between quantity and quality in ecological datasets on movement. Taken together, the consistency of results from these two independent datasets along with the similarity to patterns reported to-date along the southern United States Atlantic coast, including use of the ICW (Cummings et al., 2014), suggests broad use of ship channels as migratory corridors by manatees throughout their range in the United States.

Implications for Risk Assessment: Manatees and Other Marine Megafauna Species

Using ship channels as migration routes and travel corridors exposes manatees to a diversity of vessel types. Boat collisions are a leading cause of manatee mortality in Florida (Ackerman et al., 1995; Wright et al., 1995; Aipanjiguly et al., 2003), and manatee use of ship channels in the nGoM increases their risk of collision in waters west of Florida, where boat strike mortalities have been increasingly documented in recent years (Hieb et al., 2017). Boating speed limits and restricted access to critical manatee habitats in peninsular Florida have decreased collisions and contributed to population recovery (Laist and Shaw, 2006; Calleson and Frohlich, 2007; Gannon et al., 2007; Udell et al., 2018). However, increased use of habitats outside of Florida imposes new collision risks in areas with fewer protective regulations, many vessel types, and decreased awareness of manatees among boat operators. Accordingly, since 2013, three manatees deaths in the nGoM were attributed to blunt force trauma typical of vessel collision (no boat-related manatee mortality was documented prior to 2013 in our study area; Hieb et al., 2017). Although our data show manatees use boat channels more frequently than fairways, two of these mortalities occurred in or immediately adjacent to the Mobile Bay ship channel (Carmichael, 2017; Hieb et al., 2017), where boat channels overlap with a shipping fairway. These examples emphasize that manatees in areas with overlapping ship channel types potentially are at greater risk of collisions with a combination of recreational, commercial, passenger, and merchant vessels that are linked to manatee mortality (Calleson and Frohlich, 2007). Importantly, our data demonstrate that exposure to large vessels and associated mortality can occur even in inland harbors such as the Port of Mobile.

While most previous studies specifically evaluating collisions between vessels and manatees focus on recreational boats and other small vessels (Aipanjiguly et al., 2003; Laist and Shaw, 2006; Calleson and Frohlich, 2007; Bauduin et al., 2013), our study is unique in documenting use of multiple channel types and exposure to vessels at the intersection of boat channels and fairways. Globally, inland harbors are increasing in number and size to accommodate larger vessels (Walsh, 2012; Tournadre, 2014). For example, the main component of the Mobile Bay ship channel, which is currently 130 m wide, is planned to be expanded to 180m during the next few years (U.S. Army Corps Of Engineers, 2018), and similar harbor expansions are ongoing or planned globally (Wang, 2017). Additionally, exposure to increased noise and chemical pollution as well as increased habitat disturbance associated with shipping industries and channel construction and maintenance, such as dredging and channel widening, may have negative impacts on manatees (Todd et al., 2014; Liubartseva et al., 2015; Pirotta et al., 2019). As shipping activities continue to expand in nearshore areas, our data suggest potential for shipping traffic to pose increasing risks to manatees and other nearshore species such as small cetaceans, pinnipeds, and sea turtles.

Manatees, as well as other marine megafauna, face an uncertain future, with risks from human interactions (Ackerman et al., 1995; Wright et al., 1995; Aipanjiguly et al., 2003), habitat destruction (Laist and Reynolds, 2005; Marsh et al., 2011), and climate change (Marsh et al., 2011, 2017). Understanding ship channel use among species is crucial to evaluating risks and developing strategies to mitigate negative impacts (Carrillo and Ritter, 2010; van der Hoop et al., 2015; Martin et al., 2016; Crum et al., 2019; Pirotta et al., 2019). We show that GAMs populated with independently sourced datasets provide a powerful predictive tool that can help illuminate marine megafauna use of ship channels. Our modeling approach provides a complex evaluation of risk that can be tailored to different challenges based on type of species and their habitat requirements. Here, we focused primarily on nearshore channels (130 and 300 m) and some fairways, but our methods can easily be applied to either nearshore or offshore ship channels and fairways. While we chose variables based on manatee ecology—temperature, freshwater discharge, time-of-year (Deutsch et al., 2003; Reep and Bonde, 2006; Marsh et al., 2011)— other variables can easily be substituted based on the ecology of other marine species. Furthermore, we were able to combine telemetry and observational data to enhance occurrence data with information on movement and behavior of animals inside ship channels. We propose that this approach be used on a wider range of species among locations to help predict when and how marine megafauna use ship channels and to evaluate risks associated with channel use.

We provide new fundamental knowledge on movement ecology of a large, protected marine species and important information for managers, civil engineers, boaters, and the shipping industry to guide future conservation practices. Because manatee use of habitat in the nGoM is limited by temperature but increasing along with expanding vessel capacity and shipping activity, this region may represent the future landscape for temperature-dependent migratory and dispersing populations of many other megafauna species. The nGoM, and Mobile Bay in particular, can serve as a sentinel site that may be a 'canary in the coal mine' for assessing future boat-related risks to manatees and other migratory species. Using multiple datasets and a similar modeling approach, other researchers can evaluate nearshore ship channel use across a wider range of species and geographic distributions.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

This study was carried out in accordance with the US Fish and Wildlife Service Permits MA107933-1 and MA37808A-0, Alabama Department of Conservation and Natural Resources Wildlife and Freshwater Fisheries Division annual permits. The protocol was approved by the University of South Alabama IACUC for protocols 581568 and 1038636.

AUTHOR CONTRIBUTIONS

CC analyzed the data, wrote the main draft of the manuscript, and participated in editing the manuscript. EH contributed to the data collection and analysis for both tagged and sighted manatees, environmental data, and writing and editing the manuscript. MC contributed to collection and calculation of environmental data and editing the manuscript. KD contributed to the data collection for tagged manatees, determined coordinates of ship channels, prepared the SI video, and edited the manuscript. RC conceived the project, collected data for tagged and sighted manatees, contributed to data analysis and writing and editing the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00318/full#supplementary-material

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Potential Benefits of Vessel Slowdowns on Endangered Southern Resident Killer Whales

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A voluntary commercial vessel slowdown trial was conducted through 16 nm of shipping lanes overlapping critical habitat of at-risk southern resident killer whales (SRKW) in the Salish Sea. From August 7 to October 6, 2017, the trial requested piloted vessels to slow to 11 knots speed-through-water. Analysis of AIS vessel tracking data showed that 350 of 951 (37%) piloted transits achieved this target speed, 421 of 951 (44%) transits achieved speeds within one knot of this target (i.e., $\leq$12 knots), and 55% achieved speeds $\leq$ 13 knots. Slowdown results were compared to 'Baseline' noise of the same region, matched across lunar months. A local hydrophone listening station in Lime Kiln State Park, 2.3 km from the shipping lane, recorded 1.2 dB reductions in median broadband noise (10–100,000 Hz, rms) compared to the Baseline period, despite longer transit. The median reduction was 2.5 dB when filtering only for periods when commercial vessels were within 6 km radius of Lime Kiln. The reductions were highest in the 1st decade band (−3.1 dB, 10–100 Hz) and lowest in the 4th decade band (−0.3 dB reduction, 10–100 kHz). A regional vessel noise model predicted noise for a range of traffic volume and vessel speed scenarios for a 1133 km^2 'Slowdown region' containing the 16 nm of shipping lanes. A temporally and spatially explicit simulation model evaluated the changes in traffic volume and speed on SRKW in their foraging habitat within this Slowdown region. The model tracked the number and magnitude of noise-exposure events that impacted each of 78 (simulated) SRKW across different traffic scenarios. These disturbance metrics were simplified to a cumulative effect termed 'potential lost foraging time' that corresponded to the sum of disturbance events described by assumptions of time that whales could not forage due to noise disturbance. The model predicted that the voluntary Slowdown trial achieved 22% reduction in 'potential lost foraging time' for SRKW, with 40% reductions under 100% 11-knot participation. Slower vessel speeds reduced underwater noise in the Slowdown area despite longer passage times and therefore suggest this is an effective way to benefit SRKW habitat function in the vicinity of shipping lanes.

Keywords: southern resident killer whales, voluntary slowdown, commercial vessel, underwater noise, acoustic disturbance, Salish Sea

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INTRODUCTION

A number of at-risk species of cetaceans (whales, dolphins, and porpoises) inhabit the straits between southern British Columbia and northern Washington State often referred to as the Salish Sea. Key among these species is the endangered southern resident killer whale (SRKW), with a population of only 78 individuals at the time of this 2017 study. This population was designated as endangered under Canada's *Species at Risk Act* in 2001, which initiated the development of a recovery strategy (Heise, 2008; Department of Fisheries and Oceans Canada [DFO], 2011, 2018) and an Action Plan (Department of Fisheries and Oceans Canada [DFO], 2017) to address the current threats to the SRKW in Canadian waters). The Canadian federal recovery strategy designates much of the Salish Sea as SRKW critical habitat, where critical habitat is defined as the habitat necessary for the survival or recovery of the species. Under the United States *Endangered Species Act*, critical habitat has also been designated in much of the Washington State waters of the Salish Sea. These designations offer the species legal protection in both the inbound and outbound shipping lanes because they overlap with critical habitat.

Killer whales use sound to navigate, communicate and locate prey via echolocation (Ford, 1989; Au et al., 2004), and underwater noise can impede these functions (Holt et al., 2009). Southern resident killer whale critical habitat in the Salish Sea includes inbound and outbound commercial shipping lanes, as well as traffic from many whale watching boats and recreational vessels. In close proximity of boat traffic (<400 m), studies of both northern and southern resident killer whales behavior have shown whales reduce time spent foraging and increase time spent transiting (Williams et al., 2006; Lusseau et al., 2009). Vessel proximity has been shown to induce changes in SRKW surface-active behaviors (Noren et al., 2009), respiration rate, swim speed, and path directedness (Williams et al., 2009). Elevated noise levels from vessel traffic can hinder the opportunities for killer whales to echolocate and find food, as well as limit opportunities to share information and maintain group cohesion within a foraging group. The result is a reduction in the whale's acoustic space and foraging efficiency, making it harder for whales to find their prey. The SRKW population is believed to be undergoing nutritional stress due to ongoing changes in both the number and size of returning Chinook salmon (Ford et al., 2009; Ward et al., 2009; Department of Fisheries and Oceans Canada [DFO], 2011; Williams et al., 2011), and exposure to low-frequency ship noise may be associated with chronic stress in whales (Rolland et al., 2012). Therefore, increased underwater noise in key foraging habitat areas could have important implications to this endangered population.

Conservation of the SRKW population is challenged by increases in international trade, for which 90% is facilitated through marine shipping (United Nations Conference on Trade and Development [UNCTAD], 2018). Over the last century, there have been substantial increases in shipping worldwide and future projections indicate that this trend will continue. Marine traffic generates low frequency, high-energy noise in the ocean and can propagate across hundreds of kilometers, and even whole ocean basins (National Research Council of the U. S. National Academies [NRC], 2003). Recent evidence has documented commercial vessel noise in the Salish Sea has raised the background broadband noise levels significantly, including noise in the frequency range that SRKW use for communication and echolocation (10–40 kHz band, Veirs et al., 2016). With noise projected to continue increasing (Hildebrand, 2009), there comes an increasing potential for adverse effects on the underwater noise field of the Salish Sea (Veirs et al., 2016). The potential implications are lower survival rates and lower reproductive success of individuals that could, in the long term, have population level consequences for the SRKW. Indeed Lacy et al. (2017) identified acoustic disturbance from ships and small boats as a threat to SRKW recovery in the Pacific Northwest.

The Enhancing Cetacean Habitat and Observation (ECHO) Program, led by the Vancouver Fraser Port Authority, aims to better understand and mitigate the effects of shipping activities on at-risk whales in the Salish Sea. The ECHO Program is guided by the advice and input of an advisory working group that brings together a broad spectrum of relevant backgrounds, perspectives and interests from both Canada and the United States. The advisory working group consists of members from the marine transportation industry, environmental conservation, First Nations, government, and academia. This group identified acoustic disturbance to the endangered SRKW from vessel noise as a top priority for program research and mitigation[1]. Ultimately the advisory working group identified that slowing vessels down in a geographic area of importance to SRKW should be the priority mitigation measure to trial. Thus, during the summer of 2017, a multi-stakeholder voluntary commercial vessel Slowdown trial was conducted within 16 nm of the international shipping lanes for vessels calling into Canadian and United States ports in the Salish Sea. This 16 nm Slowdown region is a key summer foraging habitat for SRKW in the Salish Sea. Commercial vessels were requested to slow to a target speed-through-water of 11 knots across the 16 nm section to quantify underwater noise reduction and evaluate potential benefits to SRKW. The trial was designed to answer the following three key questions: (1) How does reduced vessel speed change the underwater noise generated by a specific vessel (vessel source level) and by type of vessel? (2) How does reduced vessel speed change the total underwater ambient noise received at a specific location of importance to the SRKW? (3) What are the predicted resultant effects on SRKW behavior and foraging given the changes in noise as answered by questions (1) and (2)?

MATERIALS AND METHODS

Trial Location

Southern resident killer whales are present year-round in the Salish Sea, but are concentrated off the west coast of San Juan Island in the core of their critical habitat in Haro Strait (Seely et al., 2017; Olson et al., 2018). This SRKW hotspot region intersects with the major international shipping lanes between

[1]https://www.portvancouver.com/echo

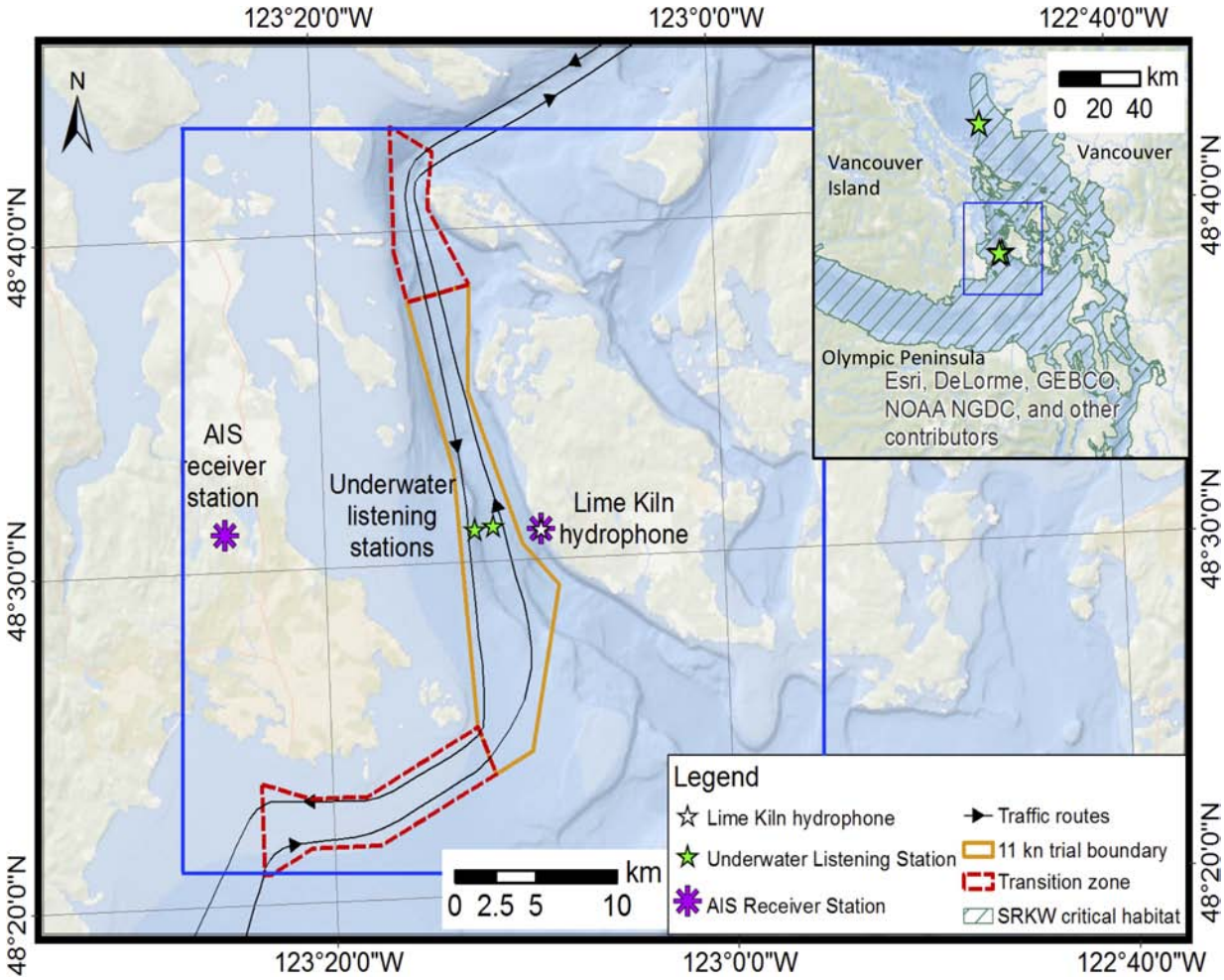

FIGURE 1 | Map of the Slowdown region. The two AIS receiver stations are shown as purple stars. The orange polygon depicts the 16 nm section of shipping lanes through which vessels were requested to slow to 11-knots speed-through-water. There are 4 underwater listening stations (ULS) as part of this study, three are within Haro Strait, two of which are located under the shipping lanes, and the third is cabled to shore and located at Lime Kiln Point on San Juan Island. The fourth ULS is in Georgia Strait, 30 km south of Vancouver, and shown only in the small map panel. The extent of the 'regional vessel noise model' for the 'Slowdown region' is depicted by the blue box, and includes cross-boundary waters of British Columbia, Canada and Washington State, United States.

the Pacific Ocean and ports of call in southern British Columbia, the largest being the Port of Vancouver, as well as the Washington State Ferries route from Anacortes, Washington to Sidney, British Columbia (**Figure 1**). The boundaries of the Slowdown region constituted an approximately 16 nm distance of shipping lanes through Haro Strait that overlaps with the core of SRKW critical habitat (**Figure 1** inset).

Trial Timing

The trial took place during two lunar months (61 days) between August 7 and October 6, 2017. The trial encompassed the period of the year when SRKW are typically at their highest presence in Haro Strait coinciding with increased availability of Chinook salmon (Ford and Ellis, 2006). Lunar months (as opposed to calendar months) were selected to evaluate total ambient noise in the region while accounting for the low frequency flow noise that is typically associated with tidal cycles (Lee et al., 2011). The ECHO Program has been collecting and analyzing ambient noise on a lunar month cycle in Haro Strait since 2016, thus a comparative evaluation of the potential reduction in ambient noise resulting from the Slowdown trial could be more effectively assessed using the same timeframe.

Determining Trial Speed and Length

In evaluating what may be an appropriate speed for conducting the Slowdown trial, several factors were considered. These included an evaluation of the potential benefits of noise reduction to SRKW, potential economic impacts to industry from reduced speed, lessons learned from other jurisdictions (Parrott et al., 2016), and most importantly, navigational safety.

An evaluation of what would be considered a safe speed for navigation of deep-sea vessels in Haro Strait was conducted in consultation with the ECHO Program's advisory working group and vessel operators committee, which includes the BC

Coast Pilots, Pacific Pilotage Authority and Canadian Coast Guard. Given the constrained waters of Haro Strait, combined with the high currents frequently encountered in this area, a target speed of 11 knots (measured as speed through water) was requested to achieve maximum potential benefit to underwater noise reduction, without compromising navigational safety.

The trial Slowdown region is located within an established Compulsory Pilotage Area as defined by the Pacific Pilotage Authority Regulations (Pilotage Act, 1985). These regulations require that every commercial vessel that is over 350 gross tons, and every pleasure craft over 500 gross tons requires pilotage while in Compulsory Pilotage Areas. The BC Coast Pilots guide ships coming in or out of ports that traverse Canadian waters to ensure safety, efficiency and environmental protection. In this report, we refer to these deep-sea commercial or larger pleasure crafts as "piloted vessels." Based on historic vessel traffic data, at least 400 piloted vessels a month would be expected to transit Haro Strait. As we didn't know ahead of time how many pilots would participate in the slowdown, we estimated that a 2-month trial period would provide an adequate number of vessel transits to allow statistical analysis of the effects of the slowdown on vessel noise emissions and total ambient noise, while also balancing the likely impact to industry and the potential benefit to SRKW at a time of year when whales are historically present (Olson et al., 2018).

Verification of Vessel Speeds

The equipment used to monitor vessel speed during the trial included two Automated Identification System (AIS) receivers to provide information such as vessel type, speed and draft on each AIS-enabled vessel transiting Haro Strait. One AIS receiver was positioned atop Observatory Hill, approximately 17 km to the west of the Haro Strait hydrophone deployments and one in Lime Kiln State Park on San Juan Island. These data were used to monitor piloted vessel tracks and were used to assess vessel speed and participation within the 16 nm Slowdown region during both the Baseline and Slowdown trial periods.

The international shipping industry AIS data records only speed over ground, thus vessel transit data was adjusted for tidal current to yield speed through water. Using the two AIS receiver stations (**Figure 1**) and a NOAA reference site at Kellet Bluff (48.588 N, −123.237 W) at the north east end of Haro Strait contributing modeled tidal current, we adjusted average vessel speed over ground to vessel speed through the water by incorporating vectors of tidal current over the designated trial area.

Acoustic Monitoring of Slowdown Trial

Four fully calibrated hydrophone stations collected acoustic data during the study. Two underwater listening stations (ULS) were equipped with JASCO Autonomous Multichannel Acoustic Recorders (AMARs) and placed directly adjacent to the inbound (northbound) and outbound (southbound) shipping lanes in Haro Strait at water depths of 203 m, and 248 m (**Figure 1**). A third cabled hydrophone was installed in 23 m water in front of the Lime Kiln State Park lighthouse, a point on the western side of San Juan Island, Washington State (48.51° N, −123.15°W, **Figure 1**). This cabled hydrophone recorded ambient noise levels at the core of SRKW summer foraging area. The Strait of Georgia Underwater Listening Station, a fourth ULS, was used in the evaluation of the Slowdown trial. This station was situated outside of the Slowdown region on the seabed at approximately 170 m water depth, in the northbound traffic lane, approximately 30 km southwest of Vancouver in the Strait of Georgia (49.04° N, 123.32°W; **Figure 1** inset), and 30 km past the Slowdown trial region. Noise data from all four stations were digitized and post-processed in a similar manner following methods described in Merchant et al. (2013) to allow for comparison across sites and time.

Vessel Source Level Methodology

Underwater source levels of marine vessels generally increase with speed due to associated increases in machinery vibration and propeller-induced cavitation (Ross, 1976; Arveson and Vendittis, 2000; McKenna et al., 2013). The most widely applied formula for scaling source levels with vessel speed is Ross's power-law model (Ross, 1976), which relates changes in source level (SL) to relative changes in speed (v) according to the following formula:

$$\mathrm{SL} - \mathrm{SL}_{\mathrm{ref}} = C_{\mathrm{v}} \times 10\ \log_{10}\left(\frac{v}{v_{\mathrm{ref}}}\right) \tag{1}$$

In this equation, 'SL' is the source level at 'v' speed through water, '$\mathrm{SL}_{\mathrm{ref}}$' is the source level at 'v_{ref}' some reference speed, and 'C_{v}' is a scaling coefficient corresponding to the slope of the curve. Different trends of source level versus speed (including negative trends) may be accommodated by adjusting the value of the scaling coefficient, C_{v}. Scaling coefficients for different categories of vessels were collected during the Slowdown trial. A power law relationship between source level and vessel speed was strongly supported by vessel noise measurements collected on the two AMARs situated adjacent to the northbound and southbound vessel traffic lanes (MacGillivray and Li, 2018a).

Vessel source levels were calculated before, during, and after the trial on three of the ULS hydrophones to determine the effect of slowdowns on noise emissions for five vessel categories (Bulker/General Cargo, Containership, Car Carrier, Tanker, and Cruise). Source levels were calculated using an automated system that tracked vessels on AIS as they passed the ULS hydrophones. The system analyzed 1/3-octave band SPL from each vessel, inside a data window encompassing ±30° of its closest point of approach to the hydrophone, according to the methods specified in the ANSI ship noise measurement standard American National Standards Institute [ANSI] (2009). A monopole SL was calculated for each vessel measurement by adjusting the received SPL for the propagation loss, using a hybrid propagation model from 10 Hz to 64 kHz. The hybrid model computed transmission loss in 1/3-octave bands, using the parabolic equation method, wavenumber integration method, or image ray method (Jensen et al., 2011), depending on the frequency and distance of the vessel from the hydrophones. Additional details regarding the automated source level measurement system are given in Hannay et al. (2016).

To ensure high data quality, only source level measurements with closest points of approach of less than 1,000 m from the hydrophones were accepted for analysis. In addition, an experienced acoustic analyst performed a manual quality review of every source level measurement. Source level measurements that contained interference from other vessels, had high levels of background noise, or traveled at irregular speeds on indirect (not straight) tracks were rejected.

Analysis of 1317 source level measurements from the Haro Strait ULS hydrophones provided the estimates of sound level reductions for five categories of piloted vessels transiting the Slowdown region (**Table 1**, MacGillivray and Li, 2018a). The source level reductions for the piloted vessel categories were calculated by comparing measurements of vessels traveling at normal speed before and after the trial (i.e., the Baseline group) with measurements of vessels traveling reduced speed during the trial (i.e., the Slowdown group). The estimated reductions were also crosschecked against measurements of piloted vessels outside of the Slowdown region at normal speeds on the Georgia Strait ULS hydrophones and found to be consistent. Frequency-dependent speed-scaling coefficients were calculated from the estimated reductions for each vessel category (**Table 1**). These frequency ranges were based on the three frequency bands that were identified by the Coastal Ocean Research Initiative working group as relevant to the acoustic quality of SRKW habitat (details in Heise et al., 2017):

1. Broadband (10–100,000 Hz), for evaluating behavioral or physiological effects.
2. Communication masking (500–15,000 Hz), for evaluating effects of noise on communication space.
3. Echolocation masking (15,000–100,000 Hz).

The frequency divisions between these three bands do not line up exactly with the divisions between the standard 1/3-octave bands used in the regional vessel noise model, thus scaling coefficients from the trial were assigned to the closest matching frequency bands (**Table 1**). Scaling coefficients in the 10–400 Hz bands approximately correspond to the broadband value since overall vessel source levels are dominated by noise below 500 Hz. These frequency-dependent scaling coefficients were used to model the effect of speed reductions on vessel source levels and used as input to the regional vessel noise model (MacGillivray and Li, 2018b).

Average frequency-dependent source levels were calculated for 14 different vessel categories (including the 5 Slowdown trial categories, described above), based on a database of 2,705 source level measurements collected by the ECHO program during 2015–2017 (MacGillivray et al., 2018), supplemented with additional source level measurements for small passenger and recreational vessels (Erbe, 2002; Veirs et al., 2016). The limited number of vessel categories applied in this study could not, of course, completely capture the large variety of different ship classes and designs in the study area (a full list of vessel types can be found in **Supplementary Material**). Nonetheless, we expect these source levels model to accurately represent average noise emissions of the different vessel categories in the study area.

TABLE 1 | Power-law scaling coefficients of monopole source level (SL) versus speed (C_v) for the bulk carrier and general cargo, container ship, car carrier, tanker, and cruise/passenger vessel categories.

Vessel type	1/3-Octave frequency bands (Hz)		
	1st octave band (10–400 Hz)	Communication bands (500–12500 Hz)	Echolocation bands (16,000–63,000 Hz)
Bulk carrier + general cargo	8.2	4.2	7.0
Container ship	5.1	4.1	7.9
Car carrier	5.2	4.1	7.7
Tanker	7.7	4.5	9.9
Cruise/passenger	4.9	5.4	8.2

These were determined from measurements taken during the slowdown trial. The 1/3-octave band frequency band ranges are specified in accordance with the CORI bands for SRKW, as described in the text.

Regional Vessel Noise Model

The regional vessel noise model combines vessel tracking data, vessel sound emission data, ambient noise levels (without vessels present), and environmental data describing how sound attenuates through the water column for the study area, to predict the vessel noise on a computational grid. Vessel sound emissions are determined by referencing a database of source levels (according to vessel type and speed), and the transmission of the sound from each AIS-enabled vessel according to a database of pre-computed propagation loss curves for the Slowdown region. Both the time of departure and the choice of inbound or outbound route were randomly selected for each simulated vessel movement. Each trip was displaced slightly from the center of the route in a randomized fashion, to represent the observed distribution of traffic along the shipping routes (rms width of the vessel traffic varied from 440 m at the north end of the 11-knot Slowdown boundary to 600 m at the south end). Other vessel traffic, which included non-piloted vessels and piloted vessels bound to and from the United States, were simulated based on actual historical AIS vessel tracks for a representative day in summer. AIS data were obtained from the community-based MarineTraffic ship tracking service[2]. Vessel tracks from the AIS data were assigned to one of 14 different source level categories, based on their vessel type classification from MarineTraffic (**Supplementary Material**). Movements of other vessels were held constant between the Baseline and Slowdown model scenarios except for the ferries sailing between Sidney and San Juan Island that participated in the Slowdown trial. This ensured that the contributions of those vessels to the soundscape were constant and did not affect the relative metrics of the trial results.

We applied the cumulative vessel noise model developed by JASCO (described in MacGillivray et al., 2014) to develop

[2]www.marinetraffic.com

time-dependent noise maps from merged vessel tracking data, piloted vessel sound emission estimates, vessel speed-scaling data for this region, ambient noise levels (without vessels present), and environmental data. These inputs describe how sound attenuates through the water column for the study region. The noise model does not account for non-AIS enabled vessels (primarily small boats under 350 gross tons), since insufficient information on the movements of these types of vessels was available for the study area.

Simulation scenarios of movements of piloted vessels through Haro Strait were designed to help inform management of realistic operational scenarios under "average" (14 vessel transits/day) and "high" (21 vessel transits/day) traffic volumes at a range of transiting speeds (**Table 2**). The "Baseline" vessel speed categories describe piloted vessels under normal operating conditions without slowdown restrictions. The "11-knot" vessel speed scenarios assume all piloted vessels observed the 11-knot slowdown speed. The "trial mean speeds and participation percentages" was matched to the same vessels moving at the participation rates and associated transit speeds recorded during the Slowdown trial. The number of piloted vessel transits for each vessel category on an "average" and "high" volume day were derived from ship traffic data provided by the Pacific Pilotage Authority. The regional vessel noise model used a 24-h time period to describe vessel transits, thus the participation rates used for modeling vary slightly from the average reported participation of the 61-day Slowdown trial. For example, if an average traffic day has eight bulker transits, and trial participation rate over all trial days for bulkers was 55%, this translates to 4.4 bulkers per day participating. As a portion of a vessel cannot be described in the regional vessel noise model, instead the closest integer number would represent the "participation percentage" in the model (i.e., 4 of the 8 bulkers in this example, or 50% participation rate, **Table 3**).

Time-snapshots of underwater noise levels were then simulated to generate sequences of two-dimensional maps, or "snapshots," of the dynamic sound field, providing sound pressure level as a function of easting, northing, frequency, and time of day. These time snapshots of simulated underwater noise levels provide broadband (9 Hz to 78,000 Hz) sound pressure level (SPL, dB re 1 μPa) and 1/3 octave band SPL centered on (high frequency) 50 kHz (PSD, dB re 1 $\mu Pa^2/Hz$) for vessels transiting the Slowdown region. We refer to these time snapshots as the Broadband and the 50 kHz noise distribution maps. The temporal resolution of these noise distribution map files was 1-min duration covering all 1,440 min in a 24-h period. The 1-min map files were then processed into 288 5-min summaries using the 5-min maximum for each time interval for each of the Broadband and 50 kHz maps. As killer whales are moving in three dimensions, and the propagation of noise necessarily involves inherent uncertainties due to imprecise environmental data (Weilgart, 2007), this was considered a conservative approach to account for variability and errors over a 200 m grid region occupied by a foraging SRKW. The spatial resolution of both broadband and 50 kHz noise distribution map files was matched

TABLE 2 | Six Modeling Scenarios described by the regional vessel noise models.

Scenario	Traffic volume	Vessel speeds	Slowdown participation rate (%)
(1)	Average volume	Baseline	0% (baseline)
(2)	Average volume	Trial mean speeds and participation percentage	57% (trial percentages)
(3)	Average volume	11 knots	100% (all vessels observe 11 knot speeds)
(4)	High volume	Baseline	0% (baseline)
(5)	High volume	Trial mean speeds and participation percentages	57% (trial percentages)
(6)	High volume	11 knots	100% (all vessels observe 11 knot speeds)

*Scenarios 1, 2, and 3, represent a day with 'Average Traffic Volume' transiting the slowdown region at 3 participation rates with slowdown speeds. Scenarios 4, 5, and 6, represent a day with 'High Traffic Volume,' similarly transiting the slowdown region at various slowdown participation rates. See **Table 5** breakdown of 57% trial participation.*

TABLE 3 | A comparison of average observed number of piloted commercial vessels per day during the 61 days slowdown period, and the assumed transit numbers for average and high traffic volume scenarios.

Vessel category	57% participation 11-knot speed average vessel traffic 2017		57% participation 11-knot speed high vessel traffic 2017	
	# of ships	# of slow ships	# of ships	# of slow ships
Bulk carrier + general cargo	8	4	10	6
Container ship	4	2	6	3
Car carrier	1	1	2	1
Tanker	1	1	2	1
Cruise/passenger ship	0	0	1	1
Grand total	14	8	21	12

The counts of commercial vessel transits for the average and high traffic volumes in the regional vessel noise model were based on a review of historic commercial traffic data provided by the Pacific Pilotage Authority. Average Traffic corresponded to the historic median, while high traffic volume corresponds to the 95th percentile.

to the 200 m by 200 m spatial resolution of the SRKW density surface.

Lime Kiln Listening Station, and Ambient Noise Assessment

At the Lime Kiln listening station, received SPLs and PSD were calculated for each 1-min period across two Baseline lunar months (9 July to 8 August 2017 and 18 August to 16 September 2016) and two Slowdown trial lunar months and then linked to 1-min AIS vessel transit information. Lunar month broadband SPLs were compared using all recorded 1-min noise data, but this perspective does not take into account differences in the number of vessel transits, the speed compliance level observed, nor the effect of weather, tidal currents or the influence of small boat presence.

We therefore undertook a fine-scale comparison focusing on periods when piloted vessel were within a 6 km detection zone around the Lime Kiln listening station. The effects of key confounding covariates were minimized by excluding times when (a) small boats were detected by the acoustic detector, (b) current speed was high (values above 25 cm/s), and (c) wind speed was high (values above 5 m/s). Rainfall for both time periods was found to be infrequent and not considered confounding, with just a handful of days of precipitation in both periods (all <6.5 mm). Cumulative distribution functions (CDFs) were plotted for each month and mean differences and differences in 5th, 50th, and 95th percentiles of SPL dB were determined.

Southern Resident Killer Whale Noise-Exposure Model

To evaluate the effect of the commercial vessel Slowdown trial, we built a simulation model based on empirical results from species-specific studies in the region. These studies describe the behavioral and masking effects of noise-exposure from passing vessels on SRKWs. Across each day of the Slowdown trial, the noise-exposure model accumulated the number of occasions each (simulated) whale received noise at levels that were assumed to temporarily inhibit or disrupt its ability to forage, either from an associated change in behavioral state (i.e., from foraging to traveling, e.g., Lusseau et al., 2009; Goldbogen et al., 2013), or alternatively via masking of echolocation clicks (Au et al., 2004). Simple assumptions around the 'dose' (received level) then inform the severity of the whale's response which relates directly to the decrease in SRKW foraging opportunities (time).

The model requires fine-scale information on SRKW habitat use and monthly presence for the model region. A relative SRKW spatial density surface at 200 m grid resolution was estimated from an 11-year synthesis (2001–2011) of effort-corrected sightings within the Salish Sea (methods follow those in Olson et al., 2018). Members of the SRKW population belong to one of three socially distinct units (i.e., J, K, or L pods). For each of the three SRKW pods, we summarized their occurrence according to the pod's presence in August, September and October within the Slowdown region.

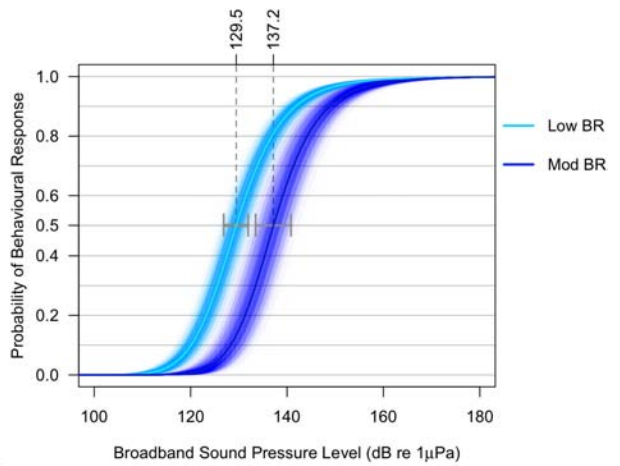

FIGURE 2 | Simulated dose-response curves for low and moderate severity responses. Variability in the dose-response relationship was included in the noise-exposure model as seen in this figure. The 95% CI are shown as gray horizontal error bars at 50% probability of a Low and Moderate (Mod) behavioral response (BR), and are derived from a regression equation of northern and southern resident killer whales responding to commercial vessel noise (see **Supplementary Figures** and **Supplementary Methods** for more details on approach).

We developed two SRKW-specific sigmoidal dose-response functions (**Figure 2**; US Navy, 2008, 2012; Finneran and Jenkins, 2012) using empirical studies collected in the coastal waters of the Salish Sea, and/or Johnstone Strait just north of the Salish Sea. The functions were based on 'low' (Low) and 'moderate' (Mod) severity responses (Southall et al., 2007) to ambient noise levels observed on 45 occasions corresponding to three regional resident killer whale datasets (surface observations via theodolite, movement and vocal behavior using suction tags and vocal compensatory behavior based on hydrophone data) (**Supplementary Material**).

Using a theodolite, the swim speed, dive time, and surface-active behaviors including respiration rates of the closely related northern resident killer whales were measured when tugs, cargo vessels and cruise ships transited past the whales (Williams et al., 2014). Changes in observed behavior were scored based on Southall's severity scale (Southall et al., 2007) and related to the noise level during the passing vessel. In a second study, Wright et al. (2017) used digital acoustic recording tags (DTAGs) in conjunction with GPS field measurements to record dive depths, whale movement and respiration rates of northern resident killer whales. These data were similarly analyzed for behavioral response to tugs, cruise ships and commercial fish transport vessels and similarly scored based on Southall's severity scale. In the final study, data from a 2009 passive acoustic monitoring study at Lime Kiln listening station measured changes in frequency and amplitude of SRKW calls in response to passing commercial ship traffic. These amplitude changes were also scored using an adaptation of Southall's severity scale. (Further details of the data and approach can be found in the **Supplementary Material**).

Underlying the dual dose-response relationship is the concept that at higher received noise levels, there is a higher probability of a disruption in behavior and that this disruption has the potential to last longer than the time period of the dose (e.g., through a switch in behavior). In other words, the nearer an SRKW is located to a noise source the louder the whale's received level, and the higher the likelihood a behavioral response occurs. A "Moderate" severity behavioral response (Mod BR) is defined as a moderate to extensive change in locomotion speed, direction and/or dive profile, moderate or prolonged cessation of vocal activity, and/or potential avoidance of area (Southall et al., 2007). Re-analysis of the DTAG data described in Wright et al. (2017) indicated that these effects have an average duration of ∼25 min (SMRU, 2015). At lower received levels (decreased vessel-whale proximity) the probability of a behavioral response declines to zero. If no moderate severity behavioral response is predicted to occur, the model assesses if noise levels are sufficient to trigger a "Low" severity behavioral response (Low BR). A Low BR is defined in the literature as minor changes in respiration rates, locomotion speed, direction or deviation by Southall et al. (2007), but can encompass lost foraging opportunities. The duration of these low severity behavioral responses (BRs) were considered short-term (5 min, or the time it takes a commercial vessel traveling at 18 knots to transit through a 1.4 km radius circle around a SRKW).

The SRKW-specific dose-response relationships had broadband received noise level median threshold values of 129.5 and 137.2 dB re 1 μPa for low severity and moderate severity BRs, respectively (**Figure 2**). Low and moderate severity BRs had a 1% probability at received noise levels of 111 and 120 dB re 1 μPa, respectively, resulting in approximate response zones of up to 3.8 and 1.4 km from a 320 m container ship traveling at 18 knots. Uncertainty around these dual dose-response relationships was derived from the combined results of the three data input studies (Williams et al., 2014, DTAG, SMRU, 2014; Figure in **Supplementary Material**), and contributed to the parameter uncertainty in the inputs of the dose response function (**Figure 2**).

'Acoustic masking' is defined as an interference with an individual's ability to detect, recognize, and/or discriminate sounds such as echolocation clicks. Masking of echolocation clicks can occur even at low broadband noise levels if noise levels in high frequency critical bands are exceeded. The SRKW noise-exposure model aimed to capture this possibility by calculating the degree of additional or residual high frequency masking when no 'Mod' or 'Low' BRs were predicted. Foraging related echolocation clicks have a peak intensity centered at 50 kHz (Au et al., 2004) and this frequency band from the regional vessel noise model was selected to assess the degree of click detection range reduction due to masking. We followed the Au et al. (2004) approach to modeling echolocation click masking and assumed a maximum click detection range of 250 m based on estimates made from acoustic data collected at Lime Kiln. Modeled click detection ranges were then converted to a proportion of the 250 m maximum click detection range.

The SRKW noise-exposure model methodology is summarized in **Figure 3**, with expanded details in the

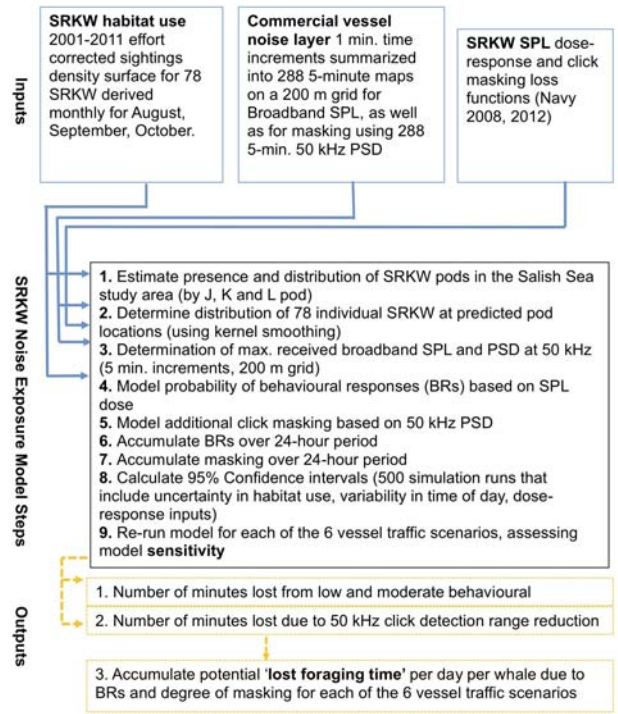

FIGURE 3 | Explanation and pseudocode of noise-exposure simulation model. More details for each of the 9 steps can be found in the **Supplementary Material**.

Supplementary Material. In summary, the dose-response function is what probabilistically determines whether a Low or Mod BR occurs when the whale is exposed to noise from a passing commercial vessel (e.g., Lusseau et al., 2009). The severity of a single BR (i.e., low vs. moderate) determines the length of time the individual whale is disrupted from foraging. The intensity of the high-frequency (50 kHz PSD) sound levels determines the degree of residual high-frequency masking implied by a proportional reduction in the distance that echolocation is fully inhibited, i.e., complete masking of echolocation clicks (Au et al., 2004). These BRs and residual masking minutes are subsequently converted into a metric termed 'potential lost foraging time,' meant to represent the time a whale is potentially inhibited or disrupted from its ability to forage due to excessive received noise levels (with 95% confidence intervals derived using simulation re-sampling). The simulation model acts on individual whales at 5-min resolution, but can be integrated over time, over space or across whales into pod or all-SRKW summaries. In this manuscript, we report the all-SRKW summaries.

The assessment of whether slower vessels had a positive effect on the behavior and foraging of killer whales was determined by comparing 'potential lost foraging time' as the output metric from the SRKW noise-exposure model. This lost-time metric was calculated for each of the six traffic scenarios (**Table 2**), and allowed a comparative exploration of the relative value of various noise mitigation and slowdown participation rates. This delta

approach minimizes the effect of the assumptions made as they are applied equally across scenarios.

RESULTS

Trial Vessel Speed (AIS) and Participation

The Pacific Pilotage Authority reported 951 piloted transits of commercial vessels through Haro Strait during the 61-day trial period, from August 7 to October 6 (**Table 4**). The most common piloted vessel type was bulk or general cargo ships (51.6%), followed by container ships (27.3%), car carriers (9.0%), tankers (7.8%), and cruise ships (3.2%). There were fewer transits during the 2-month Baseline period with 863 piloted transits. The ordering of vessel types is the same for the Baseline as it was for the Slowdown period with most common to least as follows: general cargo ships (53.8%), container (28.2%), car carriers (8.5%), tankers (7.1%), and cruise ships (0.9%).

Vessel speed and participation was monitored using Automated Identification System (AIS) receivers to identify vessel names, vessel type, speed, and location. Median speed reductions varied by vessel type from a 1.8 knot reduction in speed for bulk/general cargo ships and as high as a 7.2 knot reduction in speed for container ships (**Table 5** and **Figure 4**). Speed reductions were paired with increased time to transit the 16 nm of shipping lanes through the Slowdown region, and this ranged from 10.8 to 32.6 extra minutes for bulk/general cargo ships and container ships, respectively, (**Table 5**). Source level reductions resulting from Slowdown trial participation significantly reduced underwater noise emissions for vessels transiting the Slowdown region (**Table 5**).

'Participation' in the Slowdown trial meant aiming for a speed-through-water of 11 knots and AIS vessel tracking data showed that 37% (350 of 951) piloted transits achieved this slowdown speed-through-water. If we relax the participation cutoff to include any vessel able to maintain speed-through-water of <12 knots, this resulted in an overall vessel participation of 44% (421 of 951 piloted transits; **Table 4**). Giving leeway for the uncertainty of real-time speed-through-water measurement on ships and *post hoc* single location current speed estimates used for validation, we considered an additional category for vessel participation that included vessels transiting the region at speeds-through-water of <13 knots. By using this criterion, 55% of vessel transits through the Slowdown region participated in the Slowdown trial (526 of 951 piloted transits; **Table 4**). The comparative Baseline period logged fewer (866) vessel transits through the study region, and of these 9% transited at <11 knots speed-through water, 19% at <12 knots, and 36% at <13 knots.

For the purpose of modeling 'observed' participation in Traffic Scenarios 2 and 5 in the regional vessel noise model (**Table 2**), the overall observed participation rate of 55% was adjusted upward by 2% to be 57% vessel participation. This was necessary as

TABLE 4 | Baseline transits, and slowdown participation counts and rates of participation by vessel category.

	Baseline	Slowdown	AIS calculated speed through water < 12 knots (within 1 knot of target)		AIS calculated Speed through water < 13 knots (within 2 knots of target)	
Vessel type	**Total transits**	**Total transits**	**Count**	**Overall participation rate**	**Count**	**Overall participation rate**
Bulk carrier + general cargo	464	491	199	41%	250	51%
Container ship	243	260	123	47%	155	60%
Car carrier	73	86	36	42%	52	60%
Tanker	61	74	34	46%	39	53%
Cruise/passenger ship	8	30	26	87%	27	90%
Other piloted vessels (e.g., yachts and large tugs)	17	10	3	30%	3	30%
Grand total	866	951	421	44%	526	55%

Overall percentage of vessels transiting at speeds-through-water <12 knots was 19% for Baseline compared to 44% during the Slowdown, and 36% of Baseline vessels transitted at <13 knots compared to 55% during the Slowdown trial.

TABLE 5 | Median reductions in vessel speed and source level by vessel category during the Slowdown trial period, based on median measured speeds of participating vessels inside the 11-knot Slowdown boundary.

Vessel type	Median baseline speed (knots)	Median slow-down trial speed (knots)	Median speed reduction (knots)	Extra time (min) to transit 16 nm trial region	Median reduction in source level of participating vessels (dB)
Bulk carrier + general cargo	13.6	11.8	−1.8	10.8	−5.0
Container ship	18.6	11.4	−7.2	32.6	−10.8
Vehicle carrier	17.1	11.5	−5.6	27.4	−10.4
Tanker	13.6	11.4	−2.2	13.6	−5.8
Cruise/passenger ship	17.0	10.8	−6.2	32.5	−8.8

*Extra time is calculated by comparing time to transit 16 nm while traveling at median Baseline and median Slowdown speeds. Broadband source level reductions were calculated based on the speed scaling coefficients measured during the trial (**Table 1**).*

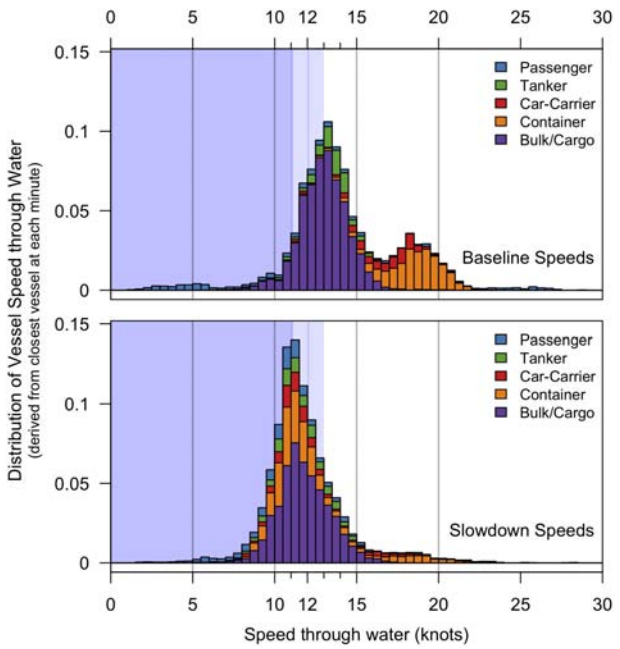

FIGURE 4 | Stacked plot of commercial vessel speeds through water at baseline **(Top)**, and during the slowdown trial period **(Lower)**, by vessel type. For the slowdown participation levels, the target vessel speed through water was set at 11 knots, and AIS-calculated speeds within 1 knot and 2 knots are assessed in the panels relative to *x*-axis ticks at 12 and 13 knots.

TABLE 6 | Quantiles of the broadband ambient SPL (10 Hz to 100 kHz dB re 1 μPa) for two Baseline months and the 61-day Slowdown trial measured on Lime Kiln hydrophone.

Frequency range	SPL dB difference between Slowdown and Baseline periods	
	Median p50% (p5%, p95%)	Mean (L_{eq})
[1]Broadband 10–100,000 Hz	−2.5 (−0.3, −1.4)	−2.0
[2]Broadband (bulk/cargo) 10–100,000 Hz	−1.5 (−1.6, +0.6)	−1.2
[3]Broadband (container) 10–100,000 Hz	−6.1 (−5.8, −2.5)	−5.2
[4]SRKW communication band 500–15,000 Hz	−2.1 (−1.1, −1.8)	−1.9
[5]SRKW echolocation band 15,000–100,000 Hz	+0.4 (+1.2, −0.2)	+0.4
[6]1st decade 10–100 Hz	−3.1 (+0.1, −1.1)	−2.1
[7]2nd decade 100–1,000 Hz	−2.3 (−0.1, −2.3)	−2.0
[8]3rd decade 1,000–10,000 Hz	−2.2 (−1.9, −1.3)	−2.0
[9]4th decade 10,000–100,000 Hz	−0.3 (+0.7, −0.4)	+0.7
[10]Unfiltered broadband 10–100,000 Hz	−1.2 (+0.2, −1.3)	−1.1
[11]Slowdown modeled SPL: average traffic at Lime Kiln	105.6 (96.3, 122.2)	107.0
[12]Slowdown modeled SPL: high traffic at Lime Kiln	109.1 (96.5, 123.4)	108.8
[13]Slowdown filtered broadband at Lime Kiln 10–100,000 Hz	107.9 (91.9, 125.1)	108.6

The dB difference between Baseline and Slowdown noise is negative for most measures, indicating a reduction in noise during the Slowdown trial period. The median dB difference was positive for high frequency bands including the 4th decade, and the echolocation bands. Footnotes 1 through 9 include filtered data with an AIS enabled vessel within a 6 km detection zone of the Lime Kiln hydrophone; periods of high wind, current, or with small boats present were also removed. When all minutes of data are included (unfiltered; footnote 10), the dB difference remains negative, across all measures except the 5th percentile (p5%). Finally, we provide a comparison of broadband model SPL (dB re 1 μPa) predictions for the Lime Kiln hydrophone location, under average and high baseline conditions (footnote 11 and 12, respectively), compared with night-time SPL data (21:00–06:00 PDT) recorded at Lime Kiln over two lunar months (August–September 2017; footnote 13).

commercial vessels were broken down by vessel type, and only integer values (counts of whole vessels transiting) were possible as inputs to the regional vessel noise model (**Table 3**).

Ambient Noise Measurements at Lime Kiln Listening Station

Ambient noise levels were lower during the Slowdown trial compared to Baseline despite the longer transit times past the hydrophone during the Slowdown trial (**Table 6**). We found reductions in broadband RL when commercial vessels were transiting within 6 km of Lime Kiln (and filtered to exclude confounding noise effects, n = 76,608 min) as well as when comparing all unfiltered data (n = 165,182 min). This is indicated by the divergent cumulative distribution functions of broadband noise (CDFs; **Figure 5**). The median broadband noise reduction at Lime Kiln for the Slowdown period when large vessels were within 6 km of the site (and filtered to remove heavy wind/current periods and periods of small boat noise) was 2.5 dB (from 116.9 dB re 1 μPa to 114.4), with a corresponding mean (or Leq) reduction of 2.0 dB (from 116.6 dB re 1 μPa to 114.6; **Table 6**). To put these results into perspective, a noise reduction of 2.5 dB is the equivalent of a 44% reduction in acoustic intensity. At the higher amplitude levels of the broadband noise distribution, noise levels were shifted by 1.4 dB, or a 28% reduction in acoustic intensity (i.e., p95% −1.4 dB).

There were sufficient numbers of transiting vessels to allow for Slowdown-Baseline comparisons by vessel type for bulk and general cargo carriers (bulk/cargo combined) as well as container vessels. Median noise levels at Lime Kiln during the Slowdown period were 1.5 dB lower for bulk and general cargo carriers, and 6.1 dB lower for container vessels during the Slowdown period. This reflects the larger (7.7 knots) reduction in average speed for container vessels compared to bulkers and cargo carriers (2.1 knots; **Figure 4**).

Analysis of the frequency bands that SRKW use for communication (500–15000 Hz; Heise et al., 2017) showed a clear benefit from the Slowdown trial (median reductions of 2.1 dB). However, at the frequencies used by SRKW for echolocation (echolocation masking bands 15–100 kHz; Heise et al., 2017), the noise distribution had shifted upward with a median peak +0.4 dB higher during the Slowdown, but with smaller variance as noise increased at the lower dB levels (p5% +1.2 dB), and decreased at the upper dB range (p95% −0.2 dB; **Table 6**).

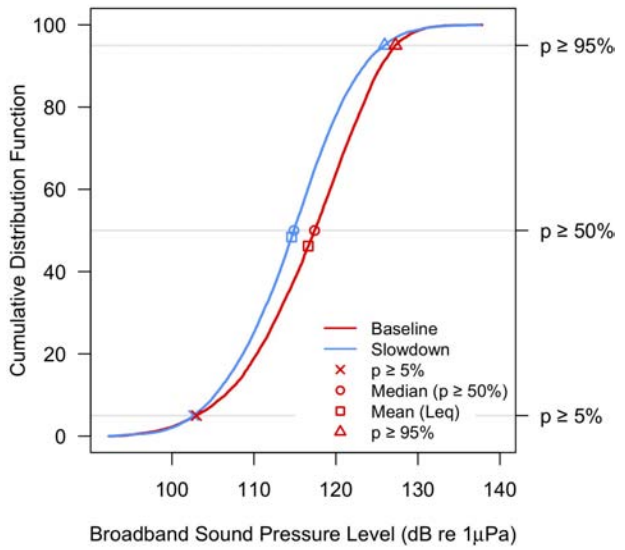

FIGURE 5 | Cumulative Probability Distribution Function integrates the area under the cumulative distribution of 1-min broadband SPL measures (10 Hz to 100 kHz dB re 1 μPa) at the Lime Kiln Hydrophone. Red line and symbols represent data collected during the Baseline period; blue line and symbols corresponds to the 61 days Slowdown period. Only minutes with an AIS-enabled vessel within a 6 km detection zone were included. Times with high wind, current, or with small boat noise present were removed. The dB difference at the 5%, 50th%, and 95th percentile of the CDF (p5%, p50%, and p95%) between Baseline and Slowdown periods is provided in **Table 6**.

Decade band analysis showed that median Slowdown noise reduction was highest in the 1st decade band (3.1 dB reduction in 10–100 Hz band; **Table 6**), and lowest in the 4th decade band (10 kHz to 100 kHz band; −0.3 dB). Reductions were 2.3 dB in the 2nd decade band (100–1,000 Hz) and 2.2 dB in the 3rd decade band (1–10 kHz).

Predictions of the noise model were validated by comparing against ambient noise data from the Lime Kiln hydrophone, pooled across the lunar months of August and September 2016 (61 days). In recognition that noise predictions from the model cannot replicate the number and frequency of small, non-AIS boats transits at Lime Kiln during the day, we performed the comparison using night-time Lime Kiln data only, between 21:00 and 06:00 PDT (**Table 6**). The mean and median model predictions were found to be in good agreement with the data recorded at Lime Kiln, although the overall spread of the model (i.e., difference between 95th and 5th percentiles) was smaller than that of the data. The greater spread of the data was expected since actual range of vessel traffic conditions in the data was greater than the two conditions considered in the model. Furthermore, ambient noise recordings at Lime Kiln will be affected to some extent by flow noise and variations in wind and wave noise that were not included in the noise models. Thus, we consider the SPL predictions as a good representation of noise conditions at Lime Kiln, especially when comparing the night-time period, in which the effects of small boat noise is considerably reduced.

Noise reduction results inferred from comparing Baseline to Slowdown speeds at the Lime Kiln listening station helped parameterize the regional vessel noise model. Compared to Baseline, the regional noise model indicated that the speeds and participation rates achieved during the trial resulted in noise reduction at Lime Kiln of 0.6 dB on an "average" traffic day (14 piloted vessel transits), and 1.5 dB on a "high" traffic day (21 piloted vessel transits). These values fall on either side of a median reduction of 1.2 dB reported at the Lime Kiln listening station (unfiltered data; **Table 6**), suggesting that for this one location the measured vessel noise reduction was between the average and high vessel traffic day of the equivalent grid output from the regional vessel noise model output.

SRKW Noise-Exposure Model (Behavioral Response Model)

The analysis of potential effects of slower vessels on the behavior and foraging of killer whales was undertaken using the SRKW noise-exposure simulation model. For a piloted vessel to have an effect on a killer whale, the whale must occur in the Slowdown region at the same time a vessel is in transit. The spatial depictions of the median (top panels) and loudest 95th percent noise exposure maps are shown in **Figure 6**. These panels show, under the assumption of uniform distribution of SRKW, the likelihood for a low behavioral response to occur under different regional vessel noise model scenarios (left to right). The striking difference in probability of observing a behavioral response between the top and bottom panels reflect that the region is without a commercial vessel for >50% of the day and the median response to transiting vessels is similarly low. When a vessel is transiting the region (lower panels D, E, F), the potential for a Low behavioral response is at least 30% across a large spatial region adjacent to the shipping lanes, with important reductions in that footprint as the participation in the 11 knot Slowdown is increased (i.e., less likely in right panels of **Figure 6**), particularly off of Lime Kiln Point.

The SRKW noise-exposure model stochastically includes SRKW occurrence according to the spatial model of habitat use (2001–2011 data). As SRKW predominately occur in the Slowdown region in the waters off Lime Kiln Point in Haro Strait, the region of maximum effects on SRKW behavior occurs here (**Figure 7**). Similarly, by slowing ships transiting past this hotspot of SRKW occurrence, the reductions in the effects of commercial vessel traffic is highlighted by the reductions in the numbers of Low BRs over Baseline scenarios (**Figure 7**, compare left panels to middle and right panels).

Collectively, results of the SRKW noise-exposure model indicated that the number of behavioral disturbances from commercial vessel noise declined when vessel speeds were reduced (**Figures 8**, **9**). The median 'potential lost foraging time' from low-severity BRs (5-min disruptions) and moderate severity BRs (25-min disruptions) decreased by 29 and 20% per day per whale for the modeled trial (i.e., under 57% participation) compared to Baseline conditions with average traffic volume and speeds. This 'potential lost foraging time' was particularly reduced in moderate BRs declining by >50% for a model scenario when all vessels observe speeds of 11 knots (compared to Baseline;

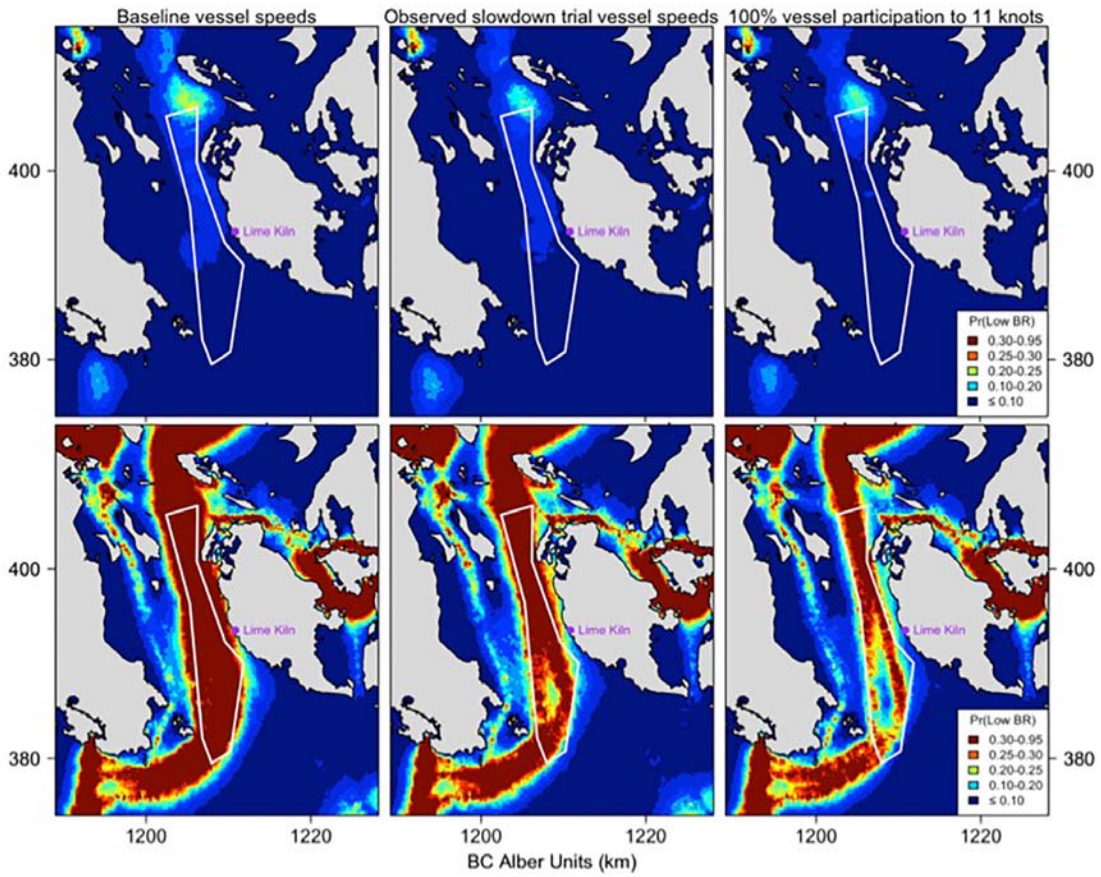

FIGURE 6 | The top panels are the spatially indexed probability of observing a low behavioral response, or Pr(Low BR), by applying the dose-response function to the 'median' (p50%) broadband noise level for an average day, under scenarios 1, 2, and 3. The bottom panels are the spatially indexed probability of observing a Low BR given the noisiest 95th percentile (p95%) of the daily broadband noise-exposure under the same three scenarios (1, 2, and 3). If SRKW were uniformly distributed across the region, we would expect the Pr(Low BR) to be related to the color scale of these maps. The three panels represent the different speed scenarios within the slowdown region with Baseline speeds on the left, observed participation in the center, and 100% participation to 11 knots depicted on the right panel. Note that the same color scale is used for all map panels, and the Slowdown trial boundary is represented by the white polygon boundary.

Figures 8, **9**). There were negligible changes in residual click masking across scenarios reflective of the smaller shifts in SPL in the 4th decade band (**Table 6**).

Overall, the expected consequence of moderate severity BRs had the greatest influence on the total 'potential lost foraging time,' particularly under baseline conditions, and had the greatest benefit from slower vessels (compare black to red bars in **Figures 8**, **9**). Moderate BRs accounted for 41–58% of the potential time lost whereas low severity responses accounted for 12–27% of the loss. This is a consequence of assuming low severity BRs of SRKW had an effect-duration of 5 min in the noise-exposure model, while moderate severity BRs were assumed to have an effect duration of 25 min. Despite requiring a higher received noise level for a moderate response to occur, the fivefold impact of moderate relative to one low severity BR translates to the biggest hindrance in potential foraging time in our model. Thus, noise reductions from any vessel that no longer results in a moderate severity behavioral response will translate into increased potential benefits to SRKWs.

The uncertainty in model outputs was high (i.e., note 95% confidence intervals in **Figure 8**). This variability in outcomes corresponded to wide 95% confidence intervals (95-C.I.) around model estimates 'potential lost foraging time' of 36.8 h (95-C.I. 16.5, 64.9) for Baseline, 29.5 h (95-C.I. 13.1, 51.6) under 57% participation, and 22.0 h (95-C.I. 10.1, 41.5) with all vessels transiting at <11 knots (**Figure 9**, left 3 bars, average traffic volume). Similarly, under high traffic volume, estimated 'potential lost foraging time' of 49.0 h (95-C.I. 20.6, 84.4), 37.4 h (95-C.I. 15.9, 65.5), and 26.7 h (95-C.I. 11.7, 49.8) for Baseline, under 57% participation, and 100% participation at <11-knot speeds (**Figure 9**, right 3 bars, high traffic volume). The wide confidence intervals around model outputs reflect the uncertainty around the inputs to the noise-exposure model. Model inputs included the spatial variability of habitat use across the study area by months August, September, October, the uncertainty in the probability of eliciting a behavioral response from the received level of noise at the whale's location, the parameter uncertainties in the dose-response function (reflecting that of the data sources

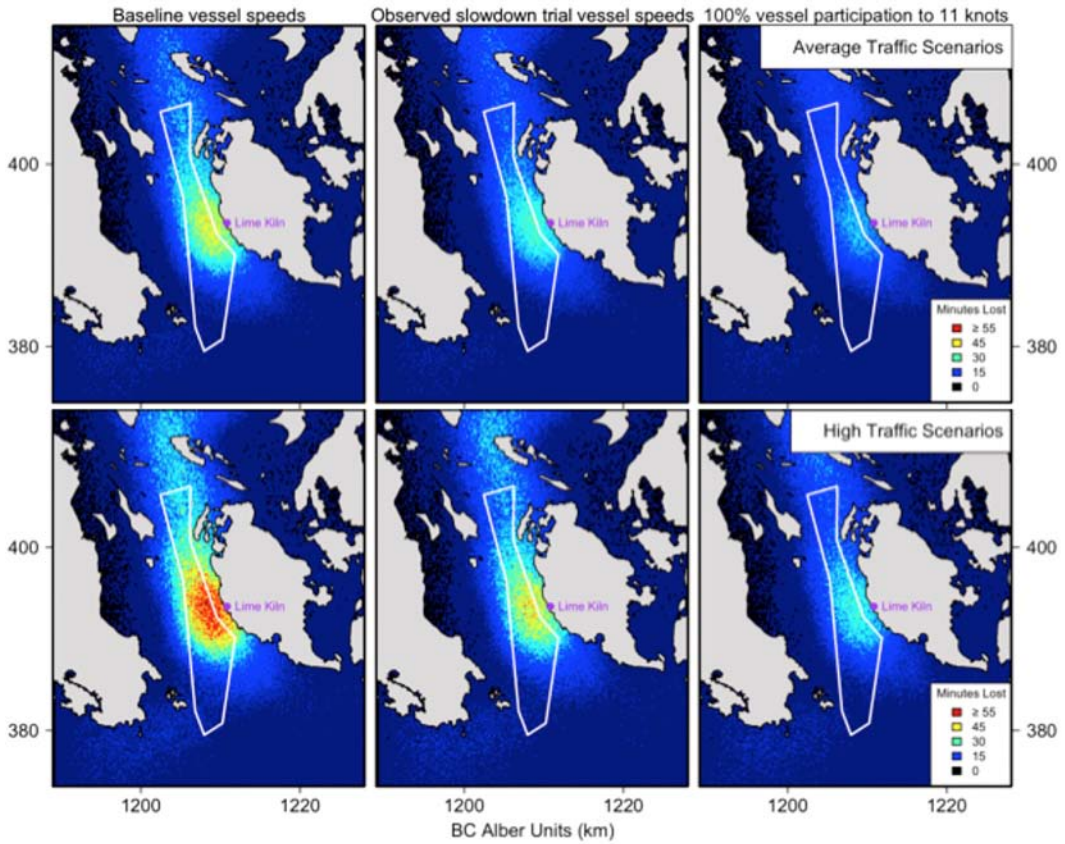

FIGURE 7 | Results of the SRKW noise-exposure model for the Slowdown region accumulated over each of the 288 regional acoustic maps that depict commercial vessel broadband noise over an average traffic day (top three panels, scenarios 1, 2, 3) and high traffic day (bottom three panels, scenarios 4, 5, 6). For a Low BR to occur in the Slowdown region, a SRKW must be present and a commercial vessel must be transiting close enough to the SRKW to elicit a behavioral response. The six panels show the spatially indexed number of minutes of 'potential lost foraging time' from a Low severity behavioral responses by a southern resident killer whale. The 11-knot slowdown trial boundary is shown in white.

from which it was derived), and the stochasticity of both ships and whales moving in time and space.

Despite these sources of input variability, the results of the noise-exposure model collectively suggest important benefits to SRKW through reductions in 'potential lost foraging time' at the core of their foraging habitat in Haro Strait (**Figure 7**). For an average traffic volume day at the participation rates observed in this trial, 'lost foraging time' would be decreased by 21.5% and decreased by 39.6% if full participation of 11 knots for all commercial vessels was achieved (**Figure 9**). For high traffic volume days, these reductions are increased to 21.6% for the observed participation rates, and 44.1% for 100% vessel participation.

DISCUSSION

Trial Vessel Speed (AIS), Reported Participation

Low-frequency energy from commercial ships is the principal source of ambient noise below 1 kHz within the deep ocean (Wenz, 1962; Urick, 1984; National Research Council of the U. S. National Academies [NRC], 2003), and noise in the low frequency bands dominates the broadband spectrum of ambient noise in the Salish Sea (Bassett et al., 2012; Cominelli et al., 2018). Motivated by an interest in better understanding and reducing the effects of commercial vessel traffic in SRKW critical habitat, this study was proposed by an industry-led multi-stakeholder initiative of the ECHO program of the Vancouver Fraser Port Authority. Participation by vessel owners and operators was voluntary throughout the 2017 Slowdown trial, facilitated by the BC Coast Pilots. During the trial, 44% of 951 piloted transits achieved a speed of less than 12 knots, and 55% achieved a speed of less than 13 knots. Given this rate of *trans*-boundary participation for transits through the international shipping lanes of Haro Strait, this study highlights the benefits of voluntary (non-regulatory) vessel Slowdowns as a meaningful noise reduction measure.

Vessel Source Level Reductions

Marine traffic generates high-energy noise in the ocean that can propagate across considerable distances underwater. A positive

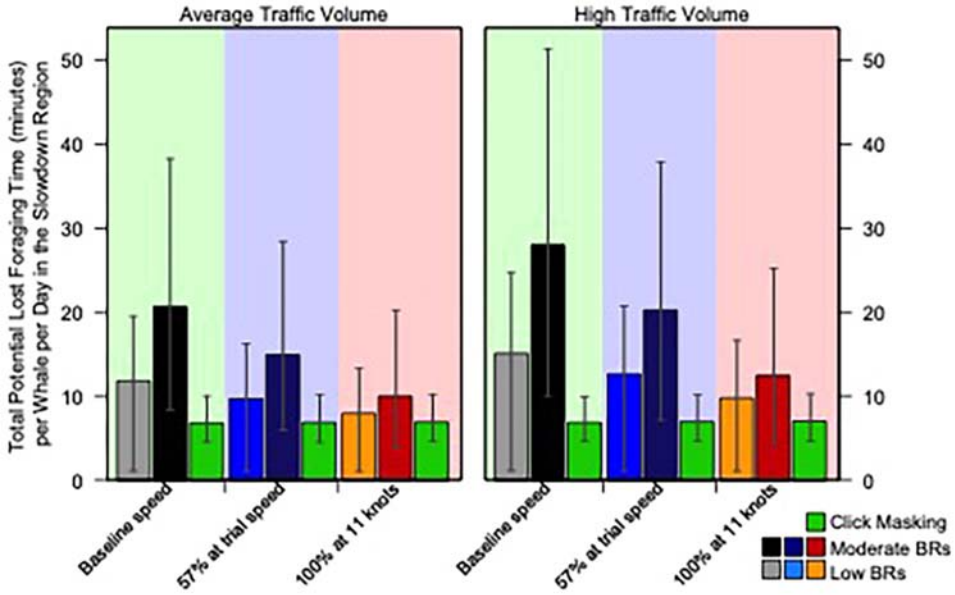

FIGURE 8 | Modeled SRKW 'Potential Lost Foraging Time' for each of the six Vessel Speed and Traffic Scenarios (Scenarios 1 through 6, **Table 2**). Bars show 'Potential Lost Foraging Time' for each of Low BR, Mod BR, and Click Masking as determined by the SRKW noise-exposure simulation model. Average and High Traffic Volume Scenarios corresponds to the descriptions in **Table 2**. The reduction in 'potential lost foraging time' attributable to moderate BRs declined by >50% in model scenario when all vessels observe speeds of 11 knots (compared to Baseline; i.e., by comparing heights of black to red bars).

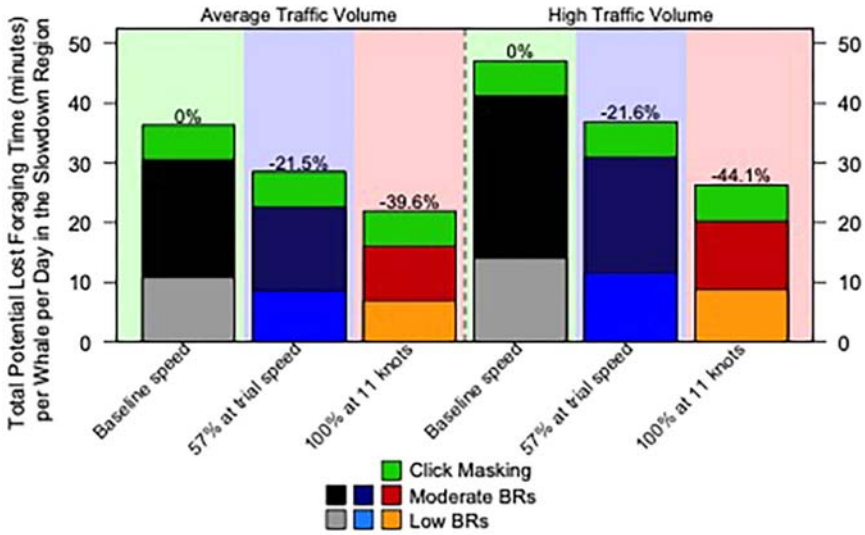

FIGURE 9 | Cumulative 'Potential Lost Foraging Time' for each of the six Vessel Speed and Traffic Scenarios (Scenarios 1 through 6, **Table 2**). Bars are stacked by the number of lost minutes for each of Low BR, Mod BR, and Click Masking. 'Potential Lost Foraging Time' is determined using the SRKW noise-exposure simulation model. Average and High Traffic Volume Scenarios corresponds to descriptions in **Table 2**.

relation between source level reductions with decreased speed was assumed, prior to the Slowdown trial, based on intuition and documented elsewhere (e.g., McKenna et al., 2012; Houghton et al., 2015; Frankel and Gabriele, 2017), but the relation was not well understood. Particularly, there was a lack of speed scaling data for the Salish Sea shipping lanes, and the relations available were based on a limited number of historic post-World War II commercial vessels (Ross, 1976). Extensive source level reduction data collected during the trial demonstrated that the biggest reductions in source levels were for container ships that reduced their speed by 7.2 knots, corresponding to a 10.8 dB reduction in median source levels (MacGillivray and Li, 2018b). This vessel type made up 27.7% of transits, compared to 51.6% of transits attributed to bulk/general cargo vessels, that move slower

and have more modest median source level reductions (5.0 dB). The greater reduction in noise from container vessels reflects the larger speed and source level reduction of faster-moving container vessels. However, due to the larger speed scaling coefficient of bulk/cargo vessels, the mean per-knot reductions are greater for this class of vessels (2.8 dB per knot) compared to container vessels (1.5 dB per knot). As bulk/cargo ships comprise >50% of the commercial fleet, the biggest reduction in ambient noise would be to reduce speeds of this vessel class.

Ambient Noise Measurements at Lime Kiln Listening Station

Ambient noise in the ocean is the sound field against which signals must be detected. As southern resident killer whales must send and receive sound waves in order to navigate, communicate and forage successfully, the level of ambient noise can have important effects on the function of their habitat Department of Fisheries and Oceans Canada [DFO] (2018). With a growing global shipping industry and with the rate of noise continuing to rise (Hildebrand, 2009), there is an increasing potential for adverse effects on the underwater noise field of the Salish Sea (Veirs et al., 2016). Additionally, multiple proposed fossil fuel-related and port development projects in the Salish Sea have the potential to further increase marine vessel traffic and negatively effect ambient noise levels (Gaydos et al., 2015). Recent work has shown that commercial vessel noise in the Salish Sea significantly increases not only the ambient broadband noise levels, but also includes significant acoustic energy in high frequency bands used for echolocating and finding prey (Veirs et al., 2016). A population viability analysis explored the relative importance of the primary anthropogenic threats to southern resident killer whale survival, and after Chinook prey availability (their primary threat), commercial vessel noise and disturbance was identified to be of sufficient magnitude to shift SRKW population trajectories from slow positive growth into decline (Lacy et al., 2017). Given this context, it is therefore important that we found that slowing piloted vessels reduced ambient broadband sound pressure levels at the core of SRKW summer foraging habitat. Using unfiltered data, the median broadband reduction of 1.2 dB corresponds to a 24% reduction in sound intensity, despite the Slowdown period having 8.7% more piloted vessel transits than the baseline period and slower vessels taking longer to transit the 16 nm section. Large commercial vessels generate noise with most energy being emitted at frequencies below 1,000 Hz, with substantial tonal contributions as low as 10 Hz (Ross, 1976; Arveson and Vendittis, 2000; McKenna et al., 2012). When isolating noise comparisons of Baseline to Slowdown to periods when piloted vessels were within 6 km of the listening station and removing key confounding influences such as boat noise and wind, the result of slowing commercial vessels was a reduction in vessel noise emissions and lower received SPL over the entire frequency range (broadband), with the greatest relative reductions observed below 100 Hz at frequencies that commercial vessels are typically loudest. Therefore, despite slower vessels taking longer to transit the 16 nm section of the shipping lanes, the net result was lower vessel noise footprints particularly at low frequency bands.

Comparison of SRKW echolocation-related frequency bands at Lime Kiln (15,000–100,000 Hz, Heise et al., 2017) showed an increase of 0.4 dB re 1 μPa in the median noise level during the trial period compared to the Baseline period. However, the distribution was narrower (had a smaller variance), with the upper tail of the distribution having a lower SPL during the trial compared to Baseline (i.e., the quietest periods were louder, but the loudest periods were quieter during the Slowdown compared to Baseline). Due to the ∼4 km distance between passing ships and the hydrophone, much of the high frequency sound coming from vessels is attenuated below background and internal hydrophone system noise by the time it reaches the Lime Kiln listening station. Similarly, Veirs et al. (2016) found commercial ship noise in bands used by SRKW for communicating and foraging (echolocation bands) could be detected at ranges of at least 3 km, however, the limitations of the hydrophone to accurately measure high frequency sounds at such low intensity (<85 dB mean value) is questionable. Nonetheless, as an important SRKW foraging hotspot is located near the shipping lanes (Olson et al., 2018) offshore from Lime Kiln Point, and with many gaps still in our understanding of echolocation click making (Erbe, 2002), or to what degree SRKW can compensate for high noise levels (and when they can no longer) (Holt et al., 2009; Zollinger and Brumm, 2011), more targeted studies than the one we describe here are required to understand the effect of the slowdown on these high frequency bands.

SRKW Noise-Exposure Model (Behavioral Response Model)

Studying whale behavior in the presence of vessels is challenging. The SRKW noise-exposure model is a temporal and spatially explicit approach designed to evaluate the potential effects on SRKW of multiple moving noise sources within their preferred summer foraging habitat off Lime Kiln Point. The noise-exposure approach used in this study, uses a 2-d surface density of SRKW habitat use coupled to the probability of a change in behavior by the whale (e.g., stops foraging) for a combined broadband received level and high frequency echolocation click masking. Underlying the dose-response relationship is the concept that at higher received noise levels (i.e., a high noise dose), there is a higher probability of a behavioral response or disruption, and that this disruption has the potential to last longer than the time period of the dose (e.g., through a switch in behavior). The scientific procedure for estimating and predicting biological impacts from noise exposure has been based traditionally on the dose-response paradigm (see, for example, Southall et al., 2007; Finneran and Jenkins, 2012). This paradigm assumes that the extent of the biological impact or consequence can be predicted by the noise received level at the animal. Our joint analysis of resident killer whale noise-exposure datasets supported the hypothesis of an increase in severity of behavioral response in response to increasing SPL, albeit with large variance associated with the relationship. There are other studies with additional supporting evidence that this relation does persist for this and other species in other areas. For example, sperm whales exposed to low frequency active sonar (LFAS, 1–2 kHz) changed from foraging to non-foraging behavior (Isojunno et al.,

2016), and killer whales, sperm whales and Blainsville beaked whales have all been shown to respond to sonar noise with increasing severity of response (e.g., Miller et al., 2012, 2014; Harris et al., 2015). Additionally, we derived the dose–response functions in our study from local killer whales exposed to vessel noise, and included the uncertainty in the observational input data as recommended by Miller et al. (2014). This was to acknowledge the variability in individual responses to different noise levels and sources. Others have found behavioral responses can depend on a number of covariates including an individual's prior experience to noise (Constantine et al., 2015), the habitat quality (Robertson et al., 2013), the distance from the sound source (Madsen et al., 2006; deRuiter et al., 2013; Dunlop et al., 2017), and perhaps by such factors as age, sexual condition, and gender (female killer whales seemed to be more likely than males to respond to the passage of a ship; Williams et al., 2014; Gomez et al., 2016). Context-specific dose-response functions with separate functions for different behavioral states (e.g., Ellison et al., 2012), could reduce uncertainty in the predicted behavioral effects, but such an approach would require increased understanding of these contexts (Harris et al., 2018). By taking a conservative approach and assuming that any behavioral response results in a change from foraging to a non-foraging behavior, our average traffic volume results suggest that the voluntary slowdown resulted in median gains of 21.5% over Baseline losses, whereas a fully participating (to 11 knots) commercial fleet might be expected to have median gains of 39.6% or more against Baseline 'potential lost foraging time.' It is notable that during high traffic volumes (an increase of 50% in vessel transit number) and participation rates of 57%, the 'potential lost foraging time' is similar to the time lost during an average traffic volume and baseline vessel speeds although we add a precaution that there is uncertainty associated with the range of expected results.

The reductions in noise levels during the Slowdown trial at Lime Kiln, and also more widely in the Salish Sea based on modeled predictions is not simply due to changes in the long-term ambient noise levels, but rather a result of short-term reductions in high levels of noise. For this reason, the simulation design evaluated the maxima over each 5-min time increment and observed the changes in risk of a short-term low or moderate severity behavioral response. Veirs et al. (2018) estimated that half of the ship noise in the Salish Sea comes from just 15% of the commercial vessel fleet, and since the results of the noise-exposure model found reducing the loudest sounds, or those most likely to induce a moderate behavioral response, conferred the greatest benefit to SRKW, emphasis should be placed on design modifications of these ships (Leaper and Renilson, 2012; Merchant, 2019). There remain, however, many gaps still in our knowledge of how ship design modifications might stack up relative to other potential mitigations such as shipping lane alterations, and commercial vessel convoys, and how these potential modifications would interact with commercial vessel slowdowns.

In this simulation study of vessel slowdowns, short-term reductions in high source levels translated to the least lost foraging time. Short-term noise reductions result in lower associated received levels by whales in the vicinity, and since the probability of a behavioral response decreases in a non-linear way according to the dose-response function, there are disproportionate benefits at particular noise levels. For example, slowing a bulk/cargo vessel past a SRKW that otherwise would receive a 130 dB re 1 μPa noise level, and assuming the speed reduction causes a 2.8 dB reduction in noise received by the whale, this will reduce the whale's probability of a Low BR by 14.5% (from 52.7 to 38.2%). Likewise, a 2.8 dB reduction in received noise levels when ambient noise levels are 117.4 dB, (i.e., the median ambient noise level when a commercial vessel is within 6 km of Lime Kiln), would result in only a 4.9% reduction in SRKW probability of experiencing a Low BR. As SRKW are typically 100–300 m offshore of Lime Kiln toward the shipping lanes (Veirs et al., 2016), the median ambient noise level for a transiting commercial vessel would be louder, and expected to have a greater benefit (than −4.9%) for SRKW.

In evaluating benefits of slower vessels to SRKW, commercial vessels transiting at the 11-knot slowdown speed pass through the study area more slowly. Therefore, despite lower instantaneous sound intensity and probabilities of BRs, the net benefit must consider that the exposure duration will be longer. As a vessel moves through an area, there is a moving acoustic footprint around the vessel. Low and moderate severity behavioral responses can occur within these acoustic footprints. As the vessels decrease speed, this footprint decreases in area and therefore, at locations more distant from the shipping lane the exposure duration for a given exposure level decreases. For example, a whale located within the Lime Kiln grid square for a 24-h day during average traffic conditions (normal speed, average number of vessels), would be exposed to noise levels of at least 121 dB re 1 μPa for fourteen, 5-min time windows (of 288 possible daily windows). If 100% of the same number of vessels participated in an 11-knot slowdown, there would be only four, 5-min time windows at 121 dB re 1 μPa despite the longer passage times. Therefore, the additional 3.3 min it takes for a container vessel to transit a 1.4 km radius circle around a SRKW at 11 knots (compared to time at 18 knots), is offset by the reduction in the probability of an adverse reaction to the vessel. As the dose-response function is affected by the maximum noise-exposure during the passage time period and not by the median value of that interval, the longer passage time does not result in more time periods of risk for a whale as the lower source levels are less likely to lead to functional disturbance of SRKW. When viewing results spatially, slowing vessels down reduces the relative risk of excessive noise exposure such that when vessels transit at 11-knot slowdown speeds, the width of the 'red' footprint with expected probability of a low BR $\geq$ 0.30 (**Figure 6**) dropped by half in the SRKW hotspot area adjacent to Lime Kiln Point. Therefore, slower vessels have smaller footprints and lower risk of eliciting behavioral responses, implying an important improvement in the function of that habitat. As SRKW are predominately found in this region off Lime Kiln Point, the 61-day Slowdown trial demonstrated important relative improvements to their summer foraging habitat in this region.

An alternate approach to converting changes in behavior to 'potential lost foraging time' is one that takes into account

the context of the interruption. There are many studies that demonstrate the importance of ecological context when assessing noise induced behavioral responses (for a review see Gomez et al., 2016). One approach to better understanding the importance of these behavioral responses may be to view them in the context of a biologically meaningful currency such as an energy budget (Harris et al., 2018). Such changes in energy budgets can then be used in models that extrapolate short-term effects to long-term effects (Christiansen et al., 2013). For example, for seasonal feeders such as blue whales that rely on dense prey aggregations, the energetic consequences of foraging disruption during periods of high prey availability can be significant (Goldbogen et al., 2013). Or equivalently, a disruption of a SRKW in its summer foraging grounds may have a higher energetic cost due to a lost prey capture, than a disruption in an area where active foraging is not common. A study of juvenile European eels showed experimentally that the effects of (playback) noise from passing coastal ships were condition-dependent, with individuals in worse condition most affected (Purser et al., 2016). Due to recent evident of multiple individuals showing signs of nutritional stress, or 'peanut head' syndrome (Durban et al., 2009), there is considerable variation in SRKW body condition. If animals are already in poor body condition as a consequence of poor Chinook salmon availability (Lacy et al., 2017), additional lost foraging opportunities could have both direct nutritional (energy) cost as well as indirectly through increased risk of parasite infection and disease, and/or reproductive performance. It is therefore worth designing field studies to collect the data to better understand these individual context-specific details, and thereby facilitate its inclusion into any model. Unfortunately, the energetic cost of any kind of behavioral response in marine mammals is not well quantified (Harris et al., 2018). Thus, there remain many uncertainties as to how a behavioral response to noise exposure could be translated into energy loss and then applied to real-world scenarios, and how these responses could be quantified into long-term individual and/or population-level effects remains an open question. By focusing our results from the simulation model on the percent reduction rather than the absolute reduction in 'potential time foraging lost' and applying these assumptions equally across scenarios, we have shown that the overall risk of behavioral responses was lower during the Slowdown trial compared to comparable baseline periods.

Understanding both the response and the variation in the response to commercial vessel disturbance is important both for assessing population consequences for SRKW and in management decisions. There are other examples of how consensus decision-making and sound science can be used to reduce the effect of marine shipping on whales in Canada (e.g., St. Lawrence estuary, the Bay of Fundy; Laist et al., 2014; Parrott et al., 2016), and internationally (e.g., Hauraki Gulf, New Zealand; Constantine et al., 2015). In this study, the use of spatially explicit models based on local data sources with transparent assumptions provided significant advantages over unsubstantiated opinion to the decision-making process by providing quantitative support in the form of maps and outputs (with both temporal and spatial variability) assessing the 'potential lost foraging time' in response to management decisions.

CONCLUSION

The speed reductions achieved by commercial vessel pilots participating in this study resulted in significant reductions in broadband noise exposure from all commercial vessel types, as well as noise reductions across most frequency bands. These reduced vessel speeds translated to important reductions to noise exposure risk for whales in an area of importance for foraging whales. By assuming that fewer negative behavioral responses to noise exposure from SRKW translates to fewer lost foraging opportunities and better foraging success, the results of this voluntary trial showed that reducing vessel speeds is likely to improve the habitat quality of a summer foraging hotspot in a region that overlaps with commercial shipping lanes.

This study was motivated by the aim to better understand, quantify, and manage shipping impacts on marine fauna. This work contributes to our understanding of how noise from commercial vessel traffic affects an important region of summer foraging habitat for SRKW, and how slowdown mitigations may benefit SRKW at frequencies important to this population. Taken together, the transparent assumptions behind a regional vessel noise model combined with a dose-response noise-exposure model allows a comparative exploration of the relative value of various noise mitigation options. This approach has provided a framework for making decisions about how to reduce the effect of vessel noise on these endangered whales. However, there remain many gaps still in our knowledge of how SRKW are affected by commercial ships and how speed reductions stack up relative to other potential mitigations such as shipping lane alterations, commercial vessel convoys, and ship design modifications. The advantage of this approach may be in allowing a comparative exploration of the relative value of various noise mitigation options. As noise-producing activities in the ocean are likely to continue to increase, there is a pressing need for better understanding and mitigation of sound-producing activities. In the future, we recommend efforts be put into doing a simultaneous observational study on SRKW to quantify effects of habitat displacement, and/or duration of behavioral responses to commercial vessels. This will ensure there is data to support the assumptions of any future simulation model of noise effects on this population.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and/or the **Supplementary Files**.

AUTHOR CONTRIBUTIONS

OR, DT, and KT designed and directed the project. ZL, AM, and JW performed all AIS, acoustic measures, and models. RJ and

JW developed the theoretical framework for the noise-exposure simulation model. All authors equally wrote the article.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00344/full#supplementary-material

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Caribbean Sea Soundscapes: Monitoring Humpback Whales, Biological Sounds, Geological Events, and Anthropogenic Impacts of Vessel Noise

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Assessing marine soundscapes provides an understanding of the biological, geological and anthropogenic composition of a habitat, including species diversity, community composition, and human impacts. For this study, nine acoustic recorders were deployed between December 2016 and June 2017 off six Caribbean islands in several Marine Parks: the Dominican Republic (DR), St. Martin (SM), Guadeloupe east and west (GE, GW), Martinique (MA), Aruba (AR), and Bonaire (BO). Humpback whale song was recorded at five sites on four islands (DR, SM, GE, GW, and MA) and occurred on 49–93% of recording days. Song appeared first at the DR site and began 4–6 weeks later at GE, GW, and MA. No song was heard in AR and BO, the southernmost islands. A 2-week period was examined for the hourly presence of vessel noise and the number and duration of ship passages. Hourly vessel presence ranged from low (20% – DR, 30% – SM), medium (52% – MA, 54% – BO, 77% – GE) to near continuous (99% – GW; 100% – AR). Diurnal patterns were observed at BO, GE, and MA with few to no vessels present during night time hours, possibly reflecting the activity of recreational craft and fishing vessels. At the DR and GW sites, vessel traffic was ubiquitous for most of the day, likely reflecting heavy cruise ship and container ship presence. Soundscapes were diverse across islands with persistent fish choruses, sporadic sperm whale (*Physeter macrocephalus*) and dolphin (*Delphinidae*) presence at BO, minke whales (*Balaenoptera acutorostrata*) from late December to late February at MA and an earthquake recorded across all sites. These analyses provide an important first step in characterizing the

health and species richness in Caribbean marine parks and demonstrate a surprising high anthropogenic foot print. Vessel traffic in particular contributes adversely to marine soundscapes, masking marine mammal sounds, potentially changing typical animal behavior and raising the risk of ship strike.

Keywords: passive acoustic monitoring, soundscape, marine mammal, humpback whale, anthropogenic noise, Caribbean

INTRODUCTION

Sound has low attenuation and travels effectively in sea water, moving approximately five times faster than it does in air. Where vision fails (at night, at depths where sunlight does not penetrate, and in turbid water), sound production and hearing function as efficient communication and sensory mechanisms. Since sound is efficient in light-limited habitats, marine animals have evolved to rely heavily on their use of sound for communication, foraging, and navigation (e.g., Benoit-Bird and Au, 2009; Radford et al., 2011; Janik and Sayigh, 2013, respectively). Passive acoustic monitoring (PAM) lets us exploit this key sensory modality and overcome some of the challenges of traditional visual surveys to learn about marine animals and their environment simply by listening. When we listen, we have the opportunity to determine species presence (e.g., Heenehan et al., 2016), distribution (e.g., Davis et al., 2017), migration (e.g., Risch et al., 2014), and abundance (e.g., Van Parijs et al., 2002). We can also characterize the acoustic environment or soundscape of an area. A soundscape is considered to be all sounds present in a given place over a certain period of time (Krause and Gage, 2003; Pijanowski et al., 2011).

The types of sounds that comprise a soundscape vary between sites and vary on multiple timescales (for examples see McWilliam and Hawkins, 2013; Erbe et al., 2015; Haver et al., 2017; Heenehan et al., 2017) but usually include a combination of sounds from three broad categories: (1) sounds from geological or physical processes (e.g., from earthquakes, wind, and rain), sometimes referred to as the geophony; (2) sounds from non-human living things (e.g., sounds from marine mammals and fish), sometimes referred to as the biophony; (3) the sounds produced by humans (e.g., vessel noise, seismic surveying, and sonar), sometimes referred to as the anthrophony (Krause and Gage, 2003; Pijanowski et al., 2011). Identifying the different components that comprise a soundscape can provide insights into the composition of a given marine environment. This soundscape characterization can prove a useful way for monitoring long term changes within a given environment, such as increases or decreases in species composition and/or anthropogenic noise impacts that may affect the ecological diversity and health of a given site (e.g., Haver et al., 2017).

Sounds within these three soundscape categories overlap in time, space, and frequency. Therefore, acoustic recordings capture multiple sound sources within and across these categories (Van Opzeeland and Boebel, 2018). Although the specific components of a soundscape preserved in a recording depends on many factors including the location of the recorder, the location of the sound source(s), and the sampling rate, recordings may be used to characterize spectral and temporal overlap between sounds and explore potential masking (Van Opzeeland and Boebel, 2018). Masking occurs when a sound not only overlaps with but actually affects or interferes with the ability to receive another sound (American National Standards Institute, 1994; Clark et al., 2009; Erbe et al., 2016). For example, a well-studied masking relationship is the one between shipping noise and baleen whale calls in Stellwagen Bank National Marine Sanctuary off the coast of Massachusetts, United States. The temporal, spatial, and spectral overlap of these sounds has resulted in high levels of masking (Cholewiak et al., 2018) and a large loss of communication space for these animals (Hatch et al., 2012, 2016).

In recent years, the importance of managing acoustic habitats, such as marine parks, sanctuaries or areas of biological importance, in order to minimize anthropogenic impacts has become increasingly recognized (e.g., Hatch and Fristrup, 2009; Hatch et al., 2012; Williams et al., 2014, 2015; Merchant et al., 2015; McKenna et al., 2017). Several designated marine parks exist within Caribbean waters, established primarily for the protection habitat such as coral reefs, mangroves and sea grass, as well as marine protected species or species of local importance, such as humpback whales, *Megaptera novaeangliae* (Knowles et al., 2015; di Sciara et al., 2016).

Marine shipping, particularly large ocean container ships, hydrocarbon transport, and cruise ships, is a recognized and persistent anthropogenic source of low-frequency ocean noise, contributing to the masking of essential sounds produced and heard by marine animals and fish (e.g., Weilgart, 2007; Hatch et al., 2008; Hildebrand, 2009; Erbe et al., 2012; McKenna et al., 2012; Merchant et al., 2014; Williams et al., 2015). The Caribbean region is largely made up of small and specialized open island economies, which import a large proportion of their consumer goods. In addition, any local production of goods and services is heavily dependent on the import of raw materials and unfinished parts. In 1996, the Caribbean region's level of foreign trade, as a proportion of GDP, was 78%, compared with 25% for Latin America for the same period (Hoffmann, 1997). Imports arrive either by air or by sea, resulting in heavy marine traffic throughout the Greater Caribbean Sea. As a result ship-generated noise presents a significant threat to the regions' marine ecosystems and their underwater soundscapes.

For this study acoustic recordings were collected across seven sites throughout the Caribbean, of which six sites were situated in marine parks. The primary aim was to identify the main soundscape contributors in order to establish an acoustic baseline and understand levels of anthropogenic noise and evaluate potential impacts on marine mammals both within and between sites. As humpback whales are an identified species of importance

across most marine parks, we used the extensive research on humpback whale utilization of the Caribbean breeding grounds to provide a basis for determining the locations of our study sites (Kennedy, 2018). Male humpback whales sing on breeding grounds (Herman, 2017), and their song is loud and persistent (e.g., Payne and McVay, 1971; Vu et al., 2012), making it an excellent acoustic indicator of these whales' occurrence in an area. The extent to which vessel noise overlaps with humpback whales' communication space can provide insight into potential masking and interference of noise with breeding behavior. Besides impacting whales' communication space, vessel noise can also affect the stress hormone levels of whales (Rolland et al., 2012), which is of conservation concern, especially in an area where whales are calving (e.g., Bejder et al., 2019)[1].

A secondary focus of this study was on the timing of arrival and departure of humpback whales at different sites in the Caribbean. Several thousand humpback whales migrate from feeding grounds in the northern North Atlantic to breed in the Caribbean in winter and spring (Stevick et al., 1998; Bettridge et al., 2015). The only other known breeding ground for these whales in the North Atlantic is in the waters of the Cape Verdes Islands, off West Africa (Ryan et al., 2014), used by less than 300 individuals. It appears that humpback whales that migrate to the southeastern Caribbean (Stevick et al., 2018) mostly do so late in the breeding season (mid-March to May), and are distinct from those occurring earlier (January – early March, Stevick et al., 2018). Some individual whales have been photographically identified in separate years off the Cape Verdes islands and in the southeastern Caribbean (Stevick et al., 2016). Further, it appears that the feeding grounds of the humpbacks occurring later in Caribbean waters are off Europe and Scandinavia, rather than in the western North Atlantic (Stevick et al., 2018). The abundance of this later-migrating population of humpback whales is likely much smaller than the earlier migrating whales, and so is of greater conservation concern, hence our interest in their overlap with vessel noise.

MATERIALS AND METHODS

Nine passive acoustic recorders were deployed between December 2016 and January 2017, and recovered between May and June 2017 at seven recording sites in waters throughout the Caribbean island chain from Dominican Republic (DR), St. Martin (SM), Guadeloupe east and west (GE and GW) and Martinique (MA), to Bonaire (BO) and Aruba (AR) (**Figure 1** and **Table 1**). All sites, with the exception of AR, were located within marine park waters, with four sites on the ocean side of the islands and three on the leeward side within the Caribbean Sea. The DR site was located within the Silver and Navidad Bank Sanctuary (Mattila et al., 1989). The GW, GE, and MA sites were all located within the Agoa Sanctuary, with the GW site also located within the National Park of Guadeloupe. The SM site was located within the Natural Reserve of Saint Martin, and the BO site was located within the Stichting Nationale Parken (STINAPA) Bonaire National Marine Park. Two types of autonomous, bottom-mounted recording devices were used, Marine Autonomous Recording Units (MARUs; Cornell University, Ithaca, NY, United States) and SoundTraps (Ocean Instruments, Auckland, New Zealand[2]). MARUs were deployed at six locations (DR, GE, GW, MA, BO, and AR; **Figure 1** and **Table 1**) and programmed to collect continuous recordings at a sampling rate of 2 kHz, suitable for recording low-frequency sounds including baleen whale calls and vessel noise. SoundTraps were deployed at three locations (SM, GE, and BO; **Figure 1** and **Table 1**), and programmed to record at a sampling rate of 48 kHz, allowing a broader characterization of the soundscape and the detection of higher-frequency sounds, such as dolphin whistles and sperm whale echolocation clicks. In order to obtain recordings across the full deployment period at this higher sampling rate, the SoundTrap recordings were collected using a duty-cycled recording schedule of 1 h of recording every 4 h for two sites (SM and BO). A less frequent recording schedule of 1 h every 4 days occurred for the SoundTrap at GE, due to a programming error (**Table 1**). Sites were chosen based on a variety of factors including past observations of humpback whales, marine park waters, oceanographic conditions, and human activities. All units were anchored at depths ranging from 16 to 59 m with the hydrophone suspended or mounted 1–2 m above the sea floor (**Table 1**). The MARUs were anchored with metal weights and contain a built-in acoustic release mechanism for remote release (Calupca et al., 2000), thus were used at deeper sites such as DR and MA. SoundTraps were strapped to a cement or metal-based anchor, with deployments and retrievals assisted by divers, and thus were used at shallower sites such as GE and BO.

All recordings were visually reviewed by an acoustic analyst for the daily presence of marine mammals and vessel noise using the sound analysis software Raven Pro 1.5 and 2.0 (Bioacoustics Research Program, Cornell University, Ithaca, NY, United States). For low-frequency analyses of humpback whale song, minke whale pulse trains, vessel noise and an earthquake event, spectrograms were viewed across a frequency range of 0–1000 Hz and a time window of 3 min.

Low Frequency Analyses – Baleen Whales

In order to identify low frequency sounds across all sites, the MARU data was preferentially used, rather than the SoundTrap data, since these data consisted of continuous, rather than duty cycled, recordings. This was possible for all sites except for SM, where no MARU was deployed. SoundTrap recordings for SM were decimated to a sampling rate of 2 kHz to match the sampling rate of the MARU recordings and standardize the analyses across recorder types. For each day, daily presence of humpback whale song was marked at the first observed occurrence of song, and absence was marked if no song was found in the entire day. During the humpback whale analysis, minke whale (*Balaenoptera acutorostrata*) pulse trains (Risch et al., 2013) were observed in

[1]https://www.nature.com/articles/s41598-018-36870-7

[2]https://www.oceaninstruments.co.nz/

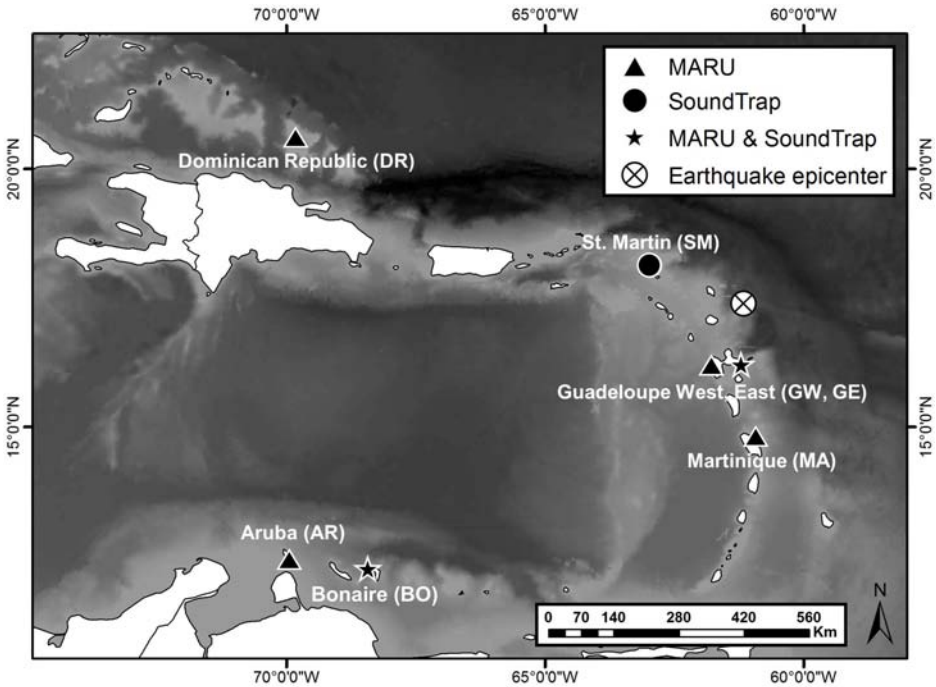

FIGURE 1 | A map of the seven passive acoustic recording sites and equipment deployed in the Caribbean, December 2016 through June 2017. The recorder type, Marine Acoustic Recording unit (MARU) or SoundTrap acoustic recorder are marked as well. X marks the epicenter for a large earthquake that occurred on 17 April 2017 at 17.450 N 61.142 W.

the MA recordings and an identical analysis for daily presence of these pulses was further conducted at that site. Each clear pulse train detected was then categorized by pulse train type. These different types of minke whale pulse trains included speed-up trains, in which the inter-pulse-interval decreases across the duration of the pulse train; slow-down trains, in which the inter-pulse interval increases across the duration of the pulse train; and constant trains with no clear change in inter-pulse-interval as described in Risch et al. (2013).

Low Frequency Analyses – Vessel Noise

A detailed analysis of the presence of vessel noise was carried out over a 2 weeks period between 12 March 2017 and 25 March 2017 across all seven sites. This period was selected to correspond with

TABLE 1 | Summary of passive acoustic recording effort across seven sites in the Caribbean Sea from December 2016 to June 2017.

Site	Instrument	Sampling Rate, Recording Schedule	Location	Depth (m)	Recording dates	Recording days (n)
Aruba (AR)	MARU	2 kHz. continuous	12° 26.318 N 69° 56.387 W	38.1	10 December 2016–15 June 2017	188
Bonaire (BO)	SoundTrap MARU	48 kHz; 1 h/4 h 2 kHz, continuous	12° 15.635 N 66° 25.265 W	35	14 December 2016–06 May 2017	144
Dominican Republic (DR)	MARU	2 kHz. continuous	20° 36.514 N 69° 49.168 W	29.9	07 December 2016–03 June 2017	179
Guadeloupe East (GE)	SoundTrap MARU	48 kHz, 1 h/4 h* 2 kHz, continuous	16° 12.749 N 61° 12.360 W	29.5	17 December 2016–02 June 2017	168
Guadeloupe West (GW)	MARU	2 kHz, continuous	16° 11.537 N 61° 47.292 W	28	16 December 2016–13 June 2017	180
Martinique (MA)	MARU	2 kHz, continuous	14° 48.627 N 60° 55.383 W	59.2	24 December 2016–06 June 2017	165
St. Martin (SM)	SoundTrap	48 kHz, 1 h/4 h	18° 07.301 N 62° 58.106 W	16	21 January 2017–06 June 2017	137

*Instrument types were either Marine Acoustic Recording Units (MARU) or SoundTrap recorders, with sampling rates that matched their capabilities. *A programming error resulted in a reduced recording schedule for the Guadeloupe East acoustic recorder.*

a period in which humpback whale song was consistently present across five recording sites. Each recording hour was marked as containing either vessel noise, humpback whale song, both or neither sound. The percentage of hours with vessel noise and humpback whale song is presented for each site. Next, selection boxes were made around all discernible vessel noise events and the duration of each vessel passages was calculated. For the sites where it was possible to identify discrete vessel passages, the number of vessel passages per hour were compared with night and day times in order to look for diel patterns in vessel activity.

High Frequency Analyses – Odontocetes

Long-term spectral averages (LTSAs) were created for all high-frequency data using the Triton software (Scripps Whale Acoustic Lab, Scripps Institution of Oceanography, La Jolla, CA, United States) developed in MATLAB (The Mathworks, Inc., Natick, MA, United States). LTSAs provide a compressed spectrogram view with a 5 s time resolution and 100 Hz frequency resolution, and facilitate efficient visual review of large datasets. The prevalence of snapping shrimp across all sites in the high-frequency recordings complicated this analysis and tended to mask other sounds in the LTSAs. Odontocete sounds such as clicks, echolocation click trains, whistles and burst pulses were clearly visible and distinguishable from the snapping shrimp in the background as their frequency range, click patterns and sound levels were distinctive. The daily presence of dolphin (*Delphinidae*) whistles and sperm whale (*Physeter macrocephalus*) clicks was analyzed at sites where they were present.

Acoustic Analysis of an Earthquake

The acoustic presence of an earthquake was discovered on the acoustic recordings and examined across all seven sites. The distance from the epicenter of the earthquake to each recording site where it was heard was measured using ArcGIS 10.3.1[3]. The duration, frequency, and timing of the earthquake and the number of aftershocks were measured using Raven Pro 1.5. Earthquake acoustic signatures were analyzed using a time window of 10 min and only events occurring between the hours of 00:45 and 03:00 UTC-4 were included.

RESULTS

Recorders were deployed between 7th December 2016 and 13th June 2017, in waters between 16 and 59 m depth, for 137 – 188 days in total. Details of deployments at each individual site are given in **Table 1**.

Low Frequency Analyses – Baleen Whales

Humpback whale song was present across five of the seven recording sites (DR, SM, GE, GW, and MA; **Figure 2**). Song was not present in the recordings from the AR or BO sites.

There was variation among the sites in the seasonal timing (beginning and cessation) of song presence as well as the percentage of days with song. Humpback whale song first began on 09 December 2016 at the DR site, our northernmost Caribbean recording site. Song was then present at the DR every day until 13 May 2017, then continued more sporadically until 25 May 2017 (**Figure 2**). Singing activity was often intense, with multiple singers recorded simultaneously. Song was present in the DR for a total of 166 (93%) of the 179 recording days.

Song was recorded next in GW, beginning on 15 January 2017 and occurring intermittently through 05 May 2017. Humpback whale song was present on 88 (49%) of the 180 recording days. Song was detected at GW a full month earlier than EG, and was present more intermittently at GW than at the other sites. At SM, song was recorded throughout the deployment period including the first and last recording day from 21 January 2017 through 06 June 2017 (**Figure 2**). Humpback whale song was present on 123 (90%) of the 137 (duty-cycled) recording days and occurred sporadically in June, which was the latest occurrence of song recorded anywhere in the study. However, it is highly probable that song may have started earlier and ended later at the SM site as the SoundTrap recorder could not be deployed sooner due to poor weather conditions and the recordings ended prior to the end of song presence. Song started latest in MA and EG, starting on the 03 and 13 of February 2017 respectively, and ending 27 May 2017 in MA and 29 May 2017 in EG. For MA, humpback whale song was present in 110 (67%) of the 165 recording days, and for EG, on 97 (58%) of the 168 recording days.

Minke whale pulse trains were detected almost every day throughout the MA recordings from 24 December 2016 to 20 February 2017, and on 3 days between 12 and 19 March 2017 (**Figure 3A**). Three different types of pulse trains produced by minke whales (*Balaenoptera acutorostrata*) were seen over the 56 days when minke whales were recorded. Of the 3,265 pulse trains, 71% were speed-up trains, reflecting similar vocal trends reported by Risch et al. (2014) in the broader Atlantic, who also reported a preponderance of speed-up trains. The pulse trains consisted of 25% slow-down trains, and 4% constant trains.

Low Frequency Analyses – Vessel Noise

A total of 336 h of data were analyzed for this 2 weeks period. The two sites that were most dominated by vessel noise during the 2 weeks detailed analysis were the AR and GW sites with vessel noise present on 100 and 99% of hours at these sites, respectively. These were followed by GE (77%), BO (54%), and MA (52%) with the lowest hourly presence of vessel noise at the SM (30%) and DR (20%) sites (**Table 2** and **Figure 4**). The AR recording site was the only site not located in a marine park, sanctuary or protected area and was in a very heavy vessel traffic zone near a port with constant vessel passage and no discernible humpback whale song (**Supplementary Figure S1**). It was not possible to calculate the number or duration of vessel passages at this site, since vessel presence was so constant that it was difficult to distinguish when one vessel passage started and another one ended.

At the GW site, both humpback song and persistent vessel noise were present each day. The majority of recording hours were characterized by overlapping humpback song and vessel noise characterized the majority of the hours recorded (96%). Only 12 h across 5 different days, included bouts of only vessel noise or only whale song. There were 9 h without whale song and

[3]http://www.arcGIS.com

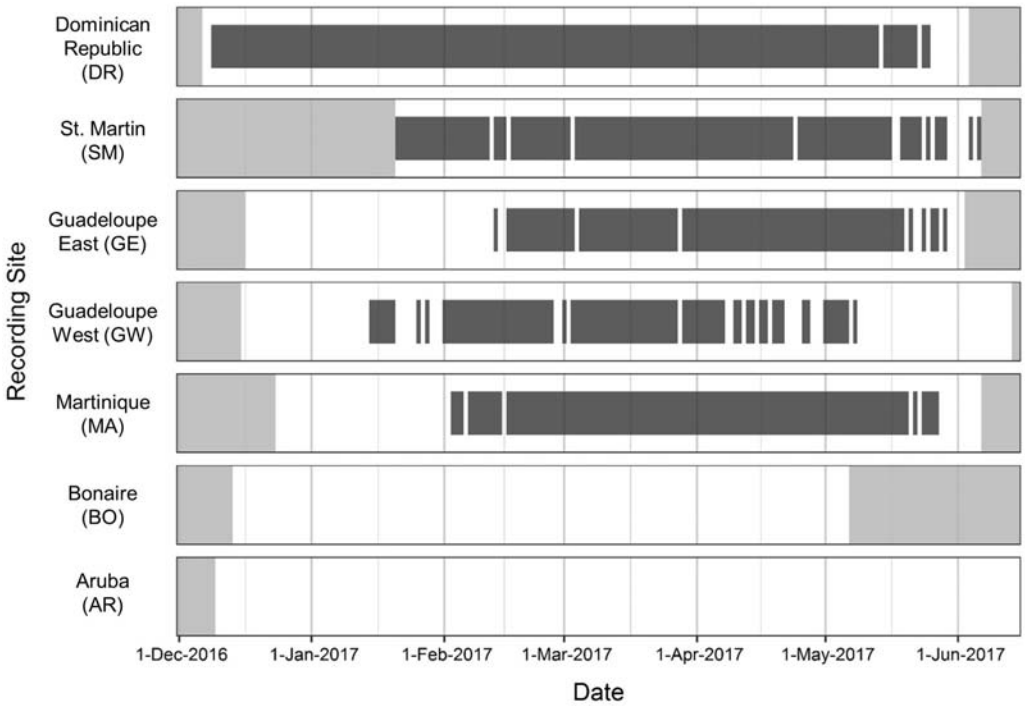

FIGURE 2 | Seasonal occurrence of humpback whale song from December 2016 to Jun 2017 across all recording sites, shown from north **(top)** to south **(bottom)**. Dark gray bars indicate recording dates with humpback song present; light gray shading indicates periods with no recording effort. Recordings were collected continuously at all sites except for St. Martin, where a 25% duty-cycle (1 h/4 h) was used.

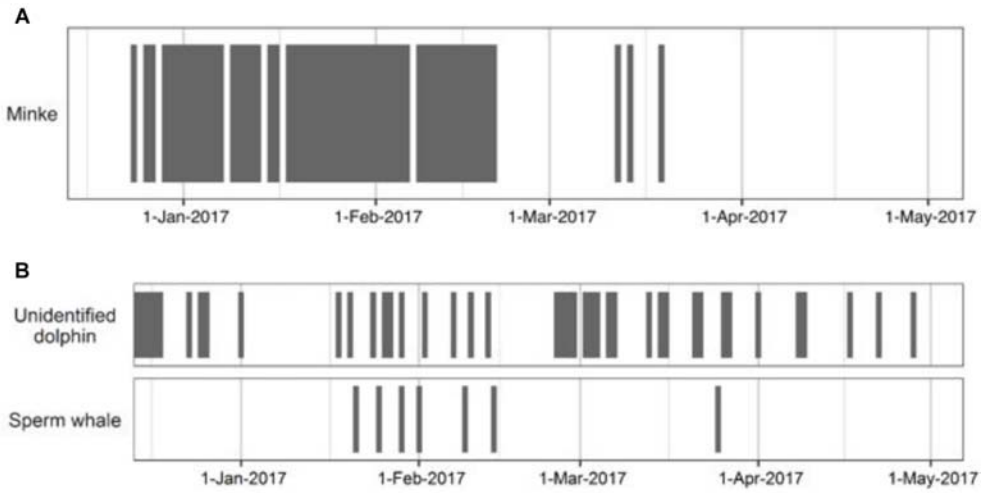

FIGURE 3 | Daily presence of **(A)** minke whale pulse trains in Martinique and **(B)** unidentified dolphin whistles and sperm whale clicks and social sounds in Bonaire.

only 3 h without vessel noise at the GW site (**Figure 4**). Similar to the AR site, it was not possible to estimate the number of vessel passages or their duration at GW due to heavy ship traffic and continuous vessel noise (**Supplementary Figure S1**).

Vessel activity and song presence was more variable at the GE site, with overlapping whale song and vessel noise being predominant (vessel noise 3% of time), humpback song (21.1% of time), both sounds (73.8% of time), and neither sound (2.1% of time) (**Figure 4**). This eastern site was further removed from the ports reflecting the slight decrease in vessel noise for this area (**Supplementary Figure S1**). Diel patterns in the number of vessel passages were examined by plotting the number of vessel passages without humpback song for the four sites, GE, BO, MA, and GE (**Figure 5**). SM was excluded from the diel pattern

TABLE 2 | A summary of the hourly vessel presence at each recording site over a 2-week period between 12 March 2017 and 25 March 2017.

	Hourly Vessel Presence (%)	Vessel passes (n)	Average vessel duration (min)	Vessel duration range (min)
Aruba (AR)	100%	NA	NA	NA
Bonaire (BO)	53.57%	441	7.01	0.09–105.10
Dominican Republic (DR)	20.23%	209	0.33	0.02–2.08
Guadeloupe East (GE)	76.78%	1084	7.56	0.02–119.46
Guadeloupe West (GW)	99.11%	NA	NA	NA
Martinique (MA)	52.38%	1241	2.22	0.02–66.78
St. Martin (SM)	29.76%	60	0.55	0.02–5.43

The number of vessel passages, average passage duration, and range of passage durations were measured for all sites where it was possible to visually distinguish individual vessel passages in spectrograms (Aruba and Guadeloupe West had near-continuous vessel presence and individual passage durations could not be measured).

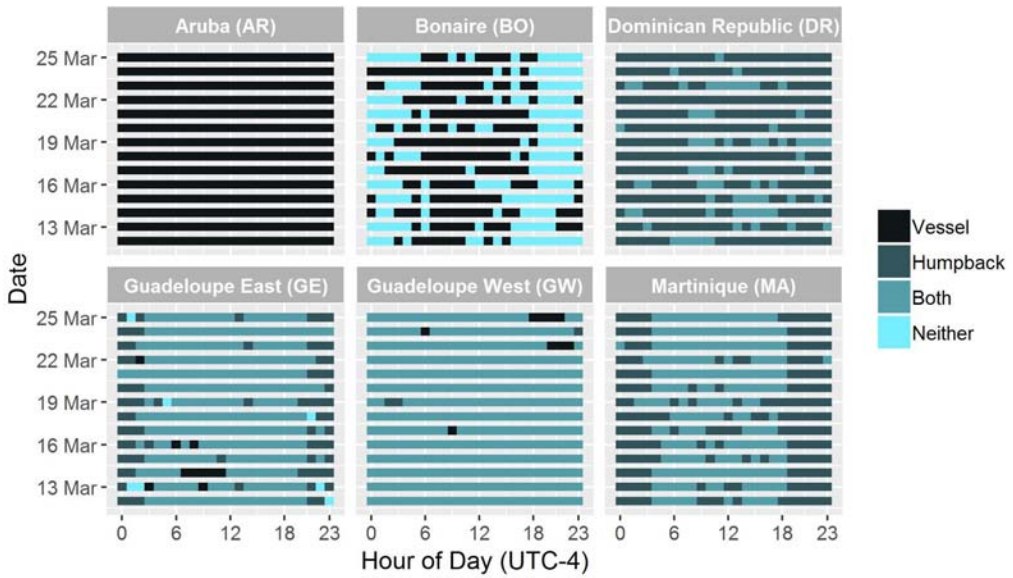

FIGURE 4 | Hourly presence of vessel noise, humpback whale song, both or neither at each recording site from 12 March 2017 to 25 March 2017.

analysis because of the duty-cycled nature of the data. At GE, a diel pattern in the number of vessel passages was observed with vessel noise receding late at night between the hours of 21:00 and 3:00 UTC-4 (**Figure 5**). The greatest density of vessels passages occurred during daylight hours.

Similar to AR, humpback whale song was not detected at the BO site. Instead, the recordings at BO consisted of hours with only vessel noise (53.6% of time) or neither humpback song nor vessel sound (46.4%) (**Table 2** and **Figure 4**). The BO site was located in the marine park and away from the main ports (**Supplementary Figure S1**). A less clear diel pattern was present in the number of vessel passages at BO compared to GE. Although the change in number of vessel passages was less distinct, fewer vessels were present during night between the hours of 18:00 and 06:00 (**Figure 5**).

MA had both hours with humpback song only (47.6%) and both humpback song and vessel noise (52.4%, **Figure 4**). The MA location was in deeper water and on the opposite side of the island to the main port (**Supplementary Figure S1**). There was a very distinct diurnal pattern with vessel passages being almost completely absent between 18:00:00 and 05:00 (**Figure 5**).

SM data were duty-cycled sampling for 1 h every 4 h, therefore only 6 h of recordings were available each day (n = 84 h) for analysis. Humpback song presence was notable at this site, with 67.8% of the sampled data containing only humpback song. Vessel noise was also present and overlapped with whale song for 29.8% of the time. The location of this recorder was also far from the islands main ports (**Supplementary Figure S1**). Due to the duty cycle of the recorder and the limited number of hours for analysis a diel comparison was not attempted.

Lastly, the DR site had no hours with solely vessel presence while hours with humpback song made up the majority of the soundscape (79.8%); humpback song and vessel noise did overlap occasionally (20.2%) (**Table 2** and **Figure 4**). Humpback whale song was present in every hour of the 2-week sample period and was often present without vessel noise in those hours. The DR site was located the furthest offshore compared to all the sites, likely reflecting the decreased vessel noise present on these recordings (**Supplementary Figure S1**). Although the total number of vessel passages was greatest during daylight hours, there were vessels present at night as well, and no clear diel pattern was apparent due to sparse vessel presence at this site (**Figure 5**).

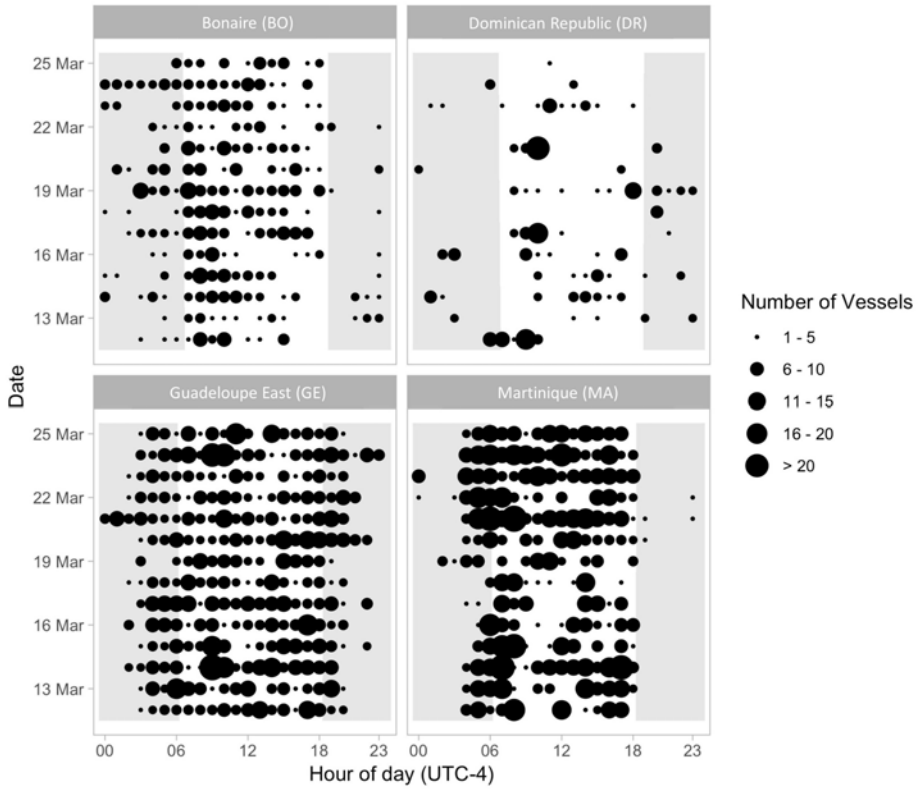

FIGURE 5 | Diel patterns in the hourly number of vessel passages at each recording site from 12 March 2017 to 25 March 2017. Dot size indicates the number of vessel passages acoustically detected for per hour on each day. Light gray shading indicates hours before sunrise and after sunset.

High Frequency Analyses – Odontocetes

At the BO site, sounds from odontocetes, namely unknown delphinid species and sperm whales (*Physeter macrocephalus*) detected sporadically throughout the dataset (**Figure 3B**). Dolphin sounds usually consisted of whistles. Sperm whale detections included echolocation clicks as well as sounds used in social communication, including slow clicks and a trumpet-like sound. Fish sounds were constantly present but were not analyzed for this study. At the SM site, possible dolphin sounds were noted only once in the SoundTrap recordings on 30 April 2017. Snapping shrimp dominated all high-frequency recordings.

Acoustic Analysis of an Earthquake

A suspected earthquake was initially discovered in the GE recordings and was subsequently detected at all other recording sites. A United States Geological Survey (USGS) website[4] and local news posts[5,6] were used to verify that a 5.6 magnitude earthquake had occurred at 05:23 UTC on 17 April 2017. The epicenter of this 5.6 magnitude earthquake was in the Flinn Engdahl region of Antigua and Barbuda, Leeward Islands (**Figure 1**). The sound from the earthquake was recorded within an 8.75 min window at all recording sites (**Table 3**). The relative sound energy and duration of acoustic signal of the earthquake was highest near to the epicenter and decreased as it was received on recorders further away (**Supplementary Figure S2**). The number of aftershocks recorded varied across sites, with the highest numbers recorded at GE and GW and fewer at MA, SM, DR, AR, and BO.

DISCUSSION

This study provides a broad scale view of the composition of marine soundscapes across seven sites located throughout the Caribbean region. All but one site was located within marine park waters and for most of these areas, this study provides the first comprehensive look at the acoustic environment, species composition and anthropogenic footprint within each park. By quantifying the soundscape at each site, this study demonstrates the value of this approach and the importance of continued long-term monitoring of each distinctive marine environment. Continued data collection would support the capacity to track changes in species presence and increases or decreases in anthropogenic noise, as well as provide an index for evaluating and understanding the health of each underwater environment

[4]https://earthquake.usgs.gov/earthquakes/eventpage/us10008iaa/executive#executive

[5]https://www.themontserratreporter.com/leeward-islands-jolted-by-strong-tremor/

[6]http://dominicanewsonline.com/news/homepage/news/general/src-records-series-of-earthquakes-off-antigua/

TABLE 3 | Summary of 17 April 2017 earthquake and aftershock analysis.

	Hydrophone distance from quake (km)	Start Time (UTC-I)	Duration (min)	Relative energy (dB)
		Main earthquake		
Aruba (AR)	1091.7	01:27:43	0.5	94.8
Bonaire (BO)	965.6	01:27:43	1.7	89
Dominican Republic (DR)	978.7	01:21:12	2.6	104.8
Guadeloupe East (GE)	130.4	01:19:57	9.1	129.5
Guadeloupe West (GW)	148.5	01:20:02	7.4	127.2
Martinique (MA)	286.6	01:19:01	3.1	102.2
St. Martin (SM)	209.2	01:23:59	4.2	107.4
		Aftershocks		
	Total number of aftershocks (*n*)	**Average aftershock duration (min)**	**Average aftershock relative energy (dB)**	**Total duration of event (min)**
Aruba (AR)*	2	0.6	82.2	1.8
Bonaire (BO)	0	NA	NA	MA
Dominican Republic (DR)	2	0.9	90.9	4.5
Guadeloupe East (GE)	25	1.1	89.2	37.6
Guadeloupe West (GW)	31	1	85.8	74.3
Martinique (MA)	8	1.4	72.5	14.1
St. Martin (SM)	2	0.4	76.9	5

*Analysis began at 00:45:00 AM UTC-4 and ended at 4:00:00 AM UTC-4. Distance between the hydrophone and quake was determined using ArcGIS. Duration, center frequency, and energy measurements were determined using Raven 2.0 robust measurements. *Two possible quake-related signatures were detected before the analysis main event in Aruba and were not incorporated in aftershock incorporated in aftershock analysis.*

over time (Dumyahn and Pijanowski, 2011; Nedelec et al., 2015; Schmeller et al., 2017).

Humpback whale sound was present across all islands other than BO and AR. There were clear differences in the timing of song presence at the five sites where humpback whale song was recorded, with singers arriving later in the year at the eastern Caribbean sites (GW, EG, and MA). Song was detected at all five sites until mid-to late-May, greatly extending the time over which humpbacks are known to occur at DR and SM. A detailed analysis of song patterns was beyond the scope of this study, and further analyses of song recorded at all sites would be informative. So, the data collected from this study demonstrates that the preponderance of whale photo-identifications made at the eastern Caribbean sites is not a sampling artifact, but is a true indication of the timing of these whales' presence in the region. This reinforces Stevick et al.'s (2018) contention that this is a second breeding population of humpback whales occurring in Caribbean waters.

Minke whales, sperm whales, dolphins, snapping shrimp and fish choruses made up the rest of the biological composition of the soundscapes at these sites. Minke whales in the Caribbean were documented previously in one PAM study off Saba (Risch et al., 2014) and visually confirmed by Debrot et al. (2011, 2013) before the 6-month acoustic monitoring effort off MA in the present study. The seasonal presence of minke whales around MA was similar to that previously reported, with pulse trains occurring from the beginning of the deployment in late December through mid- March, further supporting a likely wintering ground for minke whales in Caribbean waters.

The BO site, in particular, had a rich and diverse presence of fish sounds, and future research into chorusing patterns and spatial and temporal variation of fish sounds in these recordings would provide further insight into this important biological component of the soundscape (e.g., Staaterman et al., 2013; Nedelec et al., 2015). Sperm whales and dolphins were also present off BO, a coral reef island and is surrounded by fringing reef. The Caribbean Sea basin plunges to depths of more than 5000 m not far from BO, which is likely why sperm whales, a deep-diving species, could be heard at this recording site. Sperm whales have been included in many descriptions of marine mammals throughout the Caribbean (Debrot and Barros, 1994; Jefferson and Lynn, 1994; Debrot et al., 2011, 2013; Luksenburg, 2014), and at least six different species of dolphins have been sighted in the waters around Aruba, Bonaire, and Curacao (Luksenburg, 2014; Geelhoed et al., 2014).

AR was located on the northern side of the island and appeared to be our least biologically rich site, as the location of the site was near the main shipping channel for vessels coming to and past the island (**Figure 1**). No marine mammals were detected on these recordings and further efforts should be made to evaluate where areas of higher biodiversity, and less shipping noise, may occur around the island.

Our geological finding of the 17 April 2017 earthquake was an unexpected discovery and demonstrates the ability of soundscape monitoring to also record major geological and weather events in an area (Speeth, 1961; Locascio and Mann, 2005). The acoustic signal of the earthquake traveled to all our recordings sites over a distance of more than 1000 km. Earthquakes have

been shown to travel underwater for at least 1000–5000 km (e.g., Astafyeva and Afraimovich, 2006).

Vessel traffic in the Caribbean was a considerable contributor to the soundscape of each site, recorded on 20–100% of hours during the 2-week recording period that was analyzed in detail. Since the early 1960s, there has been a dramatic increase in worldwide ship traffic, both commercial and recreational (Eyring et al., 2005; McDonald et al., 2006). Vessel noise levels and spectral signatures vary considerably depending in vessel characteristics and operations (McKenna et al., 2012, 2013). Noise produced by large commercial vessels has repeatedly been shown to impact the ability of endangered whales, fish and other marine animals to maintain acoustic contact, especially in protected areas such as designated areas of special biological interest such as sanctuaries and marine parks (e.g., Hatch et al., 2008, 2016; Stanley et al., 2017). In addition, smaller vessels such as fishing boats and tourism such as whale watching have been shown to contribute to a significant proportion of masking and in situations where both large and smaller vessels exist the effect is cumulative (Cholewiak et al., 2018). As a result, the significantly high levels of vessel noise as well as the number of vessel passages especially for AR, GE, GW, and MA ought to raise concern for the managers in charge of the marine parks in and around these sites.

Global shipping transit maps show clear routes passing through the southern Caribbean islands and the leeward part of the Caribbean sea (e.g., Halpern et al., 2008). Cruise ship tourism in the Caribbean is a large industry that has grown substantially over the past three decades (Sprague-Silgado, 2017). One way to understand the differences in vessel noise and passages across our study sites is to look at the location of the ports on each of the islands (**Supplementary Figure S1**). Where the recorders were further from ports, such as at the DR, less vessel noise was recorded (**Figure 5**). Similarly GE and GW differed, with less vessel noise in GE. In this case GW was located on the leeward side of the island, closer to a port than was GE.

GE and MA had very distinctive diurnal patterns with few to no vessel passages during the night, while vessel passages at BO and the DR were more broadly dispersed throughout both day and night. These diurnal patterns could provide an indication of the types of vessels and activities occurring at each site. Clear diurnal patterns in vessel presence suggests traffic consisting of smaller recreational vessels or fishing vessel activity, while the lack of a diurnal pattern may reflect the presence of large container ships and cruise liners. In the DR, the distant location of the recorder makes is almost certain that a considerable amount of the vessel noise heard was produced by the illegal fishing fleet which converges in this area. Members of this fleet consist of one large mother ship from which 25–50 smaller vessels are dispersed to engage in fishing throughout Silver Bank (Salas et al., 2011). For this situation, PAM offers an opportunity to monitor the level of timing of illegal fishing within the marine park.

All recorders, with the exception of the one off Aruba, were sited in designated Marine Protected Areas. The GE, GW, MA, and SM recorders were within the Agoa Sanctuary. This MPA, created in 2012, includes all the waters of the Exclusive Economic Zone of the French West Indies[7]. The BO recorder was in the Bonaire National Marine Park, established in 1979, primarily for the protection of coral reefs. This MPA comprises the coastal waters of the islands of Bonaire and Klein Bonaire to a depth of 60 m[8]. The Silver and Navidad Bank Sanctuary, now the Sanctuary for Marine Mammals of the Dominican Republic was first declared in 1986 (Mattila et al., 1989). Of the MPAs where recorders were placed, the DR recorder was the only one in an MPA that includes measures specifically established to manage human use of the area for marine mammal protection.

Apart from masking humpback song, vessel traffic risks masking the very quiet calls between female whales and their calves on the calving ground (Videsen et al., 2017). While the Sanctuary for Marine Mammals of the Dominican Republic is removed from shipping lanes, the amount of shipping into the Agoa Sanctuary and the Bonaire National Marine Park is more substantial. Recent work has considered how MPAs can include mitigating anthropogenic sound into management planning (e.g., Erbe et al., 2012; Williams et al., 2014, 2015). Zoning planning for the Agoa Sanctuary could consider ways to mitigate the impact of anthropogenic noise on humpback whales.

As well as acoustic effects, shipping traffic also poses the risk of vessel strike on whales. Humpback females with newborn calves, given their behavior, are a particular concern (Bejder et al., 2019). Spatially explicit management to reduce the likelihood of shipping collision with large whales is well developed (e.g., Conn and Silber, 2013), and tools to notify mariners of the presence of whales are also available (Baumgartner et al., 2019). Again, we encourage when planners are zoning the Agoa Sanctuary, they consider these management options.

Overall, this study has shown that vessel traffic is as much part of the soundscapes of many Caribbean marine park sites as marine megafauna, including humpback whales. It is difficult to assess how the presence of anthropogenic noise may interfere with the essential mating activities of humpback whales, however, we have documented significant overlap in humpback whale song and vessel noise at several Caribbean sites throughout the breeding season. In addition, impacts of masking on other marine animals is likely, especially at the sites with higher noise levels. Passive acoustics provides an invaluable tool for monitoring long term composition of a marine animals and anthropogenic contributors in marine parks. Marine park managers should consider this approach when designing long term monitoring and management strategies for their sites.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

[7]http://www.sanctuaire-agoa.com/

[8]https://stinapabonaire.org/bonaire-national/

AUTHOR CONTRIBUTIONS

HH was the postdoctoral candidate who organized, planned, and ran the study. GD, JS, LK, and CR were all involved in the analyses of the data from this study. PC and SVP provided the senior scientist oversight, funding, and conceptual ideas for the study. LB, JC, JK, AH, OS-R, FV, WDW, and PH were all linked to one or more of the marine parks or deployment sites and provided logistical support, divers, helped obtain permits and facilitated the collection of the data in each of their respective regions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00347/full#supplementary-material

FIGURE S1 | Detailed maps of the recording sites located near each island, with symbols indicating the type of recorder deployed at each site [Marine Acoustic Recording Unit (MARU), SoundTrap, or both], as well as the locations of all major ports.

FIGURE S2 | The acoustic signature of an earthquake detected at 01:23 UTC-4 on 17 April, 2017 off the Antigua Islands. The MARU at Guadeloupe East was closest to the epicenter of the quake at 130 km while the St. Martin SoundTrap was approximately 209 km from the epicenter. The MARU at Aruba was the furthest from the epicenter at 1091 km.

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Fat Embolism and Sperm Whale Ship Strikes

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Strikes between vessels and cetaceans have significantly increased worldwide in the last decades. The Canary Islands archipelago is a geographical area with an important overlap of high cetacean diversity and maritime traffic, including high-speed ferries. Sperm whales (*Physeter macrocephalus*), currently listed as a vulnerable species, are severely impacted by ship strikes. Nearly 60% of sperm whales' deaths are due to ship strikes in the Canary Islands. In such cases, subcutaneous, muscular and visceral extensive hemorrhages and hematomas, indicate unequivocal *antemortem* trauma. However, when carcasses are highly autolyzed, it is challenging to distinguish whether the trauma occurred *ante-* or *post-mortem*. The presence of fat emboli within the lung microvasculature is used to determine a severe "*in vivo*" trauma in other species. We hypothesized fat emboli detection could be a feasible, reliable and accurate forensic tool to determine ante-mortem ship strikes in stranded sperm whales, even in decomposed carcasses. In this study, we evaluated the presence of fat emboli by using an osmium tetroxide (OsO_4)-based histochemical technique in lung tissue of 24 sperm whales, 16 of them with evidence of ship strike, stranded and necropsied in the Canaries between 2000 and 2017. About 70% of them presented an advanced autolysis. Histological examination revealed the presence of OsO_4-positive fat emboli in 13 out of the 16 sperm whales with signs of ship strike, and two out of eight of the "control" group, with varying degrees of abundance and distribution. A classification and regression tree was developed to assess the cut off of fat emboli area determining the high or low probability for diagnosing ship-strikes, with a sensitivity of 89% and a specificity of 100%. The results demonstrated: (1) the usefulness of fat detection as a diagnostic tool for "*in vivo*" trauma, even in decomposed tissues kept in formaldehyde for long periods of time; and (2) that, during this 18-year period, at least, 81% of the sperm whales with signs of ship strike were alive at the moment of the strike and died subsequently. This information is highly valuable in order to implement proper mitigation measures in this area.

Keywords: sperm whale, *antemortem*, fat embolism, Canary Islands, ship strike

INTRODUCTION

Strikes between vessels and cetaceans ("ship strikes") have become an issue of concern in the last decades due to an increase of the number and speed of ships (Laist et al., 2001). Reports of ship strikes have been published worldwide with fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), North Atlantic right whales (*Eubalaena glacialis*) and sperm whales (*Physeter macrocephalus*) being the most affected species (Van Waerebeek and Leaper, 2008). Areas with high cetacean diversity and high maritime traffic overlap have been identified as hot spots as ship strikes may compromise the population status of some cetacean species in those areas. In Europe, these areas include the Mediterranean Sea (Panigada et al., 2006; Frantzis et al., 2019), the Strait of Gibraltar (de Stephanis and Urquiola, 2006) and the Canary Islands (Carrillo and Ritter, 2010).

The Canary Islands form a Spanish archipelago of seven main volcanic islands, located in the north-west of Africa. It is one of the richest areas for cetacean biodiversity in the Northeast Atlantic, with 30 species identified, the sperm whale among them (Tejedor and Martín, 2013). Sperm whales are present year round in Canarian waters, with higher numbers in spring and autumn due to seasonal migrations (André, 1997). They are listed as vulnerable by the International Union for Conservation of Nature (Taylor et al., 2008), and are the most affected species by ship strikes in Canarian waters (Arbelo et al., 2013; Díaz-Delgado et al., 2018). Some factors proposed to explain the susceptibility of sperm whales to ship strikes are: (1) long periods at the surface for socialization or resting after prolonged dives (Whitehead and Weilgart, 1991; André, 1997; Watkins et al., 1999; Watwood et al., 2006); (2) drift-dives, performed at a low-activity state, which will allow them to perform bi-hemispheric sleep, being unaware of approaching ships until being touched (Miller et al., 2008); or (3) possible loss of sensitivity to low-frequency sounds produced by ship engines in Canarian waters (André, 1997).

International but mainly inter-island ferry traffic in the Canarian waters has increased considerably in the last years including: normal ferries (15–20 knots), fast ferries (21–29 knots), and high-speed ferries (≥30 knots) (Aguilar et al., 2000; de Stephanis and Urquiola, 2006; Ritter, 2010). Vanderlaan and Taggart (2007), used North Atlantic right whale ship strike data to develop a model of the probability of mortality based on strikes occurring at different speeds, regardless of vessel size. The authors suggested that strikes at speeds over 18 knots were fatal almost 100% of the time.

When whales get hit by a vessel, they can present: sharp trauma lesions, generated by the propeller or the keel of the vessel, and/or, blunt trauma lesions, caused by a non-rotating feature of the vessel, like the hull or the skeg (Campbell-Malone et al., 2008; Moore et al., 2013). Injuries within the first category, usually involve the presence of one or more linear to curvilinear laminar incising wounds, that usually cause damage to the underlying soft tissue. Extreme injuries, frequently lethal, involve damage to the axial musculature or the vertebral column affecting locomotion, or even the complete separation of part of the body, with severe central nervous system (CNS) injury. In blunt traumas, areas of hemorrhage and edema in the blubber, subcutaneous tissue, and skeletal muscle are common features, as well as luxations and/or fractures, usually concomitant. In more severe cases, rupture of internal organs can be observed (Campbell-Malone et al., 2008; Moore et al., 2013). Full necropsies should be carried out as some injuries, especially those related with blunt trauma, may not be apparent externally. In very decomposed carcasses, differentiation between ante-mortem lesions and post-mortem changes can be very challenging (Campbell-Malone et al., 2008; Moore et al., 2013).

Fat embolism is defined as the mechanical obstruction of blood vessels by circulating fat particles (Watson, 1970; Hulman, 1995). In humans it is usually related with traumas involving long and pelvic bones (Watson, 1970; Fulde and Harrison, 1991). After trauma, fat cells from the bone marrow of fractured bones or from damaged soft tissues, enter the bloodstream through torn venules in the injury or fracture site, and typically first arrive to the lungs where they may get trapped within the pulmonary microvasculature (<20 μm in diameter) (Watson, 1970). For this reason, the lung is considered a target organ for fat emboli detection (Levy, 1990). The presence of fat emboli within the lungs constitutes evidence of *antemortem* injury, as cardiac function is needed, even for a short time, to allow the circulation of fat droplets to the lungs (Armstrong et al., 1955; Mason, 1968; Saukko and Knight, 2004). It is a common and usually asymptomatic finding (Watson, 1970; Fulde and Harrison, 1991), that infrequently leads to a clinical disorder known as fat embolism syndrome (Glover and Worthley, 1999). Its severity has been related to the multiplicity of the fractures, and it occurs very rapidly after severe trauma (Tanner et al., 1990), being also present in those cases in which the death occurs immediately after the trauma (Emson, 1958).

In the Canary Islands, over 57% of the sperm whales stranded since 2000 presented evidence of ship strike, and over 70% of them were in an advanced or very advanced decomposition state (Arbelo et al., 2013; Díaz-Delgado et al., 2018). Thus, we aimed to analyze lungs from sperm whales dead in Canarian waters between 2000 and 2017 with signs of ship strike to determine: (1) if fat embolism is a common finding in sperm whale's lung tissue, (2) if the presence of fat emboli within the lung vessels is a useful diagnostic tool to assess ante-mortem ship strikes, and (3) if lung fat emboli density relates to the severity or location of the traumatic injuries.

MATERIALS AND METHODS

Animals Included in the Study

Between January 2000 and December 2017, 35 sperm whales encountered dead, floating or stranded, in the Canary Islands (28°N, 16°W; Spain) were necropsied, following standardized protocols (Kuiken and García Hartmann, 1991), to find out the cause of death. Required permission for the management of stranded cetaceans was issued by the environmental department of the Canary Islands' Government and the Spanish Ministry of Environment. No experiments were performed on live animals.

Age categories were established based on total body length (Perrin et al., 2009) and histologic gonadal examination, including: neonate, calf, juvenile, subadult and adult (Geraci and Lounsbury, 2005). Decomposition code was established according to Kuiken and García Hartmann (1991) classification, with a modification for code 1: code 1 for "very fresh" was assigned to an animal that has recently died. The other codes remained the same: code 2 for "fresh dead animals" (no bloating nor changes in coloration, eatable meat), code 3 for "moderate autolysis" (may present with some skin desquamation, the carcass might have started to swollen, and organs may have changed coloration and more friable), code 4 for "advanced autolysis" (skin desquamation, swollen carcass, organs difficult to recognize), and code 5 for "very advanced autolysis" (the skin may be absent, some or all organs may be liquefied, also mummification or adipocera may be observed in some carcasses). Body condition was determined based on anatomical parameters such as the presence of certain prominent bones, the dorso-axial muscular mass, and the presence or absence of fat deposits (Joblon et al., 2014).

The lungs of 16 sperm whales with evidence of ship strike and eight sperm whales without (control group), were studied to detect fat emboli (**Table 1**) using the osmium tetroxide (OsO_4) technique.

Osmium Tetroxide Technique

A retrospective study was carried out using lung tissue samples fixed in 10% buffered formalin between 2000 and 2017 and kept in the Institute of Animal Health Tissue Bank. Formalin-fixed lung samples were cut into thin sections ($\leq$1 mm) to ensure the proper penetration of OsO_4. Post-fixation with OsO_4 is needed as lipids are soluble in the processing solvents used to embed the tissues in paraffin. The sections were then washed with running tap water for 20 min followed by 10 min in distilled water. Next, the sections were immersed in 1% OsO_4 aqueous solution (sonication was previously used to dissolve the commercial crystalline OsO_4) within hermetically sealed bottles on a shaker inside a chemical hood. Then, the sections were rinsed in running tap water for 30 min and immersed in 1% periodic acid until the dark osmicated tissues were uniformly cleared (Abramowsky et al., 1981). Samples were washed for 30 min with tap water, and rinsed three times with distilled water. Then, the samples were routinely processed and embedded in paraffin-wax, sectioned at 5 μm-thick, treated with picric acid (1% in ethanol 96%) for 24 h to remove excess formalin pigment (Abramowsky et al., 1981), and counterstained with hematoxylin and eosin (HE). Finally, slides were mounted in DPX mounting medium. Tissue sections as blubber and *rete mirabile* (which have abundant adipocytes), were used as positive controls (treated with OsO_4) and negative controls (non-treated with OsO_4) to validate the technique.

Microscopic Analysis

All lung sections were evaluated for the presence/absence of fat emboli, as well as the area occupied by those emboli within lung vessels using light microscopy (Olympus BX51).

Each lung tissue section was divided in "N" number of 100× magnified microscopic fields (MF) (Ocular: 10× and Objective: 10×). A Bootstrap analysis was carried out to determine the number of 100 MFs ("n") that needed to be studied for each tissue section (**Table 2**). We considered a good estimation of the true value if the total bound error was below 8%. Depending on the total tissue section area the Bootstrap analysis yielded results between 14 and 20 100 MFs. These fields were randomly selected and photographed using an Olympus XC30 camera (Olympus Soft Imaging Solutions GmbH©, Johann-Krane-Weg 39, D-48149 Münster) (**Table 2**). Fields containing pleura (adipocytes are normally present in the pleura of sperm whales) (**Figure 1B**), and/or large-diameter bronchi/bronchioles (empty spaces) were discarded to ensure a similar parenchyma size comparison between the different fields.

The software ImageJ (1.48v, Wayne Rasband, National Institute of Health, United States) was used to determine the area occupied by fat emboli, in pixels, in each of the photographs. Each of the 100 MFs' photographs has a total area of 1,920,000 pixels. Fat emboli are recognized as black droplets primarily in the lumen of capillaries and small- and medium-size arteries. The software ImageJ allows the quantification of areas of a certain color automatically, or the quantification of selected areas manually. We manually selected the fat emboli areas in the lung parenchyma, as not all black areas were fat emboli [e.g., various artifacts, fat in bronchioli and/or alveoli (**Figure 1C**)], and the emboli were not homogenously stained. As a result, for each animal, we ended up having an "n" number of 100 MFs (photographs), each of them with an area in pixels occupied by fat emboli.

Analysis for the Validation of the Osmium Tetroxide Technique as a Complementary Diagnostic Tool for Ship-Strikes

The 25th, 50th, 75th, and 90th percentiles of the areas (pixels) occupied by fat emboli in the "n" 100 MFs studied were calculated for each animal (**Table 3**). As fat emboli were also present in a few lungs of the "control group," a classification and regression tree (CART) was developed to obtain a cut-off value from which the probability of association with ship-strike is high. This procedure classifies data using a sequence of if–then rules. The basis of the decision tree algorithms is the binary recursive partitioning of the data. The most discriminative variable is first selected to partition the data set into child nodes. The splitting continues until some stopping criterion is reached. The tree was constructed according to the following algorithm: in the first stage, the tree grows until all cases are correctly classified, and in the second stage, we used the tenfold cross-validation method of successive pruning (Breiman et al., 1984). Finally, the tree that minimized the error measurement (deviance) was chosen. Then, the low and high-probability categories obtained were compared using the exact Fisher test. The sensitivity

TABLE 1 | Epidemiological and biological data of the sperm whales included in the present study with evidence of ship strike.

Case	Age	Sex	Strand. locat.	Strand. date (dd/mm/yy)	Length (cm)	Decomp. code	Body condition	Food content	Type trauma	Trauma location	Fractured bones
Ship strike group											
(1)	J	F	NE-T	21/08/01	600	4	4	Fresh	Sharp	Abdomen	0
(2)	C	M	SE-T	05/07/03	530	3	4	Digested	Both	Head	0
(3)	S	F	S-F	11/05/05	750	4	4	Digested	Sharp	Abdomen	0
(4)	C	F	SE-T	27/04/06	460	2	3	Milk	Sharp	Thorax and Abdomen	Ribs
(5)	C	F	SE-GC	10/04/08	500	4	NE	NE	Sharp	Thorax	Ribs
(6)	A	F	SE-T	13/03/09	915	2	3	Digested	Sharp	Abdomen	NE
(7)	C	M	E-T	13/03/09	570	3	4	Digested	Sharp	Abdomen	Vertebrae
(8)	J	M	NE-T	09/07/10	785	4	4	Semi-digested	Sharp	Head	Skull
(9)	C	M	SE-GC	29/09/11	550	4	NE	Digested	Sharp	Thorax	Ribs
(10)	C	M	N-GC	26/04/11	575	4	2	Digested	Both	Thorax and Caudal third	0
(11)	J	M	NE-H	09/02/12	NE	5	NE	NE	Both	Head and Caudal Third	Mandible and Caudal Section
(12)	C	M	NE-F	24/04/12	552	4	4	Digested	Sharp	Neck	0
(13)	C	F	NW-L	21/06/12	550	5	NE	Semi-digested	Both	Head and Caudal Third	Mandible and Caudal Section
(14)	C	F	S-T	19/10/12	365	4	NE	NE	Sharp	Abdomen	Vertebrae
(15)	C	M	NE-GC	09/05/13	435	4	3	None	Sharp	Head	Skull
(16)	A	F	NE-T	05/06/14	825	4	3	Fresh	Sharp	Abdomen	Vertebrae
Non-ship strike group											
(17)	N	F	W-LG	22/07/08	340	1	NE	None	None	None	0
(18)	A	F	SW-T	27/06/09	940	3	2	None	None	None	0
(19)	A	F	N-T	08/09/11	1050	4	2	NE	None	None	0
(20)	A	F	NE-T	15/03/13	950	3	3	Digested	None	None	0
(21)	J	F	N-GC	29/05/13	790	4	3	Digested	Blunt	Head and Thorax	0
(22)	A	F	N-L	17/03/17	900	4	NE	Digested	None	None	0
(23)	S	F	S-H	08/10/17	905	4	3	Digested	None	None	0
(24)	J	F	E-GC	25/10/17	840	4	NE	Digested	Blunt	Caudal Third	0

Age: N, Neonate; C, Calf; J, Juvenile; S, Subadult; A, Adult. Stranding location: N, North; S, South; E, East; W, West; NE, Northeast; NW, Northwest; SE, Southeast; SW, Southwest; T, Tenerife; GC, Gran Canaria; F, Fuerteventura; L, Lanzarote; H, El Hierro; LG, La Gomera. Decomposition code: 1, Very fresh; 2, Fresh; 3, Moderate autolysis; 4, Advanced autolysis; 5, Very advanced autolysis. Body condition: 1, Very poor; 2, Poor; 3, Fair; 4, Good; NE, Not Evaluated.

TABLE 2 | Number of microphotographs captured based on the total number of 100 magnifications microscopic fields (100 MFs) of the tissue section.

N	**≤20**	**20–30**	**31–40**	**41–50**	**51–60**	**61–70**	**≥70**
n	14	15	16	17	18	19	20

"N" is the total number of 100 MFs analyzed in the lung tissue section; and "n" is the number of microphotographs that should be captured to achieve <8% error bound.

and specificity were estimated by means of 95% confidence intervals (95% CI).

Exploration of Association Between Trauma-Related Variables and Fat Emboli Severity

Categorical variables were expressed as frequencies and percentages, and continuous variables, like fat emboli areas, were expressed as medians and interquartile ranges (IQR = 25th–75th percentile) (**Table 3**).

The variables age, presence/absence stomach food content and degree of digestion of the ingesta, presence of fractures and stranding location were compared between both groups using the Chi-square ($\chi 2$) test or the exact Fisher test for percentages; and the Wilcoxon test for independent data for the medians (**Table 3**).

The variables age, sex, body condition, presence/absence and degree of digestion of the stomach food content, trauma location

FIGURE 1 | Lung sections treated with OsO_4. **(A)** Appearance of a lung parenchyma negative to fat emboli (Bar = 500 μm). **(B)** Black-stained adipocytes in the pleura (arrows) (Bar = 200 μm). **(C)** Black-stained fat droplets within a bronchiole (arrows) (Bar = 200 μm).

and presence/absence of fractures were analyzed within the group with evidence of ship strike to assess potential associations with fat emboli severity (lung area occupied by fat emboli) (**Table 1**). For this aim, a linear analysis was carried out. The variables introduced in the model were age (calf/not calf), sex, body condition, presence/absence and degree of digestion of the stomach food content, trauma location and presence/absence of fractures. Then, a selection of variables based on the Akaike information criteria was performed.

Data were analyzed using the R package, version 3.3.1 (R Development Core Team, 2016).

RESULTS

Presence of Fat Embolism

A total of 83% (13/16) sperm whales with evidence of ship strike (**Figure 2**) had intravascular OsO_4-positive fat emboli. Fat emboli ranged from 67 to 59773 pixels, and were seen circulating in medium and small caliber intrapulmonary arteries and/or obliterating arterioles and capillaries, both in fresh and decomposed specimens (**Figure 3**). None but two of eight "control" sperm whales had detectable fat emboli (**Figure 1A**). Those two animals (cases 21 and 24) had rare isolated OsO_4-positive fat emboli (<650 pixels) in arterioles (**Table 3**).

Calves were "significantly" (p = 0.003) more likely to be involved in ship-strikes than other age categories. As well, the presence of fractures was "significantly" associated to ship-strikes p = 0.003. Other variables studied like the stranding location (island) or the presence/absence and degree of digestion of the ingesta, were not significantly different between both groups (**Table 3**). An association between trauma-related variables and fat emboli severity could not be established.

When assessing the probability of ship strike based on the fat emboli area, significant differences between non-strike and strike groups started to be seen in the 50th-percentile values, but the

TABLE 3 | Categorical variables studied expressed as frequencies and percentages or continuous, expressed as medians and interquartile ranges (IQR = 25th–75th percentile).

	Ship strike		
	No *N* = 8	**Yes *N* = 16**	***P***
Length (cm)	902 (828–942)	552 (515–675)	0.010
Calf	0	10 (66.7)	0.003
Fracture	0	10 (66.7)	0.003
Location			0.661
Tenerife	3 (37.5)	8 (50.0)	
Gran Canaria	2 (25.0)	4 (25.0)	
El Hierro	1 (12.5)	1 (6.2)	
Fuerteventura	0	2 (12.5)	
Lanzarote	1 (12.5)	1 (6.2)	
La Gomera	1 (12.5)	0	
Food			0.470
No content	2 (25.0)	1 (6.2)	
Milk	0	1 (6.2)	
Fresh	0	2 (12.5)	
Semi-digested	0	3 (18.8)	
Digested	5 (62.5)	6 (37.5)	
Not determined	1 (12.5)	3 (18.8)	
Fat emboli area			
Percentile 25	0 (0–0)	0 (0–1064)	0.085
Percentile 50	0 (0–0)	0 (0–1748)	0.033
Percentile 75	0 (0–0)	830 (0–3249)	0.003
Percentile 90	0 (0–8.38)	5274 (132–10044)	0.006
Probability of ship strike			<0.001
Low: No calf and Fat 0.75 ≤ 140	8	1	
High: Calf or Fat 0.75 > 140	0	15	

In the last column, P-value <0.05 was considered as statistically significant difference between ship-strike and non-ship strike groups.

FIGURE 2 | Vessel strike-related injuries in sperm whales (*Physeter microcephalus*) stranded in the Canary Islands. **(A)** Presence of a deep incision in the right flank of the animal, caudal to the pectoral fin, with soft tissue exposure, abdominal perforation and evisceration and costal fractures, case 4. **(B)** Complete amputation of the vertebral column at the level of the last thoracic vertebrae, case 14.

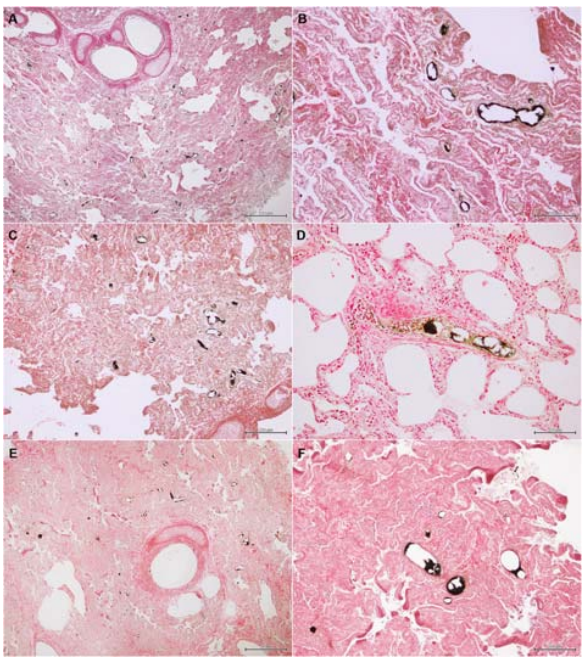

FIGURE 3 | Black stained (OsO_4-positive) fat emboli in the lung microvasculature, mainly arterioles, of some of the sperm whales stranded in the Canary Islands with signs of ship strike **(A–F)**. Bar: A = 500 μm; B = 100 μm; C = 200 μm; D = 100 μm; E = 500 μm; F = 200 μm.

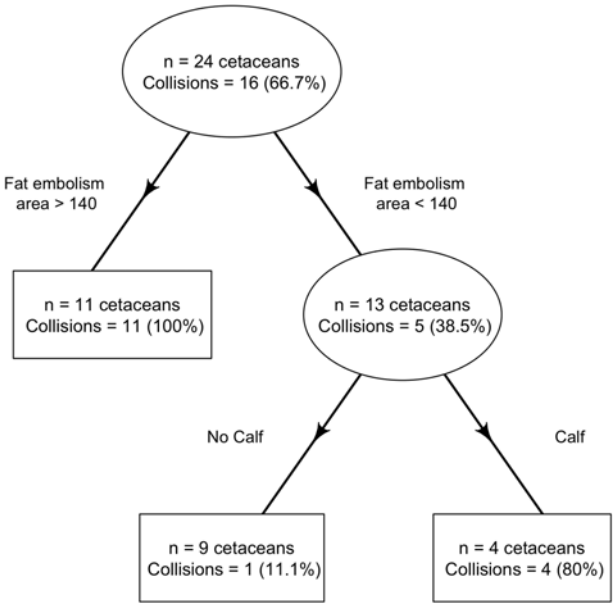

FIGURE 4 | Result of the classification and regression tree to determine the cut off for ship-strike probability of the studied sperm whales. A high or low probability of ship strike was established according to: first, the fat emboli-P75 value and secondly, being a calf or not a calf. If fat emboli area <140 and Not a Calf → Non-strike group; If fat embolism area <140 and Calf → Strike group; and if fat embolism area >140 → Strike group. Its sensitivity is 89% (95% CI = 52–100%) and its specificity 100% (95% CI = 78–100%).

highest discriminant power between both groups corresponded to the fat emboli area's value of the 75th-percentile (**Table 3**). The CART indicates that if the value of the 75th-percentile fat emboli area is greater than 140 pixels in the animal studied (cut-off value), the probability of having been hit by a vessel is high, and so the animal is assigned to the strike group. If the value of the area occupied by fat emboli in the 75th-percentile is lower than 140 pixels, and the animal is a calf, it is also assigned to the strike group. If none of the previous conditions are met, the animals are assigned to the non-strike group ($p < 0.001$) (**Table 3** and **Figure 4**). The sensitivity and the specifity were 89% (52–100; 95% CI) and 100% (78–100; 95% CI), respectively.

Factors Related to Ship Strikes in the Canary Islands

Most sperm whales with evidence of ship-strike were calves (10/16; 62.5%), followed by juveniles (3/16; 18.75%), adults (2/16; 12.5%) and subadults (1/16; 6.25%). All the adults/subadults included in this study were females.

Most of the animals with evidence of ship strike appeared floating or stranded along the east coast of Tenerife (56.25%), followed by the east coast of Gran Canaria (18.75%), east coast of Fuerteventura (12.5%), and finally east coast of El Hierro (6.25%) and west coast of Lanzarote (6.25%) (**Figure 5**).

DISCUSSION

Presence of Fat Embolism and Its Significance in Ship Strikes

The lack of detectable fat emboli in lung tissue of most control sperm whales suggests that fat embolism is not a physiological or common finding in lungs of stranded sperm whales. In addition, the presence of abundant OsO_4-positive fat emboli in most sperm whales with evidence of ship-strike indicates an association with trauma. The etiology of fat emboli in the blood stream may be trauma- or non-trauma related (Glover and Worthley, 1999). Trauma conditions may include marrow-containing bone fractures or adipocyte-rich soft tissue injuries (Watson, 1970; Fulde and Harrison, 1991;

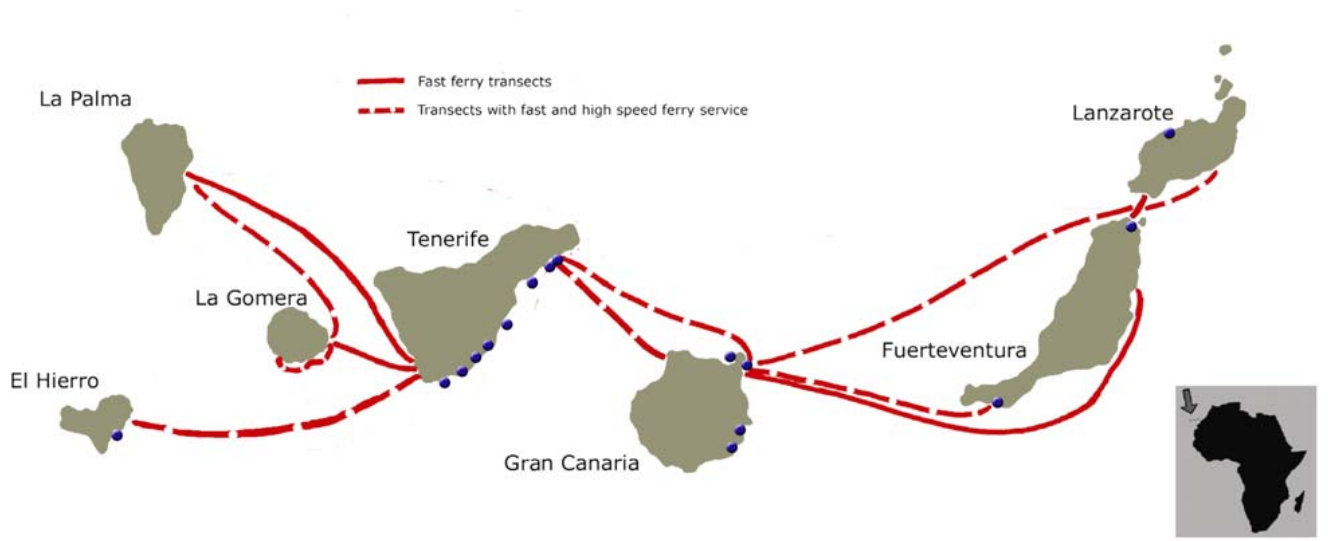

FIGURE 5 | Inter-island ferry transects in the Canary Islands (different speed-ferries are represented with different line types), and stranding locations (blue dots) of the sperm whales with evidence of ship strikes in the Canary Islands [modified from Ritter (2010)].

Gupta and Reilly, 2007). Both possibilities likely coexisted in our cases with evidence of ship-strike. The presence of fat emboli in the lung microvasculature indicates that the animal was alive at the moment of the strike and that cardiovascular collapse did not ensue immediately with successful pulmonary irrigation for an unknown period of time.

In addition to fat emboli, other typical findings of *antemortem* ship strike in cetaceans include subcutaneous, muscular and/or internal hemorrhage with hematoma formation, organ contusion and/or rupture with bleeding, e.g., in airways, in gastrointestinal tract, and edema in various organs mainly due to increased hydrostatic pressures, increased permeability due to hypoxia and vascular disruption (Campbell-Malone et al., 2008; Moore et al., 2013). Traumatic injuries on the dorsum are usually considered as *antemortem* or *perimortem* as carcasses tend to float with the ventral or lateral side upward, making a dorsal strike of a carcass unlikely (Laist et al., 2001; Campbell-Malone et al., 2008). On histopathologic examination, the presence of inflammatory response, hemorrhage or edema (Campbell-Malone et al., 2008; Moore et al., 2013), as well as acute, monophasic myocyte (segmentary, discoid) degeneration, contraction band necrosis, and/or fragmentation of the myofibers in the skeletal muscle (Sierra et al., 2014) support *antemortem* trauma.

Nevertheless, when working with decomposed carcasses it is not always feasible to assess many of the ship strike evidences described above. It is in these cases, where the detection of fat emboli in the lungs has proven to be a valuable and reliable confirmatory diagnostic tool, allowing us to conclude that, at least, 83% of the studied stranded sperm whales in the Canary Islands with evidence of ship strike were alive at the moment of the strike.

All the same, the methodology for fat emboli detection is not devoid of certain limitations that may lead to an under-estimation of fatal vessel strikes, such as the slanted and arbitrary sampling of lung tissue. Kinra and Kudesia (2004), suggested that fat emboli are not homogenously distributed along lung tissue. Although different lung areas, including cranial, medial, and caudal samples from both lungs should be routinely collected during the necropsy, there is not a specific sampling protocol to accurately assess lung fat emboli. Thus, results from small lung portions taken arbitrarily should be carefully interpreted as they may not be representative of the whole tissue. Future anatomical and topographical studies of pulmonary blood circulation and fat emboli distribution, respectively, are necessary to assess which lung areas should be sampled for an accurate fat emboli detection.

Microscopically, the severity of fat embolism has been traditionally assessed using a simple scale based on the number of emboli encountered in the tissue section studied (Saukko and Knight, 2004). Here we proposed and evaluated fat emboli area as a better estimator to assess fat emboli severity since the same number of emboli in two different lung histological sections may occupy different areas, and the one with the largest area occupied, would be a more severe case.

The rule developed for our samples based on the area occupied by fat emboli allowed us to discriminate between sperm whales that suffered strikes from those that died due to other causes, even when fat emboli was present in some animals of the control group (Cases 21 and 24). In these two cases, there was a blunt trauma of unknown origin. Possible etiologies included intra- or interspecific interactions or a potential live-stranding event (Díaz-Delgado et al., 2018). Intra-/interspecific traumatic interactions are frequent among cetaceans, and may result in blunt traumas where internal hemorrhages and/or bone fractures may occur (Arbelo et al., 2013; Díaz-Delgado

et al., 2018). In the case of sperm whales, they have been observed being attacked by killer whales (*Orcinus orca*) or male sperm whales fighting with each other (Whitehead, 2009). These interactions have been observed to occasionally cause fat emboli (Díaz-Delgado et al., 2018).

When relating the severity of fat embolism and the severity of the trauma a general positive correlation was established by Emson (1958) based on the type and number of bones fractured of 100 patients who died after injury. On the contrary, we did not find any association between variables related to trauma (i.e., presence of fractures or location of the trauma) with fat embolism severity. This could be due to the low sample size, heterogeneous distribution of fat emboli, methodological bias, immediate cardiovascular collapse with none or little pulmonary irrigation post-trauma (sudden death), or to the fact that the strike occurred *post-mortem*. Fatal lesions, often involving direct cardiovascular trauma with or without rupture of large vessels (hypovolemic shock) and/or severe neurogenic dysfunction, may cause an abrupt death, leading to immediate cease of the blood, and could explain the absence of fat emboli in different body organs, including the lung. This has been documented in aircraft fatalities, were extensive injuries were associated with lower grades of fat embolism (Mason, 1962), or no emboli at all in disintegration cases (Kinra and Kudesia, 2004), showing the importance of intact circulation for the formation of fat emboli. We surmise rapid cardiovascular collapse could explain lack of detectable fat emboli in three of our cases (cases 7, 10, and 13). These animals presented severe injuries, including abdominal evisceration or caudal amputation (Díaz-Delgado et al., 2018).

Factors Related to Ship Strikes in the Canary Islands

All the sperm whales included in this study, were either female adults/subadults, or juveniles or calves, of any sex. This is in agreement with the fact that sperm whale females and their progeny composed the main groups present all year round in Canarian waters, which are considered nursery and breeding areas (André, 1997).

The fact that young animals are not fully adapted to dive and need to spend more time at surface, together with their relatively slow swimming speed compared to adults, may explain their higher vulnerability to ship strikes (Papastavrou et al., 1989; Laist et al., 2001; Whitehead, 2009). Mothers with recent calves (cases 6 and 7) may be also at higher risk as they will spend more time in the surface with their offspring.

A previous study estimated the absolute abundance and density of sperm whales in Canarian waters, and concluded that the species would not be able to sustain the current level of strikes (Fais et al., 2016). This impact is aggravated by the female philopatry in the Canaries, as they are not genetically connected to west North Atlantic populations (Alexander et al., 2016), and by the number of calves and reproductive females affected by strikes.

Most struck sperm whales appeared in the east coast of Tenerife, in agreement with previous studies (André, 1997; Carrillo and Ritter, 2010; Ritter, 2010). A major explanation for this is that the channel between Tenerife and Gran Canaria is a prime habitat for sperm whales in the Canaries (André, 1997; Fais et al., 2016), as well as an area with a high maritime traffic density, dominated by fast and high speed ferries (Ritter, 2010). An overlap between most of the sperm whales' stranding locations and fast-ferry transects was also observed in the present study.

To conclude, this study provided compelling histochemical evidence of fat emboli as a reliable confirmatory diagnostic tool of ante-mortem ship-strike even in decomposed sperm whale carcasses. Our results demonstrated that most of the sperm whales with evidence of ship-strike and stranded in the Canaries were alive at the moment of the strike. However, this may be an underestimation as cases where immediate cardiovascular collapse (sudden death) occur, may lack detectable fat emboli in the lungs. A final diagnosis of antemortem ship-strike may considerably benefit from fat emboli detection in lung tissue, particularly when other trauma-related gross and microscopic findings are not evident.

Some future directions may include the study of fat emboli distribution within the lungs to determine if some areas are more affected by fat emboli, and based on the results, the development of a homogenized lung sampling protocol to detect fat emboli. Alternative techniques to osmium tetroxide, which is extremely toxic, should be developed to study lipids histologically. Lipid composition analyses of fat emboli may contribute to a better understanding of its pathogenesis in these animals.

Some mitigation measures have been implemented in other locations and have proven to be effective, such as a mandatory vessel-speed restriction in the United States East Coast (Conn and Silber, 2013), the establishment of Traffic Separation Schemes (TSS) in the Bay of Fundy (Vanderlaan et al., 2008) or the proposal of recommended Areas To Be Avoided (ATBA) like the Roseway Basin Area (Vanderlaan and Taggart, 2009). Similarly, mitigation measures to reduce ship-strike mortalities and guarantee the survival of the sperm whales' population in Canarian waters should be further explored and implemented.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and/or the supplementary files.

ETHICS STATEMENT

All animals included in the present study were dead, either floating offshore or stranded in the coast. Required permission for the management of stranded cetaceans was issued by the environmental department of the Canary Islands' Government and the Spanish Ministry of Environment. No experiments were performed on live animals.

AUTHOR CONTRIBUTIONS

AF: conceptualization. MArb, ES, YBdQ, JD-D, CS-S, RP-L, and MArr: sampling. MArr: laboratory analyses. PS, MArr, and YBdQ: data analyses. AF: funding. MArr: writing. All authors: review and editing. YBdQ and AF: supervision.

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frontiers in Marine Science

BRIEF RESEARCH REPORT
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Monitoring of Marine Mammal Strandings Along French Coasts Reveals the Importance of Ship Strikes on Large Cetaceans: A Challenge for the European Marine Strategy Framework Directive

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The incidence of marine traffic has risen in recent decades and is expected to continue rising as maritime traffic, vessel speed, and engine power all continue to increase. Although long considered anecdotal, ship strikes are now recognized as a major threat to cetaceans. However, estimation of ship strike rates is still challenging notably because such events occurred generally far offshore and collision between large ships and whales go often unnoticed by ship crew. The monitoring of marine mammal strandings remain one the most efficient ways to evaluate the problem. In France, a national coordinated network collected data and samples on stranded marine mammals since 1972 along the Mediterranean and Atlantic French coasts. We examined stranding data, including photography and necropsy reports, collected between 1972 and 2017 with the aim to provide a comprehensive review of confirmed collision records of large whales in France. During this period, a total of 51 ship strike incidents were identified which represents the 1st identified causes of mortality for large whale in France. It has increased since 1972 with seven records during the 1st decade to reach 22 stranded animals observed between 2005 and 2017. This issue appears particularly critical in the Mediterranean Sea where one in five stranded whales showed evidence of ship strike. This review of collision records highlights the risk of a negative impact of this anthropogenic pressure on the dynamic of whale populations in Europe, suggesting that ship strike rates could not allow achieving the Good Environmental Status of marine mammal populations required by the European Marine Strategy Framework Directive.

Keywords: ship strikes, fin whales, sperm whales, strandings, monitoring, MSFD

INTRODUCTION

Marine traffic exerts a growing pressure on marine megafauna. Ships and other sea-faring vessels cause chemical pollution, modification of habitats and animal behavior (including through noise disturbance) as well as direct injuries through collisions with animals (Pirotta et al., 2018). Although long considered anecdotal, ship strikes are now recognized as a major threat to cetaceans (Kraus et al., 2005; Douglas et al., 2008). Any vessel type may be involved in ship strikes, including tankers, cargo or cruise ships, ferry boats, whale watching vessels, and sailing vessels (Laist et al., 2001; Ritter, 2012). Ship strikes occur worldwide and have been reported in at least 11 large whale species (Laist et al., 2001). Several hotspots have been identified across the world where ship strikes seriously threaten the conservation status of whale populations, e.g., northern right whales in the Western North Atlantic, blue whales around Sri Lanka and fin whales in the Mediterranean Sea (Cates et al., 2017). The incidence of ship strikes has risen in recent decades and is expected to continue rising as maritime traffic, vessel speed, and engine power all continue to increase (Laist et al., 2001; Douglas et al., 2008).

A thorough understanding of the incidence and future threat of ship strikes is of major importance for large cetacean conservation but is challenging to achieve. Relatively little is known about the geographic distribution of ship strikes and the magnitude of their impact. The scarcity of direct reports and relevant data makes it challenging to provide quantitative indicators of absolute mortality at sea. The best source of information available on ship strike fatalities is the examination of stranded cetaceans (Laist et al., 2001).

Understanding the pressures faced by marine wildlife and implementing plans to mitigate them are crucial to achieving and maintaining a Good Environmental Status (GES) of European waters - the aim of the Marine Strategy Framework Directive (MSFD, 2008/56/EC). Good Environmental Status is defined as "the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive". In order to understand the future threat of anthropogenic pressures, like the "extraction of, or mortality/injury to, wild species, (by commercial and recreational fishing and other activities)" (2008/56/EC, Annex III), we need to study their impact on populations in the past and present, and project the observed trends into the future. For cetaceans, this requires the efficient monitoring of populations and the development of quantitative indicators that reveal the degree to which human activities impact these populations (Santos and Pierce, 2015; Authier et al., 2017).

We review five decades of whale stranding data collected along the French coasts in order to document the importance of ship strikes on populations of large whales and provide baseline data for future assessments. This is a step toward the development of a ship strike mortality indicator, which would serve as a means to better understand the importance of ship strikes in European waters in the future and identify ways to mitigate them in context of GES achievement through the MSFD.

MATERIALS AND METHODS

Stranding data was collected by the French National Stranding Network following standardized observation and sampling protocols set in place in the 1980s. This network is made up of around 400 trained volunteers distributed along the coasts of mainland France. Examination protocols include taking external measurements, photographs, and observations of all stranded cetaceans. According to the accessibility and the decomposition status of carcasses, tissues are regularly but not systematically sampled and examined (blubber, skin, internal organs, muscles, and skeleton).

Ship strike was determined as the cause of death if animals were recovered on ship bows or behind propellers, or with strong evidence of ship strikes. Evidence of ship strike includes: deep propeller cuts, significant bruising, oedema, internal bleeding radiating from a specific impact site, fractures and ship paint marks (Jensen and Silber, 2004; Douglas et al., 2008).

Observation effort has been stable since the late 1980s (Authier et al., 2014), so trends can therefore be interpreted with greater confidence for the last three decades. The earlier stranding records (1972–1982) must be carefully interpreted.

Because of low number of records, data were collated in histograms by 10-year intervals to improve the understanding of trends. Spatially, results are described following different marine sub-areas used in MSFD: the Western Mediterranean Sea (WMS), the Bay of Biscay (BB), the Celtic Sea (CS) and the Channel and North Sea area (CNS).

RESULTS

Species Composition of Strandings

A total of 396 large whale strandings were recorded in France between 1972 and 2017, of which 51 (12.9%) were diagnosed as being caused by ship strikes (details provided in **Supplementary Table S1**). Balaenopterids represented 79.5% of the total strandings, 315 in total, and sperm whales represented the remaining 81 strandings (20.5%). Ship strike incidents included 39 fin whales (76.5%), 4 minke whales (7.8%), 2 humpback whales (3.9%), 4 sperm whales (7.8%) and 2 unidentified baleen whales (3.9%). Of the fin whales killed by ship strikes, 16 were males, 13 females, and 9 were not identifiable. The average length of males was 14.6 m (± 3.5 m) and 16.1 m (± 2.6 m) for females.

Due to the high representation of fin whales in the total sample of ship strike events (83% of balaenopterids), all balaenopterids (fin whales, minke whales, humpback whales, and unidentified balaenopterids) were collated for analysis.

Temporal Trends

The total number of strandings have increased over the last 46 years for sperm whales in the Atlantic Ocean and Mediterranean Sea as well as balaenopterids in the Atlantic Ocean. Strandings of balaenopterids in the Mediterranean Sea increased steadily until the last decade during which the number of strandings slightly decreased (**Figure 1**).

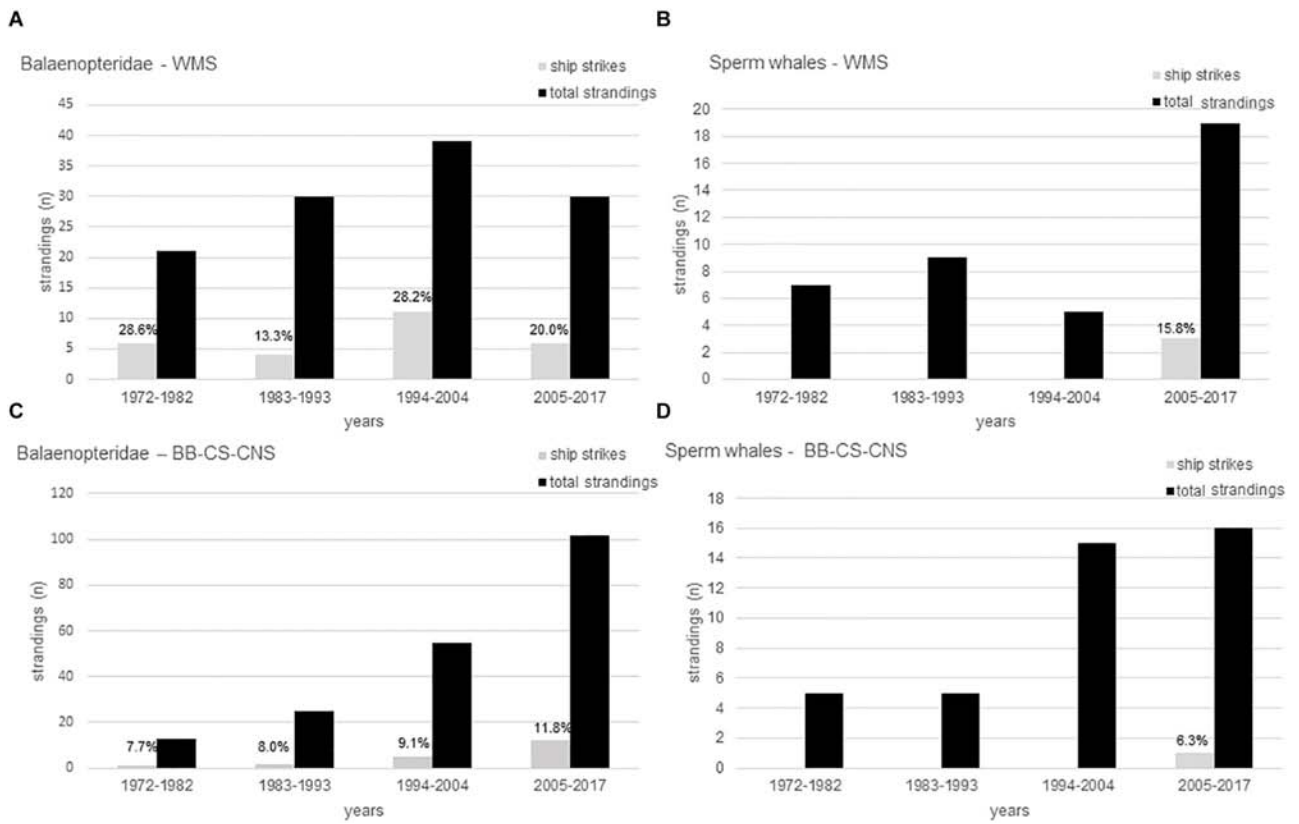

FIGURE 1 | The number of strandings with evidence of a lethal ship strike (gray bars) and the total number of strandings (black bars) per decade interval for balaenopterids **(A,C)** fin whales, minke whales, humpback whales, and undefined baleen whales) and sperm whales **(B,D)** along the Western Mediterranean Sea (WMS) and Bay of Biscay (BB), Celtic Sea (CS) and Channel-North Sea (CNS) coasts of France. The percentage of stranded carcasses with evidence of ship strike as a proportion of all strandings are indicated.

Evidence of ship strikes were only reported for sperm whales in the last decade, while balaenopterid strikes are documented as early as 1972 and increased over the decades: a total 18 baleen whale strikes were reported along French coasts between 2005 and 2017 (**Figure 1**). The proportion of balaenopterid strandings caused by ship strike per decade was variable in the Mediterranean Sea (22.5% ± 7.3%) and increased over the decades along the Atlantic coast. The proportion of total strandings caused by ship strikes remained stable over the decades. Ship strike induced strandings occurred throughout the year but more frequently between the months of July and November (67% of ship strike strandings) (see **Supplementary Table S1**).

Geographic Distribution of Ship Strikes

Strandings due to ship strike were more frequent along the Mediterranean coast than the Atlantic coast (**Figure 2**). 28 whales were struck and found stranded in the Mediterranean Sea (including 24 fin whales, one humpback whale and three sperm whales) compared to 21 whales in Atlantic Ocean and English Channel (15 fin whales, four minke whales, one humpback whale, and one sperm whale). The majority of ship strikes reported on the Mediterranean coast were recovered on the eastern part of the Gulf of Lion and the Ligurian Sea (97%), an area which includes the Pelagos Sanctuary.

DISCUSSION

Ship strikes were the predominant anthropogenic cause of death identified in large cetaceans along both Atlantic and Mediterranean coasts. This is in line with the results of studies in other parts of the world that report a direct correlation between the global increase of shipping activity, engine power, and vessel speed (Laist et al., 2001; Ritter, 2012; Cates et al., 2017). Vessel speed correlates positively with the probability of ship strikes and the severity of injuries (Vanderlaan and Taggart, 2007; Douglas et al., 2008; Ritter, 2012; Conn and Silber, 2013).

The increase of large whale strandings does not appear to be related to an increase in public awareness or reporting pressure, at least for the last three decades during which observation effort was stable. Temporal changes in strandings are therefore likely due to changes in cetacean abundance and distribution, and/or changes in the intensity of pressures (Peltier et al., 2012).

High densities of large cargo vessels in major shipping routes create a serious risk of ship strikes. The Ushant Traffic Separation

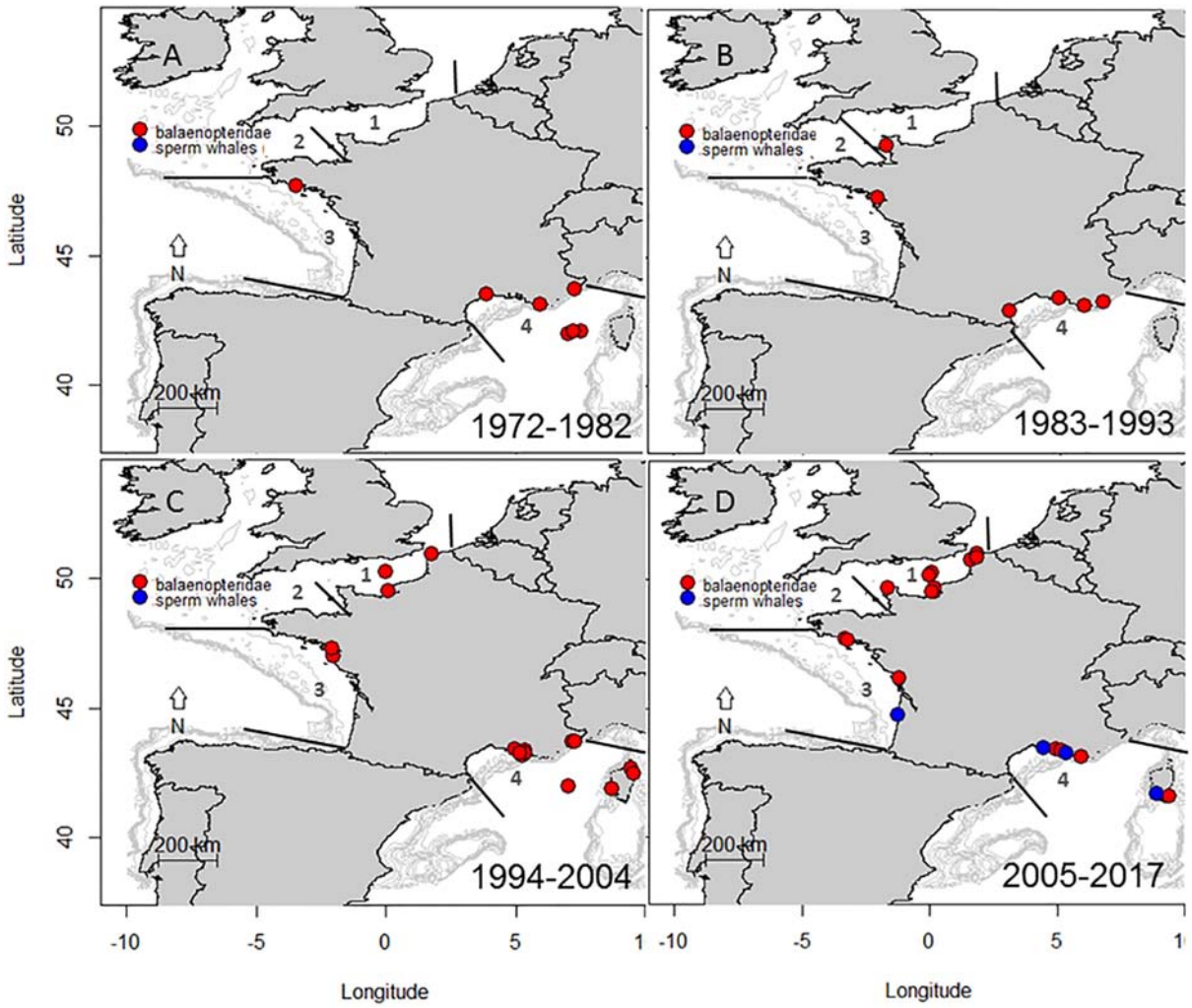

FIGURE 2 | Geographic distribution of balaenopterid (red dots) and sperm whale (blue dots) strandings with evidence of lethal ship strikes per decade (**A**: 1972–1982; **B**: 1983–1993; **C**: 1944–2004; **D**: 2005–2017). Dots depict an individual stranding event. Marine sub-regions are presented (1 = Channel and North Sea; 2 = Celtic Sea; 3 = Bay of Biscay; 4 = Western Mediterranean Sea).

Scheme in the English Channel and the Mediterranean Sea are two of the most important waterways of the world (Lloyds Maritime and Intelligence Unit, 2008). Individuals injured in these shipping routes may, however, strand great distances from the location they were struck or never strand at all. One study described a fin whale carcass being dragged over 1100 km by a cruise ship after a ship strike (Laist et al., 2001). Therefore, despite stranding data being the best source of information available to determine ship strike incidence, the degree to which they are representative of actual ship strikes is limited and the threat of ship strikes may be under- or overestimated based on stranding numbers.

Based on their known distribution, fin whale densities were expected to be highest on the continental slope of the BB and the oceanic area of the Pelagos Sanctuary in the summer months, and absent in the English Channel (Laran et al., 2017a,b). The relative number of ship strikes in the English Channel may have been over-estimated: animals injured in adjacent areas of the Atlantic Ocean may have drifted to shores along the English Channel. Ship strikes in the BB may have been under-estimated: the majority of collisions would likely have occurred in the dense shipping routes far from the coast beyond the continental shelf. Animals that stranded along the shores of the BB after collisions in these parts would have drifted long distances to the shores. The bad drift conditions in the BB in summer (Peltier et al., 2013) (when whale density and therefore collision risk is high) could have prevented some carcasses from reaching the coast at all. Moreover, because of the long travel time coupled with the aggravated decomposition of the carcass after blunt force trauma, the animals could have reached the shores in too bad a state to be able to identify evidence of ship strike.

Fin whales are of particular concern in French Mediterranean waters. A recent study estimated that the fin whale population

in the French Mediterranean Sea numbered only 2,500 individuals [CI 95% = 1472–4310] (Laran et al., 2017a) in the summer. The small population is characterized by limited gene flow (Palsbøll et al., 2004) and is thus particularly vulnerable to anthropogenic pressures (Panigada et al., 2006; Panigada and Notarbartolo di Sciara, 2012).

Aguilar et al. (1988) reported fin whale sexual maturity at a size of 17.4 m for females and 18.5 m for males. The majority of fin whales that stranded following a ship strike had not yet reached maturity: eight of the 13 females and 15 of the 16 males. Immature fin whales seem to be more vulnerable to ship strikes than mature animals, a result that is supported by the findings other studies (Laist et al., 2001; Panigada et al., 2006; Douglas et al., 2008). Laist et al. (2001) also reported that a high proportion (75%) of northern right and humpback whale fatalities due to ship strikes were calves and juveniles. Immature whales may be more naïve to ships and spend more time surfacing when vessels are in the vicinity.

Scientists need to identify the main pressures within each MSFD component in order to develop monitoring schemes and indicators to assess the condition of European marine environments, to evaluate the efficiency of mitigation measures, and to recover GES. A challenge for the second MSFD 6 years cycle will be to complete the currently used set of indicators (focused on cetacean abundance and distribution, bycatch, and contaminant issues) to allow for a broader assessment of species and the pressures they face, especially large whales (e.g., ship strikes). The development of quantitative indicators to monitor the levels and the impacts of ship strikes on large whale populations is vital for future cetacean studies under the MSFD. Such indicators could document the criteria on biodemographic parameters (called D1C3), which require estimating mortality rates.

In the context of the next GES assessment in 2024, the proper assessment of ship strike mortality on cetacean populations in European waters through the MSFD requires: (1) the stimulation of transboundary collaboration at a basin scale to collect enough standardized data over a large enough area to be relevant for such large and mobile species (Authier et al., 2017), and (2) the development and use of quantitative indicators and thresholds adapted to the low occurrence of ship strike records. Overcoming these challenges would be an important step in integrating ship strike risk and impact in the future assessment of GES for cetaceans.

ETHICS STATEMENT

This work was carried out in the respect of European regulation regarding the use of stranded dead cetacean for scientific and conservation purposes. The authors have therefore adhered to general guidelines for the ethical use of animals in research, the legal requirements in Europe. No living animals were used for this study, only dead cetaceans found stranded along European coasts by several organisations were considered. No samples were used for this study.

AUTHOR CONTRIBUTIONS

HP performed the analyses and wrote the manuscript. WD, CD, FDe, GD, and OVC coordinated the French stranding network and collected the stranding data. AB, CC, FDh, and HL collected the stranding data. JS supervised and corrected the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00486/full#supplementary-material

TABLE S1 | Details of the 51 large whale strandings with evidence of lethal ship strikes along the coasts of mainland France for different marine sub-areas between 1972 and 2017. Total numbers par MSFD reporting cycles according to marine sub-regions are presented [number of individuals (reporting cycle)].

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REVIEW
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The Assessment and Management of Marine Pest Risks Posed by Shipping: The Australian and New Zealand Experience

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Ships have been translocating species around the world for hundreds of years but attempts to understand and manage this issue date back only three decades. Here we review the assessment and management of risks from vessel biofouling and ballast water over this time period from an Australian and New Zealand perspective. We detail a history of successes and failures at the science-policy interface that include international guidelines for biofouling management and the recent ratification of a ballast water convention. We summarize the efficacy and costs of current treatment options, and highlight the practical challenges and policy implications of managing the diffuse and succinct bio-invasion risks that shipping creates pre- and post-border. We then use the lessons learnt over the last 30 years to recommend a future empirical strategy.

Keywords: marine pest, risk assessment, monitoring, ballast water, biofouling

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INTRODUCTION

The role of ballast and biofouling as vectors for the translocation of species globally has been known for over 60 years (Elton, 1958; Medcof, 1975). The magnitude of the consequences for human health, the economy and environment, however, was not widely publicized until the mid-1980s (Carlton, 1985; William et al., 1988). At this time, and in the decade that followed, the gravity of the problem was underlined by the introduction of Cholera to Peru (McCarthy and Khambaty, 1994), the global increase in the frequency, intensity and distribution of paralytic shellfish poisoning (Hallegraeff and Bolch, 1992), the role of the Atlantic comb jelly *Mnemiopsis leidyi* in the collapse of the Black Sea ecosystem (Shiganova, 1998), the economic and environmental impacts of the zebra mussel *Dreissena polymorpha* (Holland, 1993), and a growing awareness that the number of species being translocated around the globe was much larger than previously realized (Carlton, 1992).

The international community was initially quick to respond to the marine pest threat. In 1991 the Marine Environment Protection Committee (MEPC), a subcommittee of the International Maritime Organization (IMO), adopted the International Guidelines for Preventing the Introduction of Unwanted Aquatic Organisms and Pathogens from Ships Ballast Water and Sediment Discharges (**Figure 1**). Progress after that was significantly slower. The MEPC guidelines began a series of international efforts to manage ballast water. It took 13 years, however, before the International Convention for the Control and Management of Ships' Ballast Water and Sediments

was adopted by the IMO in 2004, and another 13 years before the convention was ratified in September 2017[1].

Policies that seek reasonable assurance of a net benefit must balance the often readily quantifiable costs of management against the more uncertain economic, environmental and health benefits. Scientific risk assessment is often used to help find this balance. Whilst some cost-benefit calculations for ballast water management suggested clear net benefits (World Wildlife Fund, 2009), uncertainty around this process contributed to the delay between recognizing the ballast water threat and managing it.

The principle of risk versus return is implicit in the ballast water convention. A key component is regulation A-4 which lays out the nature of exemptions that can be given to vessels. These require the application of defined risk analysis methods which are described in Guideline G7, significant elements of which reflect thinking and approaches that were originally developed in Australia and New Zealand. Both countries were leaders in introducing controls to manage the risks posed by introduced marine species. This was supported by their well-developed terrestrial biosecurity systems and island status which reduces certain complications faced by other jurisdictions.

The recent ratification, and entry into force, of the convention provides an impetus to review current knowledge around the risk-based management of marine pests. In the 15 years since the convention was adopted, Australian and New Zealand authorities have gained significant practical experience in applying different risk assessment techniques, and theoretical developments in estimating risk have taken place. This paper performs that review. It considers risk management for threats posed by both hull fouling and ballast water and for completeness considers the risk-based management of both domestic and international ballast water. We first review the Australian and New Zealand experience of using risk-based approaches, and then propose a conceptualization of the problem that is consistent with the convention but respects the relevant uncertainties and knowledge. We close by discussing the implications of this conceptualization and recommend future research directions.

RISK ASSESSMENT AND MANAGEMENT

Ballast Water

At the outset of the marine pest issue, scientists and regulators naturally looked to existing analogs for guidance on how risk assessment for marine pest introductions might be conducted. Many analysts (including ourselves) turned to three situations that, at least initially, looked analogous to the problem of risk assessment for marine pest incursions via ballast water and biofouling: (i) assessing the pre-border risk posed by accidental introductions of pests of agricultural production systems, including aquaculture; (ii) the risk of spread and impacts following post-border outbreaks of pests in these systems; and (iii) pre-border assessments of the incidental risks posed by deliberate introductions of biocontrol agents.

[1]http://www.imo.org/en/About/Conventions/ListOfConventions/Pages/International-Convention-for-the-Control-and-Management-of-Ships'-Ballast-Water-and-Sediments-(BWM).aspx

With exception of biocontrol agents, where assessment requirements and guidelines were developed relatively late (Barratt et al., 2010), risk guidelines and procedures were well established for each of the other situations, and this material often carried the imprimatur of respected national and international organizations (Kohler and Stanley, 1984; Kellar, 1993; Morley, 1993; Office International des Epizooties, 1996; Food and Agricultural Organization, 2006).

A common feature of these guidelines and procedures was that each adopted a species-specific perspective to the risk assessment problem, advocating that each assessment be treated on a case-by-case basis, using the characteristics of the species and receiving environment to guide the assessment process. Unsurprisingly, species-specific approaches for risk assessment of pests associated with biofouling and ballast water were identified as a possible approach to vector management, in-keeping with international expectations for trade.

There were, nevertheless, some misgivings about this approach at the time. Public submissions to a 1996 New Zealand government discussion paper on proposed regulatory approaches for the management of ballast water and biofouling highlighted the lack of detailed information available to underpin species-specific assessments, including on the species likely to be transported into New Zealand, the prospects for their establishment and likely consequences (New Zealand Government, 1998).

In the absence of this information, the Government supported a precautionary approach which assumed that all ballast discharges from vessels entering New Zealand could contain unwanted species, and in May 1998, New Zealand was among the first countries to implement mandatory requirements for ballast water management that reflected the three management actions – mid-ocean exchange preferably 200 nautical miles from land and in water over 200 m deep, onboard treatment, or discharge to an on-shore facility – identified in the 1991 IMO guidelines, without any provisions for exemption based on risk assessment (McConnell, 2002). Australia implemented similar mandatory requirements for international ballast water in July 2001 (**Figure 1**), but also sought to offer exemptions based on a species-specific, risk-based, Decision Support System, supported by a target-species list and port baseline surveys (McConnell, 2002; Hayes and Sliwa, 2003; Campbell et al., 2007) due to concerns about the feasibility and cost of mid-ocean exchange, particularly for domestic voyages between Australian ports.

Species-specific methods were not the only approach advocated for biofouling and ballast water risk assessment. Carlton et al. (1995) raised the notion that environmental distance could act as a proxy for probability of survival, suggesting that translocations of ballast water between saline and fresh water, and between polar and tropical, environments would be safe because of the severe environmental dissimilarity between source and recipient regions. Hilliard and Raaymakers (1997) extended this idea and proposed a method of assessing ballast water risks based almost entirely on the environmental distance between the source and recipient ports measured with 37 variables, rather than just temperature and salinity.

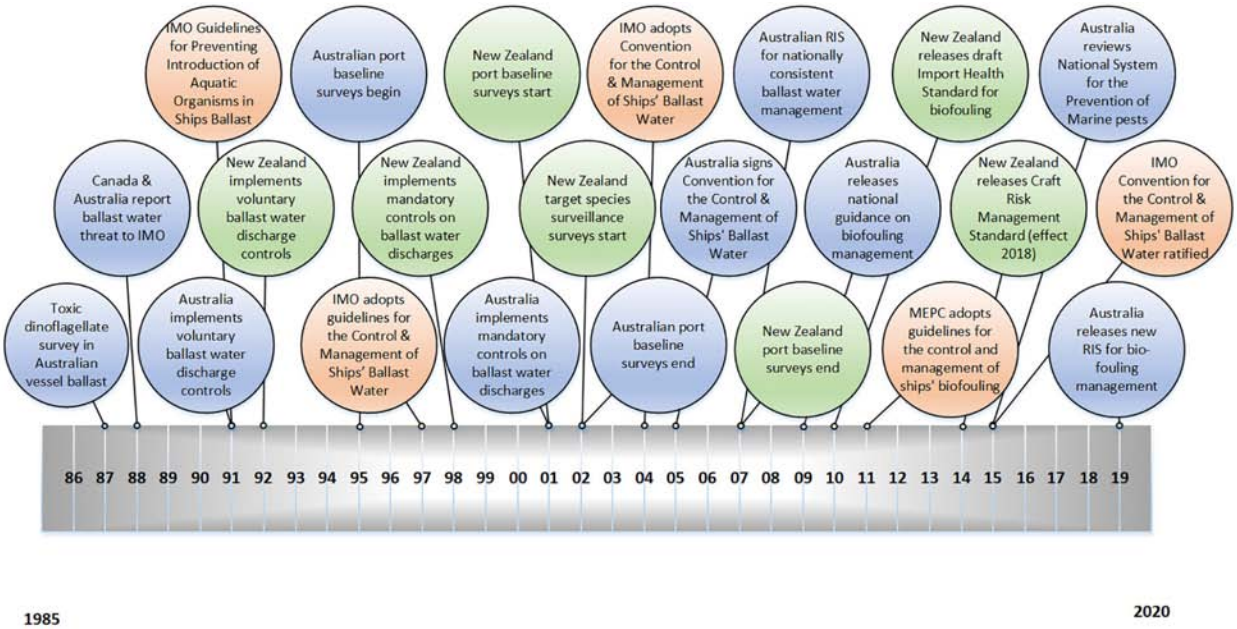

FIGURE 1 | Timeline of major international (orange), Australian (blue) and New Zealand (green) management actions associated with marine pest introductions by ballast water and hull fouling, from 1985 to the present day.

Considerations and recommendations for how to conduct risk assessments using both species-specific approaches and environmental matching, for journeys within and between bioregions, were eventually published by Barry et al. (2008). Early drafts of this document were shared with members of some of the national delegations to the MEPC, and the overall approach and recommendations were taken up in the IMO G7 Guidelines for ballast water risk assessment (Marine Environment Protection Committee [MEPC], 2007). The guidelines provided for three types of assessments – environmental matching, species-specific and species' biogeography – and were subsequently revised in 2017 to include the concept of "Same Risk Area" to recognize the possibility of species-specific, low risk scenarios where target species are already present in all ports within an area or had a high probability of establishing in all locations via natural dispersal (Marine Environment Protection Committee [MEPC], 2017).

Importantly, when the IMO guidelines were released only one (mid-ocean exchange) of the three management options identified by the guidelines was practically available because no shipboard treatments were available globally and no shore-based reception facilities were approved in either Australia or New Zealand. There was therefore some trepidation among Australian and New Zealand officials at this time that each country's trading partners might not accept blanket regulation on ballast water discharges because of the possible ramifications for trade (New Zealand Government, 1998). Many other countries, however, adopted similar regulations (Bailey, 2015) and mid-ocean exchange quickly became standard practice, routinely performed by international vessels whenever journey conditions permitted. According to information provided to the Australian Decision Support System, for example, virtually all international vessels arriving in Australia were able to complete mid-ocean exchange in accordance with the guidelines.

A strong rationale for risk-based exemptions from the blanket-management described in the guidelines remained, however, for vessels undertaking short coastal journeys between ports within the same nation, or in other regions such as the North Sea and Baltic Sea, where it was not possible to meet the D-1 performance standard because there was not enough time, depth or distance to perform 95% volumetric exchange of ballast in water 200 m deep and/or more than 200 nautical miles from shore (Behrens et al., 2005; David et al., 2013). In Australia, target species risk-based management shifted focus to domestic ballast water transfers, culminating in the Australian Domestic Ballast Water Risk Assessment tool[2].

Biofouling

Biofouling threats were originally conceived as arising primarily from fouling organisms that accumulated over time on a vessel's hull. Biofouling increases frictional drag which adversely impacts the vessel's performance (Schultz et al., 2011), hence it is standard practice for marine coatings to be applied to vessel hulls at intervals of between 18 months to 5 years to prevent corrosion and retard biofouling growth (Almeida et al., 2007). Coatings are tailored to the operational profiles of different vessel types to maximize performance and to extend the time the vessel can remain in the water.

Empirical analysis of vessel biofouling confirms that the amount of fouling material on submerged surfaces is correlated

[2]http://www.agriculture.gov.au/biosecurity/avm/vessels/ballast#quick-domestic-ballast-water-risk-assessment-tool

to the number of days since the vessel was last cleaned and anti-fouled and the pattern of use of the vessel, including how often it is used and any significant periods of lay-up, but the effect of the last two factors may vary according to vessel type, the type of anti-fouling paint used, and how good the coating application was (Floerl et al., 2005; Davidson et al., 2009; Inglis et al., 2010; Lacoursière-Roussel et al., 2012; Lane et al., 2019).

Biofouling on the hulls of commercial and recreational vessels can be managed if owners apply anti-fouling coatings that suit their vessel's operational profile, and perform regular maintenance in accordance with the manufacturers specifications (Davidson et al., 2016). Biofouling, however, is not evenly distributed over the hull, but tends to be most abundant in areas where the anti-fouling coatings are damaged or degraded and in recesses that are protected from the drag created by the vessel moving through the water ("niche areas") (Hayes, 2002). On modern merchant vessels, these niche areas can comprise up to 27% of the wetted surface area of a vessel (Moser et al., 2017) and contain >80% of the biofouling on a vessel (Coutts and Dodgshun, 2007; Inglis et al., 2010).

Concerns about vessel biofouling risks and the problem of niche area fouling were raised by Australia, New Zealand and Friends of the Earth International in 2005 at the 54th session of the IMO MEPC. In 2007, the MEPC approved the development of international measures for minimizing the transfer of invasive aquatic species through biofouling as a new high priority item in the work program of the Bulk Liquids and Gases Sub-Committee. The resulting Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species (Marine Environment Protection Committee [MEPC], 2011) were subsequently adopted by MEPC for commercial vessels in 2011 [MEPC.207(62)] and recreational craft in 2012 (MEPC.1/Circ.792).

The MEPC biofouling guidelines present best practice for choosing, applying and maintaining anti-fouling systems for vessels and include recommendations for regular in-water inspection and cleaning of problem areas, training and record keeping. Member States were requested to take urgent action to apply the guidelines and to report back to MEPC on experience gained through their implementation (MEPC.1/Circ.811). Although voluntary, the guidelines do not preclude individual States from applying other mandatory measures to provide additional protection from invasive biofouling species within their jurisdiction.

In 2004, the New Zealand government commissioned a multi-year research survey of over 500 international merchant, recreational, passenger, fishing, and slow-moving vessels to characterize the biofouling risks associated with arriving vessels (Inglis et al., 2010; Piola and Conwell, 2010). The survey showed that most vessels (>70%) from all the major types examined conveyed some biofouling into New Zealand on arrival. Over 65% of the 187 biofouling species identified in the study were non-indigenous to New Zealand and >70% of them had not yet established in New Zealand (Inglis et al., 2010).

This research was used, with other sources of information, to inform a qualitative import risk analysis of vessel biofouling (Bell et al., 2011), which eschewed a species-specific approach in favor of analysis of risks from 20 broad taxonomic groupings of biofouling organisms. This was due to: (i) the large number of species that had been recorded from studies of biofouling worldwide (>2000); (ii) the relatively poor quality of information available on their global distributions, ecology and potential impacts; and, (iii) the difficulty in predicting from this large species pool which species may be problematic. Twelve of the 20 taxonomic groups that were assessed contained species adjudged to present non-negligible risks to New Zealand's marine ecosystems and for which risk management measures could be justified. This finding, and the need for a simple, streamlined clearance procedure for vessels at the border, meant that a precautionary approach was again proposed, with any macro-organisms found on the hull of an arriving vessel considered to be risk organisms (Bell et al., 2011).

Following the risk assessment recommendations, the New Zealand government initially proposed an Import Health Standard that required vessels arriving into New Zealand to meet a clean hull standard that was defined as having "no visible aquatic organisms on the hull, including niche areas, except as a slime layer". The practicality of the clean hull standard, however, was subsequently questioned during consultation because even well-maintained vessels will accumulate some biofouling while operational (Inglis et al., 2010).

The Import Health Standard was revised and released in May 2014 as a Craft Risk Management Standard with an initial 4-year lead-in period before becoming mandatory in 2018 (Ministry for Primary Industries, 2018). The "Clean hull" requirement was altered to allow some macro-fouling with the amount depending on the intended length of stay within New Zealand waters. These allowances were explicitly described as biofouling thresholds within the standard. Vessels intending to remain in New Zealand for more than 21 days or which intended to visit areas not designated as "Places of First Arrival" are permitted to have no more than a slime layer and goose barnacles present on any area of the hull and niches. Vessels visiting Places of First Arrival and intending to remain in New Zealand for less than 21 days are permitted an additional allowance of some early-stage macro-fouling, specifically macroalgae, barnacles, tubeworms and bryozoans. The standard specifies thresholds for the maximum allowable size (in the case of macroalgae), cover and richness of these four taxonomic groups on the wind/water line, general hull area and niches, with justification for these thresholds described by Georgiades and Kluza (2017).

Subsequent debate about the Craft Risk Management Standard has been mostly directed at the biofouling thresholds and their practicality. This debate, however, overlooks the sections of the standard that outline a range of acceptable measures for meeting the "Clean hull" standard (section 2.3). These include continual maintenance of the vessel in accordance with the MEPC 2011 biofouling guidelines, cleaning within 30 days of arrival in New Zealand, the application of approved treatments or submission of a Craft Risk Management plan that outlines steps taken to reduce risk sufficiently to meet the standard. In practice most vessels resort to these other forms of acceptable evidence to demonstrate compliance with the standard rather than the biofouling thresholds.

In Australia progress on managing biofouling lagged behind ballast water because: (i) it was considered, at least initially, to be adequately managed in commercial vessels by extant management practices; (ii) at least partially managed in the recreational sector by extant practice, and; (iii) because the stakeholder groups in the recreational sector are more diffuse and therefore more difficult to represent within the biosecurity governance arrangements of the time.

In 2011, Hewitt et al. (2011) completed a qualitative assessment of the likelihood of entry, establishment and impacts of more than 1781 individual biofouling species. They then identified 56 Species of Concern that were not currently known to be present in Australia, had a high probability of arriving in Australian waters as biofouling on international vessels and had the potential to cause unacceptable impacts to environmental, economic, social/cultural or human health values. Western Australia and the Northern Territory had corresponding schedules listing Species of Concern that currently include 82 and 44 species, respectively, (Northern Territory of Australia, 2009).

The Commonwealth Government of Australia subsequently released a Regulatory Impact Statement for consultation (Price Waterhouse Coopers, 2011) that proposed new regulations, whereby commercial and recreational vessels would be assessed using an online tool to assess biofouling risk. Vessels assessed as "high" or "extreme" risk would be subject to restrictions on the time they could operate at any one port (48 h), at a series of ports (8 days total) or within Australian waters (14 days). If the vessel was unable to conduct its business within these restrictions it would be required either to leave Australian waters or be subject to a hull inspection to determine if any quarantine pests (Species of Concern) were present.

In 2015, a national review of arrangements for marine biosecurity highlighted significant concerns among stakeholders about the species-based risk assessment and approach to biofouling regulation. High among these concerns were the costs of developing and maintaining lists of Species of Concern, the evidential basis for assessing risk and the administrative burden to vessel operators of implementing this regime. The review recommended biofouling requirements for international vessels that were more closely aligned to the (Marine Environment Protection Committee [MEPC], 2011) guidelines and to the approaches taken by New Zealand and California. A new Regulatory Impact Statement, issued in April 2019, reflects this outcome. It proposes three regulatory options for consideration, with the preferred option being the requirement for vessels to implement vessel-specific biofouling management practices consistent with the Marine Environment Protection Committee 2011 guidelines (Australian Government Department of Agriculture and Water Resources, 2019). These practices require vessels to develop and maintain a Biofouling Management Plan and BioFouling Record Book, without which vessels will be targeted for inspection on arrival and required to provide evidence that their biofouling risks had been "appropriately managed." The regulations also provide for a 5-year soft-start period, starting in September 2020, during which vessels without a BioFouling Management Plan may use other options, including treating or cleaning the hull and niche areas less than 30 days prior to arrival, to demonstrate effective biofouling management.

At a state level currently, only Western Australia and the Northern Territory have formal policies on biofouling. Vessels arriving to Western Australia ports from outside the state are required to ensure that marine pests and diseases are not being carried in biofouling and inspectors accredited by the West Australian Department of Fisheries routinely inspect vessels. This Department also maintains a voluntary, online risk assessment tool[3] that allows operators of commercial vessels, non-trading, petroleum and commercial fishing vessels to assess the biofouling risk associated with any planned international or interstate movements, and thereby assist in the endorsement of any planned activities with the relevant state agencies.

EFFICACY AND COSTS

Ballast Water Exchange

Shipboard studies of ballast water exchange show that the efficiency of volumetric exchange can vary from 66% to more than 99%, depending on ship type and method of exchange (Ruiz and Reid, 2007; Molina and Drake, 2016). Using the empty-refill method, exchange efficiencies typically exceed 98%, but if not managed correctly this method can create hazards for the vessel including instability and excessive shear forces on the hull (Endresen et al., 2004). Although safer, the flow-through method can result in much lower exchange efficiencies because of mixing between the influent and effluent water (Noble et al., 2016). Meta-analysis of empirical studies shows that concentrations of zooplankton are reduced by both methods of ballast water exchange by between 34 and 100%, with higher variability in outcomes for protists, bacteria and virus-like particles (Molina and Drake, 2016).

Ballast water exchange involves two types of costs: (i) the cost of operating the pumps, including fuel, energy, labor, and maintenance; and, (ii) the opportunity costs associated with slowing ship speed or diversion to areas that meet the D1 Standard of the Ballast Water Management Convention. The pumping costs depend on the type and size of the vessel, the ballast tank configuration and the method of exchange. Arthur et al. (2015b) estimate the average operational cost of exchange for vessels entering Australia at between USD $0.017 and $0.029 per tonne, with an average cost per vessel of USD $2790 for bulk carriers and USD $2020 for other vessel types. The total annual cost of ballast water exchange by vessels entering Australia was estimated at USD $29.3 million (Arthur et al., 2015b).

Costs associated with delay or diversion will vary according to the length of voyage and ship's ballast water capacity. For most vessels traveling to New Zealand from other overseas ports these are not likely to be significant because most routes allow sufficient time to perform exchange in suitable areas. For shorter, coast-wise voyages, however, they can be significant. For example, mandatory diversion of ships traveling between Australian ports to areas that are at least 50 nautical miles from the coast and

[3]https://vesselcheck.fish.wa.gov.au/

200 meters deep (i.e., less stringent than the D-1 standard) has been estimated to cost more than USD $46 million per year (CIE, 2007).

Ballast Water Management Systems

To comply with Regulation D-3 of the Ballast Water Convention, ballast water management systems must have Type-approval from the Flag State administration. Type approval requires both land-based and shipboard testing of performance relative to the D-2 discharge standard (IMO, 2018a). As at January 2019, 14 administrations had advised Type approval of 76 ballast water management systems, 29 of which used active substances involving chemical or biological treatment (IMO, 2019). Although manufacturers continue to refine their systems to improve performance, a 2010 review of available technologies showed variable efficiency in removing dinoflagellates, phytoplankton and zooplankton from ballast water (Tsolaki and Diamadopoulos, 2010). Physical separation techniques (filtration and cyclonic separation) reduced concentrations by between 8 and 95% of original values. Mechanical treatments (ultraviolet, heat treatment and electric pulse applications) were generally more effective at reducing phytoplankton and zooplankton (reductions of 40% to more than 95% relative to controls) but less effective at treating dinoflagellate cysts (6–40% reductions). Treatments with active substances had the greatest efficacy, generally achieving reductions in excess of 80% across a range of organisms (Tsolaki and Diamadopoulos, 2010). Vessels entering the coastal waters of the United States are increasingly reporting the use of ballast water management systems but as of December 2015, 20 months before the Convention entered into force, they accounted for less than 2% of all ballast water discharged in the United States per month, (Davidson et al., 2017) whereas all ships must now meet the D-2 standard by the 8th September 2024.

The costs associated with ballast water management systems vary according to the type of system, vessel type and size, and whether it is to be installed on a new build or retrofitted to an existing vessel. King et al. (2012) reviewed information provided by vendors for a range of system types and ship types. Purchase prices of the units ranged between USD $640,000 and $947,000. Estimated costs of installation varied from USD $27,000 to $70,000 for new builds, depending on vessel type, and from USD $48,000 to $173,000 for retrofits. For most systems, the annual operating costs for maintenance were typically between USD $9,000 and $17,000, depending on vessel type and size, but technologies that used active substances had a much wider range (USD $31,000–$296,000) because the use of chemicals varied widely between different ship types and sizes.

Biofouling

In contrast to ballast water, there does not appear to be any comprehensive analysis of the compliance levels or efficacy of the Marine Environment Protection Committee 2011 biofouling guidelines. Guidance released in 2013 specified a range of performance measures for evaluation and a questionnaire *pro forma* to aid in the review (IMO, 2013). The MEPC has recently committed to review the guidelines in 2020–2021 based on these measures (IMO, 2018b). Data is emerging, however, in New Zealand. In 2015, the Ministry of Primary Industries initiated surveys of bio fouling on 40 arriving vessels to assess compliance with the (then voluntary) Craft Risk Management Standard. Thirty-nine vessels (greater than 95% of all arrivals) had some biofouling in niche areas and 16 (40%) were non-compliant with the short-stay biofouling thresholds (Kluza, 2018). Since the standard came into force in 2018, New Zealand authorities have taken action against 14 high-risk vessels, representing less than 1% of all arrivals. Six of these vessels were ordered to leave New Zealand waters within 24 h, while the others faced restrictions on the number of ports they could visit.

Section 7 of the MEPC biofouling guidelines recommends cleaning, maintenance and periodic inspection of ships to remove biofouling, which implies costs to vessel owners, operators and relevant authorities. Although technologies exist for in-water cleaning of vessels, most do not contain and capture material removed during the cleaning process so that biological material and contaminants from the anti-fouling coatings are released into the surrounding environment (Morrisey and Woods, 2015). As a result, in-water cleaning is banned or tightly regulated in many jurisdictions. The cost of removing the vessel to dry dock and applying anti-fouling coatings ranges from around USD $100,000 per vessel weighing up to 5,000 tonnes to over USD $464,000 per vessel (for vessels >200 meters in length) and may take 5–7 days with average opportunity costs per vessel per day of USD $4,400 for bulk vessels, USD $9,600 for general cargo vessels and USD $11,200 for container vessels (Branson, 2012; Inglis et al., 2012).

Cost estimates for inspection regimes are currently more difficult to come by in part because there are no internationally agreed protocols for inspection and documentation of biofouling. The Australian 2019 Regulatory Impact Statement for biofouling management estimates that the additional dive time needed to add biofouling considerations to the scheduled class survey, in-water inspections (that occur on average once every 2.5 years) will cost AUD $667, whereas specific in-water biofouling inspections are estimated to cost AUD $7,000 per vessel (Australian Government Department of Agriculture and Water Resources, 2019). The statement also estimates that the regulatory burden of developing and maintaining a vessel's biofouling Management Plan and biofouling Record Book will be approximately AUD $15 per vessel per year.

PRACTICAL CHALLENGES AND THEIR IMPLICATIONS

Risk assessments for ballast water and biofouling face a number of similar challenges, with biofouling presenting additional difficulties because unlike ballast water, whose source can be determined relatively precisely, biofouling can be acquired from multiple locations during a vessel's in-service period, making its source pools harder to identify. Some of these challenges where foreshadowed by Simberloff (2006), and several others identified by Barry et al. (2008).

Environmental Variables, Distance and Invasion Risk

Two primary problems occur with environmental matching: the choice of environmental variables, and the distance metric itself. Firstly, the analysis by Hilliard and Raaymakers (1997), and the GloBallast risk assessments that subsequently adopted this approach (for example Clarke et al., 2004) include a large number of variables in the distance calculation. Barry et al. (2008), however, demonstrate that introducing any environmental variables into the calculation that are not truly predictive of introduction or establishment success creates noise which diminishes the signal of the true predictive distance.

Secondly, environmental distance is not currently well calibrated with any of the potential ballast water risk assessment endpoints of survival, establishment or impact. Several studies use environmental distance to identify donor regions that may act as sources of successful invaders, or incorporate environmental distance into parameters of invasion models (see for example Keller et al., 2011; Seebens et al., 2013), but the empirical relationship between distance and survival or establishment of non-native species has not, in our opinion, been adequately determined. Furthermore, as Ruiz et al. (2013) demonstrate the relationship between these variables and historical introductions may no longer be discernible, and without systematic surveys the relationship into the future will remain obscured by uncertainty about the date of location, the source of inoculation and the responsible vector.

Data Needs and Saturation of Species-Specific Assessment

The primary problems associated with species-specific risk assessment are: (i) the potential for the risk assessment to identify all vessels as high risk (i.e., offer no potential for low risk outcomes) as more and more species potentially transported by the vector are added to the assessment - a process that we refer to as "saturation," (ii) the target species data needs and potential complexity of the modeling task; and, (iii) the availability and currency of information on the distribution of non-native species.

An analysis of an early proposal to develop a risk-based Decision Support System for international ballast water discharges in Australia showed that saturation can occur quite rapidly: 97% of international arrivals were deemed to be high risk with a target species list that was at that time restricted to only 12 species. This was largely due to the lack of information on the presence or absence of the target species in the international donor ports. In the absence of this information the risk assessment took a conservative approach and assumed target species were present, and risk reductions were not forthcoming in other parts of the assessment process.

Saturation did not occur, however, in the analysis of domestic ballast water translocations because the target list was smaller (reduced from 12 to 9), the Australian port baseline surveys (Campbell et al., 2007) and literature (for example Sliwa et al., 2008) provided better data on the distribution of non-native species in Australian ports, and risk reductions were forthcoming in the survival probability models that the assessment used.

The collation of data on the distribution and ecology of non-native species around the world (Katsanevakis and Roy, 2015) provides a growing information platform that can be used to populate parameters in species-specific risk assessment, such as the probability that the donor port contains a target pest. The number of different regional databases, only some of which are actively maintained, presents challenges, however, and the cost of acquiring information on the infection status of port, and maintaining the currency of this information, could become an important impediment to the long-term maintenance of species-specific risk assessments because: (i) as data ages the status of donor ports becomes increasingly uncertain; and, (ii) defensible (conservative) approaches to uncertainty lead to saturation.

Simberloff (2006) highlights a number of difficulties with species-specific approaches to invasion risk, in particular the data requirements, suggesting "*Knowledge on most species is simply in-sufficient to enable more than educated guesses about the likelihood that a species will establish and impacts it might cause*." Assessing the invasion risk of the Pacific oyster (*Crassostrea gigas*) through ballast water or biofouling highlights the data and modeling complexities that can arise: The risk assessment supporting the Australian ballast water Decision Support System minimizes the complexity of the modeling task by choosing a survival endpoint over the alternatives of establishment or impact (Hayes, 2003), on the grounds that the latter endpoints required more complex models, and for species deemed *a priori* to be environmental pests the probability of survival was sufficiently close to stakeholders' concerns to allow decision makers to make management decisions.

The probability of survival was assessed by simulating a species completing its life-cycle, and comparing the temperature tolerance at life-stage against simulated time series of daily temperature extremes in ports across Australia (Hayes et al., 2007, 2008). For sub-tidal species, the data and calculations proved tractable and have since been improved following an independent review (Arthur et al., 2015c). For inter-tidal species, such *C. gigas*, however, the analysis hit significant impediments. Firstly, the daily temperature regime experienced by inter-tidal organisms is substantially different to that of sub-tidal organisms because of their periodic exposure to the interacting elements of sunlight, wind speed and air temperate. Secondly, the body temperature of inter-tidal organisms is also influenced by body size and shape (Helmuth, 2002). These factors substantially raise the number of uncertain parameters in what was otherwise a relatively simple survival model. Whilst interest in these types of biophysical models is increasing in order to predict the effects of climate change (Levy et al., 2015), the data required to develop these types of models for target species is likely to be prohibitive.

Port Surveys

Species-specific risk assessments also require good information on the distribution of target species. Section 6.4.8 of the IMO G7 guidelines, for example, stipulates "*if a target species is already present in the recipient port, it may be reasonable to*

exclude that species from the overall risk assessment for that port unless that species is under active control." In effect, if all of the species that a vessel is assessed as being high risk against are already present (and not being actively controlled) in the recipient port, then the vessel defaults to low risk. Obviously, this approach makes no provision for species other than the target species, but it is based on the reasonable expectation that treatment costs should not be imposed on vessels if they are translocating species to locations where they already exist.

Section 6.4.8 can be problematic because of the high degree of power it imposes on the validity and currency of port survey information. In 1995 Australia embarked on a series of port baseline surveys following the "CRIMP protocols" (Hewitt and Martin, 1996, 2001). By 2005, 39 Australian ports had been surveyed to an accredited standard with these protocols, and the results entered into a National Port Survey Database (McEnnulty et al., 2005). The protocols were used extensively in New Zealand and elsewhere in the world, and have been the most widely implemented method of baseline survey for invasive species (Campbell et al., 2007).

The Australian baseline surveys helped define the distribution of non-native species in Australia. The surveys, however, are relatively expensive, and require a very high level of taxonomic expertise to successfully post-process the samples that are collected (Bishop and Hutchings, 2011). For example, only about 27% of the 15,412 taxonomic records in the Australian National Port Survey Database include a complete species name. The remaining records represent identifications to the level of Genus only (McEnnulty et al., 2005).

Australia responded to the requirement to maintain up-to-date records of the distribution on non-native species in its ports by developing a set of port monitoring guidelines that stipulated not only which ports should be regularly monitored but also to what standard (Australian Government Department of Agriculture Fisheries and Forestry, 2010a,b). The guidelines identified 18 ports around Australia as the minimum required to form an effective National Monitoring System.

Many of these ports, however, are large, complex regions that are difficult to sample for reasons such as low visibility and/or strong currents. Furthermore, the sample sizes and hence costs of sampling these locations to a standard with high statistical power proved to be too high (reportedly between AUD $175,000 and AUD $355,000), to repeat on a regular basis, and only 5 of the 18 locations, together with seven other locations, were surveyed to the recommended standard.

A lack of clearly understood objectives among stakeholders, together with the high cost of the surveys, coinciding with a significant reduction in resources allocated to marine pest research and development in Australia at this time, stalled the implementation of the National Monitoring System, and as part of a wider review of Australia's marine biosecurity systems (Australian Government Department of Agriculture and Water Resources, 2015), it was recommended that the system be abandoned and replaced with a surveillance system based on cheaper methods designed with clearer objectives (Arthur et al., 2015a).

One of the cheaper methods recommended by Arthur et al. (2015a) – the use of eDNA probes based on species-specific primers – holds promise but requires extensive testing to determine the relationship between presence and abundance of pests in the environment and the availability of eDNA to sample. Currently these probes may suffer high rates of false negatives (at the survey level), because eDNA is not sampled even though the species is present (Wood et al., 2018). These types of probes are also, by construction, species-specific which again raises the prospect of an inevitably restricted target list. eDNA probes based on meta-barcoding may offer the opportunity for "screening level" assessments of multi-species assemblages, but these probes are currently beset with challenges in the supporting infrastructure (i.e., incomplete sequence databases) and analytical pipelines that render them prone to high rates of false positives and false negatives (Ammon et al., 2018). The other method suggested by Arthur et al. (2015a) – sampling only in areas where ballast water is discharged or loaded – are unproven, and in large ports with significant circulation due to strong currents or tidal ranges may not be beneficial.

In 2001 New Zealand embarked on a national series of port baseline surveys based on the CRIMP protocols, and by 2007 had completed 43 surveys, including repeat surveys of 13 ports and 3 international marinas. The baselines surveys ended in 2007 and were considered at the time to provide a good indication of the then current distribution of non-indigenous species. In 2002 New Zealand also commenced with a series of targeted (species-specific) surveys initially designed to provide early detection of 7 high risk species. Surveys are now implemented in 11 harbors nationally every 6 months at an annual cost of approximately NZ $2 million (Arthur et al., 2015a). A risk-based stratification of environments within each harbor is used to prioritize allocation of sample effort based on the likely distribution of founding populations of the primary target species (Inglis et al., 2006). These surveys adopt a more pragmatic approach to the problem of achieving a high statistical power (sensitivity) and use survey methods that are generic, quick and efficient to implement, with minimal processing of samples post-collection. Provisional identification of target species occurs in the field with only specimens of target and unrecognized taxa being retained for verification by taxonomic experts. In this way, relatively large sample sizes and coverage can be achieved in each survey at low sample cost. Because the surveys are principally focused on specific taxa, other groups of organisms (e.g., pelagic species, fish and benthic invertebrates) are not well sampled. The sensitivity of an individual survey varies among target species and is optimized across them using a risk-based Stochastic Scenario Tree model (Martin et al., 2007; Morrisey et al., 2012). For some species the per-survey sensitivity is less than the standards typically used to provide Proof of Freedom in veterinary and disease surveillance systems and, in these cases, the New Zealand surveys use Bayesian temporal discounting to provide Proof of Freedom based on non-detection in repeat, ongoing surveys.

FUTURE APPROACHES TO MARINE PEST RISK ASSESSMENT AND MANAGEMENT

Many of the current approaches to marine pest risk assessment are based on the conceptualization that risk can be resolved at the species level, with detailed and sufficiently accurate assessments of individual risks. This view is realistic in certain circumstances. In aquaculture systems, for example, the number of identified pests is typically limited and therefore the knowledge base required to implement species-specific assessment is potentially manageable. This approach is also assisted by the similarity of aquaculture production systems internationally, so that knowledge developed in one location will readily translate to other locations. Moreover, aquaculture systems will have a commercial imperative and financial base to fund the required research and to balance the costs and benefits.

The utility of species-specific approaches is also clear in managing post-border incursions. A key example would be attempting to eradicate an invasive species. In this case the threat is identified and knowledge can be developed from direct observations of the species' behavior in its native and/or previously introduced range. In the case of post-border management, previous experience with arthropods suggests that taxon-specific information can be used to develop specific control plans and thereby improve the probability of successful eradication (Tobin et al., 2014). In developing response plans for specific target species, it is useful to cover a range of functional groups so that a response plan can be adapted easily for an incursion by an unexpected, but functionally similar species.

Pre-border, species-specific approaches to manage invasive species that pose a potential threat to environmental values are more problematic for several reasons. Firstly, the suite of potential invasive species is large but their distribution and ability to invade is poorly known. In addition, our ability to predict impact is limited, relying either on the species behavior in other non-native locales, or theoretical predictions as to the outcomes of the (potentially very) complex interactions between the species and the receiving environment. The Australian and New Zealand experience is that the inherent uncertainty and magnitude of this task can lead to policy impasses: there is potentially some level of risk in all vessels movements but its magnitude is uncertain. Hence any stakeholder can interpret it in a way that is consistent with their position. Moreover, currently there is no agreed way of determining a threshold of acceptable risk.

If we accept that species-specific approaches will not provide a practical basis for managing pre-border environmental pests we need to consider alternatives. Our starting point is to accept that all translocation pathways potentially carry some risk of environmental impact. If mitigation of this risk via a treatment was effectively cost free we would mandate it and the risk would be managed. In reality the costs of mitigation can be high. Hence there is a need to trade off the costs of mitigation versus the potential impact on the environment in a transparent and rational way.

Theory and experimental trials indicate that removing biological material, via ballast water exchange, ballast water treatment or increasingly stringent hull and niche area cleaning, will reduce inoculation pressure and therefore invasion risk (Bailey, 2015; Molina and Drake, 2016). In the limit, the elimination of all biological material on or contained in a vessel will achieve zero invasion risk. Thus we can conceptually rank treatments in terms of the amount of biological removed. It is important, however, that any such procedure accounts for operational conditions and distinguishes between application of a treatment - i.e., the management endpoint - and the actual amount of total biological inputs that are removed - i.e., the biological endpoint. Empirical analysis of the efficacy of ballast water exchange in Chesapeake Bay, for example, illustrates that operational parameters, such as changing trade patterns, can lead to an overall increase in inoculation pressure despite implementation of the management endpoint (Carney et al., 2017).

The key question is how stringent management activities should be in order to mitigate the risk to an acceptable level. Intuitively we believe that a biological threshold exists, we just are uncertain about where it is. Adaptive policy settings and the use of expert panels to define the settings and monitor performance against them are increasingly important means for dealing with complex systems with uncertain management outcomes (Walker et al., 2001). Our key observation is that if we impose a management action, based on expert opinion for example, then we must monitor its biological efficacy, observe the number of incursions that occur and adjust the action if this rate is unacceptable. In this way, we analyze the system outcomes across all species rather than a bottom up assessment of individual species. Evaluation and monitoring of performance at the system level, measured in a way that is meaningful to decision makers, provides the key feedback to assess how well risk has been mitigated (Amendola, 2002).

In essence we propose that a treatment standard is defined that is underpinned and enforced by a rigorous compliance monitoring system. Adherence to this standard potentially results in ecological, economic and social impacts on the jurisdiction due to the establishment of new marine pests. These changes are detected by the system monitoring and that is used by management to assess the acceptability of the rate and nature of these changes. If these changes are unacceptable (because the standard is not stringent enough) the standard is modified.

In practice we recognize several practical challenges. Firstly, setting an expert based threshold involves pragmatism and can appear *ad hoc*. The alternative, however, may be that no action is taken at all while more information is sought to reduce uncertainty. Unfortunately, some of these uncertainties may be irreducible. The proposal converts the problem into one that can collect explicit data (e.g., incursions) to assess performance. This provides a more transparent basis to explore costs and benefits of a system. The threshold can also be tailored to different sectors and jurisdictions where needed. To break any impasse about the practicality of the approach, the threshold could be aligned

to outcomes that are achievable with current best practice and technology in the first instance, and then modified as further information became available.

Secondly, the proposed approach require effective compliance and compliance monitoring is therefore a key component of the system. As noted, compliance monitoring should consider both management and biological endpoints, and could be targeted at the highest risk vessels based on (for example) environmental-matching risk assessment (Barry et al., 2015). We stress that the estimate of risk referred to here relates to the relative risk the vessel may pose, compared to all other vessels. This is a simpler task than assessing absolute risk.

Thirdly, the proposed approach relies on monitoring ecosystem outcomes. This is arguably the most significant challenge. The appropriate level of system monitoring is not solely a science issue but should also reflect the information needs of decision makers. At one extreme a jurisdiction could rely on passive monitoring to provide information about the impacts of new incursion on marine systems. Passive monitoring would rely on members of the public or industry to identify significant impacts as and when they become significant enough to be detected, and also relies on the media and/or government agencies to champion the issue and raise public awareness. Such a strategy, however, exposes jurisdictions to a number of threats such as creeping baselines (Papworth et al., 2009) and potentially an inability to identify incursions until they occupy large areas and are thereby costlier or impractical to eradicate (Tobin et al., 2014). Nonetheless, we believe this is the default approach in many jurisdictions.

At the other extreme, a jurisdiction could use detailed monitoring at a large number of sites to provide an information base for decision making. As discussed previously, detailed surveys in complex port environments are currently expensive and have been hard to justify to funders so careful consideration of the purpose and design of monitoring is needed. This is an area that we believe requires additional attention. The New Zealand experience is that this challenge can be met with a pragmatic approach to survey sensitivity that relies on repeat surveys conducted by a core team who become familiar with the system and are thereby able to identify unusual occurrences. This approach, however, still costs approximately NZ $180,000 per port per year. It remains to be seen if the New Zealand experience could be scaled up to Australia but we believe this is worth further examination.

Finally, an adaptive, standard-based approach implies that the standard could change, and this could result in cost increases (or decreases) as industry moves to a more (or less) stringent standard. Such an approach also requires a standard that ideally: (i) can be assessed, at least in principle, by non-specialists to assist compliance and minimize associated costs; and (ii) describes what an acceptable level of biological contamination is - i.e., what is an acceptable level of fouling or acceptable concentration of organisms of different size classes in ballast water.

This type of standard is prescribed by the IMO D2 Ballast Water Treatment standards and is a feature of the New Zealand Craft Risk Management Standard for hull fouling (Georgiades and Kluza, 2017). The development of such a standard has the advantage of providing an unambiguous definition of what is and is not acceptable that can be directly related to observed environmental outcomes. The proposed biofouling management regime in Australia, which follows the MEPC guidelines, however, has moved away from this approach toward a standard based on a management endpoint not a biological endpoint. This could make it more difficult for an adaptive regime to respond to new information on incursion rates.

In a globalized world it is obviously helpful if standards are consistent world-wide. The ballast water convention is a clear example that, given time, a biological standard, and the technology needed to implement it, can be adopted around the world. The long lead times for international consensus mean that it is important that a similar approach for biofouling commence as soon as possible. In the interim countries can still apply their own standards consistent with their country-specific acceptable level of protection as defined by the World Trade Organization. Different countries are free to apply their own standards as long as they apply equally to all vessels and are, therefore, not trade restrictive.

The adaptive approach outlined above is consistent with the risk management measures adopted by countries for terrestrial biosecurity threats. For example, the importation of soil is tightly controlled in Australia, not on the basis of species-specific assessment but instead by a recognition that soil is a potential vector for a large number of organisms and is therefore unacceptable. Similarly, there are standards applied to the required cleanliness of shipping containers rather than relying on species-specific assessments.

In marine systems where the pathways are capable of sampling from a large species pool and where there is large uncertainty about which species will be transported and prosper in any given region, a species based approach to pre-border risk assessment will not scale well because it becomes impractical as the number of species to be assessed rises. This problem is compounded by the potentially large range of values that are held in marine environments (environmental, social and cultural) and uncertainty in how introduced species may affect them.

Species-specific approaches can, however, help sharpen thinking and preparedness and should not therefore be discarded altogether. Species with a well-established track record of invasion and impact should be identified as unwanted, and they provide an opportunity to refine post-border preparedness - i.e., develop species-specific tactics for detection, eradication and control. If these tactics can be also be developed with a view to how they might be adapted to incursions by unknown/unanticipated species, by example considering the functional group that high profile invaders belong to, then this will improve a jurisdictions ability to respond to new incursions quickly as they are detected.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Lessons From Placing an Observer on Commercial Cargo Ships Off the U.S. West Coast: Utility as an Observation Platform and Insight Into Ship Strike Vulnerability

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Ship strikes of whales are a growing concern around the world and especially along the U.S. West Coast, home to some of busiest ports in the world and where ship strikes on a number of species including blue, fin, and humpback whales have been documented. This trial program examined the feasibility, logistics, industry cooperation, and effectiveness of placing an observer on board a commercial ship. An experienced marine mammal observer went on five voyages, spanning over 8 days on ships operating between U.S. West Coast ports. Daylight observations were conducted over 68 h and covered over 1300 nm as ships transited between three ports [Seattle, Oakland, and LA/Long Beach (LA/LB)]. Sightings of large whales were reported on all (n = 42), totaling an estimated 57 individuals that included humpback, blue, fin, and beaked whales. Close encounters of large whales occurred (on one occasion a near miss, estimated at 40 m, of two humpbacks), and on another, a ship chose to alter course to avoid whale sightings in its path identified by the observer. All ships personnel cooperated and voluntarily aided in the observations even after initial skepticism by some crew about the program. While most effort on mitigating ship strikes along the U.S. West Coast has focused on shipping lanes, the vast majority of these sightings occurred outside these lanes and on the transit routes, emphasizing the need for added attention to these areas. This experiment demonstrates the effectiveness and promise of observations from ships providing critical information on whale locations at risk to ship strikes.

Keywords: ship strike, ship observers, blue whales, humpback whales, fin whales, shipping lanes

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INTRODUCTION

Ship strike of large whales have become an issue of growing concern worldwide and along the U.S. West Coast in particular (Redfern et al., 2013; Rockwood et al., 2017). The eastern North Pacific Ocean (ENP) and specifically coastal waters of the U.S. West Coast are utilized by a number of threatened or endangered large whales including blue whales, fin whales and two

(Mexico and Central America) distinct population segments (DPS) of humpback whales (Carretta et al., 2017). Ship traffic along the U.S. West Coast operate from major ports including those in the north based in Washington and British Columbia that transit out the Strait of Juan De Fuca, those based in San Francisco Bay including Oakland, Richmond, and San Francisco and in the Southern California Bight (SBC) home to Los Angeles/Long Beach (LA/LB), one of the world's largest ports. Ships transiting to and from these ports can go through five west coast National Marine Sanctuaries (NMS), areas identified as biologically important areas (BIAs) for the endangered blue and humpback whales, and for high densities of feeding gray whales (Calambokidis et al., 2015). With large ports and large commercial vessels transiting through these rich marine areas, comes the potential for ship strikes (Redfern et al., 2013; Rockwood et al., 2017; Moore et al., 2018).

Ship strikes of blue, fin, humpback, and gray whales have been frequently documented along the U.S. West Coast (Laist et al., 2001; Douglas et al., 2008; Rockwood et al., 2017) and second only to entanglements in the leading cause of human caused mortality to large whales along the U.S. West Coast (Carretta et al., 2017). Blue and fin whales appear to be particularly susceptible to ship strike mortality along the U.S. West Coast (Berman-Kowalewski et al., 2010; McKenna et al., 2015) though Monnahan et al. (2015) question whether ship strikes threaten the recovery of eastern North Pacific blue whales. Mitigation efforts increased after detection of seven blue whale carcasses triggered an unusual mortality event in 2007 (Abramson et al., 2009), and four of those deaths were attributed to vessel strikes (Berman-Kowalewski et al., 2010). In 2009, the Channel Islands National Marine Sanctuary (CINMS) recommended shipping lane changes (Abramson et al., 2009), which were put into place in 2013. In 2011, the National Oceanic and Atmospheric Administration's (NOAA) Office of Marine Sanctuaries also established a Joint Working Group (JWG) to assess ship and whale interactions including ship strike and acoustic impacts in the Gulf of the Farallones and Cordell Bank, to create a list of recommended mitigation measures (Joint Working Group, 2012). The JWG concluded that in order to reduce lethal ship strikes, a reduction in the co-occurrence of ships and whales had to occur. In their 2012 report, their recommendations included modification of shipping lanes, creation of dynamic management areas (DMA) in regions of high whale concentrations, education and outreach initially engaging and informing the commercial industry, and the creation of a real time monitoring and whale sighting network with commercial ship participation (Joint Working Group, 2012).

Placing dedicated marine mammals observers on board vessels has been proposed as an effective method for getting sighting information and helping to avoid collisions with large whales (Weinrich, 2004; ACCOBAMS, 2005; David et al., 2005; Weinrich et al., 2010; Gende et al., 2011; Couvat and Gambaiani, 2013). We report on a pilot program putting an observer on ships transiting between U.S. West Coast ports to document sightings along these routes, help quantify the threat to whales from ships and also evaluate the feasibility of using these platforms as an expanded source of sighting reports which were all part of the recommendations of the JWG.

MATERIALS AND METHODS

With the assistance of the Pacific Merchant Shipping Association, we worked with two shipping companies that transit between ports on the U.S. West Coast. While many ships using U.S. West Coast ports including the ships we worked with transit to more distant ports including Hawaii and Asia, we only sought transits between U.S. ports. All trips were conducted by the senior author, an experienced marine mammal observer and a licensed captain with professional experience in the maritime industry, which was valuable in negotiating with the companies with assurances she would not disrupt ship operations. The bridge height of commercial vessels is set up to provide the vessel operators the most unobscured 360° view possible for safe operations regardless of the position of the bridge (forward, mid, or aft). Ships we worked with included vessels with both forward and mid-aft bridges (**Figure 1**), for the latter the height of the containers carried on the ships never extends above the bridge height but did partially obscure visibility directly in front of the ship.

Shipping companies and the ship's officers agreed to provide access to the vessel bridge to the observer during all daylight hours. Ship passages occurred from May to September, periods of known occurrence of feeding baleen whales off the U.S. West Coast. Specific voyages were selected based on the vessel having the highest likelihood of being in a shipping channel during the most amount of daylight hours to allow for either entering or exiting shipping lanes into San Francisco and LA/LB, California (**Table 1**).

The observer was on effort approximately 4 h at a time during all daylight hours in all conditions, with 30-min breaks in between observation periods. Environmental (Beaufort, visibility, cloud cover, and precipitation) and effort data were recorded at the beginning and end of each on-effort period and any time conditions changed. During on-effort periods, continuous scans of the horizon were made from the bow to approximately 60° port and starboard with the naked eye and occasionally with a 7 × 50

FIGURE 1 | Examples of ship configurations and views from ships that were part of this study including a ship with a forward bridge, voyages (3–5) **(top)** and a ship with a mid-aft bridge **(bottom)** during voyages 1 and 2.

TABLE 1 | Voyage numbers, dates of travel and ship specifications of ride-alongs.

Voyage	Date	Company	Transit	Vessel	Bridge position	Length (m)	Bridge deck Height off water (m) loaded
1	May 29–30, 2013	APL	San Pedro, LA to San Francisco, CA	Thailand	Mid-aft	276.3	40
2	August 1, 2013	APL	Long Beach, CA to Oakland, CA	Singapore	Mid-aft	276	40.29
3	September 26–27, 2013	Matson	Oakland, CA to Long Beach, CA	Mokihana	Forward	262.2	34.7
4	June 1–2, 2014	Matson	Seattle, WA to Oakland, CA	Manoa	Forward	262.2	34.7
5	June 5, 2014	Matson	Oakland, CA to Long Beach, CA	Mokihana	Forward	262.2	34.7

reticle compass binoculars. Species identification and behavior determination was aided with the help of binoculars. The sighting position and distance from ship of the whales was estimated for 5 of the 6 voyages, using the ship's GPS position, reticle reading of the whale's position below the horizon, angle off of ship's bow in 10° increments and height above the water. Group size and behavior if known was also recorded for any marine mammal. Data were collected both on an app based platform, SpotterPro, on a tablet (to test and demonstrate such a system to the crew) as well as on datasheets or a laptop. During observational watches, engagement with the crew offered informal discussions and educational opportunities for both crew and observer.

RESULTS AND DISCUSSION

Sighting Locations

A total of 42 large whale sightings of an estimated 57 individuals were made during 68 h and 1,387 nm of observation effort during five voyages spanning 8 days as ships transited between U.S. West Coast ports (**Tables 2**, **3**). While 60% of large whale sightings were of unconfirmed species identity (species sometimes too distant even with the binoculars to identify), those that were identified included humpback, blue, fin and beaked whales (**Table 3**). Large whales were sighted on all five voyages representing six of the 8 days observations were made. Two days without sightings were the shortest days (2 and 5.5 h of observation). Even though sighting conditions varied widely and included some very poor weather (45 + knots of wind at one time and 25 foot seas) the bridge height above the water still provided good visibility. In addition to large whales there were also sightings of pinnipeds and small cetaceans.

The majority (76%, 32 out of 42) of the large whale sightings were on coastal ship transit routes along the coast outside the designated traffic separation lanes leading to major ports (**Figure 2** and **Table 3**). Average ship speed while traveling coastally and outside traffic separation schemes ranged from 18 to 23 kts while inside the designated shipping lanes speed was slower, 11–18 kts (**Table 2**).There were six close encounters with large whales on five of the six trips with four of these outside the shipping lanes and two in the lanes. A sighting was deemed a close encounter if the whale was estimated to be less than 300 m from the vessel. One of these (outside the lanes) was considered a near miss (estimated at 40 m) of two humpback whales.

Our results demonstrate the potential large role that ships traveling along the coast may play in ship strikes and how most of the risk for these ships may be outside of the designated shipping lanes near ports (versus ships that head offshore toward principal destinations in Asia after leaving the shipping lanes). Most of the focus of management efforts on the designated shipping lanes around major ports (San Francisco and Los Angeles/Long Beach). Our findings are consistent with encounter models of ships and whales that demonstrated a majority of strikes likely occur outside the designated traffic separation lanes even though the rate may be highest in the lanes (Rockwood et al., 2017).

Experience Working on Vessels and Feasibility for Future Use

There was a high degree of cooperation from ship's personnel to the observer. Once introduced to and educated on the topic of whale strikes and explanations of the presence of a marine mammal observer not being a regulatory professional, all ships personnel on the bridge assisted and aided in the sightings, even when the observer was off effort. This cooperation came where there was initial skepticism by some crew on sighting whales with several crew saying they had never seen a whale while on watch on the bridge and were surprised at the amount of sightings detected on our trip, while others indicated that they see whales often and maneuver around them periodically. During down times on the bridge, discussions with crew included the feasibility of their recording sightings without a dedicated marine mammal observer and the best ways to achieve this. There was a repeated suggestion by crew that this could best be achieved by integrating marine mammal observations and recording into existing reports they already were conducting. Even though a poster to help identify whales and report sightings had been circulated to shipping companies operating out of LA/LB, it was not present on most vessels and in the cases where present crew seemed unfamiliar with it.

On two occasions, the ship's crew took initiative to alter course to avoid whales informing the observer of their intent. One of these involved some distant sightings in the ships path that prompted a small course alteration so they were no longer dead ahead and the other, the mate employed hand steering to avoid a closer sighting of a whale. Though not requested by the observer, these responses could have been influenced by our presence. Observations of whales directly ahead on board the vessels with a mid-aft bride were more difficult since the bow was over 180 m ahead of the observer position and containers blocked the view immediately in front of the vessel as seen in **Figure 1**.

This pilot study was more successful than anticipated both verifying the value of ships as an observation platform and

TABLE 2 | Dates, region, and on-effort length, sighting conditions, and speed of ship ride-alongs.

Date (Voyage)	Region	Effort (h)	Beaufort	Visibility (nm)	Total on-effort track length (nm)	Speed outside shipping lanes (kts)	Speed inside shipping lanes (kts)
May 29, 2013 (1)	Southern California	2	3–4	10–15	15.8	na	10–13
May 30, 2013 (1)	Central California	8.5	4–6	5–15	168.5	20–24	10–14
August 1, 2013 (2)	S. Central California	12.5	1–6	3–15	225.4	15–18	12–17
September 26, 2013 (3)	Central California	7.5	5	15–20	101.6	18–23	na
September 27, 2013 (3)	Southern California	5.5	1	3–10	97.1	18	10–12
June 1, 2014 (4)	N. WA to Central OR	13	1–4	5–20	355	22–25	22–24
June 2, 2014 (4)	Central OR to Central California	7.5	4	0.5–10	172.2	17–25	14–22
June 5, 2015 (5)	Central to S. California	12	2	0.25–5	251.9	14–25	11–18

TABLE 3 | Numbers of sightings and individuals of large whales, inside and outside shipping channels.

Voyage	Large whales	Large whale outside shipping channels	Large whale inside shipping channels	Humpbacks	Blue whales	Fin whales	Cuvier's	UnID'd whales
1	8 (9)	7 (8)	1 (1)	2 (3)				6 (6)
2	5 (6)	5 (6)		1 (2)				4 (4)
3	3 (4)	3 (4)			1 (2)			2 (2)
4	13 (18)	7 (11)	6 (7)	2 (4)	4 (5)			7 (9)
5	13 (20)	10 (15)	3 (5)		1 (2)	1 (1)	1 (3)	10 (14)

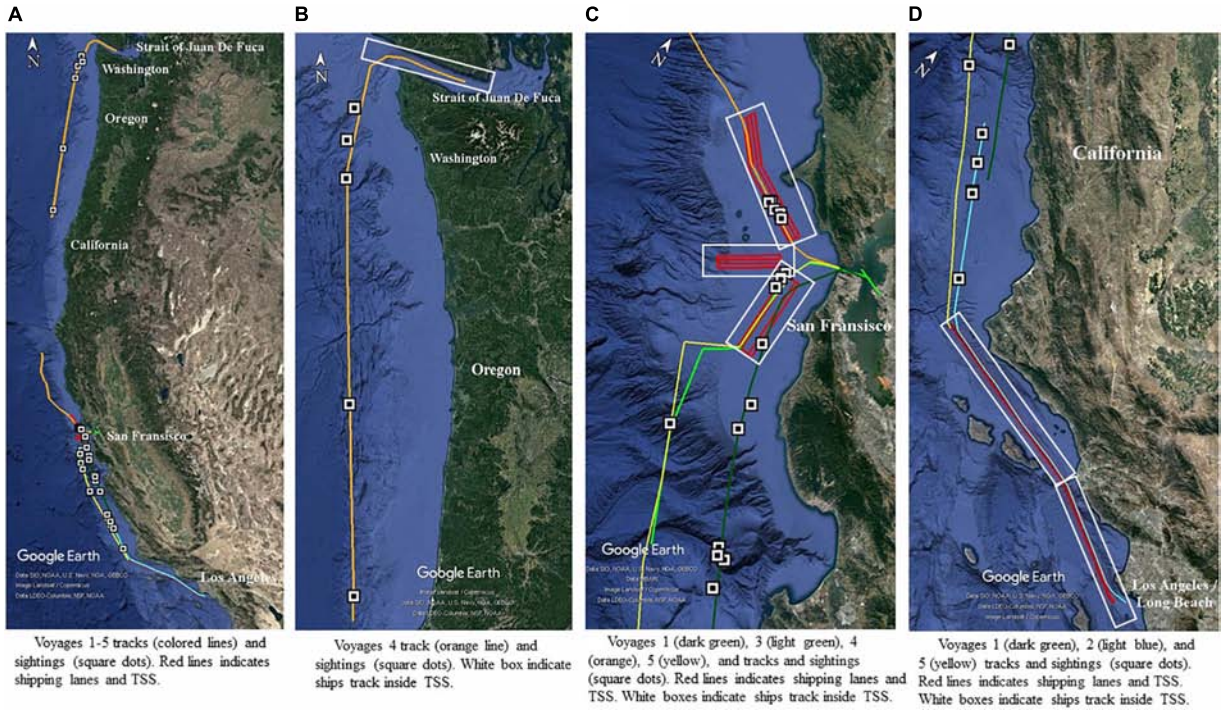

FIGURE 2 | Tracks and sighting locations of voyages 1–5. **(A)** shows an overview of all ships tracks and sightings and **(B–D)** show detail of specific areas including shipping lanes shown in red and region outlined in white.

showing the frequency of encounters of large whales for vessels transiting along the coast. This is consistent with findings in other areas regarding the value of dedicated marine mammal observers placed on board ferries and cruise ships (Gende et al., 2011; Harris et al., 2012; Williams et al., 2016). Weinrich et al. (2010) demonstrated that observers aboard fast ferries detected whales faster and at larger distances than the crew. Commercial vessels bridge height above the sea surface, the wide field of view, all designed for safe operations and the ships stability, make an excellent marine mammal viewing platform. While our

visual observations were limited to daylight conditions, other approaches (e.g., infrared) could be used to expand this to nighttime periods when whales are closer to the surface and more vulnerable to ship strikes (Calambokidis et al., in press).

CONCLUSION

Our findings demonstrated:

(1) the value of ships as an observation platform along the U.S. West Coast, providing real time data on whale occurrence along ship transit routes,
(2) the important role ships that transit along the coast between U.S. ports may play in ship strikes,
(3) the potential for ship self-reporting especially if done in conjunction with better reporting methods and training by experienced marine mammal observers accompanying some trips,
(4) the potential ability of ships to avoid whale concentrations if spotted early enough.

These findings have resulted in plans to conduct future trips with observers to improve sample size (planned for 2019–2021) with an ultimately goal of having ship personnel conduct their own observations and reporting.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

Both authors conceived and designed the study, revised the manuscript, and read and approved the submitted version of the manuscript. KF collected the data and wrote the first draft of the manuscript. JC designed the data analysis.

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Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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frontiers in Marine Science

ORIGINAL RESEARCH
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The Role of Slower Vessel Speeds in Reducing Greenhouse Gas Emissions, Underwater Noise and Collision Risk to Whales

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Reducing speeds across shipping fleets has been shown to make a substantial contribution to effective short term measures for reducing greenhouse gas (GHG) emissions, working toward the goal adopted by the International Maritime Organization (IMO) in April 2018 to reduce the total annual GHG emission by at least 50% by 2050 compared to 2008. I review modeling work on GHG emissions and also on the relationships between underwater noise, whale collision risk and speed. I examine different speed reduction scenarios that would contribute to GHG reduction targets, and the other environmental benefits of reduced underwater noise and risk of collisions with marine life. A modest 10% speed reduction across the global fleet has been estimated to reduce overall GHG emissions by around 13% (Faber et al., 2017) and improve the probability of meeting GHG targets by 23% (Comer et al., 2018). I conclude that such a 10% speed reduction, could reduce the total sound energy from shipping by around 40%. The associated reduction in overall ship strike risk has higher uncertainty but could be around 50%. This would benefit whale populations globally and complement current efforts to reduce collision risk in identified high risk areas through small changes in routing.

Keywords: speed, GHG, emissions, underwater noise, ship strike

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INTRODUCTION

In April 2018, the International Maritime Organization (IMO) adopted an initial strategy on the reduction of greenhouse gas (GHG) emissions from ships. The target is to reduce the total annual GHG emissions by at least 50% by 2050 compared to 2008 values, while at the same time pursuing efforts toward phasing them out entirely. The strategy lists candidate short term measures to reduce GHG emissions in order to meet the agreed targets. IMO has developed the Energy Efficiency Design Index (EEDI) which requires that new ships become increasingly more energy efficient according to the year in which they were built. This is the only legally binding energy efficiency regulation for international shipping and only applies to new builds. This means that in order to reach the IMO 2050 target, measures for improving operational efficiency of existing ships will also be required. One element of the strategy is to "consider and analyze the use of speed optimization and speed reduction as a measure, taking into account safety issues, distance traveled, distortion of the market or to trade and that such measure does not impact on shipping's capability to serve remote geographic areas" (IMO, 2018). It has been suggested that speed reduction is perhaps the

only short-term regulatory option capable of achieving the necessary reductions in GHG emissions to meet IMO targets (CSC, 2017).

Here, I consider the implications of speed optimization and reduction not only for GHG emissions but for two further environmental impacts of shipping: ship strike risk to whales and underwater noise. Collisions between cetaceans and ships occur worldwide where vessel activities overlap with cetacean habitat. Collisions can cause damage to vessels and lead to injury and/or death of cetaceans. In response to this threat, the IMO issued guidance on minimizing the risk of ship strikes to cetaceans (IMO, 2009) in 2009. The impacts of underwater noise from shipping have also become increasingly recognized. IMO agreed to guidelines for the reduction of underwater noise from commercial shipping to address adverse impacts on marine life in 2014 (IMO, 2014).

I examine the implications of changes in vessel speeds at a global level, which is the scale most relevant to GHG emissions; however, ship strike risk and the impacts of underwater noise will depend on the spatial overlap between shipping and the distribution of sensitive species. The International Whaling Commission (IWC) has concluded that the only proven, effective mitigation measures to reduce ship strikes to whales are to avoid areas with known concentrations of whales, or reduce speed while transiting those areas (IWC, 2016). The IMO has recognized that small changes in routing are the most effective way to reduce ship strikes in identified high density whale areas (IMO, 2016). IWC (2016) lists the priority "high risk" areas identified by the IWC, and ongoing research will likely identify more of such areas. In some cases, speed restrictions have been put in place in specific areas where routing options are not possible (e.g., Seasonal Management Areas (SMAs)) on the east coast of United States for North Atlantic right whales; Hauraki Gulf in New Zealand for Bryde's whales; approaches to the Panama Canal in conjunction with routing for humpback whales (IWC, 2016). However, most whale populations are widely dispersed, and distribution patterns are not predictable enough to allow routing measures. These situations would benefit from more general risk-reduction measures. The greatest impacts from underwater noise will also occur where there is most overlap between shipping and marine species that are particularly sensitive to underwater noise, but shipping has raised ambient noise levels across ocean basins (McDonald et al., 2006; Andrew et al., 2011; Miksis-Olds et al., 2013). Hence, most marine life will be affected to some extent and hence benefit from global measures that reduce noise output.

From a ship strike and noise perspective, speed optimization to minimize impacts could take into account the distribution of species known to be vulnerable to ship strikes or to be particularly sensitive to shipping noise, and be adjusted accordingly. However, this relies on data that are frequently not available, is operationally complex and also may conflict with optimization for other purposes including minimizing GHG emissions. For example, Doudnikoff and Lacoste (2014) showed that CO_2 emissions will increase if ships slow down in certain areas and then increase speeds to compensate for the longer sailing time. Hence, the focus in this paper is on a simple assessment of global speed reductions where speeds are optimized with respect to total GHG emissions which will undoubtedly be the overriding concern from a legal and policy perspective. In contrast to mandatory energy efficiency requirements, guidelines related to ship strikes and underwater noise are entirely voluntary. Hence, I have taken proposals to reduce GHGs as a starting point and then examined the impacts of these for ship strike risk and underwater noise.

The 73rd meeting of the IMO Marine Environment Protection Committee (MEPC) in October 2018 considered speed reductions as a proportion of "business as usual" (BAU) speeds. This allows for the requirements of different shipping sectors with different operating requirements and vessel speeds. Vessel speeds will vary with ship type, with container ships and vehicle carriers generally having the highest speeds compared to oil tankers, bulkers, and general cargo (e.g., Bassett et al., 2012).

Faber et al. (2017) considered a range of speed reductions of 10, 20, and 30% compared to "business as usual" as well as the spare capacity within sectors to allow for the same volume of goods if more ships were required because of slower speeds. They note that in 2017, 3.5% of container vessels were idle or laid up and estimated that bringing these vessels back into service would allow the container fleet to reduce speeds by up to 8%. The equivalent figures for bulk carriers and tankers are 3 and 22%, respectively. Lee et al. (2015) developed an economic model which indicated that the savings in total fuel consumption associated with slower speeds were usually higher than the cost of operating the extra vessels required to transport equivalent goods. The growth in global fleet tonnage in 2017 was around 3.3%, and for the first time in recent years the expansion in ship supply capacity was surpassed by faster growth in demand and seaborne trade volumes (United Nations Conference on Trade and Development [UNCTAD], 2018). Thus, total excess capacity reduced slightly in 2017 across the global fleet. Speed reductions of greater than 10% would likely require a further increase in fleet capacity to meet current demand, but historic delivery rates of new vessels suggest that increases in fleet capacity could allow speed reductions as high as 20% or 30% for most ship types (Faber et al., 2017).

MATERIALS AND METHODS

The first stage of analysis was to examine data on the current distribution of observed vessel speeds. These speed distributions were adjusted according to a number of possible future scenarios aimed at GHG reductions. The observed and adjusted speed distributions were then used to estimate the expected change in ship strike risk and underwater noise output associated with each scenario.

Assessment of Current Distribution of Vessel Speeds

Current vessel speeds were examined from Automatic Identification System (AIS) data. I examined a year of data from two areas, the main shipping lane across the Indian Ocean south of Sri Lanka as representative of long distance oceanic traffic, and a coastal area west of Greece in the Mediterranean as

more representative of coastal traffic. These areas were chosen because of the availability and previous analyses of AIS data (Frantzis et al., 2014; Priyadarshana et al., 2016). For the Indian Ocean shipping lane, data were available from 2013/2014 to 2017/2018 for comparison. Vessel speeds were assessed within a period of 1 year for all vessels which crossed a line perpendicular to the shipping route between two waypoints at either side of the main route.

GHG Reduction Scenarios

Slow steaming practices introduced after a slowdown in global trade in 2008 prompted a number of studies of the economic implications and potential for GHG reductions (Cariou, 2011; Lindstad et al., 2011; Lee et al., 2015). Lindstad et al. (2011) found that emissions could be reduced by 19% if speeds were reduced to minimize costs and by 28% if speeds were further reduced but with no increase in costs. Lee et al. (2015) developed a model to quantify the relationship among shipping time, bunker cost and delivery reliability noting that delivery time reliability was an additional advantage of slow steaming. More recently, Mander (2017) found that additional policy measures might be required to ensure slow steaming persisted in the longer term. Between 2013 and 2015 there has been an increase in speed for some of the largest ships (ICCT, 2017).

Faber et al. (2017) estimated reductions in CO_2 emissions for speed reductions of 10, 20, and 30% across the global fleet based on the assumptions that a ship's main engine energy consumption per unit of time has a cubic relationship with its speed and that the efficiency of the auxiliary engines is not affected by speed reduction. They also allowed for the increase in the number of vessels in order to transport the same amount of cargo. However, their estimates did not include the CO_2 emissions associated with an increase in ship construction due to demand for more vessels as a result of slower speeds. Previous work (Faber et al., 2012) found the emissions associated with such ship building to be sufficiently small (in the range from 4 to 6% of the emission reductions achieved by slow steaming) that they would not make an appreciable contribution to the estimates. Reduced port call times associated with increases in cargo handling efficiency can also allow for slower speeds for the same amount of cargo transported.

Comer et al. (2018) used the same proportional values of speed reduction (10, 20, 30%) as Faber et al. (2017) but combined these with estimates assuming different scenarios for timescales of new build, technical efficiency improvements and low-carbon fuel introduction. They then used a Monte-Carlo simulation to estimate the probability of meeting the IMO targets.

Ship Strike Risk

Speed reduction has been used as a measure to address ship strike risk in a number of locations (Silber et al., 2012). Speed restrictions to reduce ship strike risks to North Atlantic right whales were first introduced in SMAs off the east coast of the United States in 2008. In the 5 years after the enactment of mandatory 10 knot speed restrictions in several SMAs there were no right whale mortalities attributed to ship strikes either in, or close to these areas. These results indicate a statistically significant reduction in right whale ship strikes in the SMAs suggesting that the speed limits have been effective (Laist et al., 2014). A number of recent studies have also confirmed an increased ship strike risk with increased speed, supporting the use of speed restrictions as a way of reducing risk. Some studies have attempted to quantify the speed-risk relationship for specific whale species (Conn and Silber, 2013) or the hydrodynamic and impact forces in relation to speed (Silber et al., 2010). Others (e.g., Wiley et al., 2011; Chion et al., 2018) have evaluated the relative risk reduction that might be achieved by speed restrictions based on these speed-risk relationships. In addition to studies based on collisions, studies based on observations of whales close to vessels have inferred greater collision risks with increases in speed (Gende et al., 2011; Harris et al., 2012). However, there are still limited data to quantify the relationship between strike rates and vessel speed.

The probability of a fatal ship strike can be expressed as the probability that a strike will occur multiplied by the probability that it will ultimately be fatal (i.e., death or serious injury) given that it has occurred.

The relationship between these probabilities and vessel speed has been studied in most detail for North Atlantic right whales. Vanderlaan and Taggart (2007) estimated the probability of lethal injury with vessel speed at the time of impact (M_v), which was later updated by Conn and Silber (2013) with additional data. In that case M_v for speed v (in knots) was expressed as:

$$M_v = \frac{\exp(\beta_0 + \beta_1 v)}{\exp(\beta_0 + \beta_1 v) + 1} \quad (1)$$

where β_0 was estimated as -1.905 (SE = 0.821) and β_1 as 0.217 (SE = 0.058).

Conn and Silber (2013) also estimated the relative instantaneous strike rate with speed. They expressed this in the form:

$$\log(\lambda) = \alpha_0 + \alpha_1 v \quad (2)$$

where λ is the instantaneous rate at which whales are struck. It was not possible to estimate α_0 (which would have allowed an absolute estimate of strike rate), but α_1 was estimated as 0.49 (SE = 0.09), giving a relative estimate of strike rate with speed. There was insufficient evidence to support a more complex quadratic effect with an additional parameter. Thus the formulation generates an exponential increase in strike rate with speed which becomes unrealistic at high speeds. In the analysis, 99% of observed ship speeds were 20.5 knots or below (P. Conn, personal communication). For the purposes of this study, and to avoid the estimates of risk being dominated by a small number of very fast vessels, I assume λ to be constant for speeds greater than 20 knots.

Conn and Silber (2013) then derived an expression for an index of the total mortality hazard based on the sum of the independent relative hazards associated with each transit through an area. The relative hazard for each individual transit is expressed as $\lambda_v M_v D_v$ where D_v is the duration of the transit for vessel speed v. Thus D_v is proportion to $1/v$.

In this case for a fixed number of vessel transits globally (i.e., a fixed amount of cargo transported) the equivalent global relative hazard (H_v) can be written as:

$$H_v = \sum \frac{\lambda_v M_v}{v} \quad (3)$$

The dominant factor affecting the variance of estimates of H_v is uncertainty in λ. At 15 knots, the difference in λ between $\alpha_1 \pm$ one standard error (i.e., 0.40 or 0.58) is a factor of over 200. The 95% credibility interval for M_v is relatively narrow in comparison (see Conn and Silber, 2013, figure 3).

Thus, any estimates based on H_v need to be treated with caution. In addition these estimates were only made on the basis of strikes to North Atlantic right whales and may not be directly applicable to other species or populations. There is no reason to expect large differences between species in the severity of injury with speed in the event that a strike occurs, but the relationship between speed and strike rate is more likely to vary between species due to different responses to vessels, swimming speeds and ability to maneuver. However, these differences are difficult to predict. For the purposes of this study I use the estimates of Conn and Silber (2013) for North Atlantic right whales as indicative for all large whales, but note that the IWC Scientific Committee has identified the need for a better understanding of the relationship between vessel speed, the risk of death or injury to the whale and damage to the vessel (IWC, 2016).

Underwater Noise

Leaper et al. (2014) reviewed known data on the relationship between vessel speed and broadband source level and concluded that the power relationship suggested by Ross (1976), which was based on vessel noise measurements and cavitation experiments, was the most widely applicable. However, considerably more data have become available since that review.

Some studies have fitted a power relationship to empirical data to estimate the relationship between broadband source level and vessel speed. The difference in source level (ΔSL) can be expressed in terms of original speed v_0, final speed v_1, and estimated power exponent z by:

$$\Delta \mathrm{SL} = 10z\log(v_1) - 10z\log(v_0) = 10z\log\left(\frac{v_1}{v_0}\right) \quad (4)$$

which just depends on the ratio v_1/v_0 and not on the original speed.

A recent study resulting in a large number of suitable measurements was associated with the voluntary slow down program initiated by the Vancouver Fraser Port Authority as part of the Enhancing Cetacean Habitat and Observation (ECHO) program. As part of this program MacGillivray and Li (2018) obtained estimates of z by ship type from a total of 2765 source level measurements including before and after the slow down trial. For broadband monopole source levels, estimates of z varied from 5.1 (containerships and vehicle carriers) to 8.1 (bulkers). Other estimates of z are 4.5 from data in Allen et al. (2012) from fishing vessels and a model from Wittekind (2014) which suggests $z = 8$ for low frequency propeller noise above cavitation inception speed.

Others (McKenna et al., 2013; Simard et al., 2016; Veirs et al., 2016; Gassmann et al., 2017) have estimated the slope m of a linear regression where:

$$\Delta \mathrm{SL} = m(v_1 - v_0) \quad (5)$$

In this case ΔSL will depend on the actual value of the speeds as well as the ratio. Estimated values of m have ranged from 0.93 (Veirs et al., 2016) to 2.38 Gassmann et al. (2017). Veirs et al. (2016) used a linear regression on a large data set to obtain a slope of 0.93 dB/knot for broadband source level with speed, but they note that most of the variation in SL is likely driven by ship class (which was not controlled for in the regression), with little change in speed within ship class.

All these relationships of noise with speed only apply to vessels with fixed pitch propellers. Substantial cavitation can occur on controllable pitch propellers when operating at slower speeds resulting in higher noise levels. However, vessels with controllable pitch propellers are only a very small proportion of the global fleet (e.g., tugs, ferries).

Leaper et al. (2014) define the acoustic footprint of a vessel as the area of sea for which the source level will be above a given value (which can be defined in terms of energy or pressure). For situations of spherical spreading (20 logR) propagation loss, the ratio A_1/A_0 of acoustic footprint associated with a change in source level of ΔSL dB is given by:

$$A_1/A_0 = 10^{\left(\frac{\Delta \mathrm{SL}}{10}\right)} \quad (6)$$

where A_0 is the original acoustic footprint for SL_0 and A_1 is the footprint associated with SL_1 where $\mathrm{SL}_1 = \mathrm{SL}_0 + \Delta\ \mathrm{SL}$.

The ratio of acoustic footprints in this case is also the same as the ratio of total sound energy. For slower vessels and longer passage times there will need to be more vessels at sea to carry the equivalent amount of cargo. If all vessels travel at a fraction k of their former speed (i.e., $k = v_1/v_0$) then the number of vessels, and the associated acoustic footprints, need to be multiplied by $1/k$ for the equivalent cargo carried.

For the purposes of this analysis, I summarize the effects of changes in vessel speed in terms of the ratio of sound energy for equivalent cargo carried. The assumption of spherical spreading loss also seems the most appropriate general approximation to apply at a global scale. In different situations propagation loss may be more or less than 20 logR (see Ainslie et al., 2014). Thus, the ratios given can also be visualized in terms of acoustic footprint which will also be roughly proportional to the number of animals affected.

RESULTS

Vessel Speeds

Vessel speeds vary between areas and routes depending on the nature of the traffic. The Indian Ocean route south of Sri Lanka is typical of long distance routes, whereas coastal traffic is more variable. In most cases the distribution is bimodal. This is more pronounced in the coastal traffic example (**Figure 1**), but the

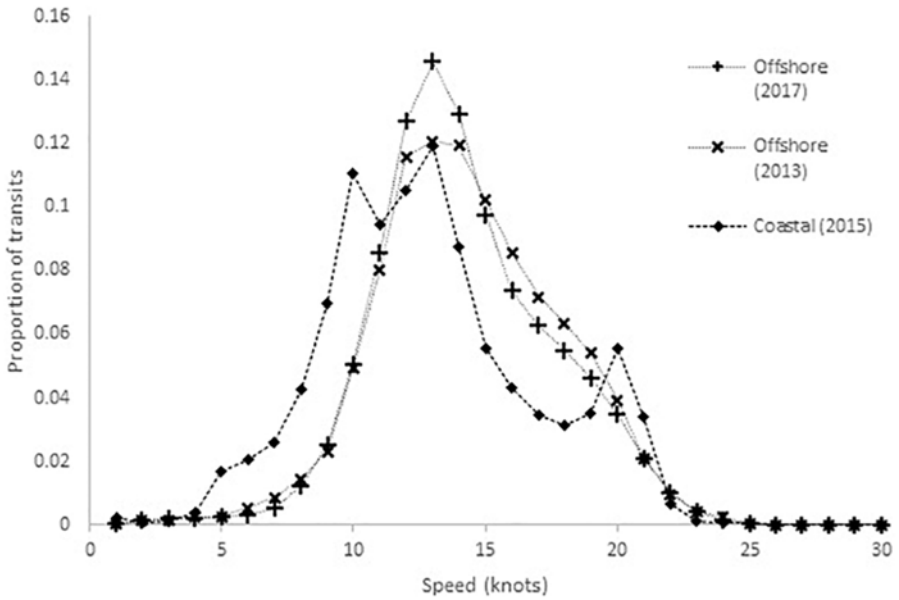

FIGURE 1 | Proportion of transits by speed for offshore traffic (crossing Indian Ocean south of Sri Lanka) and coastal traffic in the eastern Ionian, Mediterranean.

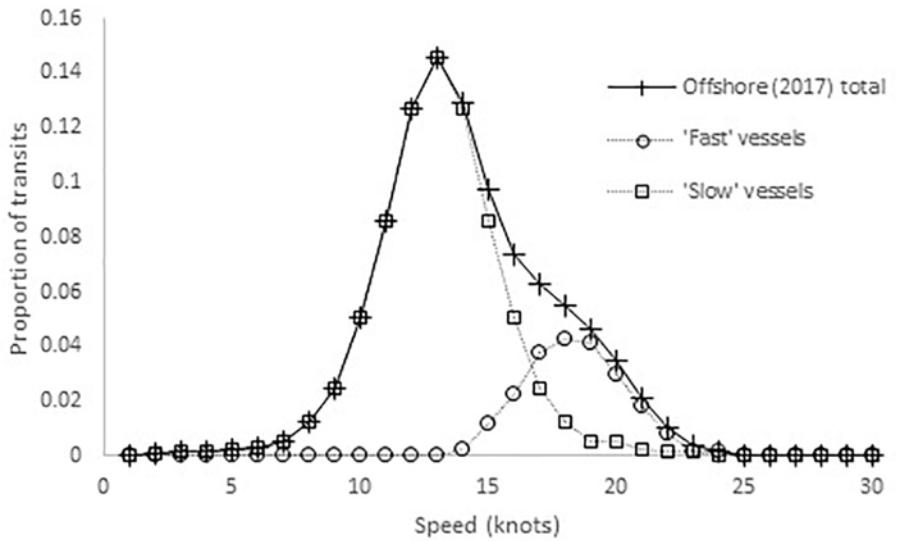

FIGURE 2 | Distribution of offshore vessel speeds in 2017 shown as the sum of two approximately symmetrical distributions representing "fast" and "slow" vessels.

offshore traffic can also be summarized as the sum of two symmetrical approximately normal distributions of "slow" and "fast" vessels (**Figure 2**). In this case the fast vessel category is dominated by container ships. Median speed for slow vessels was around 13 knots and for the fast vessel category, 18 knots. Median speed for large container ships of the major carriers was 18.4 knots. There were only small changes in median speeds between 2013 (13.7 knots) and 2017 (13.3 knots), with a smaller proportion of vessels in the 16–20 knot range in 2017 and an increase in the 12–14 knot proportion. There were 2308 transits where the same vessel (based on MMSI number) transited at least once in both 2013 and 2017. No significant difference was detected in the speeds of those vessels between years (ANOVA, $p = 0.61$).

The peak in slower vessels in the Mediterranean is dominated by recreational craft, many of which now voluntarily transmit AIS signals.

Ship Strike Risk

Ten knots has been considered as the speed at which strike risk has dropped to low levels, supported by the reduction in ship strike cases following the introduction of SMAs on the east

TABLE 1 | Changes in parameters for scenarios of 10–30% speed reductions across the global fleet adjusted for the same total cargo carried.

Reduction in speed compared to "business as usual" (BAU)	10%	20%	30%
CO_2 emissions			
Theoretical proportion of CO_2 emissions compared to BAU for alternative speed regimes based on simple quadratic relationship	0.81	0.64	0.49
Estimated proportion of CO_2 emissions compared to BAU for alternative speed regimes by Faber et al. (2017)	0.87	0.76	0.67
Probability of meeting 2050 target (>50% reduction) without accelerating the pace of new build technical efficiency (results from Comer et al., 2018)	0.23	0.32	0.42
Ship strike risk			
M_v = probability of lethal injury with vessel speed at the time of impact; H_v = global relative ship strike hazard			
Proportion of transits <10 knots	0.20	0.38	0.61
Ratio of M_v with speed reduction compared to business as usual, Oceanic shipping traffic (all vessels)	0.93	0.84	0.75
Ratio of H_v with speed reduction compared to business as usual, Oceanic shipping traffic (all vessels)	0.50	0.22	0.10
Ratio of H_v with speed reduction compared to business as usual, Oceanic shipping traffic (large container ships only)	0.52	0.22	0.09
Ratio of H_v with speed reduction compared to business as usual, Coastal shipping traffic (all vessels)	0.48	0.21	0.09
Underwater sound			
60 log (v) (independent of speed); ratio of total acoustic footprints compared to BAU	0.59	0.33	0.17
1 dB/knot (ratio of total sum of acoustic footprints compared to BAU based on observed distribution of all offshore vessel speeds)	0.74	0.54	0.40
1 dB/knot (ratio of total sum of acoustic footprints compared to BAU based on observed distribution of offshore container vessel speeds)	0.65	0.43	0.28
2.38 dB/knot (ratio of total sum of acoustic footprints compared to BAU based on observed distribution of offshore container vessel speeds)	0.36	0.13	0.05

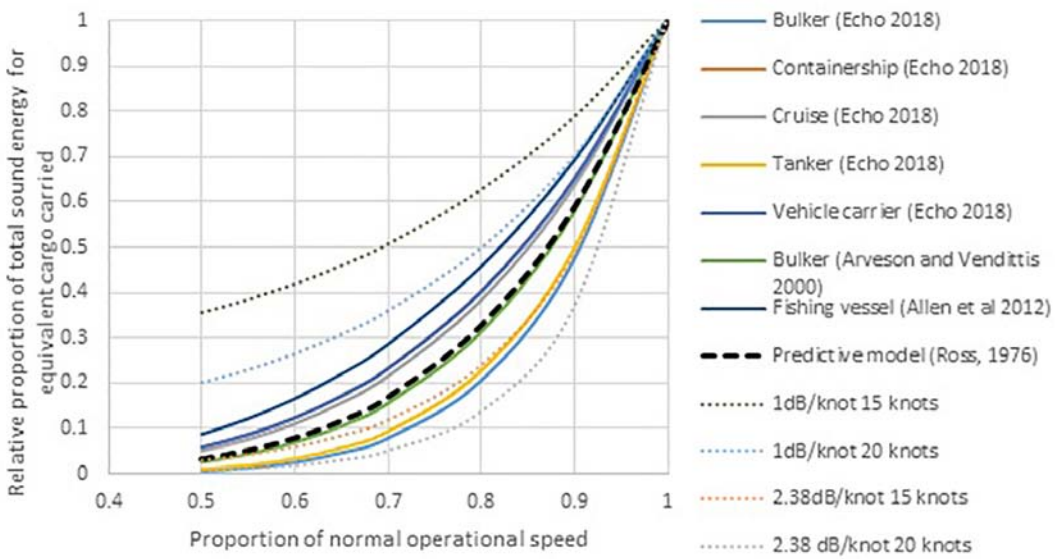

FIGURE 3 | Relative proportion of total sound energy [also equivalent to the relative total acoustic footprint as defined by Leaper et al. (2014)] from a number of studies. Dotted lines indicate relationships that are dependent on the initial speed. The dashed line indicates z = 6 in Eq. 4.

coast of the US (Laist et al., 2014). Currently, only a very small proportion (<10%) of transits occur at speeds of less than 10 knots for the oceanic traffic. This would increase to around 60% if all vessels slowed by 30% (**Table 1**).

The estimates of risk reduction associated with speed reductions of 10, 20, and 30% are shown in **Table 1**. These results are most sensitive to the assumptions about the relationship between strike rate and speed, for which there is the most uncertainty and also likely considerable variation between species. The estimates were all relatively insensitive to the overall distribution of speeds for the sector of the fleet being considered. For example, values of H_v for a speed reduction of 10% were between 0.48 and 0.52.

Underwater Noise

Studies that estimated relationships between source level and speed were divided into two categories: those of the form in Eq. 4 (a simple power relationship) and those of the form in Eq. 5 (a linear regression of source level expressed in dB on speed). These are shown in **Figure 3** for reference speeds of 15 and

20 knots for the relationships dependent on the original speed. For the case of a simple power relationship in Eq. 4 between source level and vessel speed, the ratio of sound energy associated with a proportional reduction in speed will be the same for all speeds and so can be estimated across the global fleet regardless of the original speed distribution.

It can be seen that the model of Ross (1976) with $z = 6$ in Eq. 4 falls in the middle of the more recent empirical studies. The estimates of global proportion of acoustic footprint associated with speed reductions are shown in **Table 1** for this model. A 1 dB/knot relationship would suggest a substantially lower reduction across the global fleet than the Ross model, but a closer level to the Ross model for just container vessels because of their higher speeds. **Table 1** also shows the reduction associated with the 2.38 dB/knot relationship found by Gassmann et al. (2017) for a specific class of container vessel. This shows the greatest reduction in acoustic footprint, down to 5% of the initial value for a speed reduction of 30%.

DISCUSSION

Several studies have shown that some form of speed reduction will be essential in the short-term if IMO targets on GHGs are to be met, and the IMO has identified speed reduction as a candidate short-term measure. I have attempted a simple quantification of the additional environmental benefits associated with slower speeds motivated by a reduction in GHGs of reduced ship strike risk to whales and underwater noise. These additional benefits further support the calls for effective measures to reduce speeds.

For ship strikes, there are many studies indicating a qualitative risk reduction with slower speeds but limited data are available to quantify the relationship. Only one study has attempted to do this and only for one species. There is therefore considerable uncertainty with these estimates. Nevertheless, the results indicate the potential for a 50% reduction in risk for a modest 10% reduction in global shipping speeds. The uncertainty in the risk reduction achieved means that, where it is possible to separate ships and whales by small changes in routing, this would still be the option most likely to be effective, as noted by IMO (2016).

In recent years, there has been a substantial increase in concern over the impacts of underwater noise and increased research effort including source characteristics of individual vessels (IWC, 2018). These recent studies have given a much more comprehensive assessment of the relationships between source levels and speed. Many of these studies have shown that for individual vessels, the relationship varies considerably with the characteristics of the vessel (e.g., Kellett et al., 2013; Putland et al., 2017), though consistently, slower speeds produce less noise in fixed pitch propellers. Estimates that could apply to the fleet as a whole also show a wide variation (as shown in **Figure 3**) but support the continued use of the model of Ross (1976) which has been used for some decades and falls in the middle of the more recent observations. This model indicates a 10% reduction in speed would cut global underwater sound energy from shipping by around 40%.

AUTHOR CONTRIBUTIONS

RL conceived the study, performed the analysis, and wrote the manuscript.

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frontiers in Marine Science

BRIEF RESEARCH REPORT
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Differential Vulnerability to Ship Strikes Between Day and Night for Blue, Fin, and Humpback Whales Based on Dive and Movement Data From Medium Duration Archival Tags

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We examine the dive and movement behavior of blue, fin, and humpback whales along the US West Coast in regions with high ship traffic where ship strikes have been identified as a major concern. All three species are known to feed in coastal waters near areas of high ship traffic. We analyzed data from 33 archival tag deployments representing over 3,000 h of data that were attached with suction-cups or short darts for periods >20 h and recorded depth (≥ 1 Hz), fast-lock GPS positions and other sensors. There were clear differences among the three species but all showed a distinct diurnal difference in diving behavior. While dive depth varied among animals based on where prey was located, whales spent a high proportion of their time closer to the surface where they would be more vulnerable to ship strikes at night than in the day. This was most pronounced for blue whales where vulnerability was twice as high at night compared to the day. We also found differences in movement patterns of whales between day and night. Movements were more localized to specific areas in the day near prey resources while at night these movements often involved directional movements (though sometimes returning to the same area). We show how in several specific areas like the Santa Barbara Channel, these differences in movements and locations translate to a very different overlap with shipping lanes at night compared to the daytime locations, which is the basis for most sighting data.

Keywords: ship strike, diel differences, whale behavior, movements, archival tags

INTRODUCTION

Ship strikes of larges whales have become a growing concern in many areas around the world (Panigada et al., 2006; Williams and O'Hara, 2010; Silber et al., 2012b). Along the US West Coast concern became more acute after several periods with elevated ship strikes. This included at least four fin whale ship strikes documented in the Pacific Northwest in 2002 (Douglas et al., 2008), and at least four blue whales documented struck by ships off southern California in Fall 2007 (Berman-Kowalewski et al., 2009). A number of species have been documented struck by ships along the US West Coast, and concern has focused on blue, fin, and humpback whales that often feed in coastal

waters, including in areas of high vessel traffic near the routes of ships coming and going from the major ports of Los Angeles/Long Beach, San Francisco Bay, and the Salish Sea (Calambokidis and Barlow, 2004; Calambokidis et al., 2004, 2015; Redfern et al., 2013; Douglas et al., 2014; Dransfield et al., 2014; Rockwood et al., 2017). While reported numbers of ship strikes have been of concern, these likely dramatically under-represent the true number of ship strikes occurring due to the low proportion of strikes documented or carcasses recovered (Williams et al., 2011; Rockwood et al., 2017).

Although this problem has been known for many years, solutions have proved challenging though some options have been put into place. Changes in shipping lanes have been successful in reducing overlap between areas of highest ship traffic and whale concentrations including in the major lanes off the US West Coast (Segee, 2010; Redfern et al., 2013). Vessel speed restrictions have been applied in a number of areas based on both the reduced lethality of strikes of ships going slower and the better potential for whales to avoid slower ships (Conn and Silber, 2013). Other strategies including voluntary ship slowdowns (McKenna et al., 2012) or use of acoustic alarms (Nowacek et al., 2004) have been shown to be ineffective or of limited use.

Scientific data on whale behavior and distribution has been important to evaluating strategies for reducing ship strikes but has also had some key limitations. Data on whale distribution including habitat models in most regions including the US West Coast has come primarily from sighting data based on surveys (Redfern et al., 2013; Becker et al., 2014; Calambokidis et al., 2015) which are based on daytime sighting data only. Some strategies that might help reduce ship strikes are only possible in daylight (avoidance based on visual sightings for example) and vulnerability of whales between day and night are important to understand for evaluation of mitigation strategies.

Data from tags attached to whales can provide new more detailed information on whale behavior and movements (Calambokidis et al., 2008; Johnson et al., 2009; Goldbogen et al., 2013a, 2014; Cade et al., 2016) including insights into diel differences in feeding behavior (Friedlaender et al., 2009). Tags attached to whales have provided important information on whale behavior in response to close approach of ships (McKenna et al., 2015), and alarm sounds to warn whales of ship approach (Nowacek et al., 2004), as well as other types of anthropogenic sounds like Navy sonar (Southall et al., 2012, 2019; Goldbogen et al., 2013b). Tags have also provided new information on whale distribution and movements including implications for ship strike risk (Irvine et al., 2014; Abrahms et al., 2019). Position and movement data from tags have faced some key limitations, however, with longer-term satellite tags not providing very frequent or accurate positions due to bandwidth limitations uploading data to satellites. Archival tags can record more frequent higher quality GPS positions but are limited to short durations due to the need to recover the tag and attachment limitations. These tradeoffs are beginning to be bridged with new tag developments including with archival tags making use of short darts to achieve longer duration attachment periods than could generally be achieved with suction cups (Szesciorka et al., 2016). The combination of longer duration, high resolution position information, and detailed dive behavior (especially as it related to behaviors like time near surface or reaction to ships) is needed to better assess vulnerability to ship strikes.

We use deployments from longer duration archival tags that fully sampled day and night periods to examine differences in day and night diving and movement of three baleen whale species in the eastern North Pacific and evaluate these differences in the context of risk of ship strikes.

METHODS

We have been conducting tag deployments in the eastern North Pacific along the US West Coast on blue, fin, and humpback whales using a variety of archival tags since the 1990s. For this study, we used only deployments that had at least 20 continuous hours of dive data along with high quality positions from an onboard GPS so that each tag obtained representative samples of both day and night behavior. Tags used in this study consisted of two primary tag designs:

- Wildlife Computers TDR10-F tag - Depth, Temperature, Light-level, Fastloc® GPS, and in most configurations with 3D accelerometers. The tag was modified to take a plate with darts and a satellite transmitter.
- Acousonde acoustic tag – miniature, self-contained, autonomous acoustic/ultrasonic recorder incorporating hydrophones as well as depth, attitude and orientation sensors, digital recording electronics, data storage, and battery with modified attachment plate with darts, satellite transmitter, Sirtrack FastGPS, and custom-made syntactic floatation.

Tag deployments were conducted from small 6–7 m Rigid Hull Inflatable Boats (RHIBs) equipped with a bow pulpit for a tagger to stand and use a 3–4 m pole to attach tags. Tags were attached with 3–4 stainless steel darts 4–6 cm long equipped with 1–2 rows of petals (Szesciorka et al., 2016). Tags were recovered after they detached from the animal and floated to the surface with the aid of an Argos satellite transmitter, which provided rough position as well as when tags had detached from the whales (based on the number and quality of positions) and a VHF transmitter that was used to localize on tags with a directional antenna. Tagging procedures were conducted under authority of a scientific research permit under the Marine Mammal Protection Act and Endangered Species Act and procedures were reviewed and approved by an Institutional Animal Care and Use Committee in conformance with the Animal Welfare Act.

Tags were deployed in a number of locations along the US West Coast with most frequent deployments in the: (1) Southern California Bight (primarily near the Palos Verdes Peninsula and in the Santa Barbara Channel) and (2) Gulf of the Farallones off San Francisco Bay in central/northern California (**Figure 1**). These are the areas of highest ship vessel traffic along the US West Coast corresponding to the routes to and from the ports of Los Angeles/Long Beach and those in San Francisco Bay

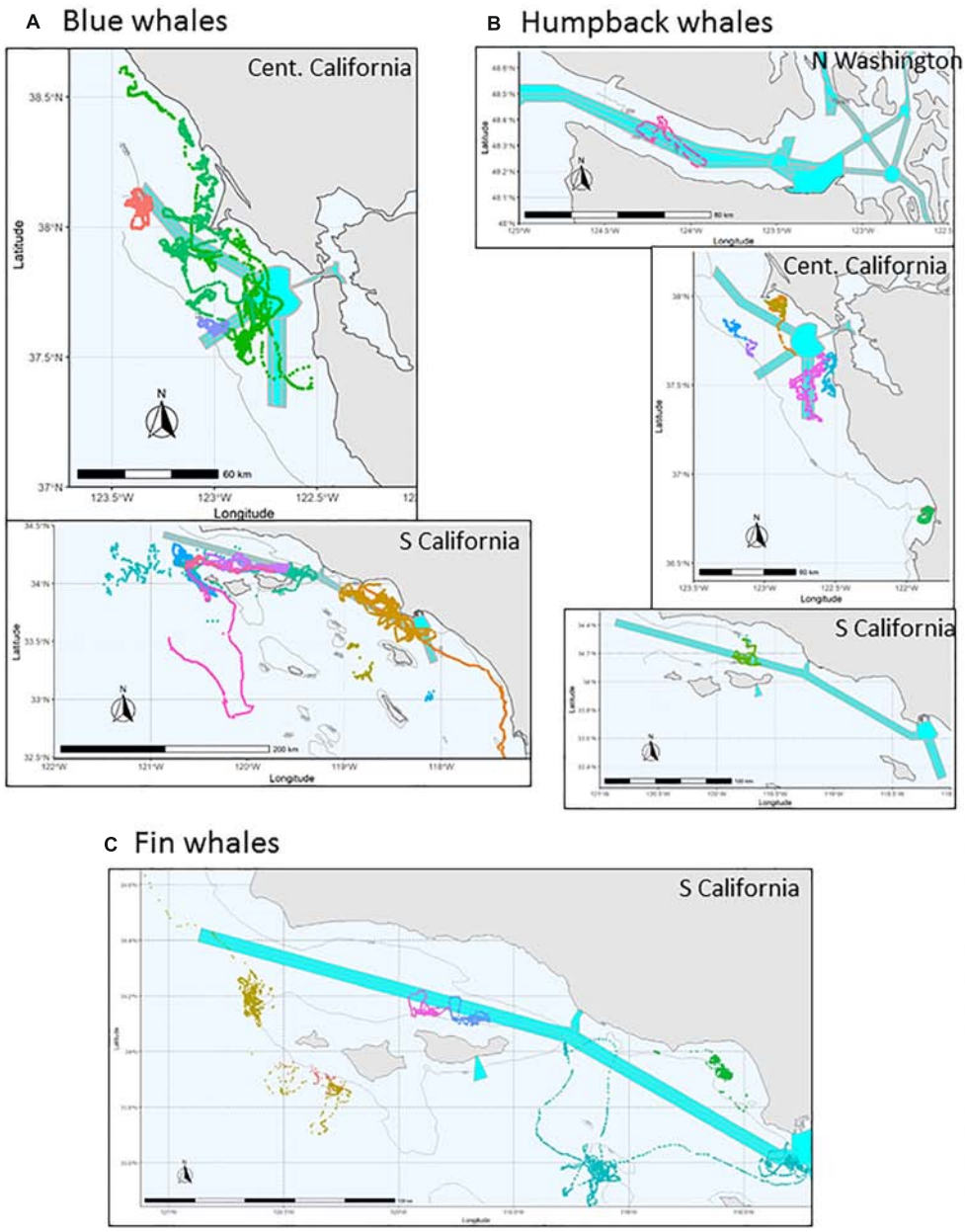

FIGURE 1 | Map of position tracks by species (color coded by deployment) for tag deployments used in this analysis. **(A)** Blue whale, **(B)** Humpback whale, and **(C)** Fin whale.

(Rockwood et al., 2017). For this analysis we focused on data gathered along the US West Coast between 32.5 and 48.5 N latitude (tag deployments on 7 additional blue whales that were tagged in this region but migrated south outside of it were not included in this analysis).

Data on whale diving behavior and movements were assigned a diel period category (Day, Night, Astronomical Dusk, Astronomical Dawn) for each 24-h cycle (Cycle) according to the time of day in relation to the sun angle, which was determined from the mean GPS location of the tagged animal for each Julian day. Day was considered sunrise to sunset, the crepuscular periods around dawn and dusk were calculated as the period between sunrise/sunset time and astronomical dawn/dusk (sun −18 degrees below horizon) as calculated by NOAA for that season and position and night was the period between the astronomical dusk and dawn.

For the geographic movement analysis, we used the GPS location data from either the integrated Fastloc GPS (TDR10 tags) or Sirtrack FastGPS on the piggy-backed GPS units (on Acousonde tags) to examine differences in movement patterns of all three whale species between day and night periods (**Table 3**). For the examination of whale movement differences during

day and night periods, we only used a period if there were continuous positions (periods were excluded if there were any gaps greater than 3 h). The remaining location data were re-discretized into a regular 15-min sampling rate using the AdehabitatLT package in R, version 0.3.23 (Calenge, 2006) for analysis. The 15-min sampling rate was selected to ensure movement metrics were not overly biased by differences in the tag determined frequency of locations which can vary by species, tag placement and animal behavior; observed intervals between positions averaged 8.9 (SD-9.3), 7.5 (SD-7.9), and 4.9 (SD-3.9) minutes and the 15-min cut-off encompassed 88, 93, and 98% of the intervals for blue, fin, and humpback whales, respectively. To determine to what degree an animal's daytime location corresponds to that of the nighttime, we calculated the geographic centroid for the day period of each 24-h cycle, then calculated the distance of each location in that cycle from the day-time centroid. We calculated a cumulative distribution of centroid distances for day and night for each individual as well as a mean cumulative distribution weighted by individual. For comparison among species, we examined proportion of positions within 2.5 and 10 km of that centroid position.

For the dive-depth analysis, we down-sampled all depth data to a common sampling rate for all tags (1 Hz) and rounded values to 0.1-m precision. All data points were assigned a period category (Day, Night, Astronomical Dusk, Astronomical Dawn). We calculated mean and standard deviation for depths for each period as well as a cumulative distribution of dive depths to examine what portion of time whales were near the surface and vulnerable to ship strikes (**Table 2**). For each species, we calculated a mean cumulative distribution of dive depths weighted by individual. We used nominal whale depths of 15

TABLE 1 | Summary of deployments used for this analysis by species.

				Hours data		
ID	**Date**	**Species**	**Region**	**Deployment**	**GPS data**	**Dive data**
Bm140719-TDR5	July 19, 2014	Blue whale	Central California	75	75	75
Bm140825-TDR5	August 25, 2014	Blue whale	Southern California	115	115	115
Bm140825-TDR6	August 25, 2014	Blue whale	Southern California	308	308	308
Bm150819-TDR5	August 19, 2015	Blue whale	Southern California	52	0	52
Bm151016-TDR5	October 16, 2015	Blue whale	Southern California	38	38	38
Bm160523-A20	May 23, 2016	Blue whale	Central California	236	236	98
Bm160523-TDR6	May 23, 2016	Blue whale	Central California	194	194	194
Bm160716-A20	July 16, 2016	Blue whale	Southern California	459	459	93
Bm160717-A21	July 17, 2016	Blue whale	Southern California	337	337	103
Bm160817-A21	August 17, 2016	Blue whale	Southern California	46	0	46
Bm160918-A08	September 18, 2016	Blue whale	Southern California	471	471	99
Bm160926-TDR7	September 26, 2016	Blue whale	Central California	22	22	22
Bm170622-TDR12	June 22, 2017	Blue whale	Southern California	425	425	425
Bm170622-TDR13	June 22, 2017	Blue whale	Southern California	118	0	118
Bm170925-TDR12	September 25, 2017	Blue whale	Southern California	98	98	98
Bm170926-TDR14	September 26, 2017	Blue whale	Southern California	103	103	103
Bp160912-A14	September 12, 2016	Fin whale	Southern California	77	77	77
Bp160914-A20	September 14, 2016	Fin whale	Southern California	168	168	102
Bp180501-A41	May 1, 2018	Fin whale	Southern California	76	76	76
Bp180502-A44	May 2, 2018	Fin whale	Southern California	197	197	197
Bp180623-A41	June 23, 2018	Fin whale	Southern California	58	58	58
Bp180623-A44	June 23, 2018	Fin whale	Southern California	58	58	20
Mn140604-TDR5	June 4, 2014	Humpback whale	Central California	65	65	65
Mn141106-TDR5	November 6, 2014	Humpback whale	Central California	46	46	46
Mn141110-TDR7	November 10, 2014	Humpback whale	Central California	20	0	20
Mn150810-TDR5	August 10, 2015	Humpback whale	Central California	42	42	42
Mn151011-TDR12	October 11, 2015	Humpback whale	Central California	46	0	46
Mn151018-TDR11	October 18, 2015	Humpback whale	Southern California	40	0	40
Mn160522-TDR11	May 22, 2016	Humpback whale	Central California	62	0	62
Mn160523-TDR12	May 23, 2016	Humpback whale	Central California	25	0	25
Mn160926-TDR12	September 26, 2016	Humpback whale	Central California	24	24	24
Mn160926-TDR14	September 26, 2016	Humpback whale	Central California	89	89	89
Mn180924-TDR11	September 24, 2018	Humpback whale	Northern Washington	30	30	30
Total				4222	3813	3006

and 30 m as two thresholds for whale vulnerability to ship strike based on:

- Average draft of medium to larger container ships (7,000–18,000 TEU) is 14–16 and 15.2 m is the minimum depth of the expanded Panama Canal under the new Expanded Panamax Standard (Rodrigue et al., 2016).
- The overall danger zone about a moving ship appears to be about one to two times the draft vertically (Silber et al., 2010).

RESULTS

We identified 33 deployments representing 3,000 h of data from the above tags that met our criteria of >20 h of continuous depth and position data (see **Table 1** for a summary by species) off the US West Coast. There were pronounced differences in dive behavior between day and night and among species (**Table 2** and **Figure 2**). In all three species, whales spent a greater portion of time near the surface during the night compared to the day. This difference was most pronounced in blue whales primarily because of their deeper average dive depth during the day compared to the other species (average daytime depth was 81 m for blue whales compared to 67 m for fin whales and 34 m for humpback whales). The night dive depths were similar among species with an average of 11.5–13.6 m by species.

These dive differences directly translated to differing proportions of time within the top 15 and 30 m zones where they would be most vulnerable to either being struck by a ship's bow or propeller. Blue whales were twice as likely to be in the top 30-m of the water column at night compared to the day, average among the deployments of 90% versus 46% of time, respectively (**Table 2** and **Figure 2**). This was similar for the proportion of time shallower than 15 m with 73% at night versus 36% during the day for blue whales. All three species were in the top 30 m close to 90% of the time at night, but for humpback and fin whales, the daytime average was 69 and 59%, respectively, not as great a difference as for blue whales (**Table 2**).

Daytime positions, typically the primary type of positions available for most data sets on whale distribution, did not very accurately reflect where whales were that night and tended to vary by species (**Figure 3**). While there was considerable individual variation, humpback whales tended to stay closest to the average daytime position including into the following night while blue whales tended to move farthest from these daytime positions. Positions for whales (for each 15 min period) were on average were within 5 km of the centroid position for that period (**Table 3**) and were generally smallest for humpback whales. On average, nighttime positions, however, were >10 km from the daytime centroid positions for blue and fin whales, indicating how daytime positions can be a poor proxy for the positions at night.

There were also differences in movement patterns of all three species in the day versus night though this was more complex and complicated by sometimes fewer GPS hits during the night — likely due to the less active surfacing patterns which limited samples sizes especially for humpback whales (**Table 3**). For periods with good positions throughout, there was little difference in speed of movement between day and night or among species. Humpback whales stayed closer to the start position or the centroid position in both day and night compared to blue and fin whales (**Table 3**).

While the overall areas over which the whales moved were similar or slightly higher during the day versus at night, the daytime periods were typical 50% longer than night periods and indicated that on a time-corrected basis whales tended to travel farther from their starting point at night (**Table 3**). Since speeds were similar between day and night this reflected a more directed course of travel at night. This was apparent in the average change in heading from each pair of positions to the next; the mean heading change averaged 49° in the day versus 30° at night for blue whales and 51° in the day versus 35° at night for fin whales, both of which were significant different ($p < 0.01$) (**Table 3**).

This was also apparent in some of the detailed tracks for blue whales that had the largest sample of day and night movement. A typical pattern for whales that fed consistently in one area over multiple days was to perform more directed movement at night that looped to bring the whale back to the feeding area by the next morning (**Figure 4**). In the Santa Barbara Channel, the main daytime feeding areas for blue whales was south of the shipping lanes (**Figure 5**) but the range of the nighttime movements seen in the tag data regularly took them into the shipping lanes.

TABLE 2 | Summary of cumulative time at depth by time period and species.

Species	Blue whales	Fin whales	Humpback whales	All
Deployments	16	6	11	33
Total hours for depth analysis	1,980	530	489	2,999
Day				
Total hours day	1122.8	301.3	241.1	1,665
Mean depth day	80.9	67.0	34.2	
SD depth day	35.3	14.4	21.9	
% Risk day (30 m)	46%	59%	69%	
% Risk day (15 m)	36%	49%	54%	
Night				
Total hours night	690.4	186.5	204.1	1,081
Mean depth night	13.6	11.5	12.5	
SD depth night	6.8	6.9	8.1	
% Risk night (30 m)	90%	90%	88%	
% Risk night (15 m)	73%	76%	76%	
Crepuscular				
Total hours crepuscular	166.8	42.2	43.7	253
Mean depth crepuscular	37.2	38.9	19.1	
SD depth crepuscular	21.1	10.6	14.4	
% Risk crepuscular (30 m)	66%	64%	79%	
% Risk crepuscular (15 m)	50%	48%	64%	

These would include proportion of time above 15 and 30 m for each species and time period. Would also include potential separation between Travel/Non-Travel modes.

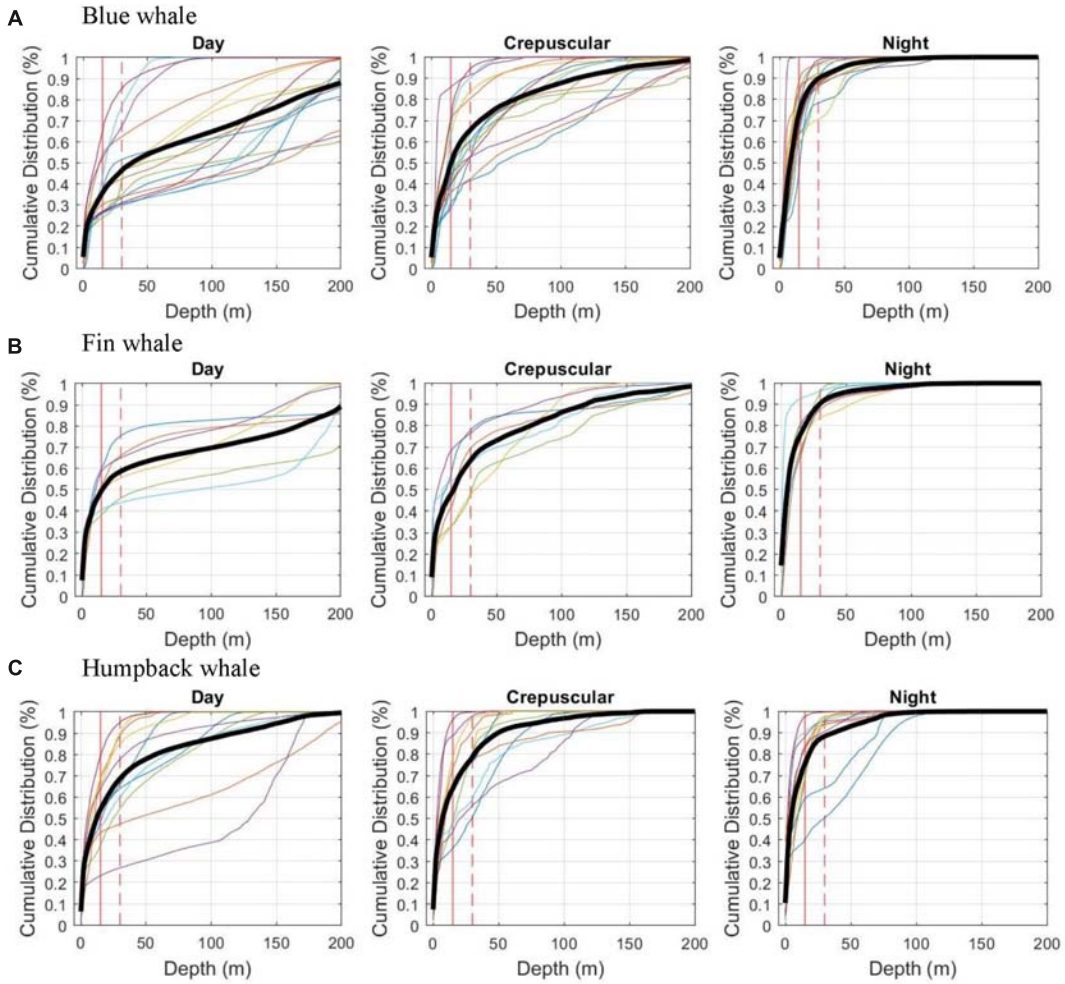

FIGURE 2 | Cumulative time at depth and shallower by species and broken into 3 periods (day, crepuscular, and night). The 9 plots (3 species and 3 time periods) show cumulative time on *Y* axis and depth on *X* axis showing faint tracings for each individual deployment and bold tracing showing average for all deployments on that species in that time period. These represent time periods where the animal was along the US West Coast and exclude migrating animals. Average for each species/period weights each individual equally (treating long deployments with equal weight as shorter ones). Vertical lines show 15 and 30 m depths as discussed in Methods. **(A)** Blue whale Day Night, **(B)** Fin whale, and **(C)** Humpback whale.

DISCUSSION

The differences between species and day versus night diving and movements have clear implications for vulnerability to ship strikes. Whales were closer to the surface almost twice as much of the time at night compared to the day and this would put them at depths where they would be in the strike zone of ships or be hit by the propeller. This was most easily seen in the day/night differences in the proportion of time within our nominal 15 and 30 m depths but this difference would hold regardless of the ship draft [15-m depth reflects the average draft of medium to larger container ships of 7,000–18,000 Twenty-foot Equivalent Unit (Rodrigue, 2017) while 30-m draft would correspond with that of some of the largest ships but could also reflect the overall danger zone about a moving ship which appears to be on the order one to two times the draft (Silber et al., 2010)].

Although the depth of the whale is important for vulnerability to ship strike, it is also influenced by when and how a whale reacts to an approaching ship. Tagged blue whales (including some of the early deployments evaluated for use in the current study) took little evasive action to the close approach of ships on near-collision courses (McKenna et al., 2015). An encounter model of the risk of ship strikes of whales off the US West Coast based on distributions of whales and ships also included dive data from some of the tag data used here (but without separation by day and night) but had to make assumptions about whale response to approaching ships to consider the probability a whale would get struck by a ship (Rockwood et al., 2017). While we do not know yet how whales might react differently to ships in the day versus the night, there is some potential for their being less reactive at night in addition to their being in a more vulnerable portion of the water column closer to the surface. The greater time whales spend farther from the surface in the day is the result of their

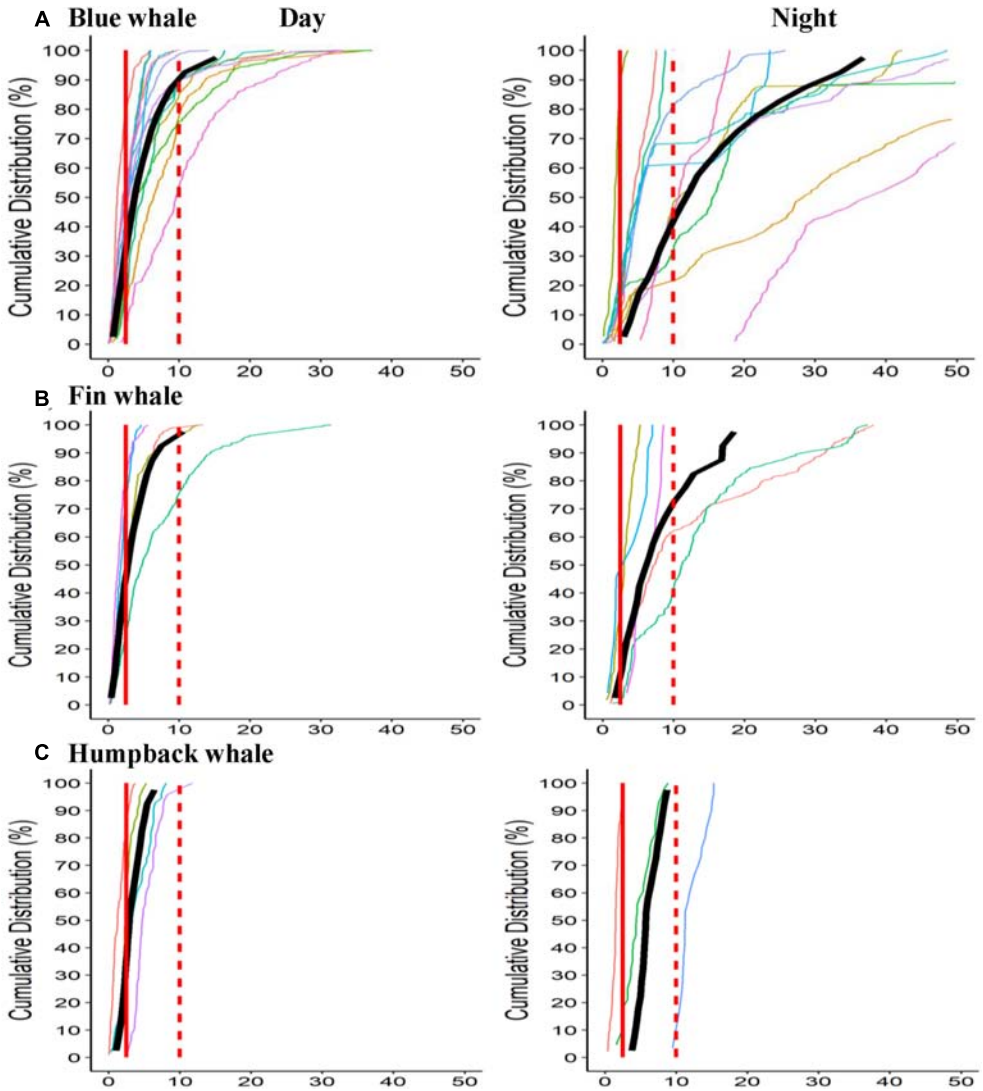

FIGURE 3 | Cumulative distance in kilometers from daytime centroid (mean of daytime positions). Positions standardized to one every 15 min interpolating as necessary (to avoid bias in sampling rate). Plots show distance from centroid point and lines show proportion of positions within that range on the *Y*-axis. Vertical lines show cut offs at 2.5 and 10 km range. Individual lines show cumulative positions for all cycles in each deployment. **(A)** Blue whale Day Night, **(B)** Fin whale, and **(C)** Humpback whale.

TABLE 3 | Summary of movement parameters by species and period.

Species	Individuals		Periods		Mean period duration (h)		Mean distance to period centroid		Mean distance to daytime centroid		Cumulative distance		Mean km/h for period		Mean heading change in period		
	Day	Night	Day	Night	Day	Night	Day	Night	Day	Night	Day	Night	Day	Night	Day	Night	*p*
Blue whales	13	12	80	77	12.9	8.2	4.6	4.0	4.6	14.1	38.9	24.9	3.0	3.0	49	30	0.0000
Fin whales	5	5	18	17	13.0	8.3	4.4	4.7	4.4	10.7	48.4	32.6	3.8	3.9	51	35	0.006
Humpback whales	3	3	4	4	13.1	8.4	2.2	1.8	2.2	4.9	39.3	20.7	3.0	2.3	65	69	NS

Mean values are averages using the average of each period as a single data point in calculation and statistics. Distances are in km.

feeding on krill prey at deeper depths (Friedlaender et al., 2014; Goldbogen et al., 2017). Humpback whales off California are more variable in their prey and switch between fish and krill depending on their relative abundance (Fleming et al., 2015). Our more anecdotal observations of whales feeding or resting at the surface including during some of the tag deployments revealed these whales are often easier to approach when surface feeding as their maneuverability is often limited due to engulfed

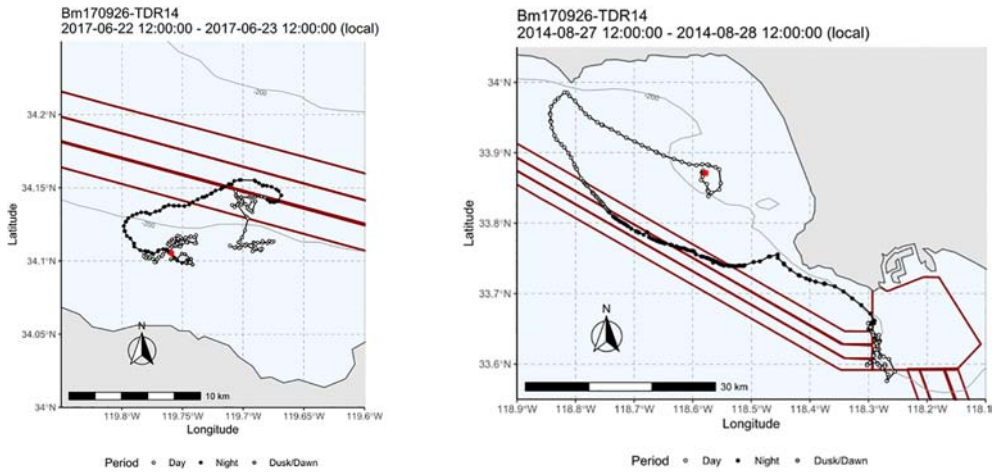

FIGURE 4 | Two examples of tracks of blue whales covering approximate 24-h periods with daytime tracks in white and nighttime in black showing movement patterns around Southern California shipping lanes (left in Santa Barbara Channel and right off LA/Long Beach). In both cases, long looping tracks were seen at night that took whales into or out of the shipping lanes (shown in red) compared to where they were feeding in the day.

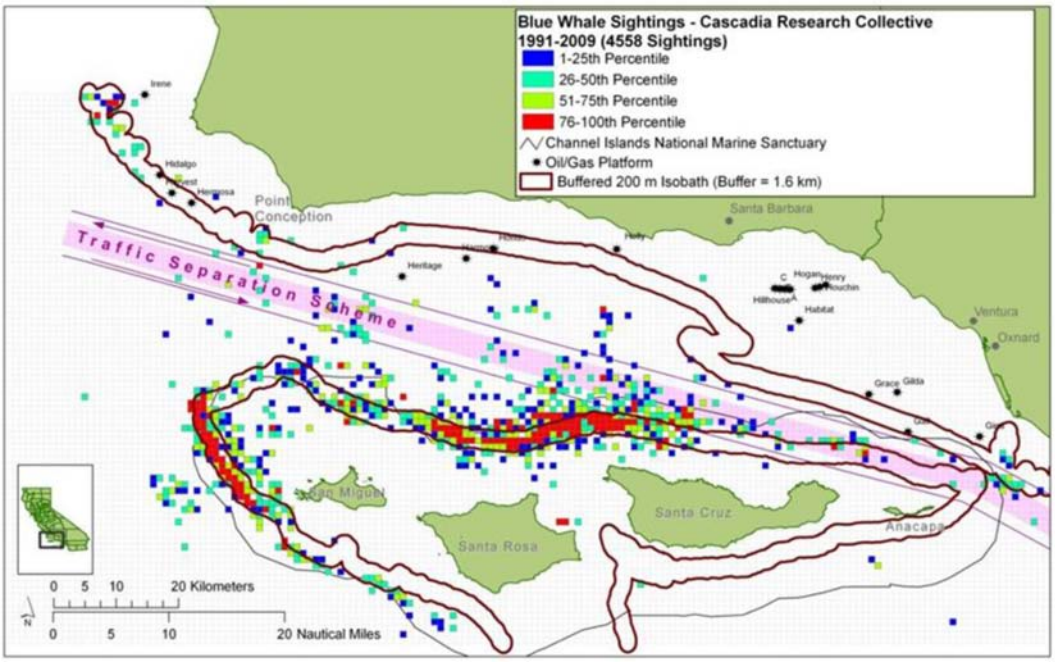

FIGURE 5 | Blue whale daytime sighting locations from Cascadia Research non-systematic surveys 1991–2009 when high densities were seen in the Santa Barbara Channel showing high density of occurrence around 200 m isobaths and mostly south of the designated shipping lanes at that time.

prey, and resting whales sometimes do not react to small boat approaches. While these observations were of resting or surface-feeding whales during the day, this might also be applicable to the night when whales are most commonly near the surface and their slower reaction during the surface periods we observed could further increase their vulnerability to ship strike than we demonstrate based solely on time at depth.

A number of studies have examined broader distributions and habitat models of blue whales in the eastern North Pacific based on long duration satellite tag data and acoustic detections though these have generally been on a broader and coarser scale. Satellite tags typically provide a few positions a day based on Argos Doppler shift and the less frequent or accurate positions these types of tags provide have been used for broader scale assessments (Mate et al., 1999; Bailey et al., 2009; Hazen et al., 2017). Lagerquist et al. (2000) also provided dive information by period of day on one blue whales with a depth-recoding satellite tags off central California but this did not reveal any

consistent diel patterns. Irvine et al. (2014) looked at home ranges based on tagging data and overlap with shipping lanes. None of these attempted or could look at differences in day versus night movement and positions that may not have been appropriate at the courser spatial scales of this data and the resulting models. Similarly, acoustic detections of blue whales have been used to examine timing and occurrence of whales in the eastern North Pacific (Stafford et al., 2001, 2005; Burtenshaw et al., 2004). Only some of these are able to localize calls to fine scale locations and calls do not appear to be representative of whale density since they vary by behavior, season, and sex of the calling whale (McDonald et al., 2001; Oleson et al., 2007a,b; Lewis et al., 2018).

Finer spatial scales are often critical to evaluating ship strike risk. One of the areas of highest risk of ship strikes are in the designated shipping lanes coming and going from major ports like Los Angeles/Long Beach and those in San Francisco Bay as well as transit routes for ships between ports along the US West Coast (Redfern et al., 2013; Jensen et al., 2015; Rockwood et al., 2017; Moore et al., 2018). Shipping lanes are often only 1 nmi wide and so whether whale presence overlaps with these areas requires position data on a very fine spatial scale. One change in shipping lanes in the southern California Bight that was made to reduce risk of ship strikes shifted one of the lanes only 1 nmi to get it farther from area of frequent blue whale feeding in the southern Santa Barbara Channel (Moore et al., 2018). Finer scale data on whale positions taking into account the differences in whale movements and distributions between day and night are required for more detailed assessments. In some areas like the Santa Barbara Channel, daytime positions would lead to a conclusion of limited overlap between some of the main blue whale feeding areas in the South Central Santa Barbara Channel, yet nighttime positions may involve more overlap as whales shift away from the specific areas (**Figure 4**). Similarly, risk may appear higher where feeding areas are concentrated in shipping lane areas during the day but are more dispersed away from those areas of overlap during the more vulnerable nighttime.

The species differences in whale shifts away from daytime positions fits both with the broader and larger movements we saw during the entire deployment durations and with other aspects of their known feeding behavior. Humpback whales which showed the most limited movements away from daytime positions are known to be fairly loyal to specific feeding areas (Baker et al., 2013; Calambokidis et al., 2015). Greater blue whale movement shifts are also consistent with their broader range of movements during the feeding season in the Eastern North Pacific (Mate et al., 1999; Calambokidis et al., 2009; Irvine et al., 2014).

The greater vulnerability of whales to ship strikes at night also has implications for management strategies to reduce ship strikes. A variety of approaches have been suggested for reducing ship strikes (Nowacek et al., 2004; Silber et al., 2012a,b; Conn and Silber, 2013; Redfern et al., 2013). Our results demonstrate that methods based on visual sightings of whales or other approaches requiring daylight would not be very effective since they would not address the primary period of whale vulnerability. Similarly, approaches to reduce ship strikes including speed or location restrictions would be most effective if they were enforced in locations when ships transit whale hot spots at night.

The use of medium-duration archival tags term has provided new insights into dive behavior and movements of whales not possible with other data sources to date. There are still challenges in use of some of this data, however. While our sample size for blue whales was fairly large, that available for humpback and fin whales was smaller and involves a limited number of individuals. Some of the parameters we report may also vary by location and season and additional deployments will be required to fully address.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

Tagging procedures were conducted under authority of a scientific research permit under the Marine Mammal Protection Act and Endangered Species Act and procedures were reviewed and approved by an Institutional Animal Care and Use Committee in conformance with the Animal Welfare Act.

AUTHOR CONTRIBUTIONS

JC drafted the manuscript. JC, JF, AS, BS, DC, AF, and JG reviewed and edited the manuscript, and collected the data. JC and JF analyzed the data.

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Vessel Operations in the Arctic, 2015–2017

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The Arctic is among the most rapidly-changing regions on Earth. Diminishing levels of sea-ice has increased opportunities for maritime activities in historically inaccessible areas such as the Northern Sea Route and Northwest Passage. Degradation of Arctic marine ecosystems may accompany expanding vessel operations through introduced underwater noise, potential for large oil spills, among other things; and may compound stressors already effecting biological populations due to climate change. Assessments are needed to track changes in vessel traffic patterns and associated environmental impacts. We analyzed Arctic-wide vessel Automatic Identification System data 1 January 2015 to 31 December 2017 to quantify the amount and spatial distribution of vessel operations, assess possible changes in these operations, and establish a baseline for future monitoring. Nearly 400,000 vessel transits were analyzed. Number of trips, hours of operation, and amount of sea surface exposed to vessel traffic were used to compare operations between 14 delineated waterways. Operations were extensive and diverse: an average of 132,828 trips were made annually by over 5,000 different vessels. Transits were made in all areas studied and all months of the year. Maritime activities were intensive in some areas, but ice-limited in others. Amount of sea surface exposed to vessel traffic exceeded 70% in all but three areas. Bulk carriers, cargo ships, passenger/cruise ships, research survey ships, and vessels supporting oil/gas-related activities were represented. However, fishing vessels, primarily in the Barents, Bering, and Norwegian Seas, surpassed operations of all other vessel types and comprised about one-half of all voyages each year. We observed no overt increasing or decreasing trends in vessel traffic volume in our limited study period. Instead, inter-year variation was evident. While the number of unique vessels and transits increased year-to-year, hours of operation declined in the same period. Abundance/distribution of fisheries resources, economic feasibility of Arctic marine travel as weighed against inherent risks, and other factors likely accounted for inter-year variation in regional activity levels. Measures have been established to protect Arctic marine ecosystems but may need strengthening to address potential ecosystem threats from existing and growing commercial and industrial activities in the region.

Keywords: Arctic, shipping operations, Northern Sea Route, Northwest Passage, Arctic routes, Arctic fishing, Arctic marine ecosystems, climate change

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INTRODUCTION

The Arctic is one of the globe's most rapidly-changing regions (Intergovernmental Panel on Climate Change [IPCC], 2018). Annual average Arctic sea ice extent has declined steadily at least since the early 1980s, and Northern Hemisphere snow and ice extent in 2016 was the lowest on record (U.S. Global Change Research Program [USGCRP], 2017). Diminishing annual and seasonal Arctic sea ice cover is expected to continue or accelerate in the foreseeable future (Overland et al., 2018). Predicted timing for seasonal ice-free waters in certain locations has ranged from within a few decades (Overland and Wang, 2013), to mid-century, (Smith and Stephenson, 2013), to as late as 2100 (Boé et al., 2009; Melia et al., 2016).

Expansion of the duration and spatial extent of seasonal ice-free water will bring changes in accessibility, and therefore the quantity and character, of maritime activities in the region (e.g., Arctic Council, 2009). Extended navigation periods resulting from decreasing ice cover are expected to encourage intercontinental transportation along routes with historically limited access (Khon et al., 2010; Smith and Stephenson, 2013; Zheng et al., 2016). Among these, the Northwest Passage (Guy, 2006; Buixadé Farré et al., 2014) and Russia's Northern Sea Route (Liu and Kronbak, 2010; Stephenson et al., 2014) are increasingly viewed as economically-viable alternatives to Suez or Panama Canal routes (Stroeve et al., 2012; Melia et al., 2016). Changing conditions also favor high-latitude tourism (Hamilton et al., 2005), new opportunities for exploitation of vast sub-seafloor oil and gas reserves (Arctic Monitoring and Assessment Programme [AMAP], 2008; Petrick et al., 2017; Wilkinson et al., 2017), and expansion of other commercial and industrial activities (e.g., Christiansen et al., 2014; Congressional Research Service [CRS], 2018).

As Arctic maritime activities increase, so are expected impacts to Arctic marine ecosystems (Arctic Council, 2009; Congressional Research Service [CRS], 2018). Expanding vessel activities will increase air-borne emissions levels (Corbett et al., 2010; Johansson et al., 2017) and contribute to the likelihood of maritime casualties (United States Coast Guard [USCG], 2010). Environmental degradation associated with increased vessel traffic will also include the effects from increased underwater noise levels on marine organisms (Moore et al., 2012; Halliday et al., 2017), introduction of non-native species (Miller and Ruiz, 2014; Nong et al., 2018), and ship strikes of marine mammals (Huntington et al., 2015; Cooke and Clapham, 2018). These, in turn heighten exposure of Arctic species already susceptible to the effects of climate change (Ragen et al., 2008; Silber et al., 2017) and negatively influence indigenous people reliant on the region's resources (Huntington et al., 2015).

Among the greatest potential threats to Arctic biological processes are large-scale oil and chemical spills (Arctic Council, 2009; Congressional Research Service [CRS], 2018; Walker et al., 2018). Substantial oil and gas reserves are being tapped in Arctic shelf waters (e.g., Congressional Research Service [CRS], 2018) increasing the likelihood of spills from well blowouts, tanker spills, or vessel accidents (Arctic Monitoring and Assessment Programme [AMAP], 2008; U.S. Committee on the Marine Transportation System [CMTS], 2013). Effects from spills are particularly acute given the region is remote, insufficiently charted, and inadequately supported by spill response infrastructure (Ivanova, 2011; National Research Council of the National Academies of Science [NRC], 2014). Long-term entrainment of oil in cold-water biological communities (Vergeynst et al., 2018) make Arctic ecosystems particularly vulnerable to spills.

Threats from increasing vessel activity on the integrity of Arctic marine ecosystems are real. However, there have been only limited efforts to quantify the amount and spatial features of Arctic-wide maritime operations (Eguíluz et al., 2016; Adams and Silber, 2017; Ocean Conservancy, 2017), and to our knowledge there are no studies of yearly changes in these operations. The magnitude of potential risks will not be fully known until the extent of maritime activities is known. Previously, we analyzed Automatic Identification System (AIS) data to quantify Arctic-wide vessel activities for calendar year 2015 and to establish a baseline of marine traffic in the region (Adams and Silber, 2017). Here, we quantify vessel operations in 2016 and 2017. Incorporating the 2015 data, and using the same metrics used in the 2015 study, we provide a 3 year, pan-Arctic characterization of current maritime activities.

MATERIALS AND METHODS

Data synthesis and analytical methods were the same as those used in Adams and Silber (2017)[1]. Descriptions of analytical approaches detailed in that report are summarized here.

Global satellite AIS data were obtained from exactEarth® for 2015 and from ORBCOMM™ for 2016 and 2017. The Transview (TV32) software application, developed by the Department of Transportation's VOLPE National Transportation Systems Center[2], was used to decode raw, global AIS data into monthly comma separated value (CSV) files. The decoded monthly data was imported into a spatially-enabled PostgreSQL/PostGIS database and linked to an enhanced vessel database (IHS Markit, 2017) to obtain verified vessel type, gross tonnage, and country of origin (flag country) information.

AIS data were then overlaid on the Arctic as defined by the Arctic Research and Policy Act of 1984 (ARPA) and data located within the ARPA-delineated boundary were analyzed. Within this area, 14 bodies of water were identified using spatial boundaries defined in the International Hydrographic Organization's Sea Areas[3] data set, namely: the Arctic Ocean, Baffin Bay, Barents Sea, Beaufort Sea, Bering Sea, Chukchi Sea, Davis Strait, East Siberian Sea, Greenland Sea, Laptev Sea, Kara Sea, Northwest Passages, Norwegian Sea, and the White Sea (**Figure 1**). These spatial designations were used to parse and characterize vessel activity in each body of water.

[1]https://www.fisheries.noaa.gov/resource/document/technical-memorandum-2015-vessel-activity-arctic

[2]https://www.volpe.dot.gov/

[3]https://www.arcgis.com/home/item.html?id=44e04407fbaf4d93afcb63018fbca9e2

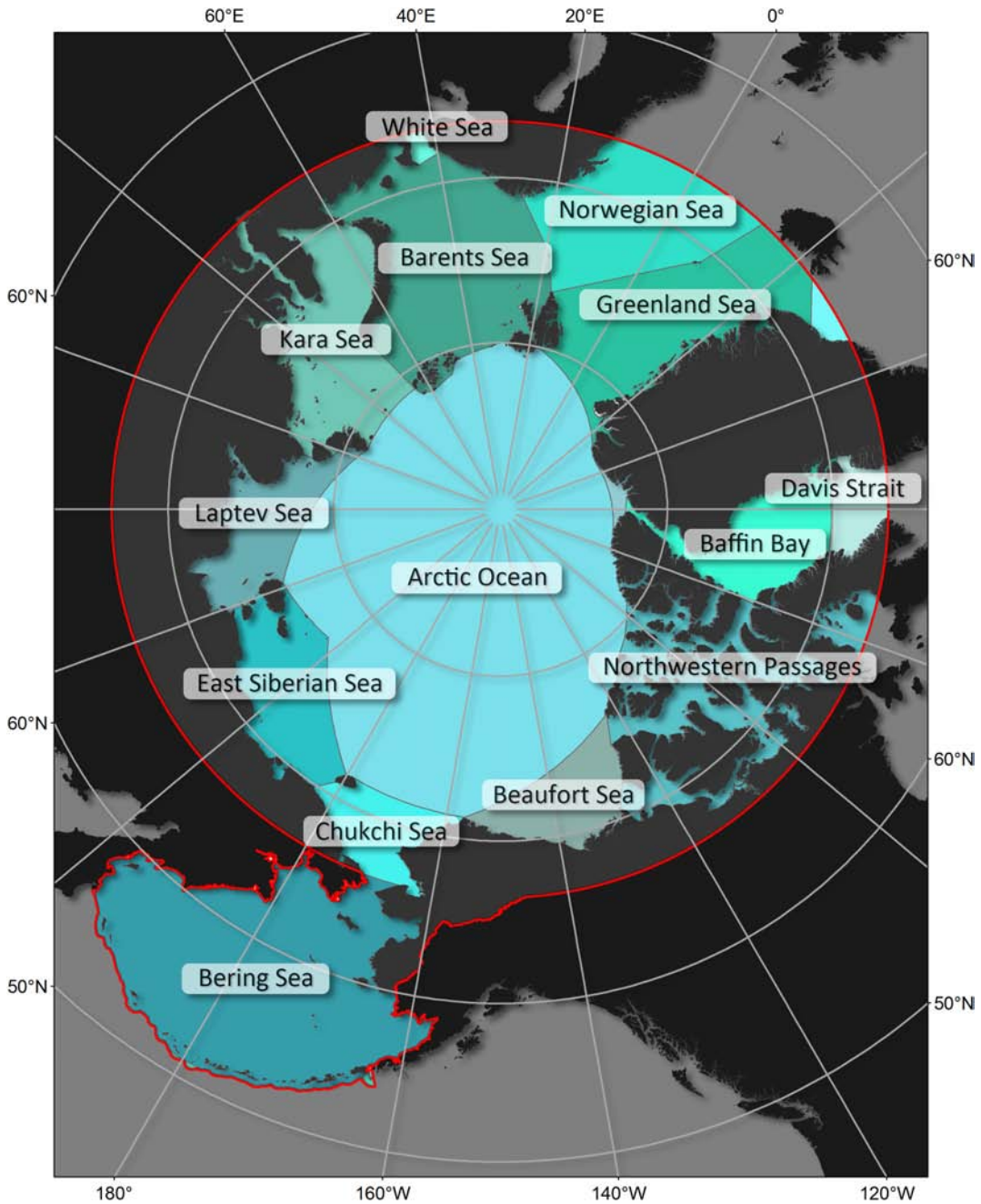

FIGURE 1 | Boundaries of study area as defined by the Arctic Research and Policy Act of 1984 and the 14 oceans, seas, bays, straits and passages within the boundary (International Hydrographic Organization Sea Areas).

Each AIS record contains a speed, position, and timestamp. We used this information to assess the spatial and temporal integrity of a given record using temporally adjacent AIS records by the same vessel. Records for an individual vessel were sorted using timestamps and then aggregated into transits (i.e., individual trips made by vessels) according to time elapsed between successive records. If elapsed time between successive records for a given vessel was <4 h, they were aggregated into the same transit; if ≥4 h, a new transit was initiated for the vessel. Distance-based computational checks were also made for two temporally adjacent records to ensure the integrity of the derived transit. To segment trips by area, we intersected transits with clipped body of water polygons. Transits that straddled the end and the beginning of a month were split between consecutive months using interpolation.

Spatial density analyses produce gridded surfaces containing cell values representing the concentration of certain features. Linear vessel transit features were used in calculating areas of relative traffic density. A Line Density tool in ArcGIS for Desktop was used with a search radius of 10km and an output cell size of 5km for all transits and specific vessel classes to determine linear distances. Color ramps were used to depict relative densities throughout the study area. We also computed the percent of the total water surface area exposed to vessel traffic by summing the area associated with grid cells containing non-zero density values. An area represented by a grid cell was considered exposed to vessel traffic when the associated vessel traffic density value was greater than zero.

Data from over 360 million individual AIS transmissions logged within the Arctic spatial boundary during the study period were analyzed. These data were first screened to remove any transmissions that contained suspect position, speed or timestamp data. Transmissions by vessels that were not actively engaged in travel were also removed. The remaining AIS transmissions were aggregated into transits. Transits that contained less than 5 AIS transmissions or represented less than 5 nm of travel were removed. The remaining transits were linked to a third-party vessel database (IHS Markit, 2017) to obtain more detailed and accurate information on vessel size and type. Transits by vessels that could not link to the above-described (IHS) database were dropped from the analysis. The remaining transits were then segmented based on sea area, year and month. It should be noted that vessels engaged in nationally-sensitive operations, involving for example military, sovereign, or other government vessels, may not routinely transmit AIS signals and these are data potentially unavailable to this study.

A set of metrics were used to characterize vessel activity: number of trips, hours of operation, and trip distance. Each of these metrics were used to compare operations between areas, between years, and IHS vessel type class. Gross vessel class designations (using AIS "Level 3" data) were used to characterize broad-scale (e.g., Arctic-wide) seasonal and inter-year variation in vessel activities, whereas finer scale ("Level 5" AIS data) vessel class information was used in describing results for all other metrics. The 14 delineated bodies of water differ a great deal in size; as a result, absolute counts of number of trips and values for distances traveled may be of limited value in inter-area comparisons. Therefore, in an effort to provide a measure of relative amount of vessel traffic volume in each of the 14 areas studied, we also determined transit densities (trip distance per unit of water surface area, or km/km^2) and extent of sea surface exposed to vessel trips (percent of total water surface area with non-zero density values).

RESULTS

Arctic vessel operations were substantial and spatially varied. During the study period, an average of 132,828 trips were made each year by over 5,000 different vessels (**Table 1**). These vessels represented more than 60 different IHS Level 5 vessel types. Annually, vessels operated an average of over 3.2M hours (yearly range 2.8M–3.7M hours). Transits were made in all areas studied (**Figures 2A–C**) and all months of the year in **Figure 3**. While the number of individual vessels and the number of transits increased steadily during our study period, the number of hours of operation and total distances traveled declined (**Table 1**). Thus, on the whole Arctic-wide, vessels engaged in an increasing number of trips, but these trips tended to be relatively shorter in duration.

TABLE 1 | Annual Arctic-wide vessel traffic metrics, 2015–2017.

Vessel traffic metric	2015	2016	2017	Average
Number of unique vessels	5,437	5,475	5,606	5,506
Transit counts	116,317	140,105	142,062	132,828
Operational hours	3,752,055	3,053,909	2,820,561	3,208,842
Transit distance (nm)	24,536,244	19,865,667	18,299,701	20,900,537

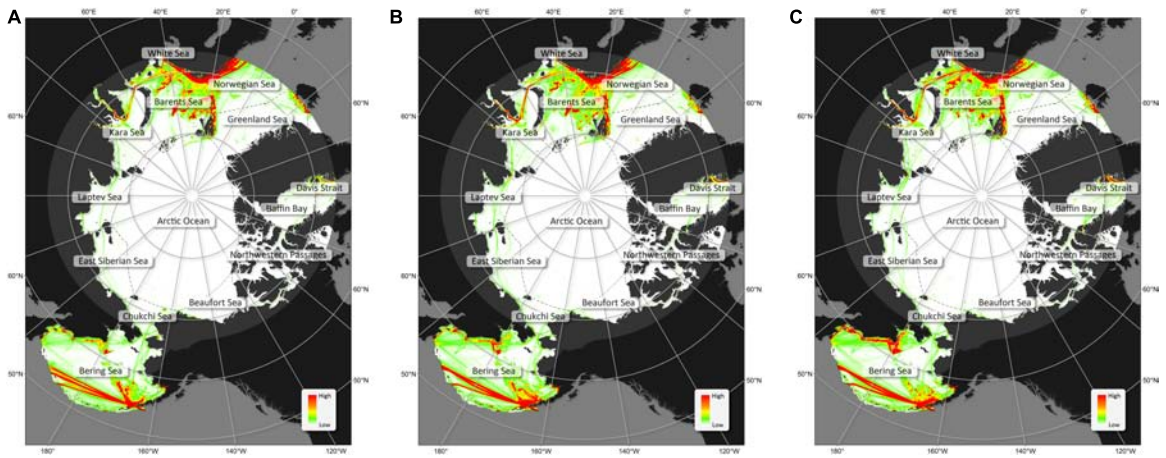

FIGURE 2 | Relative track densities of all vessel types in **(A)** 2015, **(B)** 2016, and **(C)** 2017. Areas with no color indicate no vessel traffic or low traffic volumes relative to other locations.

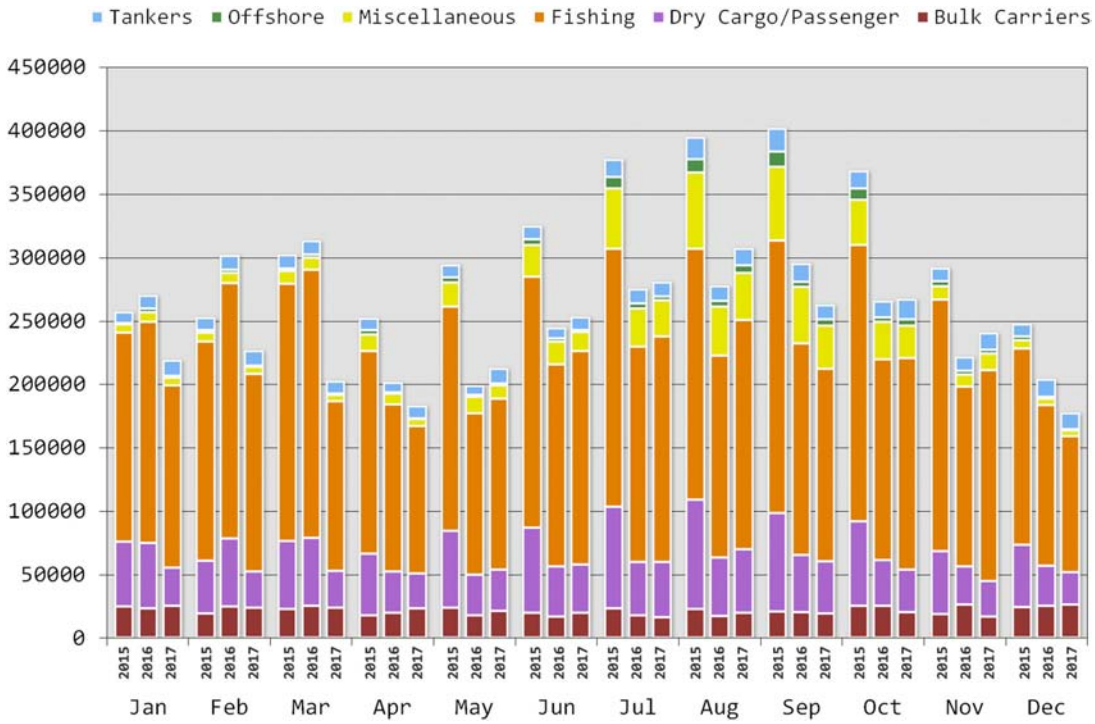

FIGURE 3 | Vessel hours of operation, by month and vessel class, in 2015, 2016, and 2017.

TABLE 2 | Nations with >100 flagged vessels operating in Arctic waters and number of registered ships, 2015–2017.

Country	2015	2016	2017
Panama	754	826	767
Russia	734	763	800
Norway	639	702	727
United States	412	419	410
China, Hong Kong	319	336	334
Liberia	290	288	332
Marshall Islands	267	268	338
Malta	212	216	263
Singapore	210	206	236
Bahamas	189	174	180
Denmark	*	127	126
Iceland	121	111	115
Netherlands	113	105	*
Cyprus	103	119	113

**66 Denmark-registered ships in 2015; 90 Netherlands-registered ships in 2017.*

Country of origin for vessels utilizing Arctic waters was diverse, with most being registered to Panama, Russia, Norway and the United States (**Table 2**). Norwegian-flagged vessels made the most trips (mean = 42,365; range 37,429 in 2017 to 49,394 in 2015) followed by vessels flagged by Russia (mean = 36,422; range 21,008 in 2015 to 48,092 in 2017) and the United States (mean = 17,536; range 16,583 in 2017 to 18,580 in 2016).

The distribution and number of nations under which vessels were registered (also referred to as its "flag state") reflect at least two types of activities in the region. One, ships from Arctic countries (Norway, Russia, the United States, etc.) were engaged in extensive commercial fishing industries, local/regional shipping activities, or resource extraction operations. A second group was comprised of commercial vessels registered in countries (e.g., Panama, Marshall Islands) that may have little or no direct interest in Arctic operation. Instead, "flags of convenience" are nations who offer attractive tax and employment regulations that confer cost benefits to owners, and our findings mirror, in part, a worldwide registry of vessels. The number of vessels registered in nations distant from Arctic waters (e.g., China, Liberia) also illustrate the importance of Arctic routes to inter-continental shipping activities.

Overall, fishing vessels, primarily <1000 gross tons (gt), logged the greatest number of trips and hours of operation (**Table 3**). In all 3 years combined, these vessels made 52.2% of all trips in the entire study area, and constituted 43.9, 54.2, and 57.0% of all Arctic vessel trips in 2015, 2016, 2017, respectively. Of the 14 waterways studied, fishing vessel activity was highest in the Bering, Barents, and Norwegian Seas, representing 66.3, 62.1, and 41.4% of all trips in those areas, respectively. Fishing vessel activities (including Stern Trawlers) in those three seas also represented 42.4–46.7% of the hours of operation of all vessel types combined (**Table 3**). Fishing vessels (primarily < 1,000gt) also accounted for more trips than any other vessel type in the Greenland Sea, Davis Strait, and Baffin Bay.

TABLE 3 | Hours of operation by fishing vessels in waterways in which commercial fisheries operations were the predominant vessel activity (as measured by hours of operation).

Location	Vessel type	2015			2016			2017		
		Hours	% Sea	% Arctic	Hours	% Sea	% Arctic	Hours	% Sea	% Arctic
Barents Sea	Fishing vessels	217,792	19.3	5.8	154,825	17.8	5.1	303,480	39.6	10.8
	Stern trawlers	599,641	43.5	13.1	487,353	43.4	12.4	259,398	26.5	7.2
Bering Sea	Fishing vessels	376,295	35.7	10.0	365,515	33.0	12.0	467,779	40.0	16.6
	Stern trawlers	228,441	11.4	3.2	275,790	9.7	3.5	249,656	10.6	4.4
Norwegian Sea	Fishing vessels	152,385	14.5	4.1	113,532	17.5	3.7	115,555	25.2	4.1
	Stern trawlers	329,685	22.6	6.3	249,436	28.4	6.0	149,036	22.8	3.7
Greenland Sea	Fishing vessels	30,194	17.2	0.8	22,084	15.7	0.7	64,262	43.5	2.3
	Stern trawlers	92,788	40.6	1.9	79,443	45.3	2.1	43,600	22.1	1.2
Davis Strait	Fishing vessels	1,310	2.8	0.0	9,485	19.2	0.3	27,842	46.8	1.0
	Stern trawlers	31,815	47.3	0.6	30,781	45.2	0.7	18,143	21.6	0.5
Baffin Bay	Fishing vessels	980	3.5	0.0	3,494	14.1	0.1	16,763	52.3	0.6
	Stern trawler	13,449	30.7	0.2	13,062	30.0	0.2	2,088	6.3	0.1

Stern Trawlers here include Factory Stern Trawlers and Stern Trawler AIS Level 5 (see text for explanation) vessel categories. Percent (%) Sea indicates the proportion of hours of operation by the indicated vessel class relative to all other vessel classes in the specified body of water. Percent (%) Arctic values are the same proportion relative to all vessel classes operating in the entire study area.

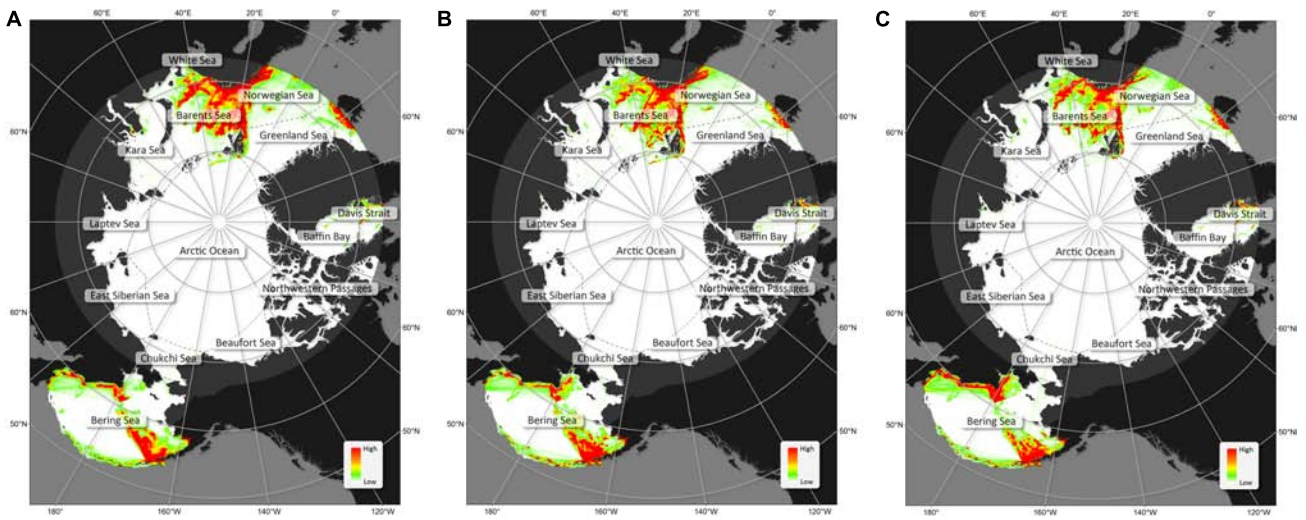

FIGURE 4 | Relative track densities of fishing vessels in **(A)** 2015, **(B)** 2016, and **(C)** 2017. Areas with no color indicate no vessel traffic or low traffic volumes relative to other locations.

In 2017, fishing vessels in the Bering Sea alone accounted for 16.6% of all operational hours for all vessel types in the entire study area combined, the highest such value in our dataset. However, total hours logged (as well as proportions of hours of operation within a given body of water) by fishing vessels varied inter-annually both within those seas and across the Arctic (**Table 3**). Tracks of fishing vessel routes were numerous and spatially diffuse (**Figures 4A–C**), indicative of the magnitude of this industry and the importance of fisheries resources in these areas. The spatial distribution of these trips also underwent inter-annual changes (**Figures 4A–C**).

Excluding all vessels engaged in fishing operations, the greatest number of transits occurred in the Norwegian, Barents, and Bering Seas (**Table 4**). (Note, we excluded fishing vessel activities in this table and in other representations to better highlight/discuss activities and metrics involving other vessel classes). Well over 100,000 trips were made in 3 years in the Norwegian and Barents Seas alone. The number of trips in these three Seas exceeded those in the Greenland Sea (the location with the fourth highest number of trips) by fourfold or more. In comparison to other locations, trips in these areas exceeded the number of trips made in the Northwest Passage, the Beaufort and East Siberian Seas and other locations by nearly two orders of magnitude.

Average sea surface area exposed to vessel traffic exceeded 96% in the Norwegian, Barents, and Bering Seas and over 90% of the sea surface was exposed to vessel transits in the Davis Strait and Chukchi Sea (**Table 4**). The Arctic Ocean had the

TABLE 4 | Total number of transits and percent surface area exposed to vessel traffic, excluding all fishing vessel operations, for each body of water in the study area.

Waterway	Total transits (3 years)	Water surface exposed (%; avg/year)
Norwegian Sea	149,271	99.1
Barents Sea	109,441	96.4
Bering Sea	98,526	98.3
Greenland Sea	25,559	77.5
Kara Sea	17,279	85.3
Davis Strait	9,447	91.8
Baffin Bay	5,350	80.1
White Sea	4,682	88.9
Chukchi Sea	3,999	90.2
Arctic Ocean	3,528	18.6
Laptev Sea	2,969	76.2
East Siberian Sea	2,015	75.4
Northwest Passage	1,847	36.5
Beaufort Sea	1,433	48.0

greatest overall surface area but was also at least 50% ice-bound for 12 months and 75% ice-covered for 11 months of the year. It ranked last in both density and percent sea surface covered by vessel transits. The Beaufort Sea and the Northwest Passage also exhibited low sea surface exposure and also hosted the fewest trips.

Vessel Types

Excluding Fishing Vessels, General Cargo Ships were among the vessel types making the most trips in nine of the 14 bodies of water studied (**Table 5**). Research/survey vessels, supporting primarily academic oceanographic and geophysical seismic research, were also among the most common vessel types, totaling an average of over 79,000 h of operation in each year of study period. Most of this work was conducted, in rank order, in the Barents, Kara, Norwegian, and Greenland Seas totaling between 21,000 and 93,000 h of operation during 2015–2017. In addition, vessels engaged in fisheries research logged an annual average of over 6,500 h; the majority occurring in the Greenland, Norwegian, Bering, and Barents Seas, in that order.

Vessel operations associated with the oil and gas industry were also strongly represented in our records, both numerically and spatially. Level 5 vessels types associated with the production and transport of oil and gas included, but were not limited to, Platform Supply Ships, Anchor Handling Tugs, Standby Safety Vessels, Drilling Ships, and Tankers. The Norwegian, Barents, and Kara Seas exhibited the greatest extent of oil/gas exploration- and extraction-related activities. Collectively, oil industry ships (excluding tankers) accounted for a total (for all 3 years) of 4,633, 2,821, and 671 transits in the Norwegian, Barents, and Kara Seas, respectively. Tankers moving hydrocarbon products [crude oil, Liquified Natural Gas (LNG), and Liquid and Petroleum Gas (LPG)] made 2,089, 1,930, and 596 transits in these three seas in 2015, 2016, and 2017, respectively.

Ships in the Passenger/Cruise vessel class made the greatest number of trips in the Greenland Sea (3 year total = 3,740 trips; mean = 1,246.7/yr), followed by the Barents (mean 630.0/yr) and Norwegian Seas (mean 484.0/yr) and the Arctic Ocean (mean 263.3/yr) (**Table 5**). Passenger/Cruise vessels comprised 22.4 and 20.8% of all vessel trips in the Arctic Ocean and Northwest Passage, respectively. Tugs made more trips than all other vessel classes in the Chukchi and Beaufort Seas (1,464 and 570 transits, respectively) and ranked among the top three vessel types in trips made in the Barents, Bering, and Kara Seas (**Table 5**). Transits by Icebreakers were most common in the Arctic Ocean and Northwest Passages, but trips by this vessel class were relatively few elsewhere.

Vessel Activities by Location

The Norwegian Sea exhibited the highest overall number of transits in 2015 (n = 56,952 trips) and 2016 (n = 47,103); whereas the greatest number of trips occurred in the Barents Sea in 2017 (n = 45,578 transits). Highest overall number of vessel operating hours occurred in the Bering Sea in 2015, 2016, and 2017, followed by the number of hours logged in the Barents Sea in those same years.

Most transits in the Bering Sea were logged by fishing vessels (exceeding 11,000 trips in each of the 3 years; mean = 13,520; range 11,089–15,082). These were generally short in duration (3 year mean = 30 h/trip) and distance (3 year mean = 143 nm). In addition, stern trawlers and factory stern trawlers collectively logged over 20,000 trips in the 3 year study period (mean = 3,464/yr; range = 2,146–4,559), a mean of 38 h/trip. Bulk Carriers in the Bering Sea logged more hours of operation than all other (non-fishing) vessel types in all locations, exceeding the second-highest, Cellular Container Ships in the Bering Sea, by over 280,000 h (**Table 5**). Transits by these vessels involved considerable distances (Bulk Carriers: mean = 712 nm/trip; Cellular Container Ships: mean = 669 nm/trip) many having been engaged in lengthy intercontinental voyages between western North America and Asia using the North Pacific Great Circle Route (**Figures 2A–C**).

In the East Siberian, Kara, and Laptev Seas (along with portions of the Chukchi, Beaufort, and Barents Seas), collectively considered as segments of the Northern Sea Route, General Cargo Ships made the greatest number of trips (**Table 5**). Research Survey Vessels and Products Tankers were also among ship classes making the greatest number of trips along this shipping route. Relative to other locations in the study area, comparatively small portions (75.4–85.3%) of these waterways were exposed to vessel traffic.

Most transits through the Canadian archipelago including the Northwest Passage, Baffin Bay and Davis Strait were made by General Cargo Ships (1,375 total, in these three areas in all 3 years) and Passenger/Cruise ships (1,048 in 3 years). Container Ships, Refrigerated Cargo Ships, Bulk Carriers, Chemical/Products Tankers, and Icebreakers were also among vessel types making the greatest number of trips in these waterways (**Table 5**). In the Northwest Passage, alone, total number of trips increased from 443 in 2015 to 644 in 2016 to 760 in 2017. In this location, the total number of hours under

TABLE 5 | Two most common (as measured by number of trips) vessel classes in each Arctic waterway studied.

Location	Vessel type	Trips	% of total	Mean	*SD*	Operation hours
Norwegian Sea	Passenger/Ro-Ro ship (vehicles)	20,452	14	6,817	±3,498	177,261
	General cargo ship	19,305	13	6,435	±825	244,188
Barents Sea	General cargo ship	7,939	7	2,646	±345	126,207
	Research survey vessel	3,148	3	1,049	±417	93,215
Bering Sea	Bulk carrier	8,628	9	2,876	±241	554,950
	Container ship (fully cellular)	6,731	7	2,244	±121	274,134
Greenland Sea	Passenger/Cruise	3,740	15	1,247	±121	34,914
	Research survey vessel	1,156	5	385	±38	21,030
Kara Sea	General cargo ship	3,748	22	1,249	±318	75,009
	Tug	1,676	10	559	±71	27,733
Davis Strait	General cargo ship	778	8	259	±91	5,141
	Container ship (fully cellular)	670	7	223	±10	5,896
Baffin Bay	General cargo ship	597	11	199	±40	126,207
	Refrigerated cargo ship	502	9	167	±13	3,759
White Sea	General cargo ship	2,318	50	773	±210	18,942
	Products tanker	372	8	124	±21	2,532
Chukchi Sea	Tug	1,464	37	488	±122	26,944
	General cargo ship	430	11	143	±26	6,833
Arctic Ocean	Passenger/Cruise	790	22	263	±65	7,191
	Research survey vessel	551	16	184	±32	12,673
Laptev Sea	General cargo ship	1,284	43	428	±99	17,481
	Research survey vessel	400	13	133	±62	8,329
East Siberian Sea	General Cargo Ship	876	43	292	±82	15,835
	Research survey vessel	253	13	84	±49	5,474
Northwest Passage	Passenger/Cruise	384	21	128	±8	4,282
	Bulk carrier	341	18	114	±53	2,767
Beaufort Sea	Tug	570	40	190	±81	6,600
	Pusher tug	204	14	68	±12	2,482

Values here include all vessel types, except fishing vessels. Total number of transits, 3 year mean (± standard deviation) number of transits by vessel class, and percent of total trips by all vessel types in specified body of water are indicated.

operation progressed from 8,669 h in 2015 to 6,766 in 2016 to 10,158 in 2017.

Inter-Year Comparisons

We observed modest inter-year changes in the number of individual vessels (by vessel type) operating in Arctic waters. Dry Cargo/Passenger vessels comprised the highest overall number of individual vessels operating in 2015 (1,771 vessels) and 2016 (1,699 vessels); while most ships operating in 2017 were in the Bulk Carrier class (1,779 vessels).

The overall number of trips increased in each year of the study period over the previous year while hours of operation and distances traveled declined (**Table 1**). At least some observed inter-annual variability can be attributed to large-scale changes in fishing vessel operations. For example, vessels engaged in fishing operations in aggregate logged a total of 51,093, 75,892, and 80,958 trips in 2015, 2016, and 2017, respectively (constituting more trips than all other vessel types in each year). Fishing vessels also represented 39.6% of all vessel operating hours in the Barents Sea in 2017 (and 10.8% of all vessels in the entire study area), but only 17.8% in 2016; in the Bering Sea fishing vessels logged over 100,000 more hours in 2017 than in 2016 (**Table 3**).

Other regional inter-annual fluctuations in vessel operations occurred. The number of trips and hours of operation by Bulk Carriers and Cellular Container Ships in the Bering Sea exhibited increases of 3–15% in 2016 relative to 2015, and the number of trips either declined (by 7%) for Cellular Container Ships or increased (by 7%) Bulk Carriers in 2017 relative to 2016. Hours logged by Chemical Tankers, Chemical/Products Tankers, Crude/Oil Products Tankers, and Products Tankers in the East Siberian Sea (a 3 years collective total of 5,958 h) increased 4.6% in 2016 over 2015 but declined 5.3% the following year. Total operating hours by these same four vessel classes in the Kara Sea went from 10,579 h in 2015 to 18,241 h in 2016 and 12,427 in 2017.

Trips by Passenger/Cruise vessels in the Greenland Sea increased by 2% in 2016 over 2015 and increased 20% in 2017 relative to 2016. In contrast, voyages by passenger/cruise ships in the Barents Sea was greatest in 2015 and dipped by about 5% in each of the subsequent 2 years. Trips by this vessel class in the Norwegian Sea were most numerous in 2015, subsequently declined 21% in 2016 and then increased 13% the following year. In one perplexing example an abrupt and precipitous drop occurred in vessel hours of operation in the Norwegian Sea

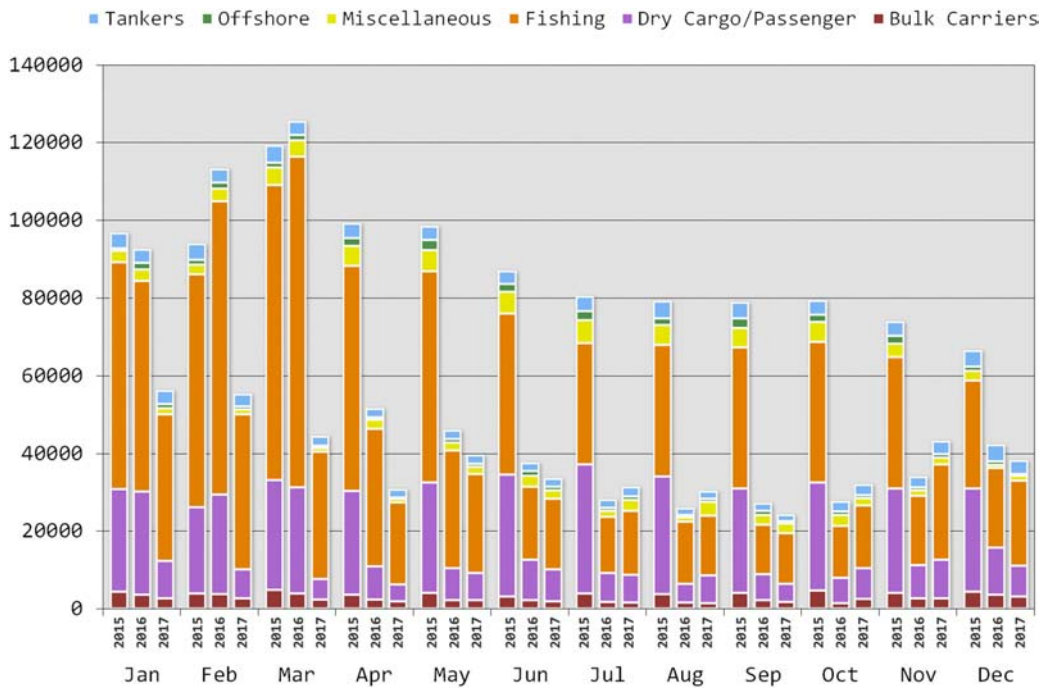

FIGURE 5 | Vessel hours of operation in the Norwegian Sea, by month and vessel class, in 2015, 2016, and 2017.

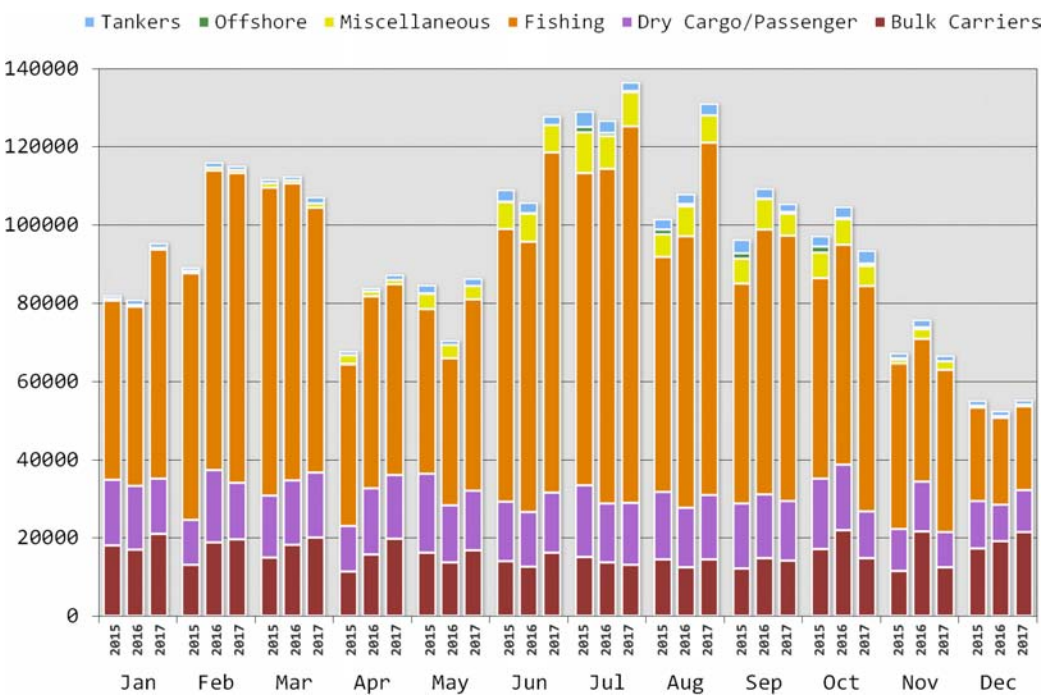

FIGURE 6 | Vessel hours of operation in the Bering Sea, by month and vessel class, in 2015, 2016, and 2017.

in mid-2016 that continued at similar levels throughout 2017 (**Figure 5**) – a pattern that did not occur in monthly hours of operation in the Bering Sea (**Figure 6**). In this Norwegian Sea case, the number of trips by Passenger/Roll-On, Roll-Off vehicle transport (or "Ro-Ro") vessels alone dropped from 11,656 trips in 2015 to 5,291 in 2016, and 3,505 in 2017 (**Figure 5**); while the number of transits by General Cargo Ships was 7,468, 6,389, and 5,448 in 2015, 2016, and 2017, respectively. In contrast, General

Cargo Ships and Bulk Carriers operating in the Barents Sea added an average of 363 and 264 trips, respectively, in each subsequent year of the study while operating hours of these two vessel classes declined between 6.1 and 35.6% each year in the same period.

DISCUSSION

Tens of thousands of voyages were undertaken in Arctic waters in each year of our study. And, yet, high latitude vessel activity is projected to grow as conditions evolve. This in turn is expected to heighten negative consequences for Arctic marine ecosystems (Arctic Council, 2009; Roach, 2018). While vessel traffic levels were extensive in some locations, they remained limited in others. It is the latter areas that may undergo the greatest amount of growth in changing ice conditions.

Characteristics of Maritime Activities in Various Waterways

Based on the volume, nature, and spatial characteristics of vessel traffic, Arctic waterways can be organized into three groups. These are the Barents, Bering, and Norwegian Seas; the Northern Sea Route and Northwest Passage and associated complexes of bays and straits; and the Arctic Ocean. In the first of these, the Barents, Bering, and Norwegian Seas include relatively low-latitude areas with extensive seasonal ice-free zones. Access to the Barents and Norwegian Seas is facilitated by warm northward-bearing North Atlantic surface waters creating vast ice-free areas during much or all of the year. Consequently, operations in these three seas were conducted by a diverse set of vessel types and represented an overwhelming majority of all vessel activity in the entire study area. Fishing activities predominated and were a major contributor to observed levels of surface area exposure to vessel activity (over 95%), a function of both the number of trips and broad spatial footprint of this industry. However, the importance of these bodies of water to both long- and short-distance transport of goods was also evidenced by the presence of large vessels such as General Cargo Ships, Bulk Carriers, Container Ships, and Passenger/Ro-Ro (vehicle) transport vessels, among others (**Table 5**).

In the Bering Sea, for example, long-distance, inter-continental trade was indicated by Bulk Carriers having logged the highest number of hours of operation of all (non-fishing) vessel classes, surpassing by nearly twofold the next-highest vessel class, Container Ships. These values are due, in part, to the expanse of this body of water, i.e., temporally and spatially lengthy trips were needed to traverse it. A significant inter-continental trade route exists in a portion of the Bering Sea connecting Asian and North American ports along a Great Circle Route atop the North Pacific Ocean (Burns and Poe, 2014) (a portion of this route is evident in **Figures 2A–C**). While most of the substantial ship traffic along this route passes south of the Aleutian Islands, much of it utilizes the Unimak Pass through the Aleutians and passes into waters north of the archipelago (Schwehr and McGillivary, 2007).

A second group of waterways – the East Siberian, Laptev, Kara and White Seas (or, the Northern Sea Route), and in the Northwest Passage, Baffin Bay, Davis Strait and the Beaufort and Chukchi Seas – generally exhibited ice cover that exceeded 50% (up to 9/10 in some locations) in winter and relatively little (e.g., <10%) ice cover in summer/fall (Adams and Silber, 2017). Some trips were regional in nature delivering supplies and goods to ports within the Northern Sea Route (Humpert, 2014; Zhang et al., 2016) or Northwest Passage (Haas and Howell, 2015).

A comparatively small number of unique, primarily large vessels (e.g., bulk and container ships, tankers) (**Table 5**) undertook lengthy trips in the Northern Sea Route, a finding consistent with previous studies (Zhang et al., 2016). Some of these ships were on trans-Arctic voyages delivering supplies and materials, transporting oil and gas products, or otherwise connecting distant ports (Humpert, 2014; Dalaklis and Baxevani, 2017). Arctic waterways as tourism destinations was evidenced by numerous passenger cruise ship transits in waters comprising the Northwest Passage and Northern Sea Route, although these vessels made far fewer trips than containerized ships and tankers. The number of vessels engaged in transport of energy products along the Northern Sea Route reflected the presence of a mature oil/gas industry on the Siberian coast.

Varying amounts (36.5–88.9%) of sea surface exposure to vessel traffic along the Northern Sea Route and in the Northwest Passage (including the Greenland and Beaufort Seas here) (**Table 4**) appear related to prescribed, non-diffuse routes and restricted access due to ice cover for at least part of each year. Total water surface extent available for transits was also a factor. In the Davis Strait, for example, a comparatively small body of water, a relatively modest number of trips resulted in over 90% its surface area being exposed to trips. In contrast, in the Greenland Sea, a spatially vast area, numerous trips (fourth highest of all areas studied) exposed only 77% of its surface area to vessel activities, suggesting these trips (many by passenger cruise and research survey vessels) were localized or bounded by certain routes.

Research survey vessels logged a substantial number of hours in the Greenland, Kara, Laptev, and East Siberian Seas (**Table 5**). Much of this, notably in locations such as the Kara (Smith, 2016) and Laptev Seas (Soldatkin, 2017), can be attributed to seismic surveys for sub-seafloor oil reserves evidenced as grid pattern ship tracklines in our sample. Separately, research survey cruises in locations such as the Barents Sea (and Arctic Ocean) were tied to scientific studies of the oceanography, glaciology and marine ecosystems of the region (e.g., Hopkins, 2018), and fisheries stocks research cruises (e.g., Ingvaldsen et al., 2017; Vedenin et al., 2018). Bathymetric surveys in support of Arctic nation's recent claims to Exclusive Economic Zones under the United Nations Convention on the Law of the Sea were also undertaken during our study (e.g., Shimeld and Boggild, 2017). Because they can involve visits to remote sampling locations and may include extended data-collection periods, research and scientific cruises can be long in duration, thereby inflating hours of operation values.

Studies indicate the Northern Sea Route maintains regional and potential international importance (Pierre and Olivier, 2015). However, its economic relevance, especially regarding global trade implications, may remain somewhat limited in the near term due to market conditions (Raspotnik and Stephen, 2013; Lee and Kim, 2015; Zhang et al., 2016) and lingering

concerns regarding navigational hazards (Stephenson et al., 2014). Voyages may be similarly limited in the Northwest Passage at present, particularly as significant ice hazards still exist (Haas and Howell, 2015).

Nonetheless, commercial and logistical boundaries along these two major shipping routes are currently being tested with real trials. In July 2016, the Yong Sheng, an ice class ship owned by Chinese shipping giant COSCO, made its third trip (its first two voyages having taken place in 2013 and 2015) delivering wind turbine equipment from China to the United Kingdom through the Northern Sea Route, and then returned via this route (Humpert, 2016). The company sent several additional general cargo vessels along this same passage that same year. In addition, in fall 2018, the Venta Maersk became the first container ship to travel (albeit while still being aided by an icebreaker for a portion of the trip) from Vladivostok along the Northern Sea Route, via portions of the Bering, Chukchi, East Siberian, and Barents Seas, to the German port of Bremerhaven (Reuters News, 2018). In the same period, two LNG-powered bulk carriers owned by Finnish shipping company ESL bore general cargo loads from Japan, westbound along the Northern Sea Route, to the Swedish port of Oxelösund (also icebreaker-aided) (Corkhill, 2018). In August 2016 a cruise ship made a journey from Seward, AK to New York, NY, the first of its class to traverse the Northwest Passage (Dennis and Mooney, 2016). Tens of cruise ships capable of long Arctic voyages now exist, with several more being added each year (Nilson, 2018).

Cases involving these "firsts" notwithstanding, overall, we observed relatively small increases in trips in these key Arctic passageways. However, as changes in environmental conditions evolve toward greater accessibility, sea-borne industries now enter a critical phase of testing increased use of these areas. It is precisely these areas that may be experiencing the most growth in the coming years and may be most ecologically vulnerable as maritime activities expand.

Last, the truly polar Arctic Ocean consisted of the greatest overall surface area in our study and was characterized by comparatively low (but, not zero) ship traffic volume (<1,500 trips annually). Being ice-bound much of the year travel is limited in the Arctic Ocean and limited amounts (<25%) of the water surface was exposed to vessel traffic each year. Here, passenger cruise ships, research survey vessels, and ice-beakers, in that order, made most trips in the 3 years study period. These voyages were also highly-seasonal, most having occurred primarily June to October.

Sea Ice Cover

Amounts of ice on the sea surface and in the water column are important determinants in the location and extent of Arctic vessel traffic. Globally, the highest recorded annual global temperature occurred in 2016 (U.S. Global Change Research Program [USGCRP], 2017), exceeding a 139-year annual average by 0.95°C (NOAA, 2019b). The Earth's historical annual average temperature was surpassed in 2017 by 0.85°C. Related, in 2016 Northern Hemisphere snow and ice extent was the lowest recorded in the period 1979 to present (NOAA, 2019a), a substantial decline from 2015 coverage. Snow/ice cover was 2% greater in 2017 relative to 2016, but these 2 years represented the lowest levels in the last 39 years (U.S. Global Change Research Program [USGCRP], 2017). Likewise, in our study area sea ice coverage changed minimally (ca. <2% between years) in the course of our study (National Snow and Ice Data Center [NSIDC], 2018). Thus, relative amounts of Arctic-wide ice coverage may have been only one factor influencing observed inter-annual changes in volume and distribution of vessel traffic patterns.

Three-Year Trends

Numerous sources have indicated declining ice cover will prompt increased maritime activities in historically ice-limited areas. Sea ice extent almost certainly had a role in the observed number of trips observed in some locations (e.g., the Northern Sea Route) and by some vessel classes. However, we observed no overt increasing or decreasing trends in vessel traffic volume in our limited study period. For example, as noted, although the overall number of voyages steadily increased during our study, distances traveled and time underway decreased. Instead, a more apparent pattern involved considerable regional inter-annual variability in operations. This variation confounds conclusions about region-wide activities and defy simple, single-factor explanations regarding trends and their causes.

A suite of variables, nearly all of which are beyond the scope of this study, likely had roles in Arctic maritime activity levels. These might include, for example, abundance, availability, and locations of fisheries resources (e.g., Watson and Haynie, 2018); market forces (e.g., oil/gas demand and prices); global economic factors effecting intercontinental trade (e.g., Eurasia Group, 2014; International Energy Agency [IEA], 2019); and economic feasibility relative to navigational safety of high Arctic marine travel (e.g., Zheng et al., 2016). As one example, in late-2015 Royal Dutch Shell abandoned pursuit of further exploratory drilling in the U.S. Chukchi Sea. This decision resulted from assessments of expected risks and high project costs relative to potential return (i.e., comparatively low oil prices at the time) after finding insufficient indications of crude in the region (Schaps, 2015).

Potential Impacts

The release of air-borne emissions from shipping activities has become a growing concern for high-latitude marine transportation (Dalsøren et al., 2007; Johansson et al., 2017). Amounts of aerosols and gaseous emissions, including carbon dioxide, are expected to increase with increased shipping activities thereby accelerating the melting of ice and snow (Corbett et al., 2010). Air-borne pollutants and greenhouse gases are also emitted as by-products of ongoing oil and gas extraction activities (Peters et al., 2011). In addition, scores of non-indigenous species have been detected in Arctic ballast water exchange operations (Miller and Ruiz, 2014).

The U.S. Coast Guard has indicated an increased likelihood of maritime accidents is expected to accompany expanded Arctic vessel operations (United States Coast Guard [USCG], 2010). Fatal vessel strikes (Huntington et al., 2015; Cooke and Clapham, 2018) and fishing gear entanglement of Arctic marine mammal species (George et al., 2017) are reported conservation concerns. Adverse impacts to fish and marine mammal populations are expected as industrial activities increase

(Reeves et al., 2012) in turn impacting indigenous people reliant on the region's resources (Huntington et al., 2015). These impacts may compound the vulnerability of Arctic species as they undergo shifts in distribution and habitats change as a result of rapidly changing Arctic ecosystems (Silber et al., 2017).

Our findings indicate waters exposed to at least some vessel traffic exceeded 75% in nearly all areas studied and exposure levels exceed 90% in 5 of the 14 areas (**Table 4**). Associated vessel-related impacts, such as those related to introduced noise, may follow. Anthropogenic underwater noise sources may alter marine mammal intra-species acoustic signaling behavior (Moore et al., 2012; Fournet et al., 2018) or disrupt normal behavior (Gordon et al., 2003). These noise sources and other stimuli may compound stressors already impinging on Arctic marine mammal populations as a result of climate change itself (Ragen et al., 2008; Kovacs et al., 2011).

The possibility of an uncontained, large-scale oil spill may be the single greatest threat to Arctic ecosystems (Afenyo et al., 2017; Congressional Research Service [CRS], 2018). Effects from a spill on biological communities could be severe and long-lasting. And, while local effects may linger, high winds and moving ice fields may transport surface oil to locations other than a spill site. Among other potential impacts, Arctic seabird and mammal species (e.g., polar bears, Aleutian sea otters) are highly vulnerable to oiling whereby thermoregulation and buoyancy are effected (e.g., Renner and Kuletz, 2015). Yet, spill response infrastructures and contingency planning are generally regarded as inadequate in many poorly supported locations where severe and unpredictable weather conditions can prevail (e.g., National Research Council of the National Academies of Science [NRC], 2014; Wilkinson et al., 2017). The plausibility of a substantial spill appears high given the levels of activity devoted to oil/gas industries in our findings.

To be sure, the severity of some potential impacts identified here has not been fully quantified in all cases, particularly at the population level. However, given projections for growth of vessel operations, it is reasonable to assume that potential threats may become realities.

Fishing Vessels and Effort

Multi-billion-dollar commercial fisheries exist in the Norwegian, Bering, and Barents Seas for various species of cod, halibut, capelin, pollock, salmon, herring, and crustaceans, among others. Norwegian and Barents Sea fisheries resources destined for markets throughout Europe, Russia and Asia are supported by a vast network of fishing vessels, trawlers, and vessels engaged in processing and transporting fish products (Food and Agriculture Organization of the United Nations [FAO], 2011). Crustacean and groundfish fisheries for Walleye Pollock (*Gadus chalcogrammus*), Pacific Cod (*Gadus macrocephalus*), Atka Mackerel (*Pleurogrammus monopterygius*) among other species render the Bering Sea one of the most productive fishing regions in the world.

As measured by number of trips, unique vessels, and hours of operation, fishing vessel operations, collectively, dwarfed activity levels by all other vessel classes. Our metrics also attest to the magnitude and breadth of these fleets. For example, in the Bering Sea, alone, fishing vessels accounted for 10–17% of all vessel operating hours Arctic-wide (**Table 3**).

But industry activities were not static. Availability of fisheries resources in Arctic waters can be variable (e.g., Haug et al., 2017; Troell et al., 2017). Therefore, the extent and locations of vessel operations associated with these resources will likewise vary. Observed inter-year changes in spatial distribution may have reflected shifts in targeted species locations (**Figures 3A–C**) in turn having been driven by changing prey distribution and other factors. Evidence indicates that commercially important fish species are undergoing northward distributional shifts in response to warming water temperatures (e.g., Fossheim et al., 2015; Misund et al., 2016) and fishing practices are adjusting to these ecosystem-level changes (Watson and Haynie, 2018). The industry also responds to changing markets, seeking to maximize profits depending on fish products sought or proximity of processing centers (Watson and Haynie, 2018). Diminishing sea ice levels may also contribute to the emergence of, and increased access to, previously inaccessible fishing locations. In short, a malleable industry was apparent in our sample.

An abrupt decline in hours of operation logged in the Norwegian and Barents Seas noted here may be at least in part related to a substantial reduction in availability of capelin (*Mallotus villosus*), an integral part of this ecosystem (ICES, 2017). Landings for this species declined from 11,500 landed tones in 2015 to zero in 2016 and 2017 (but subsequently increased in 2018) (ICES, 2017). Some inter-year variability in fishing trips may also be attributed to changes in catch limits for Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*) and other species established each year for Norwegian and Barents Sea by Norwegian-Russian bilateral agreements and the International Council for the Exploration of the Sea (ICES) (Seaman, 2017).

Impacts From Fisheries

The size of the fleet and number of vessels supporting the vast industry suggest impacts from the industry may be significant. While typically smaller in size than large container, tanker, or bulk carrying ships, fishing vessels nonetheless contribute a proportional share to air-borne emissions (Dalsøren et al., 2007; Roiger et al., 2015), radiated underwater noise (Hovem et al., 2015; Peng et al., 2017), or small fuel spills during routine operations or accidents (International Maritime Organization [IMO], 1988; Richardson et al., 2016). Active and "lost" gear may entangle Arctic-dwelling marine mammals and other marine vertebrates (Reeves et al., 2014; George et al., 2017); and like all ships, fishing vessels are capable of fatally striking marine mammals (Jensen and Silber, 2003).

Factoring in incidental bycatch (e.g., Ianelli and Stram, 2014; Breivik et al., 2017) and the effects of trawling (represented by Stern Trawlers and Factory Stern Trawlers in this study; **Table 3**) on demersal communities (Puig et al., 2012; Christiansen et al., 2014), impacts on ecosystem integrity from this industry are indeed consequential. Effects on targeted species assemblages themselves and related ecosystem impacts compound negative influences on these biological communities (Zeller et al., 2011; ICES, 2018; Popov and Zeller, 2018). Therefore, considering the

volume of fishing operations, as characterized by numbers of trips in our sample, this source may dwarf ecosystem impacts typically ascribed to larger vessels.

Oil and Gas

Intensive vessel activity found in some locations reflected a dynamic Arctic energy industry. Terrestrial and undersea oil deposits in the region are expansive (US Geological Survey [USGS], 2009; Gautier et al., 2011) and they have been accessed in waters of the Norwegian/Barents Seas and off Russian coasts. Year-round ice-free conditions and relatively shallow shelf waters enhance the economic viability of drilling in the Norwegian and Barents Seas. Over 30 exploratory wells were drilled on the Norwegian Shelf in each of 3 years, 2016–2018 and 11 new discoveries were made in 2017 (Norwegian Petroleum Directorate, 2018). In the Barents Sea, alone, 15 new exploratory oil wells were drilled in 2017 north of Hammerfest, Norway (Norwegian Petroleum Directorate, 2017).

Vessel operations related to oil/gas activities were strongly represented in our sample. For example, plots of ship tracks indicated hubs of oil and gas vessel activities in Russia's Kara Sea (**Figures 2A–C**) in support of the Prirazlomnoye oil field, south of Novaya Zemlya. Production from the field began in December 2013. By the end of 2017, over 41 million barrels of oil had been extracted from the site (Gazprom Neft, 2018). In addition, construction was completed at the Russian seaport of Sabetta in mid-2016, one of the most complex LNG projects ever undertaken (Dalaklis et al., 2018), to handle LNG and gas condensate from the Yamal Peninsula. Shipments of LNG from the facility began in late 2017, and by August 2018 over 4 million tons had been removed (World Maritime News, 2017). Transport of gas from these facilities, across the Barents Sea to Murmansk, Russia (**Figures 2A–C**) via ice breaking shuttle tankers occurs year-round (Offshore Technology Focus, 2018).

Seismic survey vessels, shallow and deep-water ice islands, and ice-strengthened drillships were in use in the Alaskan and Canadian Beaufort Sea throughout the 1970s and 1980s (Timco and Frederking, 2009). By 2000, hundreds of exploratory or producing wells had been opened in the region. While at present only the Beaufort Sea has producing wells in U.S. federal waters (Bureau of Ocean Energy Management [BOEM], 2018), the region is expected to remain of interest to oil companies. For example, in December 2017 the Italian oil producer Eni began drilling a new well from an existing man-made island in the U.S. Beaufort Sea, becoming the first company to do so since 2015 (Congressional Research Service [CRS], 2018). Whereas energy production may one day ramp up again in the Beaufort Sea, we found some of the lowest levels of vessel transits and hours underway in this location.

Impacts From Oil/Gas Industries

Our findings indicate energy products are on the move on large scales and traversing great distances in some Arctic waters, thereby contributing to the vulnerability of marine environments. Potential for large spills from oil- and gas-bearing vessels are a serious concern (Arctic Council, 2009; National Research Council of the National Academies of Science [NRC], 2014). Indications are insufficient infrastructure and resources exist to respond to, contain, or clean spills in remote and hazardous locations (National Research Council of the National Academies of Science [NRC], 2014; Nevalainen et al., 2017; Wilkinson et al., 2017). In addition, long residency times of spilled oil (Hazen et al., 2016) especially in cold-water ecosystems (Engelhardt, 1985; Vergeynst et al., 2018), and long-term entrainment in marine biological processes (Hicken et al., 2011; National Research Council of the National Academies of Science [NRC], 2014; Yuewen and Adzigbli, 2018) contribute to the potential severity of impacts from a large spill.

Seismic survey exploration of sub-seafloor oil reserves introduces loud sounds into the water column. Pulsed seismic signals are among the most powerful man-made sound sources (Hildebrand, 2009). Exposure to these sounds have consequences for marine mammal populations and other organisms (Nowacek et al., 2015; Kyhn et al., 2019) by disrupting normal behavior (Richardson et al., 1995), eliciting startle responses (Heide-Jørgensen et al., 2013), and resulting in temporary and permanent hearing loss (National Marine Fisheries Service [NMFS], 2018).

Global reliance on fossil fuels assures energy production will remain a feature in Arctic waters for the foreseeable future. However, in the past, market forces have influenced the levels of Arctic oil industry activities (Harsem et al., 2015). For example, a 2007/2008 global decline in oil demand and prices slowed oil/gas production in the Arctic (Petrick et al., 2017). Economic conditions akin to the 2007/2008 downturn, or other factors, may once again slow production in the future (Lindholt and Glomsrød, 2012). Inherent hazards associated with existing long supply lines through inhospitable natural environments are areas of legitimate concern. Severe ecosystem degradation may follow.

Conservation Measures

The Polar Code

In January 2017, the International Maritime Organization's (IMO) Polar Code went into effect. Intended to protect unique environments and ecosystems of polar regions, the Code provided new requirements for vessel navigational safety and pollution prevention by addressing new ship design; operations and mariner training; and search and rescue capabilities. Years in the making, the Code applies to passenger and cargo ships >500 gt engaged in international voyages in polar regions (but does not apply to fishing vessels, pleasure yachts, or military vessels) (International Maritime Organization [IMO], 2014). At the time of this writing, the IMO's Sub-Committee on Ship Design and Construction is considering amendments to the Code to include recommended safety measures for fishing vessels ≥24 m in length operating in polar waters and pleasure yachts >300 gt (not engaged in trade)[4].

The Polar Code was a landmark set of regulations and guidelines for limiting impacts from shipping. However, some maintain it might have established far more stringent protection

[4]http://www.imo.org/en/MediaCentre/MeetingSummaries/MSC/Pages/MSC-100th-session.aspx

measures (Friends of the Earth [FOE], 2016; Ocean Conservancy, 2017; Roach, 2018). Nonetheless, as an already IMO-adopted instrument, strengthening the Code's existing framework and its measures in response to more fully understood adverse effects would be commendable.

Routing Measures

Potential increases in north-south flowing vessel traffic through the Bering Strait prompted the IMO to adopt routing measures in the region to increase navigational safety. Jointly submitted to the IMO by the Russian Federation and United States and adopted by the IMO's Maritime Safety Committee in late-2017, the measure (International Maritime Organization [IMO], 2017) established six "two-way routes" and six precautionary areas (for all ships >400 gt, excluding fishing vessels) in the Bering Strait and Bering Sea between the Chukotskiy Peninsula and Alaskan coast. The adopted measure also includes Areas to be Avoided around St. Lawrence Island, King Island, and Nunivak Island (Hobson, 2018).

In its analysis of the routing measures, the U.S. Coast Guard indicated "increased cargo traffic, passenger ship traffic, adventure tourism traffic, oil and gas exploration, and research and scientific activities" would increase the likelihood of maritime casualties and threaten environments inhabited by endangered marine species and remote indigenous communities that rely on traditional subsistence activities (United States Coast Guard [USCG], 2014). Therefore, anticipated benefits of this action include helping ships avoid numerous hazardous shoals, reefs and islands; reducing "...risk of pollution or other damage to the marine environment, including national and international recognized habitat and species; and avoiding the key areas of fishing activities and areas of subsistence activities by local indigenous communities..." It is generally accepted that these measures will also contribute to the reduction of vessel strikes of large cetaceans such as bowhead (*Balaena mysticetus*), gray (*Eschrichtius robustus*), and North Pacific right (*Eubalaena japonica*) whales (Allen, 2014; Huntington et al., 2015).

However, extensive use of the international Unimak Pass and Bering Sea route by large vessels (**Figures 2A–C**), along with activities of the spatially diffuse commercial fishing industries (**Figures 4A–C**), may have significant consequences for biological communities occurring in this region. Numerous endangered mammal species occur in this area including bowhead and the highly depleted North Pacific right whales and other whale species. Critical habitat for right whales (National Oceanic and Atmospheric Administration [NOAA], 2008) and conservation zones for the endangered Steller sea lion (*Eumatopis jubatus*) has been established in this area (National Oceanic and Atmospheric Administration [NOAA], 1993). Therefore, east-west vessel traffic near the Aleutian Islands likely rivals or exceeds the threat of ship strikes of endangered marine mammals from the relatively smaller amount of traffic on north-south trips through the Bering Strait and Bering Sea. As such, inter-continental vessel routes near the Aleutian Islands should also be candidates for imposing conservation measures to minimize impacts to marine mammals and sensitive ecosystems near the archipelago.

Oil Spill Contingency Planning

At this time, planning and preparation for a major oil/chemical spill in Arctic waters is inadequate. Cognizant of the deficiencies, several Arctic nations have met jointly or attempted independently to increase response and remediation readiness. Fortified and double-hulled tankers are routinely in use by oil companies and subsidiaries as a safeguard against spills. However, existing contingencies for such an event are almost certainly insufficient. International bodies such as the Arctic Council (Arctic Monitoring and Assessment Programme [AMAP], 2008; Arctic Council, 2009) and the IMO have highlighted and attempted to address these shortcomings. These efforts are commended, should be continued or accelerated, and where feasible, resources provided to prepare for the occurrence of a spill of significant magnitude.

CONCLUSION

Vessel operations were extensive in volume and diverse in the types (e.g., long-range shipping, fishing) and location of activity. The observed already substantial maritime activity is projected to increase. Our study period was too brief to demonstrate significant growth in maritime activities; instead, inter-annual variation prevailed. While receding ice coverages have created new opportunities for increased use of these waters, expansion of Arctic maritime traffic may not be tied solely to receding amounts of ice-covered waters. Decisions regarding utilization of new or expanded routes are likely to be a function of logistical and economic factors.

Vessel traffic volume is somewhat limited in some areas. However, maritime transport operations will likely experience the greatest growth in locations undergoing the most rapid increases in seasonally ice-free zones, which includes water bodies along the Northern Sea Route and the Northwest Passages. Increased use of these locations will be driven by assessments of economic feasibility weighed against inherent risks to hazardous high-latitude travel. These boundaries are currently being tested with real trials (both with and without ice breaker assistance) in locations with historically limited access. Cost/benefit ratios will shift with global demand for certain goods and resources (e.g., oil), and logistical costs may diminish as needs for ice-breaker assistance is diminished. A modest growth of the tourist cruise industry occurred during our study and will likely undergo additional growth as certain destinations become increasingly accessible. Commercial fisheries operations, including those engaged in stern trawler fisheries, are largely year-round in many locations and spatially vast primarily in the lower latitudes. By volume alone, these industries may have the greatest overall impact (e.g., bycatch, disruption of benthic communities) to Artic marine biological processes. Therefore, considerations regarding ecosystem protection should include direct and indirect impacts from fishing activities.

Exact threats to Arctic marine ecosystems are not known with certainty especially as related to population levels impacts. However, increased use of this region by trans-Arctic shipment of goods, oil/gas exploration and other commercial enterprises is accompanied by concurrent risks to Arctic ecosystems. Limited supporting infrastructures and constraints of response capabilities in the case of a large oil spill suggest existing protective measures are probably inadequate and should be strengthened. AIS data have been used to support maritime security, vessel traffic monitoring, and regulatory functions. Coupled with baseline descriptions of the volume and spatial distribution of Arctic-wide commercial activities, the AIS can aid in characterizing trends in Arctic vessel operations and assessing potential threats to Arctic marine ecosystems. Only through monitoring of trends in vessel activities, particularly those areas where growth is most likely to occur, will assessments of impacts be possible. As understanding of maritime activity levels and concurrent environmental impacts improves, so should a strengthening of ecosystem protection measures.

DATA AVAILABILITY

All datasets generated for this study are included in the manuscript and/or the supplementary files.

AUTHOR CONTRIBUTIONS

GS and JA helped to build the NOAA Fisheries AIS program, jointly conceived this analytical project, and worked collaboratively to complete it. JA conducted most of the data analysis. GS prepared most of the text.

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Active Whale Avoidance by Large Ships: Components and Constraints of a Complementary Approach to Reducing Ship Strike Risk

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The recurrence of lethal ship-whale collisions ('ship strikes') has prompted management entities across the globe to seek effective ways for reducing collision risk. Here we describe 'active whale avoidance' defined as a mariner making operational decisions to reduce the chance of a collision with a sighted whale. We generated a conceptual model of active whale avoidance and, as a proof of concept, apply data to the model based on observations of humpback whales surfacing in the proximity of large cruise ships, and simulations run in a full-mission bridge simulator and commonly used pilotage software. Application of the model demonstrated that (1) the opportunities for detecting a surfacing whale are often limited and temporary, (2) the cumulative probability of detecting one of the available 'cues' of whale's presence (and direction of travel) decreases with increased ship-to-whale distances, and (3) following detection time delays occur related to avoidance operations. These delays were attributed to the mariner evaluating competing risks (e.g., risk of whale collision vs. risk to human life, the ship, or other aspects of the marine environment), deciding upon an appropriate avoidance action, and achieving a new operational state by the ship once a maneuver is commanded. We thus identify several options for enhancing whale avoidance including training Lookouts to focus search efforts on a 'Cone of Concern,' defined here as the area forward of the ship where whales are at risk of collision based on the whale and ship's transit/swimming speed and direction of travel. Standardizing protocols for rapid communication of relevant sighting information among bridge team members can also increase avoidance by sharing information on the whale that is of sufficient quality to be actionable. We also found that, for marine pilots in Alaska, a slight change in course tends to be preferable to slowing the ship in response to a single sighted whale, owing, in part, to the substantial distance required to achieve an effective speed reduction in a safe manner. However, planned, temporary speed reductions in known areas of whale aggregations, particularly in navigationally constrained areas, provide a greater range of options for avoidance, highlighting the value of real-time sharing of whale sighting data by mariners. Development and application of these concepts in modules in full mission ship simulators can be of significant value in training inexperienced mariners

by replicating situations and effective avoidance maneuvers (reducing the need to 'learn on the water'), helping regulators understand the feasibility of avoidance options, and, identifying priority research threads. We conclude that application of active whale avoidance techniques by large ships is a feasible yet underdeveloped tool for reducing collision risk globally, and highlight the value of local collaboration and integration of ideas across disciplines to finding solutions to mutually desired conservation outcomes.

Keywords: vessel strike, active whale avoidance, ship operations, speed, detection probability

INTRODUCTION

Lethal collisions between large ships and large whales (ship strikes) are a recurring and common threat to whale populations across the globe (Thomas et al., 2016). In some cases, such as with the critically endangered North Atlantic right whales (Fujiwara and Caswell, 2001), and an important sub-population of sperm whales in the Canary Islands (Fais et al., 2016), ship strikes have direct implications for population persistence and biodiversity. In other cases, such as with the population of blue whales in the eastern North Pacific, ship strikes do not appear to regulate population dynamics given the frequency of (known) ship strike mortalities (Monnahan et al., 2015), although the number of detected collisions may be an underestimate of the true number that occur (Rockwood et al., 2017). Regardless, management agencies and the general public value large cetaceans and seek effective ways to reduce ship strikes, even when population persistence is not at stake (Gende et al., 2018).

To date, most management efforts aimed at reducing ship strike risk have focused either on modifying shipping lanes, which can reduce the relative and absolute risk of strikes by reducing spatial and temporal overlap between ships and whales (Knowlton and Brown, 2007; Vanderlaan et al., 2008; van der Hoop et al., 2015), and/or reducing ship speed, which may reduce the probability of a collision (Conn and Silber, 2013) or the likelihood of mortality should a collision occur (Vanderlaan and Taggart, 2007). Yet each of these approaches has limitations. Modifying shipping lanes will only be as effective as the spatial persistence of whale aggregations, can require considerable regulatory effort, or may be impractical in narrow straits or for ships arriving into ports of call (Webb and Gende, 2015; Monnahan et al., 2019). Speed restrictions can generate resistance from the shipping industry owing to economic implications of the additional at-sea time that results from lower speeds, particularly when applied over large areas, which may be one reason voluntary reductions in speeds tend to have low compliance (McKenna et al., 2012). Regardless, whales can be notably unresponsive to approaching ships (Nowacek et al., 2004; McKenna et al., 2015), and thus any action that facilitates the avoidance of whales by mariner training and active avoidance techniques (lowering the reliance on whales to avoid ships) are important to develop.

Here we describe active whale avoidance by mariners aboard large ships which serves as a complementary, but comparatively underexplored, means to reduce whale strike risk. Active whale avoidance is defined here as a mariner making operational decisions, such as a course change or speed reduction, with the goal of reducing the chance of a collision with a sighted whale. Active avoidance differs from more 'passive' regulatory approaches in that the risk- reducing action is primarily initiated by the mariner upon sighting of a whale surfacing forward of the ship as opposed, for example, to a ship entering a mandatory speed reduction area which requires a change in operational state independent of whether a whale is present in the area and/or at risk of collision.

Active whale avoidance has been developed and successfully practiced for decades by marine pilots in Alaska (and possibly elsewhere) and is not new in the maritime community. However, a more formal exploration will help clarify (1) the development and application of these techniques by other mariners, (2) the regulatory language that makes implicit or explicit assumptions about a ship's ability to avoid whales, and (3) important research questions with regard to the efficacy and effectiveness of different maneuvers under varying operational and environmental conditions. For example, the U.S. Code of Federal Regulations (50 CFR §224.103) states that it is illegal to approach [North Atlantic] right whales closer than 500 yards (457 m) with some exceptions for vessels 'restricted in her ability to maneuver.' In Alaska, federal regulation dictates that all vessels must operate at a 'slow, safe speed when near a humpback whale' (50 CFR §223.214) which assumes that the ship can take proper and effective action to avoid collision when near a humpback whale or that ship operators have advance knowledge of where whales are located. 36 CFR §13.1170 stipulates that a vessel in Glacier Bay inadvertently positioned within 1/4 nautical mile of a whale must "immediately slow the vessel to ten knots or less without shifting into reverse", and "direct or maintain the vessel on as steady a course as possible away from the whale until at least 1/4 nautical mile of separation is established" – requirements that were largely established pertaining to smaller craft and may be unattainable by large ships.

Understanding the opportunities for, and feasibility of, active whale avoidance also serves to benefit mariners by clarifying conditions and actions that may facilitate effective whale avoidance. For example, large ship operators undergo years of training, including frequent maneuver testing in full-mission bridge simulators, which are often focused on collision avoidance with objects including reefs, shoals, and other vessels. Yet we know of no simulator modules for whale avoidance, which would provide opportunities for mariners to learn from others and test new ideas for maneuvering, particularly if they incorporated state-of-the-science information pertaining to whale behavior.

Finally, clarifying research needs and models derived from active whale avoidance will help scientists prioritize and/or refine existing efforts that will have tangible conservation outcomes and assist mariners in applications of these concepts. For example, a suite of efforts currently exist to facilitate mariners sharing information on whale sightings yet it's unclear how well these sightings equate to changes in maritime operations and, ultimately, whether certain factors, such as the way the information is transmitted or when its received by the operator, equates to a reduction in ship strikes.

Our goal is to present a conceptual model of active whale avoidance derived by coupling perspectives from biologists, focused on the science of whale behavior, with the expertise of ship operators. To that end, our research team included Alaska marine pilots with over 90 years of combined experience developing and practicing active whale avoidance while piloting large ships. As proof of concept, we collected and applied data to our conceptual model focused on avoidance of humpback whales by large cruise ships transiting waters in Alaska. Data informing our conceptual model originated from (1) a study that has placed observers aboard large cruise ships in Alaska since 2006 focused on quantifying surfacing behavior of humpback whales around the ships and the ability of mariners to detect them; and (2) data collected during trial simulations in a full-mission bridge (ship) simulator to identify and quantify the practices that occur on the ship's bridge during active whale avoidance. Large ship maneuvering capabilities were further explored using SEAiq, a navigation software commonly used by marine pilots to navigate and assess maneuvering possibilities[1]. Although our work is focused on a specific type of ship (large cruise ships) and single species of whale (humpback), variations of the components of our conceptual model can be applied to whale avoidance by other types of ships and other types of whales.

We emphasize that our goal is to generate a conceptual foundation upon which specific processes, such as the relationship between whale surfacing distance and appropriate maneuver response, can be subject to more rigorous testing and replication. To that end, our findings (at this stage) are not intended to prescribe what mariners should (or shouldn't) do when in the vicinity of surfacing whales. Instead, we draw some more general but important inferences from our conceptual model and related data including the role of ship operations (e.g., speed and heading variables) in active whale avoidance. Ultimately we hope these ideas will help advance the development and application of active whale avoidance techniques on a global scale.

MATERIALS AND METHODS

Our goal for this paper was to present (1) a conceptual model of active whale avoidance, and (2) provide a proof of concept by utilizing empirical data of humpback whale surfacing behavior collected from the bow of cruise ships and from simulations of large cruise ship operations in a full-mission bridge simulator and via commonly used pilotage software. The conceptual model, generated to help deconstruct this complex and highly variable process into components that could be informed by data, was developed during a series of meetings conducted since 2013 between a team of State of Alaska marine pilots from the Southeast Alaska Pilots' Association (SEAPA), and scientists from Glacier Bay National Park, where ship strike reduction efforts have been implemented and refined since the early 1980s. The conceptual model is presented first (**Figure 1**) by describing each of the constituent processes, and factors that influence them. Components include availability and detection processes, reflecting how often and how long whales are available to be detected, and the ability of mariners to detect them once available; and command and maneuver processes, reflecting the procedures that occur on the bridge once a whale is detected, and the ability of a ship to achieve a new operational state commanded by the mariner that reduces collision risk. These components are based upon existing literature (e.g., availability and detection processes) and the collective experience of marine pilots (command and maneuver). To that end, the 'results' of the conceptual model include narrative describing how and why certain factors are important, particularly as it relates to ship operations and maneuvering, including events that transpire on the ship's bridge when a whale surfaces and is detected forward of the ship. For our proof of concept, data collection procedures are organized according to the different components of the conceptual model. While more details on the field-based methods can be found elsewhere (see Gende et al., 2011; Harris et al., 2012; Williams et al., 2016) they are described briefly below.

Availability of Whales for Detection

In the context of active whale avoidance, whales need to be available for detection in order to be avoided. Thus the availability is dictated by the type, frequency, and duration of the opportunities for perception by the mariner. In Alaska, humpback whales (and other whale species) regularly embark on a repeated cycle of a foraging dive punctuated by a surface interval. For clarification we define a *surface interval* as the time the whale first comes to the surface following a dive to the time it embarks on another dive. Therefore the surface interval encapsulates one to many *surfacing events* defined as when the whale breaks the surface of the water to respire. Surfacing events are separated by brief submergences (e.g., Dolphin, 1987; Stelle et al., 2008; Godwin et al., 2016; Garcia-Cegarra et al., 2019). During each surfacing event (surfacing) the whale may provide multiple '*cues*' that can be perceived by the mariner to infer the whale's distance from the ship and direction of travel (Hiby and Ward, 1986). Cues include spouts/blows/breaths and presentation of the head, dorsal fin, back, or tail (flukes) breaking the surface. Cues are available for only a second or two, occur in rapid succession, and often overlap in time (such as when the water vapor from a spout lingers long enough to be visible when the whale's flukes break the water's surface). In contrast, the surfacing events are separated by submergences that may

[1] http://seaiq.com/

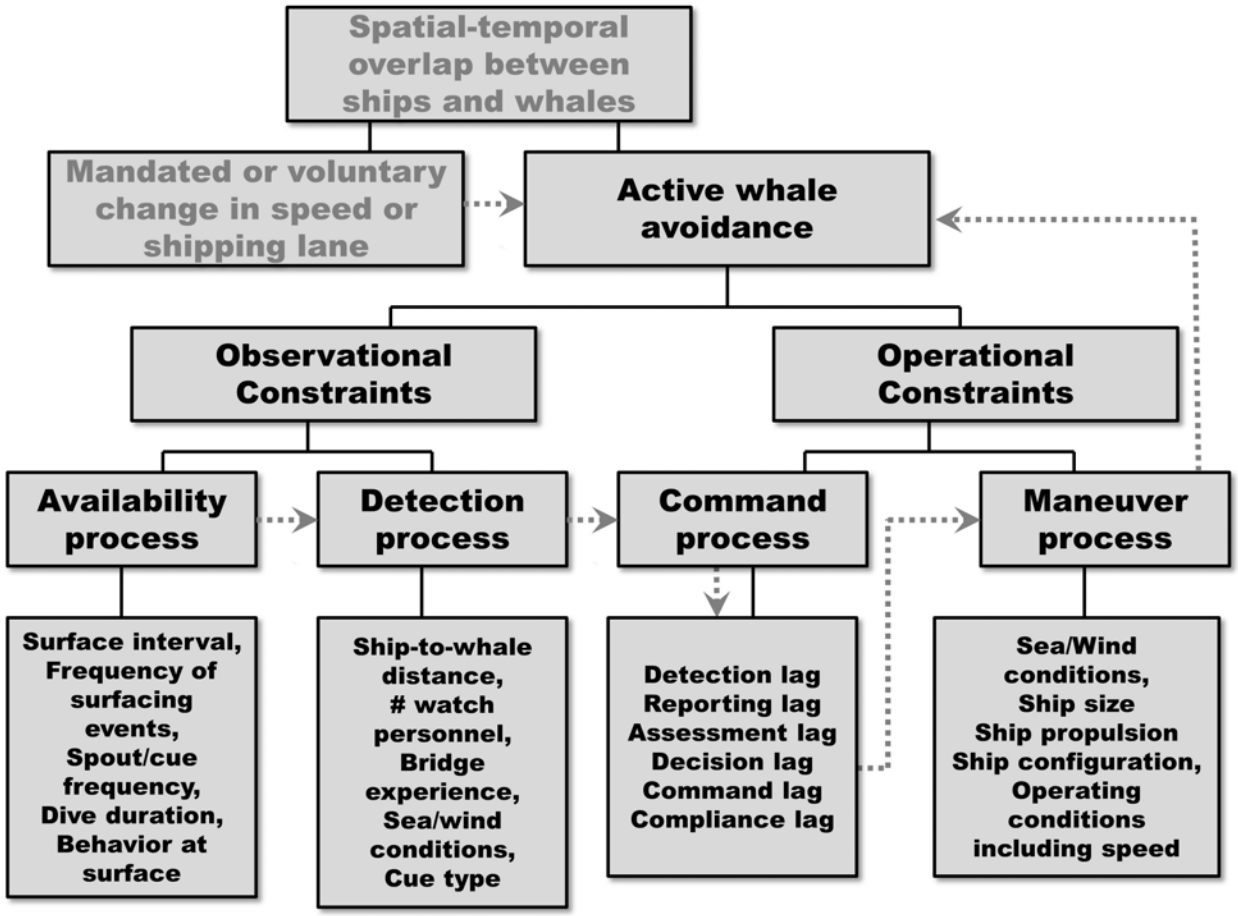

FIGURE 1 | Conceptual model depicting some components related to active whale avoidance by mariners aboard large ships. The first two rows reflect opportunity: active whale avoidance can be practiced anywhere in the world where there is overlap between large ships and large whales, including within management areas that already require (or encourage voluntary) speed reductions or within shipping lanes that have been altered to minimize ship-whale encounters. Lowermost boxes represent a sampling of important factors that may influence their related processes either singly or in concert with other factors. The arrows among the lower boxes reflect the general chronology of whale avoidance.

last 20–40 s or more, during which time the ship will move up to several hundred meters closer to the whale (depending upon speed). The change in ship-to-whale distances between cues (within a surfacing event) will thus be inconsequential (meters) whereas the change in distances between surfacing events will be sufficiently large to affect the probability of detection (see below).

To understand the nature by which humpback whales become available to be detected by mariners, we utilized data collected as part of an ongoing study that has placed an observer aboard large cruise ships in Alaska since 2006 (**Figure 2A**) to estimate (1) the frequency and duration of surfacing events throughout a surfacing interval, and (2) the probability that one or more surfacing events will be detected. We briefly summarize the relevant methods of whale detection here, but reference previously published work (Gende et al., 2011; Harris et al., 2012; Williams et al., 2016) containing more details on data collection and processing protocols.

Surfacing Behavior of Humpback Whales Near Cruise Ships

During the summers (May–September) of 2016 and 2017 a marine mammal observer embarked on $N = 67$ large cruise ship cruises (mean length = 268 m; Gende et al., 2018) while the ship transited the waters in Glacier Bay National Park, Alaska. The observer was transported out to the cruise ships just after it entered the park (only 1 or 2 ships entered per day) and boarded the ship via an NPS transport vessel. Regardless of weather, the observer proceeded to the bow (the forward-most point of the ship; **Figure 2B**) and conducted continuous naked-eye scans of the water in a 180-degree arc from directly forward to directly abeam, on both sides of the ship. Scans were assisted using Swarovski 10 × 42 binoculars and tripod-mounted laser rangefinder binoculars (Leica Viper II; accuracy + 1 m at 1 km; Leica, Charlottesville, VA, United States) to search for whales.

When the observer detected a humpback whale, the ship's position was recorded using a Global Positioning System

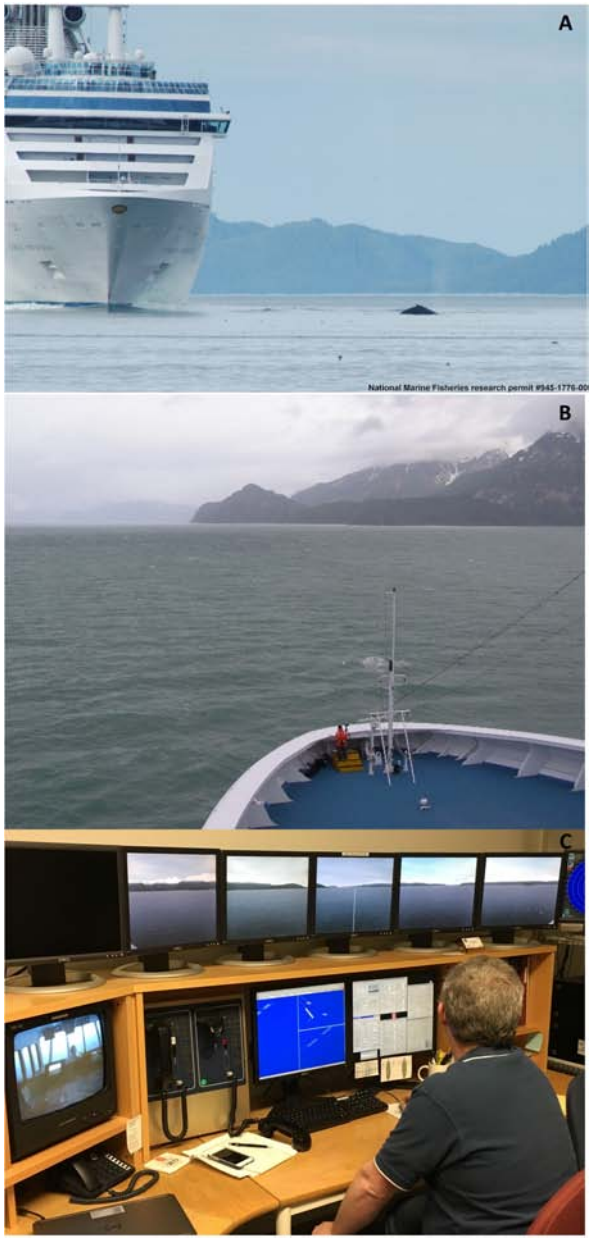

FIGURE 2 | (A) A humpback whale surfaces in front of a large cruise ship, Glacier Bay, Alaska. **(B)** An observer standing at the bow of a large cruise ship in Alaska quantifying the frequency, proximity, and behavior of humpback whales that surface forward of the ship. The Observer program has occurred since 2006 and included more than 750 cruises. **(C)** The command center at the AVTEC Full Mission Bridge simulator in Seward, Alaska, during simulations whereby marine pilots simulated whale encounters and active whale avoidance. Closed-circuit video of 2 pilots in the bridge room can be seen in lower left of the photo enacting a whale avoidance maneuver.

(Garmin 76Cx GPS, Olathe, KS, United States), and the distance between the observer and the whale was measured using tripod-mounted laser rangefinder binoculars, or estimated if the observer could not 'ping' the whale with the rangefinder. Based on training and testing throughout the study, estimated distances were deemed unbiased, and typically within 10% of the true distances (Williams et al., 2016). The relative bearing of the surfacing event was recorded using a tripod-mounted protractor along with group size, cue type (spout, fluke up, etc.), direction of travel, and sighting conditions (see Williams et al., 2016 for complete list). All data were recorded using a voice-activated recorder and transcribed following each cruise. Data were then summarized using (1) only whales with a group size of 1 (i.e., singletons) to ensure that surfacing events were not mixed in multi-whale groups (singletons constituted 91% of all groups detected in 2016 and 2017), and (2) only from a single surfacing interval per whale to insure independence. Owing to the speed of the ships (typically 14–20 kts; Webb and Gende, 2015), and foraging dives often lasting several minutes or more, only one surfacing interval was typically recorded (>90% of all sightings) before the whale passed abeam. To avoid using surface intervals that were ongoing when the whale passed abeam, the total number of surfacing events per surfacing interval was summarized across all of the surface intervals where whale flukes were displayed as the terminal cue (indicating a deep dive). In contrast, the length of submergences between surfacing events were summarized using all surface intervals, regardless of the nature of the terminal cue. Both of these parameters aid in understanding how many surfacing events are available for mariners to detect and the time elapsed between available events.

Probability of Detecting a Humpback Whale During a Surfacing Interval

Detection functions of humpback whales surfacing near cruise ships have been published previously by Williams et al. (2016) who used distance sampling applied to sighting data collected since 2008. Importantly, unlike some studies focused on estimating abundance of whales where detection functions were derived using line transects, Williams et al. (2016) derived detection functions tailored to the question of whale avoidance by using a series of instantaneous samples as point transects, with the ship-to-whale distances analyzed as radial measures from the bow. Accordingly, the proper interpretation of these detection functions is the instantaneous detection probability of a whale that becomes temporarily available at a specific ship-to-whale radial distance across the 180-degree arc forward of the ship.

In the context of active whale avoidance, the relevant inference is the probability the mariner detects at least one of the available surfacing events in a surfacing interval because whales often engage in multiple surfacing events (per surfacing interval) and mariners generally need only to detect one of the events to begin evaluating whether a whale avoidance maneuver is necessary and feasible. We thus utilized the Williams et al. (2016) estimates to calculate the *cumulative* probability of detecting one of the events in a series of surfacing events, i.e., the first or second surfacing event in a 2-surfacing interval, the first or second or third surfacing event in a 3-surfacing interval, and so on.

In this regard, the surfacing events are analogous to a series of Bernoulli trials with one of two outcomes (detected, non-detect) each of which are mutually exclusive and complementary. However, it is important to recognize two conditions when

estimating cumulative probability of detection. First, once a whale is detected, it doesn't matter (for detection) how many subsequent trials (surfacings) occur because it only takes one detected surfacing for the mariner to (1) know a whale is present and forward of the ship, and (2) begin to evaluate whether an avoidance maneuver may be necessary, effective, and safe (recognizing that the first detection may be of variable quality and that subsequent surfacings may need to occur to clarify relevant information such as the whale's direction of travel). We assumed that once the mariner has detected the whale the detection probability for any subsequent surfacing events = 1 owing to the highly concentrated search efforts that ensue in the small area where the whale is likely to resurface.

Thus, if we characterize the two possible outcomes of a surfacing event as D = Detect and N = Non-detect, assume 100% detection probability for any subsequent surfacing event following detection, and that the initial surfacing is the key parameter of interest, the five possible outcomes for detecting at least one surfacing event in the series of (for example) five surfacing events simplifies from:

DDDDD, NDDDD, NNDDD, NNNDD, NNNND

to:

D, ND, NND, NNND, NNNND

Second, and perhaps more importantly, each trial (surfacing) occurs at different distances influencing the distance-specific instantaneous (radial) probability of detection. For example, if a ship is approaching a whale at 19 knots (9.77 m/s) and the time between surfacing events (duration of submergence) is 20 s, the second surfacing event can occur at a ship-to-whale distance of nearly 200 m less than the first surfacing event, the third surfacing event nearly 400 m closer than the first, etc.

To account for these conditions, we utilized the Williams et al. (2016) instantaneous detection probability estimates for the initial surfacing event, and estimated the cumulative probability of detection across the series of N surfacing events by adding the probability of detecting the second surfacing event after the first event went undetected (because if the first was detected, the second is assumed to be detected), and so on. By extension, the cumulative probability of detecting the second surfacing event will always be greater than the instantaneous probability of detecting the event at that distance because it represents the sum of two probabilities. To illustrate, for a 5-surfacing event interval, the cumulative probability of detection was calculated as:

$$\Pr[\textit{at least } 1 \textit{ detection}] = p_1 + (1-p_1)p_2 + (1-p_1)(1-p_2)p_3 + (1-p_1)(1-p_2)(1-p_3)p_4 + (1-p_1)(1-p_2)(1-p_3)(1-p_4)p_5$$

The individual, radial distance-specific detection probabilities were defined using the hazard rate function:

$$\pi = 1 - e^{(-(\frac{x}{scale})\wedge-(shape))}$$

where the scale parameter = $e^{6.73157}$ and shape parameter = $e^{0.747}$ is based on excellent sighting conditions (see Table 3 in Williams et al., 2016). R script (R Core Development Team) written for calculating the cumulative detection probabilities across any distance is provided in **Supplementary Material**.

To illustrate the *cumulative* chance that a mariner detects a whale that initially surfaces at different distances, we then plotted the cumulative probability of detecting at least one of the surfacing events for a whale engaged in an average surfacing interval of 3 surfacing events each separated by 20 s submergences (from our data below) initially surfacing at distances of 4000, 3000, 2000, or 1000 m from a ship. Note that because the speed of the ship is relevant to the changes in ship-to-whale distances among surfacings, we modeled these probabilities based on a ship traveling 19 knots.

Surfacing, Detection and Avoidance: An Example of a Ship Strike Scenario

The combined variation from ship operations (course, speed, etc.), whale behavior (swim speed, dive duration, surfacing frequency, direction of travel, etc.), and initial whale surfacing location (distance and relative bearing from the ship) produces an extremely large number of scenarios in which a ship strike can occur (final ship-to-whale distance and bearing = 0m). These scenarios range from virtually no opportunities for avoidance, such as when a whale initially surfaces from a dive just a few meters from the bulbous bow, to scenarios where mariners have an opportunity to avoid the whale, such as when it initially surfaces at a distance sufficient to allow the mariner to complete the command and maneuver processes and potentially avoid the whale.

To understand the interplay between ship operational state and whale avoidance, we chose a scenario where the mariner has the opportunity to invoke an avoidance maneuver. For our chosen scenario, we started at the point of collision, i.e., the ship and whale are in the same place and same time (horizontal distance = 0 m, time to collision = 0 s) and worked backward in time based on defined parameters of the whale's behavior (constant course traveling adjacent from, and directly perpendicular toward, the ship's path; constant swim speed = 1.23 m/s; Barendse et al., 2010; Kavanagh et al., 2017) and ship's operational state (constant course; constant speed of either 10 knots – 5.14 m/s – or 19 knots – 9.77 m/s). Thus if the collision occurred at 0 s, at 100 s prior to collision the whale will be 123 m from the point of collision and the ship will be 514 m (slow ship) or 977 m (fast ship) from the point of collision.

Whales, however, may be at the same horizontal location of the ship but owing to their dive behavior may pass safely below the ship (vertical distance > 8m which is the average large cruise ship draft from our study). To account for the vertical movements of whales (surfacing events and dive intervals), we further modeled the whale to surface 3 times during its surfacing intervals (data from this study) with 20 s submergences (this study), followed by a foraging dive of 5.4 min (324 s; a typical dive length for foraging humpback whales in Alaska; Dolphin, 1987). For simplicity, we assumed linear travel even though the whale was diving. Using these parameters we then graphed the ship-to-whale distances and time to collision through two whale surfacing

intervals and a foraging dive for mariners approaching a whale that will ultimately be struck on a fast (19 knots) and slow ship (10 knots). To illustrate the trade-off between detection probability and available time for ship personnel to decide on, and achieve, an avoidance maneuver, the cumulative probability of detection for each of the surfacing events were also plotted.

Where Whales Are at Risk: A Mariner's 'Cone of Concern'

Our estimates of the cumulative probability of detection represent the probability of detecting at least one of the surfacing events for a whale initially surfacing at different distances within the entire 180-degree arc forward of the ship from beam-to-beam (Williams et al., 2016). However, throughout development of our conceptual model, marine pilots in Alaska noted that when assessing risk in active whale avoidance they often focus search on a narrower area forward of the ship where a whale strike is more probable, which they define as the 'Cone of Concern.' This is because the relative bearing of the whale influences risk; a whale initially surfacing directly forward of the ship (relative bearing: 000°) at 3000 m is at a higher risk of a collision than a whale that surfaces an order of magnitude closer (300 m), but directly abeam (relative bearing: 090°) because the closer whale is unable to swim fast enough into the ship's path to be struck.

We formalize this idea using simple vector analysis and a trigonometric representation of a whale crossing a ship's path at a 90-degree angle. We contrasted ships traveling at 10 knots (5.14 m/s) and 19 knots (9.77 m/s) with whales swimming at an average speed of 1.23 m/s (2.4 knots) and at fast swimming speeds of 2.46 m/s (4.8 knots) to explore how these parameters influence the size of the Cone of Concern.

Decision-Making During Active Whale Avoidance: Full-Mission Bridge Simulation

A ship's bridge represents a classic example of a socio-technical work environment because operational tasks, such as changing course or speed, must be achieved by a team requiring joint efforts of 'human and technological interlocutors' (Hontvedt, 2015). To that end, full-mission ship simulators are appropriate for understanding the decision-making process by coupling the human element with technology. To better understand the elements of decision-making and time lags related to active whale avoidance, we conducted familiarization and feasibility exercises during 2 days in 2016 using the Kongsberg full-mission bridge simulator (**Figure 2C**) at the Alaska Vocational Technical Center (AVTEC) in Seward, Alaska[2]. The full-mission simulator at AVTEC is regularly used for training Alaska's marine pilots in the maneuvering of large ships as part of (re)certification and continuing education, and mirrors the platforms used by marine pilots at other training centers around the United States.

Seven simulations were conducted whereby a team of two pilots, one serving as the pilot, the other as the helmsman, operated the bridge of a ship, which had operational parameters similar to that of the M/S *Diamond Princess*, a 115,875 gross tonnage, 288 m cruise ship that is representative of the large cruise ships calling in Alaska during the summer. Also on the bridge was an observer who recorded the time of events including (1) the start of simulation, (2) the first *detected* surfacing event of a simulated humpback whale spout (the first actual surfacing event – detected or not – was known only to the simulator operator and scenario coordinator who were located in a different room; **Figure 2C**), (3) the communications that occurred between the pilot and helmsman, (4) when a command was initiated and (5) the end of the simulation, once the ship had passed the whale. Following each simulation, a de-brief discussion was held to review the events and clarify the reasoning related to the decision-making process. During the de-brief, the elapsed time between first detection and the time of the ordered command was quantified, and the common elements related to the decision-making process were identified.

Our simulations were limited in number as was our bridge team size, which would normally include a dedicated Lookout and one or more deck officers. Thus, we did not draw inferences on detection probability from the simulator. Additional limitations existed due to the lack of fidelity of the simulated whale/cues which are the subject of further refinement and improvement. Nevertheless, the descriptive data on time-to-command and archive of commonalities that influenced decision-making were appropriate as full-mission simulations are regularly used to describe processes that occur on the ship's bridge, and can serve as realistic proxies for evaluating risk and commanding new operational states (Hontvedt, 2015).

Ship Maneuverability During Active Whale Avoidance

Once a decision is made on an appropriate avoidance maneuver (maintaining existing operations may also be an active decision; see section Discussion), the rapidity by which the new operational state is achieved can vary dramatically among ship types (e.g., bulk carriers vs. tankers vs. passenger vessels) and within similar-type ships based on technical features such as hull shape and maneuvering systems (e.g., Yasukawa et al., 2018; Zaky et al., 2018). Further, variation can occur based upon environmental conditions (e.g., Yasukawa et al., 2012; Rameesha and Krishnankutty, 2019) and/or the existing operational state of the ship (wave height, wind, ship's existing speed, acceleration/deceleration, whether or not the ship is already engaged in a turn, etc.; Chen et al., 2017). Consequently, determining how quickly a ship can achieve an avoidance maneuver is well beyond the scope of this paper (although our simulations were insightful for which factors should be prioritized for further development).

We utilized the navigation software SEAiq, a commonly used platform by pilots across the U.S. for understanding ship maneuverability, and to focus on a simple and achievable question: for a typical large cruise ship traveling at 10 vs. 19 knots, how far in advance must a turn be initiated to achieve a CPA of at least 100 m with a stationary whale while remaining

[2]https://www.kongsberg.com/digital/products/maritime-simulation/k-sim-navigation/

within defined safety parameters? We used a stationary whale because it simplified the vectors and isolated the focus on the maneuvering capacity of the ship. The 100 m CPA was also for simplicity purposes and should be viewed simply as a means to estimate maneuverability, not as a recommended CPA for mariners. We did not introduce confounding factors, instead simplifying the simulation to reflect 'best case scenarios' including unlimited visibility, calm water, no wind or current, deep water maneuvering, no other vessel traffic or whales, and the ship was initially traveling in straight line. Our defined safety parameters were guided by our working history of the ship's safety parameters and a generalized Pilot Card describing the ship's sensitivity to heading changes (e.g., maximum rate-of-turn; ROT) at varying speeds, as well as limitations in stopping distances. The turn, based on non-emergency safety parameters, conservatively did not exceed a 10-degree rate-of-turn and did not factor in progressively higher rates of turn.

We note that we did not use SEAiq to estimate how much time (and the total distance) it would take for the ship to slow down (e.g., from 19 to 10 knots) because during the full mission bridge simulations, pilots were found to avoid slowing speed in response to a single sighted whale, reflecting their normal practice. During de-briefings it was noted that while a moderate change in heading can be achieved in a relatively short time period (following whale detection), it takes much longer to achieve a moderate change in speed, reducing the effectiveness of speed reduction as a reactive response for whale avoidance, particularly avoidance of a single observed animal. Moreover, pilots never practice 'crash stops,' i.e., a rapid stopping of the ship to avoid a collision with a whale owing to the deleterious impacts it could have on the infrastructure of the ship. Instead, to get a general idea regarding how long it takes for a large cruise ship to reduce speed, and the distance covered during that non-emergency transitional state, we reproduce data from Nash (2009) and re-visit the role of speed reduction as a pre-emptive avoidance maneuver in the Section "Discussion."

Finally, in typical ship operations, while only one person has ultimate 'command' authority while on the bridge, the person directing the movement of the vessel may vary depending upon time and duties, and may be the pilot, captain or deck officer. For simplicity, hereafter, we refer to this person collectively as the Person Directing the Movement of the Vessel (PDMV).

RESULTS

Conceptual Model of Active Whale Avoidance by Large Ships

In its simplest terms, the process of active whale avoidance can be described as occurring in five sequential events (1) a whale surfaces somewhere forward of the ship where a collision with the vessel is possible; (2) bridge personnel tasked with ship navigational decisions detect the whale; (3) the PDMV evaluates the situation and decides that an avoidance maneuver is necessary, feasible, and safe; (3) the PDMV decides upon and commands a new operational state such as a change in course, speed or both; and (5) the ship obtains a new operational state resulting in a lower risk of a collision. Our conceptual model (**Figure 1**) includes Observational (whale surfacing behavior, detection) and Operational processes (commands and maneuvering) that are structured sequentially. Each of these components are described in more detail.

Availability and Detection Process

The first step in this process is dictated by whale behavior because whales need to be available for detection at the surface in order to be avoided. The availability and detection processes have been well studied owing to its relevance for abundance estimation (via distance sampling), and we refer to these studies for describing factors that influence cue frequency and behavior (Hiby and Ward, 1986; Zerbini et al., 2006). Gray, blue, and humpback whales (among many others) regularly embark on a cycle of surface intervals, consisting of several shallow submergences between respiration/surfacing events, punctuated by longer deep dives (e.g., Dolphin, 1987; Godwin et al., 2016; Garcia-Cegarra et al., 2019). Consequently, whales are infrequently but regularly available to be detected. In general the most frequent cue available during a surfacing event takes the form of the appearance of the whale's body in concert with a vertical spout/blow, which is composed primarily of water vapor, air, and lung mucosa, that may extend to several meters above the water and persist for several seconds.

Command Process

Our conceptual model lists a series of steps that we have termed the Command and Maneuver Processes. The Command Process consists of Detection, Reporting, Assessment, Decision, Command, and Compliance actions best described as time lags because any time that elapses after a whale is detected reduces the ship-to-whale distance (as the ship moves toward the whale in the scenarios modeled) and decreases the options for an avoidance maneuver to occur. The Maneuver Process represents the time it takes for the ship, once commands have been executed, to achieve the desired new operational state. We describe these steps in more detail below.

The Detection Lag represents the time between when a bridge team member detects an object in the water, confirms its identity, and formulates their report to the PDMV. Based on anecdotal observations from marine pilots aboard large cruise ships in Alaska, this lag was estimated to vary from 1–2 s, as when a whale spout is immediately recognizable, or as much 5–10 s if the nature of the perceived object is not readily apparent (e.g., a whale lying motionless on the surface or a floating log?). What's more, many Lookouts (personnel assigned to view the waters forward of the ship) are trained to simply make a report of an "object in the water," if they cannot readily identify what it is, and then continue to observe the object to develop clarifying information.

The Reporting Lag represents the time it takes for the person making the observation to vocalize the observation which, from our experience, may vary from 2 to 10 or more seconds depending upon: (1) the volume or quality of the initial sighting information (which may require dialogue with the PDMV); (2) the observer's ability to articulate the relative bearing, distance, direction of travel, or other relevant information;

(3) existing bridge communications; and (4) language or cultural communication issues. For example, the Lookout may spot a whale spout and report, "whale two points to starboard" with no additional information on distance, direction of travel, or speed. At that point, the PDMV will look in the indicated direction and engage the Lookout for information needed to make an assessment which may result in an additional 10–20 s depending upon the length of the submergence between surfacing events or other ongoing action by the PDMV (e.g., communicating on the radio with other traffic, establishing and monitoring navigational parameters, etc.). In the meantime, the initial cue is often no longer available. The elapsed time associated with Detection and Reporting Lags will be minimized if the PDMV makes the observation him/herself (with a high degree of certainty) and immediately articulates the observation to the bridge team. In these instances the total time elapsed may be as short as 5 s, but more frequently it will be closer to 15 s.

The Assessment Lag represents time needed for the PDMV to verify the information and subsequently assess if a collision is possible. In the determination of collision risk, mariners are trained not to make assumptions on the basis of "scanty" information (see US Coast Guard Rule 7 Risk of Collision, International Rules of the Road[3]), highlighting the need for quality information before taking action. If, for example, direction and travel speed of the whale are not available, the process may cycle back to the Detection Lag, awaiting another surfacing event upon which to formulate an avoidance decision. Consequently, a simple report of a whale at a relative bearing and distance may not provide sufficient information upon which to base an avoidance action, even for a whale sighted directly ahead. Consequently, the Assessment Lag, as with the other lags, may be relatively quick (3–5 s) for the "obvious" situations or it may take longer if inconsistent or incomplete information is reported.

The Decision Lag represents the time needed for the PDMV to consider the available safe avoidance options based on competing risks. The decision by marine pilots (serving in the capacity of PDMV) is founded on the principle of do-no-harm, firstly to people, secondly to the ship, and thirdly to the environment. In practice, this results in a rapid and dynamic calculation of the trade-offs in the risk of whale collision with the risks of harm to people, the ship, or the environment (or some combination such as a collision with another ship, a shoal, or even another whale). Consequently, critical factors in the Decision Lag are based on the situational awareness of the PDMV to the proximity to these hazards and the operational state of the ship; i.e., what is in the realm of possibility based on its speed, sea state, etc. Based on opportunistic assessments, Decision lags can vary from a few to 20 s, based on complexity and competing risks.

The Command and Compliance Lag is the time needed for the PDMV to articulate the avoidance decision into a specific command and for the bridge team to comply. For example, the PDMV may command the Helmsman to initiate a new heading. For some shipping entities, the bridge procedures require 'closed loop communication' whereby the command cannot be executed until the initial order is first acknowledged (by the Quartermaster for course changes, or by the deck officer for speed changes), and then confirmed by the PDMV. This lag is generally 5–7 s in situations where all involved understand and are in agreement, but can be longer (upwards of 60 s) if the command is misunderstood, not heard, not acknowledged, or there is disagreement on the appropriate avoidance action.

Maneuver Process

The Maneuver Process is the time it takes for the ship, once commands have been ordered, to achieve the desired operational state. The maneuver process is also best considered in the context of a time lag because the new commanded operational state does not occur instantaneously. The Maneuver Process can vary dramatically among ships although approximate generalizations are appropriate for estimation and/or simulation scenarios. Similar to other large ships, safe maneuvering of large cruise ships encapsulate a range of turning and slowing options based on the interaction between ship type, existing operational state, and environmental conditions. Our experience, based on informal sampling of whale avoidance maneuvers during the past several summers in best-case scenarios, has been that the maneuver process can vary from 25 to 180 s depending upon operating conditions and the type of maneuver ordered.

Proof of Concept: Large Cruise Ships Avoiding Humpback Whales

Availability

The data collected by observers stationed at the bow of cruise ships transiting waters in Alaska demonstrate that humpback whales surfacing around the ships often provide a small but variable number of opportunities for detection. For all surfacing bouts that ended with a fluke-up dive, whales embarked on an average of 2.8 surfacing events per interval (N = 156 unique intervals; range of surfacing events per interval: 1–15; **Figure 3A**). We again clarify that this average is based on the number of surfacing events per surfacing interval, not the number of cues per surfacing event. Based on the empirical cumulative density function, about 40% of all surfacing intervals included more than three events (**Figure 3A**). As we only used surface intervals that terminated in a fluke-up dive, the data on surfacing frequency was not 'right censored' in that we had confidence that the surface interval did not include unrecorded events that occurred after the fluke-up dive. However, there is a possibility that Observers may have missed a surfacing event (or two) prior to detection ('left censored' data) resulting in the true number of events likely being larger.

The time elapsed between surfacing events was also variable, although the length of most submergences were centered in groups of 10–15 and 15–20 s (**Figure 3B**). We feel confident that, once a surfacing event was observed, detection probability of subsequent events was very high as observers (and bridge teams) focus on small area where whales are likely to resurface to gain as much quality information as necessary to evaluate collision risk. Together, the data suggest that mariners engaged in active (humpback) whale avoidance in Alaska generally have about

[3]https://www.navcen.uscg.gov/pdf/navRules/CG_NRHB_20151231.pdf

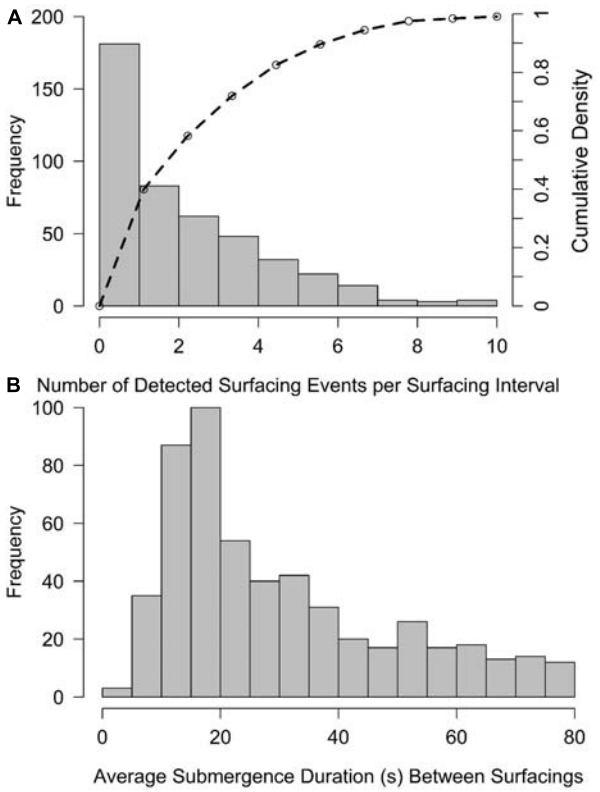

FIGURE 3 | Quantitative descriptions of whale surfacing behavior that help describe the 'availability' of humpback whales to be detected by ship personnel engaged in active whale avoidance in Alaska. **(A)** The average number of detected surfacing events per surfacing interval of humpback whales surfacing near large cruise ships, 2016–2017, with a curve representing the empirical cumulative density function. **(B)** Histogram of the average elapsed time between available surfacing events (submergence duration) during surfacing intervals of humpback whales surfacing near cruise ships, 2016–2017.

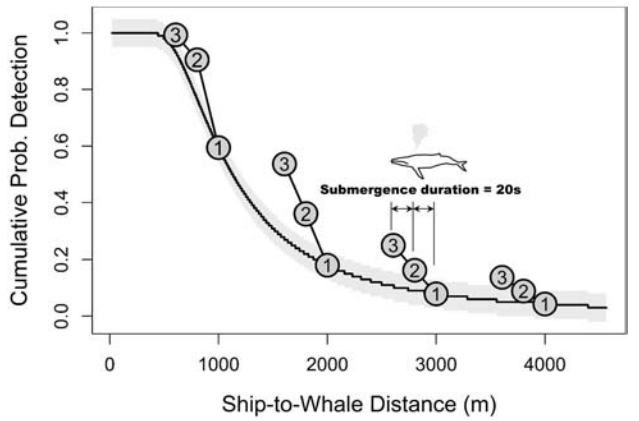

FIGURE 4 | The cumulative probability of detecting at least one of the three surfacing events for 3-event surfacing intervals that began with an initial surfacing event at 4000, 3000, 2000, or 1000 m from a ship. Circles represent the surfacing event number. The detection function used for calculating cumulative probabilities were from Williams et al. (2016) based on excellent sighting conditions.

three opportunities for detecting the whale during its surface interval, with an average of around 20 s between events.

Cumulative Probability of Detection

For a surfacing interval that included three surfacing events, the cumulative probability of detecting at least one of the events was lower at larger distances, and increased (non-linearly) with decreasing ship-to-whale distances (**Figure 4**). For example, based on detection functions for whales surfacing across the 180-degree arc in front of the ship, mariners have a nearly 60% chance of detecting at least one of the three surfacing events for a whale that initially surfaces from a dive 2000 m from the ship, but a less than 15% chance of detection for a whale that initially surfaced 4000 m from the ship (**Figure 4**). The doubling of the distance resulted in four-fold lowering probability of detection because at larger distances the cumulative increase in detection probability was more linear in nature (e.g., for a surfacing interval that begins at 4000 m) but more exponential in nature for intervals that began at mid distances (e.g., 2000 m). Whales that surface close to the ship (<1000 m) have near certainty of being detected (**Figure 4**).

Surfacing, Detection, and Avoidance: An Example of a Ship Strike Scenario

In our chosen hypothetical ship-strike scenario (ultimate CPA = 0 m) involving ships traveling at different speeds (19 or 10 knots), the whale was struck (PoC; ship-to-whale distance = 0 m, time to collision = 0 s; **Figures 5A,B**) when it surfaced to take its third respiration during its second surfacing interval (red shaded area). Working backward in time (and space) from the Point of Collision, at 40 s prior to collision the whale surfaced about 211 m from the slower ship and about 394 m from the faster ship. At both those distances, the cumulative probability of detection was near certain (>0.99). Working further backward in time, the whale embarked on a 324 s dive at 364 s prior to collision which placed it over 3500 m from the faster ship but just over 1900 m from the slower ship. At this point, which represents the last chance to detect the whale before it dives, the cumulative probability of having detected at least one of the 3 surfacing events during the first surfacing interval (green shaded area, includes fast ship-to whale distances of 3978, 3781, and 3584 m; slow ship-to-whale distances = 2135, 2029, 1924 m) was approximately 60% for the 10-knot ship but less than 15% for the 19-knot ship (red lines; **Figures 5A,B**). Owing to the near 4-fold greater (cumulative) probability of detecting at least one of the surfacing events during the first (earlier) surfacing interval, the PDMV aboard the slower ship could have an additional 324 s (post detection and during the whale's dive) to decide upon and implement an avoidance maneuver.

Note that, based on our estimates of the command and maneuver lags (see above), both the slow and fast ship would have limited (if any) opportunities to avoid the whale if it went undetected during the first surfacing interval (green shaded areas) because 40 s to collision (when the whale surfaced from its dive) exceeded the aggregate time to implement these processes.

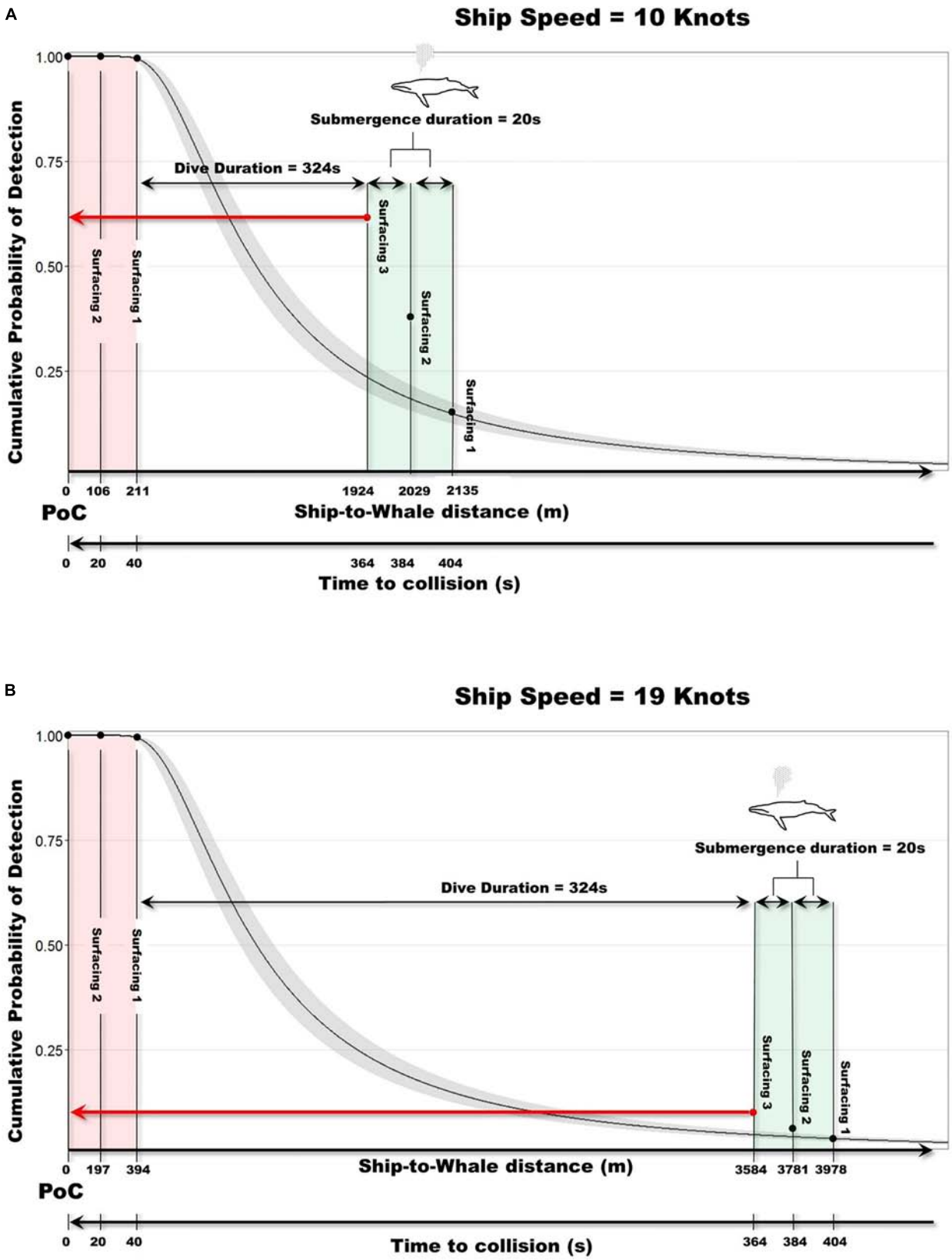

FIGURE 5 | Continued

FIGURE 5 | The cumulative probability of detecting the first, second, or third surfacing event of a 3-event surfacing interval of a humpback whale for mariners aboard a 10 knot ship **(A)** and a 19 knot ship **(B)** relative to the ship-to-whale distance and corresponding time to collision. In each scenario, the whale behavior is held constant and modeled as traveling at 1.23 m/s (2.3 knots) perpendicular to but toward the path of a ship and, following the initial surfacing interval (green shaded area) and a 5.4 min foraging dive, is struck when it surfaces a third time during the second surfacing interval (red shaded area) at the same location at same time as the ship. Surfacing events are indicated by dots and surfacing intervals by shaded areas. From the first surfacing event (Surfacing 1, green shaded area), the time to collision is held constant at 404 s for each scenario which results in an initial ship-to-whale distance of 2135 m from the slow ship **(A)** and 3978 m from the faster ship **(B)**. Curved line with gray 95% CI from Williams et al. (2016). A ship strike occurs in both scenarios unless the whale or ship deviates course or speed. Distances are approximately to scale.

Where Whales Are at Risk: A Mariner's 'Cone of Concern'

Figure 6 depicts the results of simple vector analysis demonstrating how a whale's swimming speed and a ship's transit speed influences the width of the Cone of Concern. **Figures 6A,B** depicts ships traveling at 10 knots (5.14 m/s) and 19 knots (9.77 m/s) on a collision course (toward PoC) with a humpback whale swimming at a typical speed (1.23 m/s; **6A**) or at a fast swimming speed (2.47 m/s; **6B**) perpendicular to, and toward, the ship's path. For both scenarios, the opposite angle in the right triangle (defined by the ratio of the whale's swimming speed and the ship's travel speed) is maximized because the whale is in a 'crossing' situation; i.e., it is headed directly toward the ship's path resulting in the shortest time for potential collision. For the Fast Ship/Typical Whale (6A) scenario, the Cone of Concern would be approximately 14 degrees (7.2 degrees on either side of the ship) and encapsulate a search area of nearly 0.8 km^2. For the Slow Ship/Fast Whale scenario, the Cone of Concern has a nearly equal search surface area (0.84 km^2) even though the search area is much wider (∼51.2 degrees).

Command Process

As part of the simulations conducted in the full-mission ship bridge simulator at AVTEC, the average elapsed time, resulting from the aggregate of the Command time lags, i.e., from detection of the simulated whale spout to initial compliance with an ordered avoidance action, was 23 s. This compared favorably with the informal observations conducted by several pilots during opportunistic whale avoidance efforts while navigating large cruise ships in 2016 and 2017. During the debriefing meetings, we found that, following initial detection, uncertainty in the whale's direction of travel and swim speed were common factors that contributed to the delay in a command; the PDMV needed sufficient confidence in the information (making it 'actionable'), particularly on whether the whale was swimming toward or away from the ship's heading. Consequently, the PDMV regularly communicated with the Lookout and mate and, absent good information, waited for a subsequent surfacing event before deciding on an appropriate avoidance action.

During post-simulation de-briefs several common themes were discovered. First, marine pilots rarely command a speed reduction in response to a single sighted whale owing to their familiarity with the time it takes to achieve the new speed (Nash, 2009, reproduced in part in **Table 1**) and that a course change alone is most often more effective and efficient than a potential speed reduction. Second, and perhaps more importantly, the pilot's maneuvering decisions were ubiquitously based on the evaluation of competing risks. For instance, once the pilots confirmed that a whale was within the Cone of Concern, a primary consideration was how a change in course would influence other risks, such as the risk of collision with other navigation hazards including, but not limited to, other vessels, reefs, or shoals. Likewise, in all simulations where the pilot decided that a course change was needed to reduce collision risk with a whale, an evaluation occurred whereby the efficacy of the course change was considered relative to the time needed to safely 'build up' to the required rate-of-turn. The rate-of-turn required to avoid the whale was then considered relative to that particular ship's safe rate-of-turn guidelines and heel angles to mitigate the risk of deleterious impacts to the vessel and its passengers.

Maneuverability

Using SEAiq and defined safety parameters, we found that mariners aboard a ship traveling 10 knots (5.14 m/s) would require action not less than approximately 741 m from the whale to achieve a 'near-miss CPA of 100 m' (**Figure 7A**). In contrast, a CPA of 100 m aboard the 19 knot ship occurred only after it initiated a turn at least 1121 m from the confirmed sighting (**Figure 7B**; both scenarios occurred under optimal conditions). In both cases, the Command Lag was modeled as constant (based on results from the simulator), occurring in approximately 25 s, during which time the ship traveled 241 and 130 m closer to the whale for the 19 and 10 knot ship respectively. The Maneuver Lag was achieved over the course of approximately 90 s for the fast ship when it traveled 880 m, and approximately 119 s traveling just over 600 m for the 10 knot ship. Again, this was an abstract, best-case scenario, limiting the Reporting, Assessment and Decision Lags to minimums.

DISCUSSION

We coupled whale-surfacing data collected using a ship-based observer with data from simulations in a full-mission ship bridge simulator, opportunistic data collected by marine pilots aboard large cruise ships, and simulations using typical pilotage software to generate the first holistic model of active whale avoidance by large ships. While our goal was to provide a general introduction of the constituent processes such that development and more rigorous testing can build on our efforts, our results provided some insight into the opportunities and constraints for increasing the effectiveness of active whale avoidance and some priority avenues of research, which we discuss below.

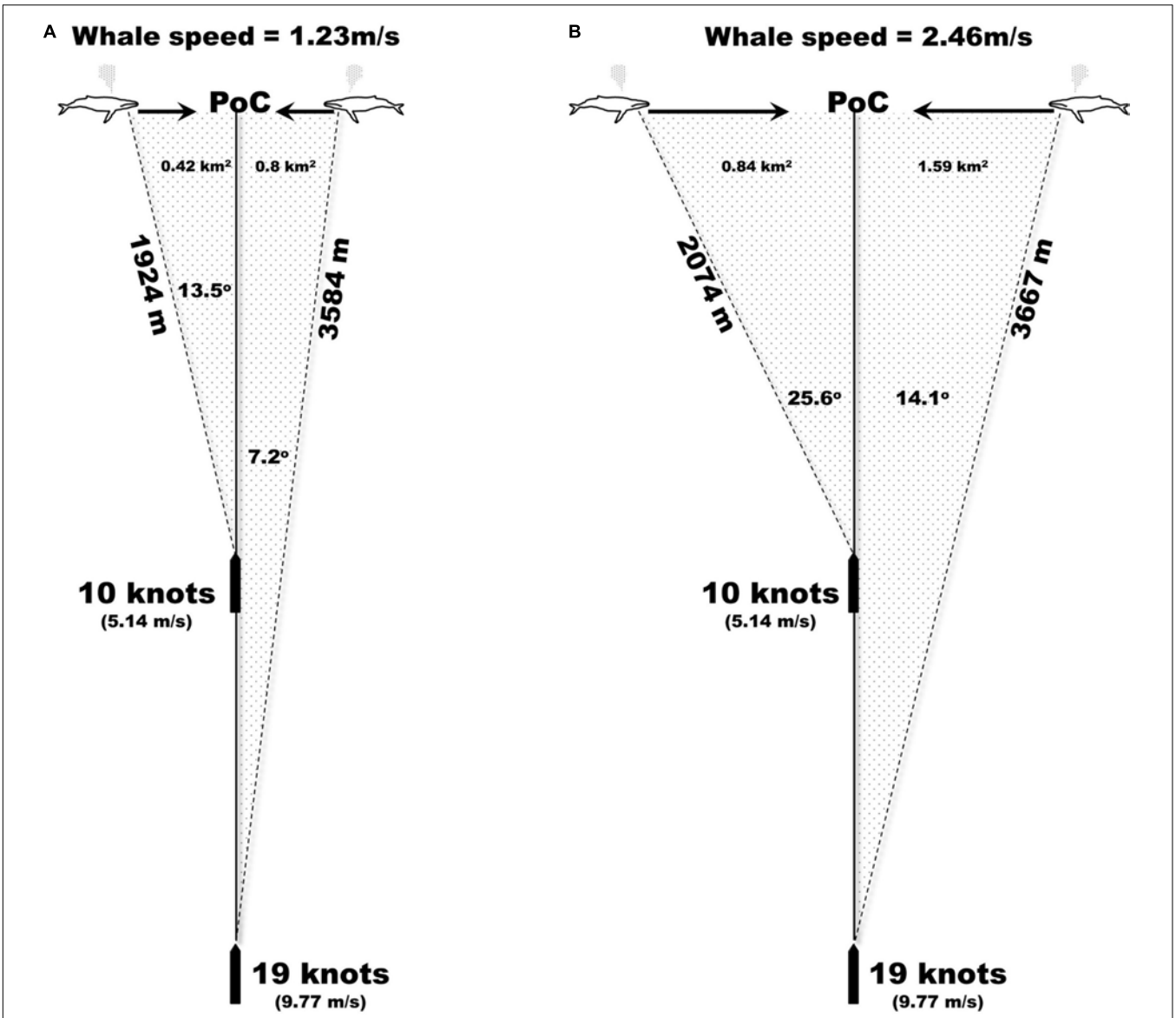

FIGURE 6 | Two scenarios encapsulating different ship transit and whale swimming speeds that define the 'Cone of Concern' (shaded area), i.e., the area searched by marine pilots to more efficiently detect surfacing whales that have the potential to be struck by the ship (at the bulbous bow). Figures depict the Cone for a 19-knot ship and 10-knot ship relative to an average swimming speed of a humpback whales (1.23 m/s; **A**) and a fast swimming whale (2.46 m/s; **B**) on a crossing pattern and on a collision course with the ship (PoC, Point of Collision; CPA = 0 m). Cones range from 14.4° (7.2° on either side of ship for fast ship and slow whale) to 51.2° (25.6° on either side of ship for a slow ship and fast whales). Distances are approximately to scale to a 284 m ship.

Availability, Detection, and a Mariner's Cone of Concern

Based on data from hundreds of surfacing events of humpback whales by observers, we demonstrate that mariners aboard large ships in Alaska typically have about three opportunities, each separated by about 20 s, to detect the whale and make a decision about whether an avoidance maneuver is necessary, possible, and safe. While these estimates were largely consistent with other studies of humpback whales in Alaska (Dolphin, 1987), we highlight that data on surfacing frequency was not corrected for any negative biases owing to the observer's chance of missing the initial (or several) surfacing events, particularly at large ship-to-whale distances or limited to the mariner's Cone-of-Concern. While we initially sought to minimize the chance for this distance bias by using only information from whales surfacing close to the ship, ultimately we decided against subsetting the data (1) because surfacing intervals that began close to the ship were often still continuing when the ship passed abeam (when observers terminated their observations of that whale) which would also underestimate the number of surfacing events per interval, and (2) to avoid biasing the inferences if, in fact, whales alter their surfacing behavior as a function of distance.

TABLE 1 | Approximate time (min) and distance (nautical miles) needed for slowing a large cruise ship with multiple engine configurations from various initial speeds to an (arbitrary) target speed of 14.7 knots using a safe slowing speed reduction of 2 RPM per minute.

# of Engines needed	Initial Speed (knots)	Propeller (RPM) at Initial Speed	Target Speed (knots)	Target Propeller RPM	Time needed (min)	Distance needed (nm)	Distance needed (m)
4 + 2	22.2	136	14.7	90	25	7.0	12,964
4 + 1	21.4	131	14.7	90	20	5.0	9,260
3 + 2	20.6	126	14.7	90	18	4.5	8,334
3 + 1	18.3	112	14.7	90	11	3.0	5,556
3	15.5	95	14.7	90	5	1.5	2,778

Note that slower (initial) speeds require less power (load) and thus fewer engines are needed to meet those power load demands for propulsion. Modified from Nash (2009).

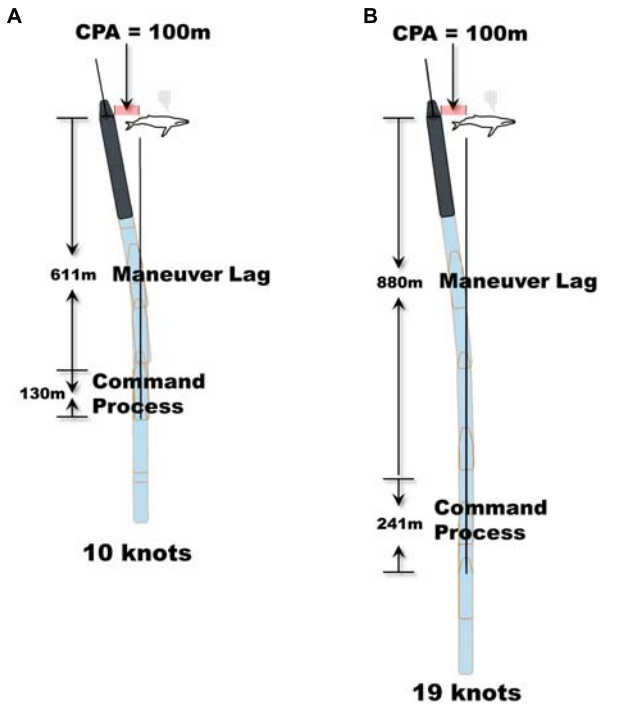

FIGURE 7 | Example of the maneuvering capability of a large cruise ship traveling at 10 knots **(A)** and 19 knots **(B)** to achieve a Closest Point of Approach (CPA) of 100 m based on defined environmental conditions and safe turning constraints and assumptions (maximum rate of turn = 10 degrees per minute; see text). These scenarios depict situations that, should a mariner want to achieve a 'near miss' of a surfacing humpback whale, the ship would have to begin its turn at 741 and 1121 m away, owing to the constant time lags (the same regardless of ship speed) related to the Command process and maneuvering capability of a large ship. This distance represents a minimum owing to uncertainty in the location, swim speed, and direction of travel for a sighted whale.

Recognizing these biases, however, helps identify possible ways in which the effectiveness of whale avoidance can be increased. For example, a key finding of our conceptual model is that processes that occur on the ship's bridge such as reporting a whale sighting, assessment of the risk, and compliance to commands, couple with maneuvering constraints to produce a variable, yet important time lag between detection and achieving a new operational state (that reduces collision risk). This aggregate lag contributes to the inverse relationship between time available to make an avoidance maneuver and the range of maneuver options available. Any activity or operation that increases the chance of detection when a whale is first available to be detected thus increases the options for avoiding the whale and the odds of successful avoidance. We identified three factors that may help PDMVs detect a whale and obtain sufficient information to actively avoid it.

First, marine pilots in Alaska, based on decades of experience encountering and avoiding (primarily humpback) whales surfacing near large cruise ships, have developed a searching pattern 'Cone of Concern' based on familiarity with approximate travel speeds of humpback whales relative to the ships' transit speeds. In doing so, pilots and other bridge personnel narrow their search efforts (by over 80% based on our simple vectors and geometry; **Figure 6**) by delineating the 'population' of surfacing whales at risk of collision vs. those that are not. This practice could easily be standardized by integrating the concept into transit planning and/or regular communications with the bridge team to focus on parameters of, and need to search within, the Cone of Concern.

Second, assigning a designated Lookout tasked solely with searching for whales in the Cone of Concern could also enhance detection probability and thus opportunities for whale avoidance. While we did not test whether different configurations of personnel (pilot, pilot + designated observer, etc.) produced different detection functions, experiments aboard large fast ferries have demonstrated that a dedicated whale 'spotter' vastly improved detection probability and the distance at which whales were detected (Weinrich et al., 2009). Research based on line transect theory and distance sampling also demonstrate that detection probability increases when additional observers are utilized (Schmidt et al., 2017) including with whales (Zerbini et al., 2006). While we recognize that transiting at night or in heavy seas may reduce or eliminate detection, we also highlight that technology continues to reduce barriers to detection in some of these conditions (Zitterbart et al., 2013) and application of the Cone of Concern may help inform development of the technology to maximize effectiveness.

The third potential way to facilitate the effectiveness of active whale avoidance is by reducing the time identified in the Command process. Pilots in Alaska are regularly conveyed unnecessary or incomplete information by members of the bridge team following a whale sighting. If the information is incomplete,

the PDMV may have to wait for another surfacing event before having information of sufficient quality to be 'actionable.' This may equate to the ship traveling several hundred meters closer to the whale (based on average submergence data and typical transit speeds in Alaska) before the PDMV can confirm the whale's location and direction of travel. Training bridge personnel with regards to what information is desirable and protocols for communicating that information (e.g., 'whale approximately 2000 m three points to starboard, moving away from the ship') can make a significant difference in time available for PDMV to assess the situation and implement the maneuver without further increasing the risk of harm to the people, the ship, or other components of the marine environment. A simple suggestion of utilizing the same training used for reporting of a man-overboard to continuously point to the person (whale) promotes the effectiveness of the PDMV's detection and decision process significantly.

Ship Speed

Throughout our effort we consistently contrasted scenarios involving fast (19 knots) and slow ships (10 knots) to explore how speed may influence the constituent processes in active avoidance of a single whale surfacing near the ship. While our objectives were not to rigorously test the role of ship speed in these processes, nor were they to identify an optimal speed that balances whale avoidance vs. transit efficiency (should one exist), we highlight some insights based on our results that warrant discussion and further development.

First, when simulations of whale encounters were conducted in the full-mission ship simulator, pilots never attempted to slow the ship in response to the sighted whale, instead preferring slight changes in course. In the de-briefs that followed, pilots communicated that, while change in speed may influence the dynamics of a whale – ship encounter, the distance necessary to slow the ship to speeds necessary for effective avoidance based on speed change alone (mariner body of knowledge relative to vessel avoidance actions) tended to exceed the sighted distance to the whale owing to potential unsafe results of rapid speed changes. Given the absence of this response, we did not produce simulations that contrasted the efficacy of slowing the ship vs. slight course changes, instead reproducing some recommendations from Nash (2009) simply to provide context with regards to the magnitude of space/distance needed to slow (and recognizing that the target speeds listed in Nash were arbitrary relative to whale avoidance). However we feel a brief description as to why rapid changes in speed are not regularly practiced is necessary owing to its prominent relevance in ship strike dynamics and context for understanding the estimates in **Table 1**.

For large cruise ships (and likely other large ships) power management plays a major role in operational decision-making (e.g., Ancona et al., 2018), not just in the context of managing fuel costs and optimal fuel efficiency (and resulting levels of air pollution; Khan et al., 2012), but also for safety reasons. For large cruise ships, power needs are met using multiple engines that are variably configured for two different power loads including the propulsion load, which is typically about 80% of total load while transiting, and the hotel load, which is the electrical energy needed to power the ship's lights, heating/air conditioning, galley, etc. Rapid changes in power use can negatively affect emissions, damage the generators (engines) and, in a worst-case scenario, cause a 'blackout' (total loss of power). To help guard against these negative outcomes, large cruise ships typically have some form of power management system, such as a 'Load Control Program' that limits dramatic fluctuations in power use. Given that propulsion is the primary power requirement, and that propulsion is a function of the propeller's rotations per minute (RPM), as a general rule, when a large cruise ship is in Load Control the propeller RPMs are generally not reduced by more than 2–3 RPMs per minute. Consequently, gradual changes in speed represent best load management practices and the gradual change may not meet the more immediate change in operational state necessary for avoiding a whale.

Pre-emptive (planned) reductions in speed are, however, regularly used by pilots in Alaska as a strike risk reduction strategy. Pre-emptive speed reductions are those initiated in anticipation of, rather than in response to, a whale aggregation, and are utilized in two general scenarios. The first is when mariners are informed of a whale aggregation recently detected along the ship's route and communicated to the ship personnel. The second general scenario is when the ship is approaching a narrow navigational area that also historically has supported whale aggregations. For example, with the cooperation of cruise ship Masters, pilots regularly slow cruise ships to 14 knots in Snow Passage, Alaska, because avoidance options are limited and the area is often characterized by small to large whale aggregations. Pilots have found that these pre-emptive speed reductions tend to produce less resistance from other bridge personnel when (1) they can be accounted for in transit planning and (2) they do not adversely affect port arrival times.

We thus encourage continued development of software applications[4,5] in which mariners participate in a sighting network that helps inform others vessels that whales have been detected in their area. The type of information conveyed, its timeliness, and receiving platform is, however, critical for its utility. Receiving information via a mobile application (often with sporadic cell coverage) is a more cumbersome means than, for example, a ship's Electronic Chart Display and Information System (ECDIS) which could overlay historical (e.g., weekly) and recent (e.g., <2 h) whale sightings to assist with transit planning. Recently, the programmer for Whale Alert and the developer for SEAiq coordinated to provide the ability to import weekly whale sighting information automatically for display on the electronic chart.

Our results also demonstrate how, in some scenarios, slower ships may have increased opportunities for whale avoidance acting through both the Maneuver and Detection processes. Faster ships, by definition, travel further distances compared to slower ships during set time periods, such as when whales are submerged between surfacing events (averaged 20 s in our study), on deep dives (324 s modeled based on literature), or during

[4]www.WhaleAlert.org

[5]www.repcet.com

time lags related to decision-making and communications on the bridge following detection. For example, if the Command processes takes the same amount of time on fast and slow ships, and total time elapsed following detection to the point the ship begins to change course is approximately 115 s (**Figure 7**), we demonstrate that the faster ship achieving a 'near miss' of 100 m from a whale would need to detect the whale over 1100 m from the Point of Collision as opposed to just over 700 m for a slow ship, simply because the faster ship moves further over the same time period given the conservative safe maneuvering limitations imposed on the initial test scenarios. An alternative way of interpreting those results is that, had the slow ship and fast ship begun the Command and Maneuver process at the same distance from the whale (as opposed to the same time), the slower ship could have achieved a greater CPA because it would have had a longer time period to continue its turn.

Ship speed can also influence whale avoidance by influencing detection probability. To be clear our results do *not* indicate that mariners on slower ships are able to detect whales any better compared to mariners on faster ships – there is no logical reason why detection probability would differ for a surfacing whale at a set distance (e.g., 2000 m from the ship) for mariners aboard a fast or slow moving ship. However, if we held the *time* to a collision constant, as in the scenario in **Figure 5**, then, by definition, the faster ship will be further from the point of collision than a slower ship at the same time to collision. Thus, a surfacing event critical for detection would occur closer to the slower ship influencing the cumulative probability of detection, providing more time for a maneuver.

Future Research and Training

The conceptual model of active whale avoidance was derived primarily from the collective experience of pilots in Southeast Alaska who have 'learned by doing,' which has required significant time on the water. We thus submit a number of ideas for priority research and training development to hasten the adoption, applicability, and effectiveness of active whale avoidance.

First, ship personnel need sufficient time to make a decision related to an avoidance maneuver and achieve a new operational state, assuming one is commanded, that reduces collision risk. For example, marine pilots in Alaska are often challenged by predicting where a whale is likely to surface following its dive because, if they waited for the whale to resurface before initiating a maneuver, the options for avoidance would be significantly reduced. During simulation de-briefs pilots communicated that they will, at times, choose to 'turn behind' a whale if they ascertain it's swimming direction based on a general rule of thumb that, informed by years of encounters, humpback whales are more likely to continue their general direction of travel than they are to turn 180° following a dive. However, pilots are less likely to enact the same maneuver if humpback whales are foraging along a tidal rip, which they've found tends to produce more unpredictable movements. Thus, a priority avenue of research could be to explore the 'linearity' of whale movements, loosely defined as the degree to which whales travel in a straight line vs. turning (see also Williams et al., 2002; Barendse et al., 2010), and how it may be influenced by environmental conditions or other factors.

Refining estimates of detection probability, particularly as it applies to the area with in the Cone of Concern also represents an important research thread. The instantaneous detection estimates of the radial ship-to-whale distances we utilized (from Williams et al., 2016) were derived based on detecting whales across the 180-degree arc (beam-to-beam) forward of the ship. We assume that the probability of detection will be much higher at a given distance, or much father at a given detection probability, if similar detection functions were derived based on search effort solely in the Cone of Concern. We acknowledge that some of the 'gains' in detection from focused search in the Cone would be offset somewhat if mariners tasked with sighting whales are also tasked with other duties (e.g., monitoring radar or responding to radio communications). However, updating estimates of detection probability based on a Lookout's focused search within the Cone of Concern would provide more reliable estimates and produce a more realistic range of feasible options of avoidance maneuvers.

Another productive avenue of research is a more rigorous examination of competing risks related to whale avoidance. During our simulations, once pilots confirmed that a whale was forward of the ship at some risk of collision, a primary consideration was how to achieve a new (avoidance) heading while not increasing other risks, such as collision with other vessels, reefs, or shoals. For obvious reasons, ship operators will rarely increase the risk of deleterious impacts to passengers or damage to the electrical system that accompanies dramatic and unsafe operations [e.g., a 'crash stop' (Wirz, 2012) or rapid turn], unless those maneuvers are offset by reduction in risk to more consequential events such as a grounding (for example). The risk of negative impacts from dramatic changes in course or speed to avoid a whale will thus always be weighed against the potential benefits of whale avoidance. In all simulations where a course change was needed, the pilot evaluated the efficacy of the course change by considering the needed time to incrementally 'build up' to the desired rate-of-turn to minimize impacts to passengers, the ship, and the environment, thereby avoiding excess 'heel.' Larger heel angles aboard cruise ships increase the chance that furniture will begin to slide and passengers will be injured from falls/by falling objects, and swim pools will spill, etc. As previously discussed, the electrical or propulsions systems can be negatively impacted in extreme instances of abrupt speed changes. We note that parameters identified as "safe" often represent general guidance and can be modified depending upon the PDMV's experience and the situation (e.g., commencing an initial rate-of-turn, within defined parameters, and then after the ship has stabilized at that rate-of-turn, incrementally increasing, and stabilizing at greater rates-of-turn, while maintaining the ship within safe heel angles).

The development of a whale avoidance module in a full-mission ship simulator can also advance whale avoidance by training mariners, through repetition and experimentation, who have less experience with conditions where whale

avoidance may be effective, avoidance techniques, and range of maneuvers that may be possible. Training modules can also lead to improved transit planning by scripting exercises within a specific operating area where whales are likely to be encountered while also accounting for local environmental conditions (e.g., wind, current, sea state), traffic situations, and other navigation hazards commonly experienced (e.g., ice).

Finally training can assist in communicating the value of whale avoidance to other members of the bridge team, such as the ship captain/mates. Pilots in Southeast Alaska have found that, upon boarding the ship, communicating with the bridge team that whales may be encountered, emphasizing the importance of whale avoidance, and discussion of avoidance techniques has increased situational awareness of whales while in transit (similar to communicating local knowledge of navigational hazards) and, importantly, often reduced resistance to implementing proactive avoidance maneuvers or temporary reductions in ship speed. A recent study in the St. Lawrence Estuary demonstrated the value that marine pilots can have in implementing strike-risk reduction efforts, in part, through elevating its importance for the larger bridge team (Chion et al., 2018).

An important caveat is that the development of training modules not generate a 'recipe' for proper maneuvers. The range of variation in avoidance maneuvers is large, based on whale swim speed, direction of travel, ship speed, and operational constraints, as are the competing risks of an avoidance maneuver. In a whale avoidance situation, mariners are often faced with making rapid decisions to prevent making an undesirable situation (e.g., risk of collision with a whale) become an even more harmful event (to the whale, the passengers, the ship itself, the environment, or all four). Marine pilots in Alaska, when asked what they do to avoid a whale, answer ubiquitously: "it depends." In "mariner-speak" avoidance actions are based upon all factors appropriate to the prevailing circumstances and conditions, and with due regard for good seamanship. And good seamanship is a direct function of good training. To that end, we follow the reasoning of De Terssac (1992), as cited in Chauvin et al. (2009) who stated that, to achieve an overarching objective (which in this case is a reduction in collision risk) the best approach is to define "*…a space of operation within which formal rules no longer specify the solution to be implemented, but list a range of permissible solutions among which the operator will have to choose the one that seems most relevant in the context.*" Within that range of potential solutions may be a decision to maintain course or speed either because the value of the information available is insufficient or the risk of a worse outcome exceeds the risk of the ship strike (e.g., the altered course resulting in coming too close to shore, increasing the risk of grounding and catastrophic oil spill).

Based on our findings and observations, we conclude that active whale avoidance is feasible and, in most cases, can be practiced without creating an increase in competing risks. What's more the practice can complement existing efforts that increase situational awareness of whales (e.g., Whale Alert) even in areas where other risk-reduction measures, such as operating at slower speeds, are in place. Most importantly, continuing collaboration between professional mariners, scientists, and natural resource managers is vital to reaching mutually beneficial reductions in whale strikes.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

SG and LV conceived the idea and initiated the collaborative efforts between the Southeast Alaska Pilots' Association and the National Park Service. SG led the writing of the manuscript and construction of the figures including the data used for detection and availability. SG, LV, CG, RP, and AH participated in the simulator activities at AVTEC. JB and LV collaborated on generating maneuvering graphics. SG, LV, CG, and JB contributed to the writing of the manuscript and idea development.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00592/full#supplementary-material

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frontiers in Marine Science

REVIEW
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The Effects of Ship Noise on Marine Mammals—A Review

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The number of marine watercraft is on the rise—from private boats in coastal areas to commercial ships crossing oceans. A concomitant increase in underwater noise has been reported in several regions around the globe. Given the important role sound plays in the life functions of marine mammals, research on the potential effects of vessel noise has grown—in particular since the year 2000. We provide an overview of this literature, showing that studies have been patchy in terms of their coverage of species, habitats, vessel types, and types of impact investigated. The documented effects include behavioral and acoustic responses, auditory masking, and stress. We identify knowledge gaps: There appears a bias to more easily accessible species (i.e., bottlenose dolphins and humpback whales), whereas there is a paucity of literature addressing vessel noise impacts on river dolphins, even though some of these species experience chronic noise from boats. Similarly, little is known about the potential effects of ship noise on pelagic and deep-diving marine mammals, even though ship noise is focused in a downward direction, reaching great depth at little acoustic loss and potentially coupling into sound propagation channels in which sound may transmit over long ranges. We explain the fundamental concepts involved in the generation and propagation of vessel noise and point out common problems with both physics and biology: Recordings of ship noise might be affected by unidentified artifacts, and noise exposure can be both under- and over-estimated by tens of decibel if the local sound propagation conditions are not considered. The lack of anthropogenic (e.g., different vessel types), environmental (e.g., different sea states or presence/absence of prey), and biological (e.g., different demographics) controls is a common problem, as is a lack of understanding what constitutes the 'normal' range of behaviors. Last but not least, the biological significance of observed responses is mostly unknown. Moving forward, standards on study design, data analysis, and reporting are badly needed so that results are comparable (across space and time) and so that data can be synthesized to address the grand unknowns: the role of context and the consequences of chronic exposures.

Keywords: auditory masking, chronic noise exposure, marine watercraft, ship noise emission, commercial shipping, marine mammal, behavioral response

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INTRODUCTION

Marine traffic in the world's oceans is increasing. This includes watercraft ranging from small boats to large ships. Commercial ships are increasing in number as well as size, linked to overall economic growth (United Nations Conference on Trade and Development [UNCTAD], 2018). Between World War II and 2008, the global number of ships rose by a factor 3.5 and the total gross tonnage by a factor 10 (Frisk, 2012). Based on satellite altimetry, global ship density increased by a factor 4 between 1992 and 2012, with the greatest increase in the Indian Ocean (Tournadre, 2014). Ship noise is rising concomitantly. In fact, ships have become the most ubiquitous and pervasive source of anthropogenic noise in the oceans. Ship traffic is responsible for the steady rise in ambient noise at low frequencies (10–100 Hz) in many ocean regions—a rate that has been reported to be as high as 3 dB/decade (Andrew et al., 2002, 2011; Chapman and Price, 2011; Miksis-Olds et al., 2013; Miksis-Olds and Nichols, 2016).

Concern about the potential effects of ship noise on marine mammals is not recent, but instead has been raised for decades (e.g., Payne and Webb, 1971; Myrberg, 1978; Geraci and St Aubin, 1980). As ship noise peaks in the low frequencies, early studies primarily focused on low-frequency specialist species such as mysticetes (i.e., baleen whales) (e.g., Eberhardt and Evans, 1962; Cummings and Thompson, 1971). Mysticetes produce and use sound at the frequencies emitted by large ships, and they are considered to be more sensitive at these low frequencies than are other marine mammals (e.g., Parks et al., 2007b; Cranford and Krysl, 2015). However, ships also emit significant energy at higher frequencies (tens of kHz) (e.g., Arveson and Vendittis, 2000; Hermannsen et al., 2014; Veirs et al., 2016), and so odontocetes (i.e., toothed whales, dolphins, and porpoises), which specialize in high-frequency sound usage, can also be affected (e.g., Marley et al., 2017b). Not only commercial ship traffic but also numbers of small boats have been increasing around the world. For example, the number of registered recreational vessels in the United States increased by 1% per annum between 1980 and 2017 (U.S. Department of Homeland Security, 2018). In the state of Florida, there is approximately one registered recreational boat per 17 people (Sidman and Fik, 2005). Similarly, parts of Australia saw increases of 3% per annum between 1999 and 2009 (Nsw Government Maritime, 2010). In Sydney Harbour, 70% of overall vessel traffic is comprised of recreational boats (Widmer and Underwood, 2004). Noise from small boats peaks at higher frequencies (e.g., Erbe, 2013; Erbe et al., 2016b) at which coastal odontocetes are more sensitive (e.g., Houser and Finneran, 2006).

The noise field around a boat or ship is not isotropic (i.e., it is not the same in all directions; e.g., Arveson and Vendittis, 2000). It depends on source frequency and the environment in which the vessel travels, and it changes with vessel speed, load, size, and other factors (e.g., Ross, 1976; Urick, 1983). Consequently, it is not straightforward to translate acoustic recordings made in one environment to others. Obtaining quality recordings of watercraft noise is a science of its own, with numerous flaws that are commonly unrecognized in the literature.

Similarly, determining the responses of marine mammals to watercraft noise has numerous challenges, including constraints in experimental design; variability in species-, population-, and individual-specific characteristics and responses; and context-specific factors that may need to be considered. For example, many studies suffer bias from observer presence in that the majority of marine mammal studies are, by necessity, vessel-based. This introduces a potential source of bias from the presence of the research vessel, as well as the noise it creates. Furthermore, many studies struggle to differentiate between the effects of vessel presence and vessel noise, resulting in confounding explanatory variables. Even if researchers can be confident in noise as the source of disturbance, measurements are often inconsistent between studies, thus complicating comparisons. Animal behavioral responses can also take many forms. Due to the challenges associated with studying these fast-moving, far-ranging, often-submerged animals, the majority of marine mammal behavioral response studies in the wild concentrate on visible changes to physical behavior at the sea surface, such as changes in occurrence or cessation of certain activities. Far fewer consider a combination of behavioral changes, including acoustical behaviors. The resulting knowledge gaps, biases, and uncertainties may be minimized by standardization and interdisciplinary cooperation.

In fact, the effects of watercraft noise on marine mammals is an interdisciplinary field: Sound generation, propagation, measurement, and modeling are physics problems, yet monitoring animals, determining impacts, and understanding biological significance are biological problems. Misinformation and miscommunication have led to numerous issues with underwater acoustic quantities, units, recording and reporting, as well as experimental design, statistical analysis, and interpretation. This review provides an overview of the field of watercraft noise impacts on marine mammals, explains the fundamental physical and biological concepts, highlights common issues and problems, identifies data gaps, and discusses research needs.

GENERATION AND PROPAGATION OF WATERCRAFT NOISE

There is a large variety of motorized boats and ships, such as recreational boats, passenger and car ferries, high-speed hovercraft, cruise ships, tug boats, dredges, dry and liquid cargo vessels, fishing vessels, oil and gas production platforms, research vessels, naval ships, submarines, etc. All of these produce noise. Source levels[1] of 130–160 dB re 1 μPa m have been reported for small watercraft such as jetskis and rigid-hulled inflatable boats

[1]In the case of ship noise, source levels (SL) are typically given as a root-mean-square (rms) sound pressure level (SPL). The sound pressure is recorded at some distance (i.e., in the far-field) from the vessel, and the root-mean-square is computed (i.e., literally squaring the pressure samples, summing, dividing by their number, and taking the square-root). Applying "20 $\log_{10}()$" converts the rms sound pressure to a level quantity (i.e., SPL) in the far-field. Propagation loss is typically modeled and a propagation loss term is added, yielding a (monopole) SL referenced to a distance of 1 m from the source. SPL and SL are thus expressed in dB relative to 1 μPa and 1 μPa m, respectively. Note that the notation of '@ 1 m' is common in the literature but deprecated by the ISO (International Organization for Standardization, 2016, 2017).

(Erbe, 2013; Erbe et al., 2016b). Large and powerful watercraft such as ferries, container ships, and icebreakers have source levels of 200 dB re 1 μPa m and more (e.g., Erbe and Farmer, 2000; Simard et al., 2016; Gassmann et al., 2017). Source levels may vary by 20–40 dB within a ship class due to variability in design, maintenance, and operational parameters such as speed (Simard et al., 2016; Joy et al., 2019).

The strongest noise source is typically the propeller when it cavitates (Ross, 1976). Propeller cavitation involves the formation of bubble clouds behind the propeller. Bubbles of all sizes are created, then grow, vibrate and collapse, producing an overall broadband noise spectrum that ranges from a few Hz to over 100 kHz (Ross, 1976). Traveling at low speed and/or great depth (hence pressure; e.g., submarines) can reduce and avoid propeller cavitation noise. Cavitation noise increases with vessel speed, size, and load (e.g., Ross, 1976; Urick, 1983; Scrimger and Heitmeyer, 1991; Hamson, 1997; Trevorrow et al., 2008; Simard et al., 2016). Cavitation noise is typically amplitude-modulated by the propeller blade rate (i.e., the number of propeller blades times the number of rotations per second; Ross, 1976). 'Propeller singing' refers to narrow-band noise that is a result of vibrating propeller blades. The engine and any machinery onboard a ship also produce noise, and this may couple well into the water through the ship's hull (Urick, 1983). The engine generates narrow-band noise consisting of the engine firing-rate plus overtones (Arveson and Vendittis, 2000). Furthermore, hydrodynamic flow past the hull can lead to vibration of appendages or cavities generating additional narrow-band noise (Urick, 1983). Overall, the noise spectrum emitted by a ship may have multiple sources that contribute noise from different locations about the ship, at different frequencies and into different directions—leading to a complicated and dynamic noise field.

The noise field varies with frequency and angle about a vessel (Arveson and Vendittis, 2000; Trevorrow et al., 2008; Gassmann et al., 2017). Given that boats and ships operate at the water surface and the propeller sits, at maximum, a few meters below the surface, emitted noise reflects at the water surface leading to a strongly downward-directed noise emission pattern (e.g., Gassmann et al., 2017). In physical terms, the source of the watercraft noise and its image source (in air) create a dipole radiation pattern. This means that watercraft noise radiates very well to great depth in the ocean. Radiation in the horizontal plane, near the sea surface, is greatly reduced because of destructive interference of the image source with the real source (i.e., the Lloyd's mirror effect; note that the interference pattern is frequency-dependent). In addition, the hull may shield sound propagation from the propeller in the forward direction. These acoustic radiation phenomena might explain why marine mammals that spend a lot of time at the water surface are prone to vessel strike (e.g., right whales and sirenians) and why bow-riding marine mammals (Würsig, 2018) are not disturbed by the vessel's noise (Gerstein et al., 2005).

As a vessel travels through different environments, from coastal to offshore waters, its noise field changes. In shallow water, the propagating noise repeatedly interacts with the water surface and seafloor, where it is reflected, scattered, and partly absorbed (e.g., Cole and Podeszwa, 1967). The directionality of the noise field is highly variable. In deep water, the directionality is dipolar (i.e., strongly downward) and interactions with, and hence acoustic energy losses at, the seafloor and sea surface are reduced. The noise from watercraft traveling in deep water easily couples into the deep sound channel [i.e., the so-called Sound Fixing And Ranging (SOFAR) channel; e.g., Williams and Horne, 1967; Shockley et al., 1982], where it can traverse entire oceans with very little acoustic energy loss. The noise from watercraft traveling over sloping bathymetry (such as the continental slope) can enter the SOFAR channel with just one seafloor reflection (**Figure 1**). Animals in coastal versus offshore waters or at low versus great depth may experience quite different noise fields—even at the same range from the same vessel.

IMPACTS OF WATERCRAFT NOISE ON MARINE MAMMALS

The effects of underwater noise from anthropogenic activities on marine mammals have been summarized in several works and include the following: behavioral responses, acoustic interference (i.e., masking), temporary or permanent shifts in hearing threshold (TTS, PTS), and stress (e.g., Richardson et al., 1995; Nowacek et al., 2007; Erbe et al., 2018). Acute effects on individual animals are more easily observed, more frequently published, and hence better understood than long-term effects on populations from chronic exposures. Watercrafts are the primary source of chronic noise exposures on marine mammals.

We set out to review the effects of watercraft noise on marine mammals by compiling the literature from a Web of Science search[2], augmented by our personal libraries. The following criteria had to be met for articles to be included in the review. Studies:

- must have dealt with marine vessels;
- must have dealt with marine mammals in water (hence excluding hauled-out pinnipeds);
- may have focused on one or the other;
- must have measured, observed, modeled, or estimated responses (i.e., articles that addressed the potential effects of vessel noise only in the Discussion were excluded); and
- did not need to have measured or modeled source levels or received levels of noise.

A total of 154 articles were included in this review. A rapid growth in the number of publications has occurred since the year 2000 (**Figure 2**). Forty-seven marine mammal species have been studied. The most studied species are the bottlenose dolphin (*Tursiops truncatus*), humpback whale (*Megaptera novaeangliae*), and then beluga whale (*Delphinapterus leucas*) (**Figure 3**). **Figure 4** maps the different study sites by species.

[2]Web of Science search information: Search string: TS = (ship$ OR boat$ OR vessel$) AND TS = noise AND TS = (marine mammal$ OR whale$ OR porpoise$ OR dolphin$ OR seal$ OR sea lion$ OR sealion$ OR dugong$ OR manatee$). Years searched: 1972–2019. Number of returned articles: 504.

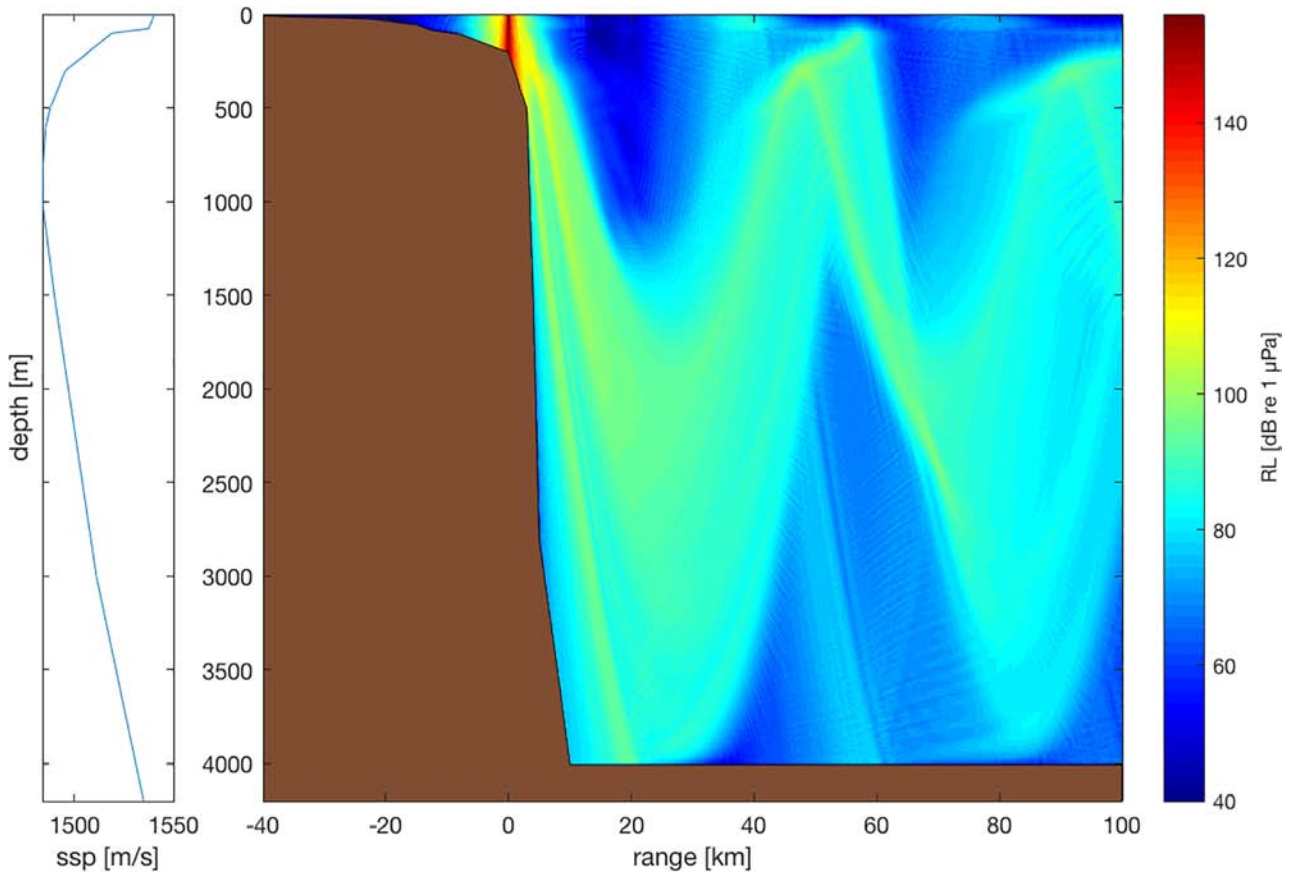

FIGURE 1 | Sketch of the noise field of a cruise ship at the continental slope [at location (0| 0)]. Source spectrum representative of vessel class 'L5' (length: 156 m, speed: 15 knot, source level: 191 dB re 1 μPa m; Erbe et al., 2012). Propagation loss model: RAMGeo in AcTUP V2.8 (https://cmst.curtin.edu.au/products/underwater/). Equatorial sound speed profile (ssp; **left** panel; taken from Shockley et al., 1982). Seafloor modeled as hard, dense limestone (Hamilton, 1980). No ambient noise is included in this plot. Broadband received levels (RL) are color-coded using the scale in the **right** panel. The dipole radiation pattern (i.e., most energy radiating downward) is clearly visible. While sound energy propagates poorly into shallow water (with received levels rapidly decreasing with increasing range), it propagates well (i.e., with little loss) into deep water.

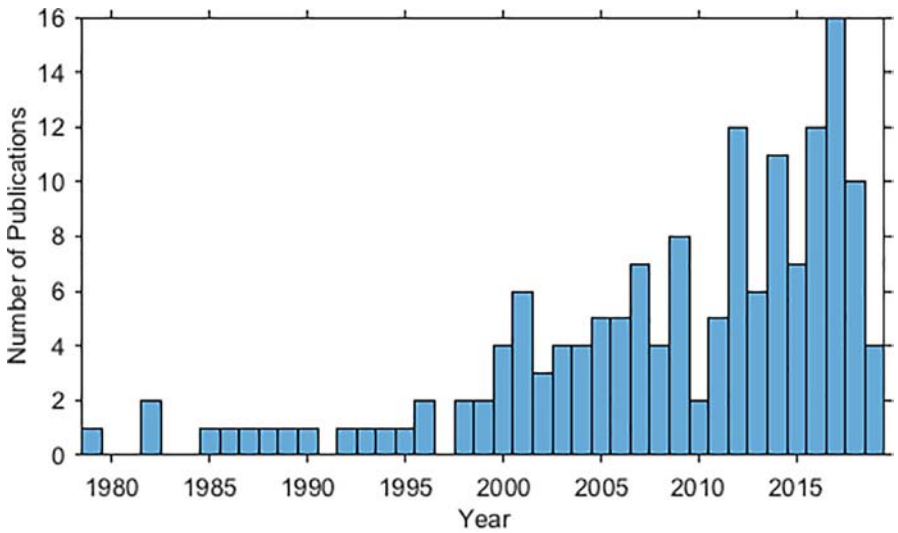

FIGURE 2 | Histogram of the number of publications meeting search criteria—by year (until mid-2019).

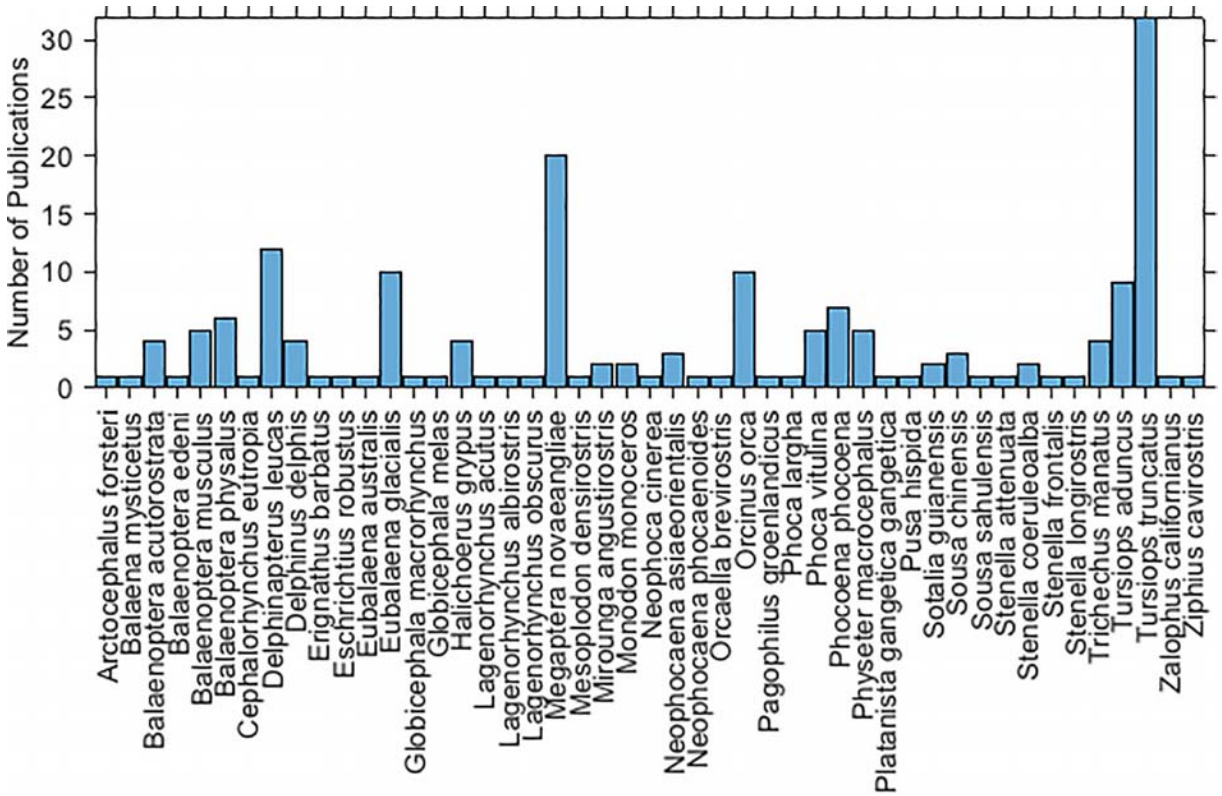

FIGURE 3 | Histogram of the number of publications by marine mammal species.

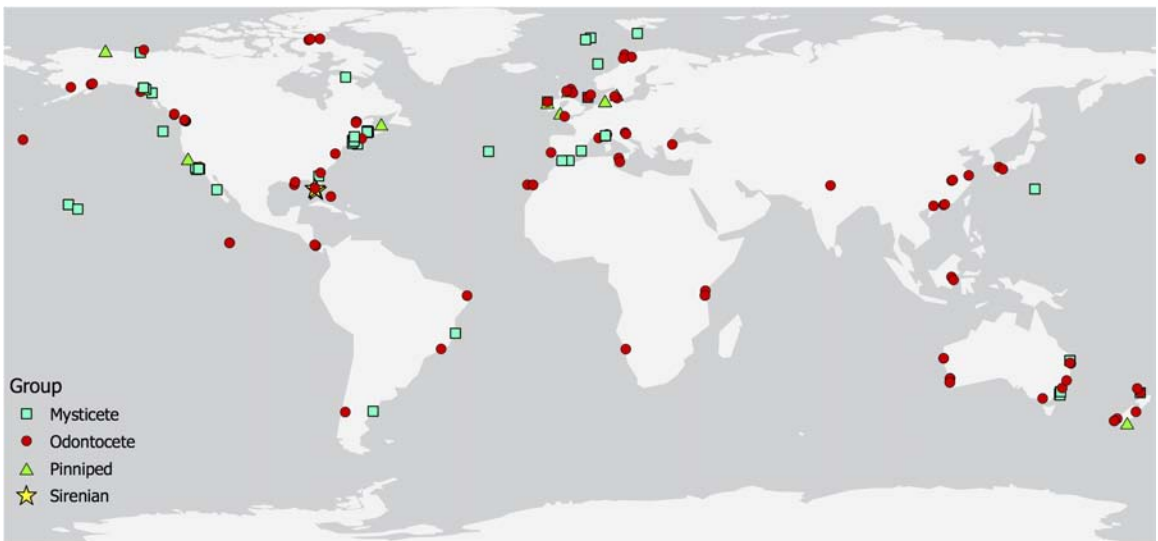

FIGURE 4 | World-map showing locations where the effects of watercraft noise on marine mammals have been studied.

The reported effects of boat or ship noise on marine mammals include changes in both physical and acoustic behavior, masking of communication and echolocation sounds, and stress.

Supplementary Table S1 lists the articles we reviewed and provides information on the following: the types of vessels and marine mammal species studied; the study location, objectives, design, and methodology; and the animal responses observed or modeled. Several interesting patterns are revealed, which are presented in the following sections, along with discussions of the key findings for particular species groups. Additionally, a number of common issues and problems are identified, which highlight research needs.

Mysticetes

In the early 1980s, concern about the effects of shipping and hydrocarbon development in the Arctic led to several multi-year studies on underwater noise effects on bowhead whales (*Balaena mysticetus*; e.g., Richardson et al., 1982; Greene, 1985; Richardson et al., 1985; Johnson et al., 1986). In these studies, experimental approaches of bowhead whales by small vessels at high speed showed that whales generally moved away, thereby interrupting foraging, socializing, and playing behavior, while spending less time at the surface. The early 1980s also saw the first and only playback experiment on the response to vessel noise by gray whales (*Eschrichtius robustus*) in their breeding and nursery habitat off Mexico (Dahlheim, 1987; Dahlheim and Castellote, 2016). Gray whales have a limited repertoire of low-frequency (40–4000 Hz) vocalizations, overlapping with watercraft noise (Dahlheim et al., 1984; Moore and Ljungblad, 1984; Dahlheim and Castellote, 2016; Burnham et al., 2018). In the presence of ships and boats, gray whales increased their vocalization rate, and at times of increased outboard engine noise, received levels from gray whales were higher (interpreted as an increase in source levels; Dahlheim, 1987; Dahlheim and Castellote, 2016).

An increase in studies on the potential effects of vessel noise on a wider range of mysticete species has occurred in recent years. The most extensively studied species is the humpback whale. Humpback whales in Glacier Bay National Park, AK, United States of America, are prone to high noise exposures from tourism vessels and have been shown to increase the amplitude of their vocalizations by 0.8 dB for every 1.0 dB increase in ambient noise, while vocalizing less frequently (Frankel and Gabriele, 2017; Fournet et al., 2018). Similarly, singing individuals near Chichi-jima Island ceased their song after a passenger-cargo vessel passed within 1400 m (Tsujii et al., 2018). Humpback whales off the Australian east coast exhibited great variation in behavioral responses to seismic survey vessels with the airguns turned off. While no behavioral change was seen in some trials, others revealed a decrease in dive duration, travel speed, and the number of breaches (Dunlop et al., 2015, 2016, 2017a,b, 2018). Most humpback whales did not respond to sonar vessels with the sonar turned off (Sivle et al., 2016; Wensveen et al., 2017). Tsujii et al. (2018) found that humpback whales moved away from large vessels, while others noted changes in respiratory behavior (Baker and Herman, 1989; Frankel and Clark, 2002) and a cessation of foraging activities (Blair et al., 2016). The large number of studies on humpback whales and the resulting variety of documented responses demonstrate that context affects behavior.

Conversely, North Atlantic right whales (*Eubalaena glacialis*) show no behavioral response to ship noise at all, or at least not to received levels of 132–142 dB re 1 μPa rms from large ships passing within 1 nm distance, nor to received levels of 129–139 dB re 1 μPa rms (main energy between 50 and 500 Hz) from ship noise playback (Nowacek D.P. et al., 2004). A lack of behavioral response of right whales to ship noise is particularly concerning due to the high levels of ship strike in this species (Laist et al., 2001), affecting their conservation status (Kraus et al., 2005). Nevertheless, analyses of North Atlantic right whale fecal samples suggested that noise from large commercial vessels might increase stress levels (Rolland et al., 2012). In addition, studies suggest that right whales have vocally adapted to environments with increased low-frequency noise through a shift in vocalization frequency and duration (Parks et al., 2007a, 2009, 2011), which may have been a response to compensate for a loss in communication range (Clark et al., 2009). Tennessen and Parks (2016) modeled the communication space of mother-calf pair up-calls in the vicinity of container vessels and found that an up-call would only be detected when the receiving whale was 25 km from the moving vessel and within 320 m of the transmitting whale. Another important social call for right whales, the gunshot, was also found susceptible to masking by vessel noise (Cunningham and Mountain, 2014).

A decrease in communication range as a result of increased levels of ship noise has also been modeled for Bryde's (*Balaenoptera edeni*), fin (*Balaenoptera physalus*), humpback, and minke whales (*Balaenoptera acutorostrata*) (Clark et al., 2009; Cholewiak et al., 2018; Gabriele et al., 2018; Putland et al., 2018). The Lombard effect comprises changes in the spectral features of vocalizations (i.e., in frequency and level) and in vocalization rates, in order to compensate for masking (Lombard, 1911). In addition to the examples from gray, humpback, and right whales above, fin whales lowered the bandwidth, peak frequency, and center frequency of their vocalizations under increased levels of background noise from large vessels (Castellote et al., 2012).

Less attention has been paid to the effects of noise generated by smaller vessels. Dunlop (2016b) predicted an increase in humpback whale social call source levels and the proportion of surface-generated sounds under increased vessel noise, as observed in response to increased wind noise. However, no behavioral changes were observed at received levels of 91–124 dB re 1 μPa rms from a recreational fishing vessel. Au and Green (2000) studied the response of humpback whales to four different whale-watching vessels, each with their own acoustic signature, approaching to 91 m distance. Individual whales responded strongest (i.e., abrupt changes in direction and longer dive durations) to the vessel with the highest received level (127 dB re 1 μPa, 1/3 octave band level at 315 Hz). Several other studies report on the behavioral responses of mysticete whales to smaller vessels in the absence of noise measurements. These studies indicate avoidance of vessels at close range (Palka and Hammond, 2001; Stamation et al., 2010). Changes in behavioral state and respiratory behavior were also observed (Jahoda et al., 2003; Morete et al., 2007), with mother-calf pairs eliciting stronger responses than adults (Morete et al., 2007).

Odontocetes

Much of the significant early work on the potential effects of watercraft noise on odontocetes was—similar to studies on mysticetes—a result of concern about Arctic industrial development (hydrocarbons, mining, and shipping) in the early 1980s (e.g., LGL Ltd., 1986; Finley et al., 1990; Richardson et al., 1990). The focal species were beluga whales and narwhals (*Monodon monoceros*). In response to icebreakers, beluga whales lost pod integrity, commenced rapid movement, asynchronous and shallow dives, and changed their vocal behavior (i.e., vocalization types) at received levels of 94–105 dB re 1 μPa rms (20–1000 Hz), while narwhals changed their locomotion (i.e., exhibited more directed and slower movement, became

motionless, and sank) and fell silent at received levels of about 124 dB re 1 μPa rms (20–1000 Hz) (LGL Ltd., 1986; Cosens and Dueck, 1988; Finley et al., 1990). Since the 1990s, beluga whale responses to boats and ships have been studied more extensively in the St. Lawrence Estuary, Canada. Here, beluga whales have shown increasing avoidance (i.e., increased dive duration and swim speed) with the number of boats, as well as other changes in both physical and acoustic behavior (Blane and Jaakson, 1994; Lesage et al., 1999). The Lombard effect has been demonstrated as an increase in source level, vocalization rate, and frequency (i.e., shift to higher frequencies; Lesage et al., 1999; Scheifele et al., 2005).

In the case of beaked whales, much effort has been spent on understanding the potential effects of ship-based sonar transmissions given coincident strandings and naval exercises (e.g., DeRuiter et al., 2013; Goldbogen et al., 2013; Sivle et al., 2015; Kvadsheim et al., 2017). The effects of ship noise without sonars have been investigated less. Using passive acoustic monitoring and acoustic tags, ship noise at received levels of approximately 135 dB re 1 μPa rms (0.1–45 kHz) affected beaked whale foraging by reducing both the horizontal area in which animals foraged and the number of successful prey captures (as indicated by the number of feeding buzzes recorded), with foraging efficiency reduced by >50% (Aguilar Soto et al., 2006; Pirotta et al., 2012). Similarly, fewer clicks were recorded of sperm whales (*Physeter macrocephalus*) during vessel passes (Azzara et al., 2013), and decreases in surface time, respiration interval, and the number of ventilations were reported in the presence of whale-watching boats (Gordon et al., 1992). A different study found no decrease of sperm whale acoustic detections in ship noise (André et al., 2017). Rather, an increase in sperm whale acoustic and visual detections was found near longline fishing vessels, and propeller cavitation noise (to be exact, changes in that noise corresponding to typical operational changes in longline fishing vessel speeds) was identified as the 'dinner bell' attracting sperm whales to depredate (Thode et al., 2007). Such diverse responses (avoidance, no response, and attraction) highlight the importance of context in assessments of underwater noise.

Killer whales (*Orcinus orca*) in British Columbia and Washington State have recently received much attention with regards to impacts from ships, given the steady decline in their population size. Changes in behavior (i.e., less foraging and increased surface-active behavior), respiration, and swim speed and direction occurred at received levels above 130 dB re 1 μPa rms (0.01–50 kHz), and the Lombard effect (i.e., increased source level and vocalization duration) has been reported in ship noise levels above 98 dB re 1 μPa rms (1–40 kHz) (Foote et al., 2004; Holt et al., 2009, 2011; Lusseau et al., 2009; Noren et al., 2009; Williams et al., 2002, 2014). This geographic area has seen a lot of ship noise recording, quantification, and impact modeling studies (e.g., Erbe, 2002; Erbe et al., 2012, 2014; Williams et al., 2015; Cominelli et al., 2018; Joy et al., 2019).

A great deal of research has also focused upon smaller delphinids. Occupying habitats from freshwater rivers to coastal estuaries and the open ocean, dolphins often experience high habitat overlap with human activities. In particular, the potential impacts from dolphin-watching tourism vessels have been investigated (e.g., Scarpaci et al., 2000; Lusseau, 2003a, 2005, 2006; Constantine et al., 2004; Lusseau and Higham, 2004; Bejder et al., 2006; Stensland and Berggren, 2007; Arcangeli and Crosti, 2009; Christiansen et al., 2010; Steckenreuter et al., 2012; Guerra et al., 2014; May-Collado and Quinones-Lebron, 2014; Symons et al., 2014; Heiler et al., 2016; Pérez-Jorge et al., 2016). Dolphins were displaced or changed their site occupancy in response to vessel traffic (Lusseau, 2005; Bejder et al., 2006; Rako et al., 2013; Pirotta et al., 2015b; Pérez-Jorge et al., 2016). They altered their movement patterns within an area in response to vessel traffic, with animals changing their direction of travel, beginning to travel erratically, or significantly increasing traveling speeds when approached by vessels (Au and Perryman, 1982; Nowacek et al., 2001; Mattson et al., 2005; Lemon et al., 2006; Lusseau, 2006; Christiansen et al., 2010; Marley et al., 2017b). Watercrafts can cause a shift in dolphin behavioral budgets, generally increasing time spent traveling whilst decreasing time spent resting and socializing (Lusseau, 2003a; Constantine et al., 2004; Stensland and Berggren, 2007; Arcangeli and Crosti, 2009; Steckenreuter et al., 2012; Marley et al., 2017b). Other changes in behavior can include alterations to dive patterns, displays of breathing synchrony, and changes in inter-animal distances (Janik and Thompson, 1996; Nowacek et al., 2001; Hastie et al., 2003; Kreb and Rahadi, 2004; Stensland and Berggren, 2007). Furthermore, dolphins have been observed to alter their whistle characteristics, such as their frequency range, in elevated noise conditions or in the presence of vessels (Morisaka et al., 2005; May-Collado and Wartzok, 2008; Guerra et al., 2014; May-Collado and Quinones-Lebron, 2014; Papale et al., 2015; Heiler et al., 2016; Rako Gospić and Picciulin, 2016; Marley et al., 2017b). Changes to whistle duration have also been reported (May-Collado and Wartzok, 2008; Guerra et al., 2014; May-Collado and Quinones-Lebron, 2014), as have increases in whistle production rates (Scarpaci et al., 2000; Van Parijs and Corkeron, 2001; Buckstaff, 2004; Guerra et al., 2014; Martins et al., 2018).

However, delphinid studies are heavily biased toward particular species, with some receiving considerably more research attention than others. The bottlenose dolphin (*Tursiops* spp.) has been the focus of the most research effort of all the odontocetes. Bottlenose dolphins have a cosmopolitan distribution, ranging from northern Scotland to southern New Zealand and occupying both coastal and pelagic habitats. As a result, they are available to marine mammalogists around the world, and so dominate the literature. Bottlenose dolphins are also the most common cetacean kept in captivity, which has facilitated a range of physiological studies regarding the impacts of noise that have not been possible for other species; e.g., studies on how behavioral and acoustical changes affect energetics. Dolphin metabolic rates increase during periods of vocal effort and sound production, with energy requirements varying according to the type of sound produced (Noren et al., 2013; Holt et al., 2015, 2016). This combined with increased energy expenditure due to more time spent traveling, moving at speed, avoiding vessels, or leaving impacted areas, results in disturbance having potential cumulative energetic consequences.

Conversely, little is known about the responses of dolphin species that inhabit relatively constrained systems that are also some of the world's busiest waterways. The river systems utilized by these species are known to have high levels of vessel traffic and, in some cases, there is evidence of river dolphins being the target of tourism activities (e.g., boto, *Inia geoffrensis*, in Brazil; de Sá Alves et al., 2012). Ganges river dolphins (*Platanista gangetica gangetica*) showed mixed responses to approaching vessels, including changing direction to orient away from the boat, prolonging dive times, and displaying attraction toward the boat, as well as no obvious effect (Bashir et al., 2013). Such variability, again, shows the importance of context in behavioral responses. Finally, there is a clear paucity of publications addressing the responses of river dolphins (Families *Iniidae, Platanistidae*, *Pontoporiidae,* and *Lipotidae*) to vessel traffic or noise.

Similarly, of the porpoise species, only harbor porpoises (*Phocoena phocoena*) and finless porpoises (Indo-Pacific, *Neophocaena phocaenoides*; Yangtze, *N. asiaeorientalis asiaeorientalis*) have been studied with regards to the impact of watercraft. Harbor porpoises moved away from vessels (Palka and Hammond, 2001), showed higher levels of porpoising in the presence of boats (Dyndo et al., 2015), changed behavioral states (Akkaya Bas et al., 2017), reduced foraging behavior (Wisniewska et al., 2018), and experienced decreased communication ranges (Hermannsen et al., 2014). Acoustic tags (DTAGs) placed on harbor porpoises in Danish waters showed that animals encountered vessel noise 17–89% of the time, and exhibited vigorous fluking, bottom diving, interrupted foraging, and cessation of echolocation during some high vessel noise events (received level > 96 dB re 1 μPa at 16 kHz 1/3 octave band; Wisniewska et al., 2018). Meanwhile the Yangtze finless porpoise has been shown to forage in busy (port) areas exhibiting high vessel traffic, with no detected impact on echolocation behavior (Dong et al., 2012; Wang et al., 2014). Wang et al. (2015) proposed that the high prey densities in the ports in comparison to surrounding areas mean porpoises need to forage there regardless of boat traffic. The closely related Indo-Pacific finless porpoise appears not to exhibit the same pattern, with echolocation behavior showing a negative correlation with ship traffic (Akamatsu et al., 2008). Porpoises may be more vulnerable to this type of disturbance due to their small size and low fat reserves, such that any disturbance that reduces foraging opportunities may result in negative fitness consequences (Nabe-Nielsen et al., 2014; Wisniewska et al., 2016).

Sirenians

Knowledge about the potential effects of watercraft noise on sirenians grew from curiosity of why these animals did not avoid approaching boats and whether they perhaps could not hear them. Fatal collision with watercraft is a serious problem that has been recognized since the 1970s (Ackerman et al., 1992; O'Shea, 1995; Marsh et al., 2001; Rycyk et al., 2018). The majority of these fatalities are a result of blunt force trauma rather than propeller cuts (Lightsey et al., 2006). Vessel strike is the main source of mortality for some populations (e.g., 25% of all Florida manatee, *Trichechus manatus latirostris*, deaths; Calleson and Kipp Frohlich, 2007). Consequently, an understanding of the hearing capabilities of sirenians has been of interest to determine the capabilities of sirenians to detect watercraft noise. There are no data for dugong (*Dugong dugon*); however, manatee hearing underwater is sensitive at 1–30 kHz (Klishin et al., 1990; Popov and Supin, 1990; Gerstein et al., 1999; Gaspard et al., 2012). This overlaps with the spectrum of noise from boats, raising the question of why manatees do not manage to avoid a vessel strike. The current hypothesis is that, as they spend a great deal of time very close to the sea surface, received noise levels from watercraft are low due to the Lloyd's mirror effect and less sound radiation toward the bow. This, combined with manatees' relatively low movement speed, leaves manatees vulnerable to vessel strikes (e.g., Gerstein et al., 1999).

Conversely, some behavioral studies have concluded that manatees (*Trichechus* spp.) are able to detect and respond to approaching boats, often changing their orientation (heading or roll), depth, diving behavior, behavioral state, and swimming speed (Nowacek S.M. et al., 2004; Miksis-Olds et al., 2007b; Rycyk et al., 2018). Such responses to vessels were more pronounced for vessels in close proximity and traveling at speed (Nowacek S.M. et al., 2004). Dugongs were also affected by close boat approaches and less likely to continue feeding when vessels traveled within 50 m (Hodgson and Marsh, 2007). Manatees foraged in habitat with lower ambient noise (that included vessel noise below 1 kHz), particularly at times with less boat density (Miksis-Olds et al., 2007a). Playback experiments simulating different boats at different speeds approaching to within 10 m supported earlier behavioral response studies that manatees swam to deeper waters in the presence of boat noise (Miksis-Olds et al., 2007b).

Pinnipeds

Pinnipeds are amphibious and haul out on land or ice to breed, pup, molt, and rest. Consequently, much of the research examining vessel traffic has focused on the easily observable reactions of hauled-out pinnipeds to approaching boats and ships. This includes the haul-out behavior of harbor seals (*Phoca vitulina*) (Andersen et al., 2012; Blundell and Pendleton, 2015), Australian fur seals (*Arctocephalus pusillus doriferus*) (Stafford-Bell et al., 2012), Saimaa ringed seals (*Phoca hispida saimensis*) (Niemi et al., 2013), Australian sea lions (*Neophoca cinerea*) (Osterrieder et al., 2017), and walrus (*Odobenus rosmarus*) (Øren et al., 2018). A small number of studies also extend observations to the water surrounding haul-out sites (Osterrieder et al., 2017). Common reactions of pinnipeds to approaching vessels include flushing off haul-out sites into the sea (Jansen et al., 2010; Andersen et al., 2012; Blundell and Pendleton, 2015), increased alertness (Henry and Hammill, 2001), and head raising (Niemi et al., 2013). However, these studies focused on the reactions of pinnipeds to the presence of a vessel rather than perceived levels of vessel noise. Studies that incorporate in-air noise generation, transmission, and reception are very rare (Tripovich et al., 2012). In-air watercraft noise and the perception of sound in air are notably different from their underwater equivalents (Kastak and Schusterman, 1998). Therefore, the remainder of this section and **Supplementary Table S1** focus on studies investigating the impacts of underwater watercraft noise on pinnipeds.

Underwater noise from watercraft has the potential to mask or alter the communication of pinnipeds. Bagočius (2014) showed that gray seal (*Halichoerus grypus*) vocalizations recorded underwater in captivity overlapped with the noise spectrum of a vehicle/passenger ship. Terhune et al. (1979) reported a decrease in the loudness of underwater harp seal (*Pagophilus groenlandicus*) vocalizations after the presence of a vessel was recorded acoustically near whelping sites in the Gulf of St. Lawrence. This may have reflected a change in seal vocalizations or the movement of seals away from the recording area (Terhune et al., 1979).

Studies on the behavioral responses of pinnipeds to shipping noise have been undertaken at a range of spatial scales. A national-scale assessment of seals and shipping in the United Kingdom showed high rates of co-occurrence between gray seals or harbor seals and shipping traffic within 50 km of the coastline near haul-out sites (Jones et al., 2017). At regional and local scales, it was estimated, using sound propagation models, that harbor seals in the Moray Firth were exposed to 24-h cumulative SEL[3] between 170 dB re 1 $\mu Pa^2 s$ (95% CI 168–172) and 189 dB re 1 $\mu Pa^2 s$ (95% CI 173–206) from shipping (Jones et al., 2017). When considering the upper limits of the 95% confidence intervals, these predicted values exceeded the estimated thresholds for the onset of TTS (Southall et al., 2007, 2019). Locally in Broadhaven Bay, Ireland, gray seals potentially varied habitat use in response to vessels as indicated by a negative correlation between the numbers of gray seals and construction vessels (Anderwald et al., 2013). A recent study using acoustic tags (DTAGs) that record sound and behavior concurrently showed that harbor and gray seals were exposed to vessel noise 2.2–20.5% of their time at sea (Mikkelsen et al., 2019). In response to vessel noise, a tagged seal changed its diving behavior, switching quickly from a dive ascent to descent (Mikkelsen et al., 2019). This observation agrees with descriptions of changes in diving reported during the development of early acoustic recording tags on juvenile northern elephant seals (*Mirounga angustirostris*) (Fletcher et al., 1996; Burgess et al., 1998). Studies using acoustic recording tags on pinnipeds demonstrate the potential opportunities, and the need, to further explore the impact of shipping noise on the at-sea behavior of pinnipeds.

APPROACHES TO STUDY DESIGN

In order to compare studies, identify focus areas and research gaps, and point out common issues and problems, we defined a 'study' as a unique combination of publication reference and species. For example, if a publication dealt with two species, then this was counted as two studies. However, if a publication investigated the same species at two different sites, then this was counted as one study.

With this definition, an approximately equal number of studies dealt with large ships as with small boats (ratio: 1.05:1). Animal responses to these vessels were observed in the wild in 82% of studies, while 4% of studies were done in captivity and 14% of studies used models instead of live animals. The majority of studies on live animals dealt with real vessels *in situ*, while 5% were playback studies of pre-recorded sound.

In terms of measuring animal responses, 34% of studies undertook vessel-based observations, 19% land-based observations, and 8% aerial observations. Passive Acoustic Monitoring (PAM) was employed in 33% of studies, and tags were used in 13% of studies. Some studies used more than one method of observation. Studies were designed as controlled exposure experiments (14%) or before-during-after observations (29%), while 21% were opportunistic in nature.

Out of all studies, 28% determined the received noise level at the study animals, 13% measured the received level, 12% used a sound propagation model to determine the received level, and 3% applied a geometric propagation loss. In addition to determining the received level, 15% of studies also considered frequency-dependent hearing sensitivity of the animals (e.g., audiograms or critical bands). A total of 41% of studies neither estimated the received level nor the range of the vessel to the animals.

In terms of context, 58% of studies considered vessel-related factors such as vessel numbers, types, speeds, distances, directions of approach, etc. Environmental factors such as location, habitat type, bathymetry, tide, sea state, temperature, prey presence, and ambient noise (in addition to vessel noise) were considered by 42% of studies. Biological factors such as group demographics, behavioral state, speed of movement etc., were considered by 46% of studies. Only 17% of studies did not consider any contextual variables. However, the majority used only very few and basic contextual variables such as range to the vessel, ambient noise, and current behavioral state.

COMMON ISSUES AND PROBLEMS

With Physics: Estimation of Exposure, Recording, and Playback of Vessel Noise

Studies on the effects of watercraft noise on marine mammals would ideally be able to determine the sound levels received by the animals and the total sound exposure (i.e., the integral of the squared sound pressure over time; International Organization for Standardization, 2017). Few studies employed acoustic recording tags on the animals, which store a record of received levels over time right at the animal. The majority of studies that determined received levels did so by modeling and estimation. In this case, watercrafts are recorded at some site, source levels are estimated, and these estimates are then applied to mostly different situations (i.e., locations, environments, and times of year) for the computation of received levels. There are common problems with all of these steps.

Measuring ship noise is not as simple as lowering a hydrophone over the side of a boat. Over-the-side deployments as well as hydrophones suspended straight from surface buoys may record noise from wave action against the boat or buoy, and show artifacts from the hydrophone moving through the water with the waves, affecting acoustic recordings at frequencies from

[3]The sound exposure level (SEL) is a measure of the total noise energy over time. It is computed as the time-integral of the squared pressure, before applying 10 $\log_{10}()$, and it is expressed in dB relative to 1 $\mu Pa^2 s$ (International Organization for Standardization, 2017).

a few Hz to a few kHz (e.g., Strasberg, 1979; Cato, 2008; Erbe et al., 2016c). Common in moored deployments, flow noise is an artifact of recording resulting from hydrodynamic flow past the hydrophone, which causes non-acoustic pressure fluctuations at approximately 0.005–1 kHz range (e.g., Buck and Greene, 1980; Erbe et al., 2015). Strong currents might set mooring ropes and legs into vibration and resonance, causing mooring noise at a few hundred Hz to a few kHz (e.g., Koper et al., 2016). Metal chains and shackles in moorings cause clonking noise in the same frequency range (∼100 Hz to a few kHz; e.g., Marley et al., 2017a). Many such artifacts can be minimized with hydrophones deployed on the seafloor (e.g., McCauley et al., 2017), though soft seafloor material such as sand moving over the hydrophone may contaminate acoustic recordings up to a few kHz (e.g., Erbe, 2009). Alternatives are arrangements that drift freely with the currents. The recorder is suspended from a buoy via a suspension system, which may comprise a drogue and a bungee that decouple the hydrophone from surface wave action. Similarly, a catenary (or distributed buoyancy) arrangement will decouple the hydrophone and spatially remove it from potential noise generated at the surface buoy (**Figure 5**). Building noise-free moorings is an art, and different designs may be required for different situations.

An international standard has recently been developed for the measurement of ship noise in deep water (i.e., water depth more than 150 m or 1.5 × ship length, whichever is greater) (International Organization for Standardization, 2016). The ship travels along a pre-defined course, and recordings are taken from both port and starboard aspects. While the standard does not specify a certain speed, it would be good to obtain measurements at multiple speeds representing typical operational speeds. Recording is done in the geometric far field (i.e., closest point of approach 100 m or 1 × ship length, whichever is greater) with a vertical array, having three hydrophones at specified inclination angles from the ship. The 'radiated noise level' (RNL, referenced to 1 m) is computed by applying a geometric (spherical) spreading loss term [20 $\log_{10}$(range)] over the slant range for each hydrophone and then averaging over all hydrophones. This averaging smooths over the Lloyd's mirror interference pattern. The RNL is useful for noise emission studies, but may lead to large errors when used to estimate received levels at animals in other environments. This is because the environment in which the ship was recorded affects RNL. The recent release of Part 2 of this standard (International Organization for Standardization, 2019) provides formulae to estimate equivalent monopole source levels that correct for surface effects.

In order to compute environment-corrected monopole source levels, sound propagation models need to be applied that translate levels recorded at long range to levels normalized to 1 m range. There are a number of sound propagation models to choose from—depending on the environment (e.g., Etter, 2003; Jensen et al., 2011). The resulting source levels can then be inputted into sound propagation models for other environments in order to estimate received levels at the animals (e.g., Erbe et al., 2013; Williams et al., 2014). If a spherical spreading loss term is applied rather than a sound propagation model, then source levels are commonly under-estimated if the recording hydrophone was at shallow inclinations from the ship. Conversely, if monopole source levels are taken from the literature and a spherical loss is applied, then received levels may be over-estimated, when the receiving animal is at shallow inclinations from the ship. These are likely common problems in the literature. For example, the RNLs (re 1 m) of cargo vessels reported by McKenna et al. (2012) and Veirs et al. (2016) were up to 15 and 25 dB less than the source levels (re 1 m) of Simard et al. (2016), respectively, likely due to an underestimation of propagation loss. This is because of the dipole radiation pattern of a ship and its image source, yielding a propagation loss well above the wrongly, yet commonly applied 20 $\log_{10}$(range) at shallow inclination angles (e.g., Ainslie et al., 2014). Using sound propagation models, Chen et al. (2017) showed that gray seals experienced step changes of up to 20 dB in the received ship noise levels as they dove throughout the water column in the Celtic Sea. This was because of environmental features such as thermoclines, which a geometric propagation loss model cannot account for.

Finally, once recordings of watercraft have been obtained, they are sometimes played back to animals in different environments for response studies. The recorded sound was affected (in frequency and level) by the environment in which the recordings were made and by the recording system. It will likely be broadcast in yet another, different environment, resulting in further affected received spectrum levels. In addition, the speaker used for playback will have a frequency response, which can distort the signal. Ideally, the speaker's frequency response is measured, and the playback signal is digitally filtered with the inverse of the frequency response before the playback study. Furthermore, the underwater speaker used will have a rather different sound radiation (i.e., directivity) pattern from the recorded vessel. Finally, it is impossible to simulate an approaching vessel with a single, moored speaker, because not only the received level changes as a vessel approaches, but also its spectrum and directionality.

With Biology: Experimental Design, Disturbance Differentiation, and Biological Significance

One of the most fundamental aspects of experimental design is ensuring that fair comparisons are made. In many response studies, this requires having some idea of 'normal' animal behavior in the form of a control group, with which treatment groups can then be compared for deviations that could imply disturbance (Johnson and Besselsen, 2002). However, here field-based marine mammal studies typically hit a problem: Despite the advancements of acoustic and visual monitoring techniques over recent decades, many fundamental questions regarding marine mammal behavior remain unanswered. As a result, the scientific community are still trying to determine the realms of normal behavior, hindered by continual new discoveries describing range expansions, diving abilities, hearing capabilities, and so on (e.g., Schorr et al., 2014; Cranford and Krysl, 2015; Accardo et al., 2018). Furthermore, all animals are individuals and the response of any given individual may change based on its current

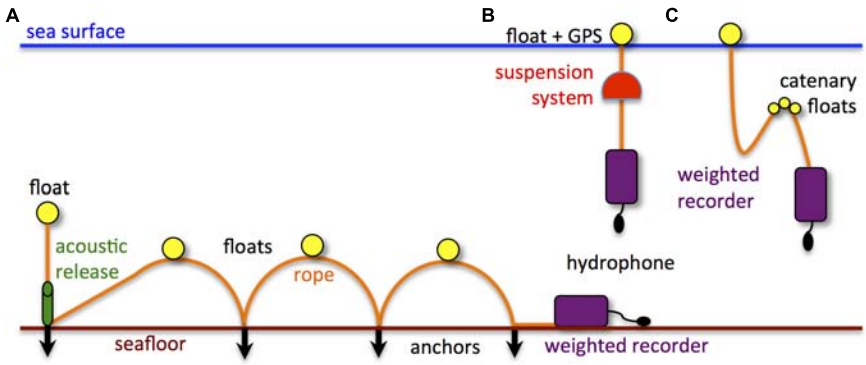

FIGURE 5 | Sketch of hydrophone deployments that minimize mooring artifacts: **(A)** recorder on the seafloor and spatially decoupled from the release mechanism; **(B)** recorder suspended from a surface float via a decoupling suspension system and freely drifting; and **(C)** recorder suspended via a catenary system. The latter could be freely drifting from a surface float of attached to a seafloor anchor.

requirements and motivational states (e.g., health, reproductive status, age, energetic requirements; Pirotta et al., 2015a). Overall, this means that within the same species, individuals may respond differently in different environments and at different times, depending upon their previous experience with man-made noise and the importance of the habitat they are occupying for their current life-function requirements. Additionally, as previously discussed, animal behavioral responses can take many forms. This can make it difficult to conclusively identify when disturbance has occurred.

Similarly, a lack of control contexts can further confound results. There are few environments globally which have not experienced anthropogenic stressors (Halpern et al., 2015). Thus, there are few 'naive' populations of marine mammals to serve as baselines in behavioral response studies. This raises the question of habituation (e.g., Cox et al., 2001). Do we see no behavioral response to noise because the population is already used to the presence of such sounds? If so, did behavioral responses ever occur or have animals developed strategies to deal with these noisy environments? And, if such strategies exist, do they evoke an energetic or reproductive cost to the animals involved?

It is possible to account for anthropogenic, biological, and environmental contexts by including a suite of additional variables. In fact, the majority of studies we reviewed tried to account for at least one form of context. Some contextual factors, however, have not been addressed in impact assessments of underwater noise, such as the role of nearby conspecifics (Dunlop, 2016a) or nearby animals of other species (e.g., Koper and Plön, 2016). Contextual data of any type may not always be available or obtainable at a sufficient spatial or temporal resolution to coincide with quick behavioral events (Mannocci et al., 2017). And so, this leads to the issue of sample size. Statistical models with too many variables and insufficient sample size will fail to converge. Consequently, there are minimum sample sizes required for different statistical tests and levels of precision (e.g., Hampton et al., 2019). Unfortunately, the optimum sample size generally cannot be calculated until after the study has been completed. Methods for estimating it beforehand require some knowledge about variation within the study population (Dell et al., 2002); but, such variation remains poorly understood for the majority of marine mammal species. While increasing the sample size is statistically preferable, the majority of marine mammal studies suffer sample-size restrictions due to logistics and economics.

Once the best possible experimental design has been implemented, there is the problem of disturbance differentiation. Firstly, impact assessment studies are often confounded by the fact that the majority of marine mammal studies are boat-based. This introduces a potential source of observer bias from the presence of the research vessel and the noise it creates. Such bias is unavoidable in many situations, although increasingly researchers are attempting to include this in their analyses (e.g., Lusseau, 2003b). In coastal settings, land-based observations are more readily implementable and may help reduce (or totally exclude) any influence from observer presence. However, this does not assist in resolving the question of whether animals respond to the physical presence of a vessel or if responses are due to the noise that vessel creates, or to any other factor in the environment.

And so, despite the best intentions, many response studies may be restricted to relatively simple analyses, such as the use of basic comparative statistics (such as *t*-tests, ANOVAs, and non-parametric equivalents) to look at one particular behavioral response with and without the presence of ships. This is not to say that such studies are of no value—every result adds another piece to the overall puzzle. But they by no means capture the full context of the situation. Now that long-term datasets are in existence, researchers are increasingly able to apply more complex analytical techniques, consider individual motivations in the study species, and even make predictions using agent- or context-based modeling (e.g., Ellison et al., 2012; Nabe-Nielsen et al., 2014; Pirotta et al., 2014).

Once analytical techniques have been applied, the final question is whether any observed response actually matters in terms of biological significance. Behavioral changes associated with anthropogenic activities are often assumed to equate to a

biologically significant effect (New et al., 2013; Curé et al., 2016). Individuals exposed to novel forms or chronic levels of disturbance may be displaced from critical habitat, disrupted from key activities, and thus suffer lower individual fitness, reproductive success, or overall survival (New et al., 2013). However, this may not be the case for infrequent disturbance resulting in instantaneous or short-term responses. For example, although animals may initially leave a site when exposed to anthropogenic activities, this may not equate to their utilizing lower-quality habitats or experiencing long-term, broad-scale displacement (Thompson et al., 2013). Recently, several studies have attempted to investigate biological significance using advanced mathematical models that allow for complexity of animal behavior, motivational state, social structure, and exposure to anthropogenic activities (e.g., New et al., 2013). Unfortunately, ground-truthing the outcomes is logistically challenging, requiring long-term studies at the individual- and population-level. Therefore, most behavioral studies are still restricted to establishing links between short-term measures and long-term population consequences (New et al., 2014).

RESEARCH NEEDS

As can be seen from **Supplementary Table S1**, research on the potential impacts of watercraft on marine mammals has been patchy—in terms of its coverage of species, geographic areas, vessel type, and type of impact. As a result, there are a number of knowledge gaps resulting in several obvious research needs.

Species Coverage

The Society for Marine Mammalogy currently recognizes 126 extant species of cetaceans, pinnipeds and sirenians. While 47 of these species have been studied regarding the impacts of vessel noise, the vast majority have received no attention or at maximum, one publication. More than half (64%) of the mysticete species have at least been the topic of a publication once, as have about half (46%) of the delphinid (Family *Delphinidae*) and half (43%) of the porpoise (Family *Phocoenidae*) species. However, of all the river dolphins (Families *Iniidae, Platanistidae*, *Pontoporiidae,* and *Lipotidae*—noting that the latter was declared possibly extinct in 2006), only one publication was found. All of the 22 species of beaked whales (Family *Ziphiidae*) are deep-diving pelagic species and rather cryptic, and so only two have been studied with regard to noise impacts. In terms of sirenians, only the Florida manatee appears in the literature on vessel noise impacts. Out of the pinnipeds, four of 18 phocids (true seals) and one otariid (i.e., eared seals) have been included in publications on responses to vessel noise at sea. Note that we did not review publications on the potential effects of approaching vessels on hauled-out pinnipeds, as underwater noise would not have been the cause.

The most-commonly studied species identified in this review were bottlenose dolphins and humpback whales. Ease of access might have played a role, as these species are widespread and at times exceptionally coastal. Thus their popularity as a target species for vessel noise impact studies does not necessarily reflect their being a research priority, although many populations do inarguably experience high levels of vessel traffic and noise. In comparison, given that river dolphins experience a multitude of anthropogenic stressors, including often-chronic noise from boats, it is perhaps surprising that these species have not had greater research focus. Rivers are among the most threatened ecosystems in the world (Tockner et al., 2010); but these systems represent problematic study sites for cetacean research. For example, the Indus River dolphin (*Platanista gangetica minor*) historically occurred in approximately 3,400 km of the Indus River and its tributaries; surveying this extensive, narrow and convoluted system is logistically challenging (Braulik, 2006; Jensen et al., 2013). Finding river dolphins and tracking them during response studies is difficult. The literature thus far has consequently focused on abundance estimates and status assessments, as well as documenting and mitigating immediately lethal threats (e.g., bycatch; Smith and Smith, 1998), as opposed to potentially less-obvious threats such as disturbance from vessels and noise. Similarly, the potential impacts on cryptic species like deep-diving, pelagic beaked whales are perhaps not always apparent or easy to study. But impacts could be biologically significant, given the sheer volume and density of ocean traffic, coupled with a vertically downward focused sound radiation pattern and a deep-ocean sound propagation environment that enables very long propagation distances.

Non-cetacean species received considerably less research attention. Sirenians are predominantly found in coastal areas, whereas pinnipeds are tied to land; both these characteristics mean these animals inevitably have high habitat overlap with human activities. Yet the impacts of those activities in terms of their physical presence and associated noise remain poorly understood.

Geographic Area

Another group of species that has been under-represented are those utilizing Antarctic waters. Annually migrating mysticetes critically depend on the Antarctic Ocean in the austral summer for feeding, as they do not feed while on their tropical breeding grounds in the austral winter. Some of the phocid species are truly Antarctic in the sense that they are present there all year round. Antarctica is predominantly governed by high-income countries, and thus might be expected to receive higher levels of research attention. Ship noise, in particular, is rapidly increasing off Antarctica due to booming tourism and heightened fisheries effort (Erbe et al., 2019). While Arctic marine mammals were first studied several decades ago, at a time when industrial development (i.e., mostly offshore oil and gas) was expected to grow rapidly, no such impetus has yielded a research increase in Antarctica. In fact, not a single publication has addressed the potential effects of watercraft noise on marine mammals in Antarctic waters, perhaps because of an absence of oil and gas exploration (as prohibited under the Antarctic Treaty) and the associated funding that accompanies such work. However, the expanding tourism and fishing industries may offer opportunities for future research work.

Not all areas have such opportunities. Marine mammal conservation at a global scale is challenged by a lack of basic

information on species presence, but this is particularly true in the developing world (Braulik et al., 2018). For instance, as noted above, river ecosystems have received relatively little research attention. However, in addition to being logistically challenging study areas, those utilized by river dolphins are all located in developing countries, and so local researchers also experience considerable socio-economic challenges when conducting even baseline research. Overall, the majority of publications identified in this review originated from developed countries. Although this likely in part reflects funding or resource availability, it could also reflect publishing practices. For example, in this review only English-publishing journals were included. Furthermore, whilst studies may have taken place on the impacts of noise on marine mammals in developing countries, this research may not have reached the international, peer-reviewed publication stage. It is likely that this information is available, but difficult to access or not publically available (e.g., internal reports, environmental impact assessments, or local conservation and management plans). Therefore, there is a need not only for greater research in particular geographic areas, but also for sharing of research outcomes with a global audience.

Vessel Type

Vessels ranging from small, rigid-hulled inflatable whale-watching boats to large, powerful icebreakers have been investigated with regards to their potential impacts on certain species of marine mammal. Some combinations of vessel type and marine mammal species are more common than others in the literature. For example, the effects of cetacean-watching tourism vessels have most commonly been studied on bottlenose dolphins, then killer whales, humpback whales, and beluga whales. As tourism vessels are directly targeting marine mammals, it is reasonable to be concerned about the impacts these may have on the animals of interest. This is particularly true in areas where multiple trips occur each day or multiple tourism vessels are in operation, as this could lead to cumulative exposure and impacts. Additionally, cetacean-tourism vessels can also act as platforms of opportunity, allowing researchers the chance to study these animals from the tourism vessel itself rather than a dedicated research vessel. However, whilst there are many studies investigating the impacts of cetacean tourism, few specifically consider noise from tourism vessels.

In comparison, small recreational watercraft, such as jetskis, have received relatively little attention. Recreational watercraft may also have cumulative impacts on marine mammals, with an individual animal potentially encountering a multitude of vessels each day. Personal watercraft are considerably more challenging to document than tour vessels, but, given the continual increase in personal watercraft ownership, these vessels are of increasing concern with regards to noise impacts on marine mammals.

Type of Impact

The types of noise impacts that have been studied are as patchy as the coverage of species, areas, and vessels. Risk assessments are often based on the assumption that affected animals will leave the area. However, as summarized above, there is overwhelming evidence that marine mammals can display a wide range of behavioral responses, ranging from the obvious (e.g., area avoidance) to the subtle (e.g., shifts in acoustic behavior or raised cortisol levels). Measuring these responses comes with a number of logistical challenges; consequently, many studies have historically focused on the former, easier-to-identify response types. Recent technological developments have facilitated a rise in the number of studies targeting subtler types of impact, which will undoubtedly continue over coming years. However, there is still a need for integrative studies that simultaneously consider multiple response types in order to capture the variation associated with different species, populations, cohorts, and individuals.

One obvious pattern is that the effects of noise on the vocalizations of dolphins have been studied more than on those of other marine mammals. Perhaps this is due to the ease at which coastal dolphins can be recorded these days and due to the stereotypical nature of their vocalizations. This does not imply that acoustic communication is more important (and hence of more concern) in dolphins than other species. In fact, a range of responses can be evidence of disturbance, and more studies simultaneously looking at both physical and acoustical behavior are needed.

A significant gap in our knowledge is our lack of understanding of the potential long-term and population-level impacts and the corresponding biological significance. It could be argued that if a response does not equate to having biological significance, then it is of least concern; such conclusions would have obvious regulatory and management implications, but require considerable ground-truthing. This emphasizes the need for long-term, broad-scale studies targeting a range of response types to examine their consequences at the individual and population level. Physical and vocal behavioral changes impact an individual's energetic costs (Noren et al., 2013; Holt et al., 2015, 2016; Williams et al., 2017), but knowledge on how these costs affect other biologically important functions (e.g., growth and reproduction) is currently absent. Even if population consequences could be ascertained, the question remains how these consequences affect the structure, function, and stability of the ecosystem of which the population is a part (Wong and Candolin, 2015). Recent research has focused on developing a framework for assessing the population consequences of disturbance (PCoD) using sparsely available data, supplementing it with expert elicitation to link changes in individual behavior or physiology to vital rates, and incorporating these into stochastic population models (King et al., 2015; Harwood et al., 2016). This methodology has the benefit of being able to model population consequences on the best available data, identifying gaps in understanding to focus research efforts, and being able to be updated as more data becomes available.

DISCUSSION

The potential for watercraft noise to impact marine mammals is considerable. Some interactions have received particular attention, such as small boats affecting coastal dolphins; cetacean-watching boats affecting the specific populations of whales, dolphins, and porpoises that they target; large commercial ships affecting threatened species such as gray and southern

resident killer whales; and icebreakers affecting Arctic mysticetes and odontocetes. Reasons for these specific combinations of vessel type and species include spatio-temporal overlap in presence, identified research needs (such as an expected rise in industrialization of the Arctic due to climate change), conservation urgency (as in the case of the southern resident killer whale), and ease of access (such as coastal and tourism-targeted species).

Other patterns, in addition to specific species-vessel combinations, emerge. For example, research looking at the effects of small vessels is primarily related to vessel behavior without mentioning noise produced by these vessels. This is in contrast to larger vessels, where the noise factor is more often taken into account. Overall, our understanding of the potential effects of watercraft noise on marine mammals exhibits a number of 'holes.'

In this article, we have summarized the information available in the literature, highlighted some of the data gaps, and identified common problems. Standards are needed for both physical and biological aspects of study design, data collection (including recording of vessel noise and animal responses), data analysis, modeling, and reporting to avoid common mistakes and make results comparable and synthesisable (Erbe et al., 2016a). Given the inter-disciplinary nature of the field of noise impacts on marine fauna, multi-disciplinary teams are needed to ensure consistent quality of outcomes.

While this article focused on the impacts of ship noise on marine mammals, ship noise also impacts other marine fauna such as fish (e.g., Slabbekoorn et al., 2010; Simpson et al., 2016) and crustaceans (e.g., Wale et al., 2013). The potential bioacoustic impacts on these species have been of concern for as long as those on marine mammals (Myrberg, 1978). However, despite the longevity of these concerns, there remains an information paucity for many species, populations, and cohorts in terms of the impacts of noise, responses invoked, and biological significance of disturbance. As well as being a concern in its own right, this topic also has biological significance for marine mammals in terms of impacts on their prey species.

Overall, ship traffic is expected to keep increasing by approximately 4% per year over the coming five years, with different rates predicted for different ship types (United Nations Conference on Trade and Development [UNCTAD], 2018). Ship noise is a loss in energy, and vibrating propellers, appendages, and cavities are a structural risk; therefore, there is a natural incentive for the shipping industry to maintain its vessels and thus reduce noise (Leaper and Renilson, 2012; Leaper et al., 2014). Reducing ship noise for environmental reasons has also been on the agenda of the International Maritime Organization (IMO) publishing guidelines on quieting technologies and methods for newly built, as well as existing, vessels (International Maritime Organization [IMO], 2014). Particularly quiet vessels have been designed for defense and research applications, demonstrating that significant reductions in a ship's noise footprint are achievable (Mitson, 1995; Fischer and Brown, 2005; Bahtiarian and Fischer, 2006; De Robertis et al., 2013; Palomo et al., 2014). The conundrum remains though, whether quieter vessels pose a higher risk of collision with marine mammals.

CONCLUSION

The impacts of ship noise on marine mammals continue to be of great concern. Despite this and increasing research attention over recent years, a number of common problems exist in terms of both the physics and biology of this inter-disciplinary issue. Consequently, a number of knowledge gaps remain. However, growing awareness, improving technology, increasing availability of multi-variate data streams, and analytical advancements have started to provide much-needed context for impact assessments. The continuing growth of long-term data sets is enabling much-needed assessments of chronic exposures at the individual and population level of marine mammals. As a scientific community, we should endeavor to address the gaps highlighted in this review to strategically target under-represented species, geographic areas, vessel types, and types of impact.

AUTHOR CONTRIBUTIONS

CE conceived, designed, and led the study, collected and analyzed the data, prepared the figures and table, authored and reviewed the drafts of the manuscript, and approved the final manuscript. SM and RS contributed to the design of the study, collected the data, prepared the figures, authored and reviewed the drafts of the manuscript, and approved the final manuscript. JS, LT, and CE collected the data, authored and reviewed the drafts of the manuscript, and approved the final manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00606/full#supplementary-material

TABLE S1 | Publications on the effects of ship noise on marine mammals: Reference, vessel type, species, study location, latitude, longitude, study objectives, study design, range to vessel, types of responses, received level, anthropogenic covariates, environmental covariates, biological covariates, and sample size.

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Managing the Effects of Noise From Ship Traffic, Seismic Surveying and Construction on Marine Mammals in Antarctica

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The Protocol on Environmental Protection of the Antarctic Treaty stipulates that the protection of the Antarctic environment and associated ecosystems be fundamentally considered in the planning and conducting of all activities in the Antarctic Treaty area. One of the key pollutants created by human activities in the Antarctic is noise, which is primarily caused by ship traffic (from tourism, fisheries, and research), but also by geophysical research (e.g., seismic surveys) and by research station support activities (including construction). Arguably, amongst the species most vulnerable to noise are marine mammals since they specialize in using sound for communication, navigation and foraging, and therefore have evolved the highest auditory sensitivity among marine organisms. Reported effects of noise on marine mammals in lower-latitude oceans include stress, behavioral changes such as avoidance, auditory masking, hearing threshold shifts, and—in extreme cases—death. Eight mysticete species, 10 odontocete species, and six pinniped species occur south of 60°S (i.e., in the Southern or Antarctic Ocean). For many of these, the Southern Ocean is a key area for foraging and reproduction. Yet, little is known about how these species are affected by noise. We review the current prevalence of anthropogenic noise and the distribution of marine mammals in the Southern Ocean, and the current research gaps that prevent us from accurately assessing noise impacts on Antarctic marine mammals. A questionnaire given to 29 international experts on marine mammals revealed a variety of research needs. Those that received the highest rankings were (1) improved data on abundance and distribution of Antarctic marine mammals, (2) hearing data for Antarctic marine mammals, in particular a mysticete audiogram, and (3) an assessment of the effectiveness of various noise mitigation options. The management need with the highest

score was a refinement of noise exposure criteria. Environmental evaluations are a requirement before conducting activities in the Antarctic. Because of a lack of scientific data on impacts, requirements and noise thresholds often vary between countries that conduct these evaluations, leading to different standards across countries. Addressing the identified research needs will help to implement informed and reasonable thresholds for noise production in the Antarctic and help to protect the Antarctic environment.

Keywords: underwater noise, Antarctica, marine mammal, Antarctic Treaty, ship, seismic survey, noise management

INTRODUCTION

The Antarctic Treaty was established for the protection of the Antarctic, allowing scientific research but prohibiting military activity. It entered into force in 1961 and has since been signed by 53 Parties. Its Protocol on Environmental Protection (the Protocol) entered into force in 1998, stipulating that the protection of the Antarctic environment and associated ecosystems be fundamentally considered in the planning and conducting of all activities in the Antarctic Treaty area (area south of 60°S, i.e., approximately south of the Antarctic Convergence, including all ice shelves). While fishing was deemed allowable by the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) in 1982, the Protocol prohibits all activities relating to Antarctic mineral and hydrocarbon resources, except for scientific research.

Parties implement the Protocol via national acts and laws. For example, in Germany, the Act Implementing the Protocol on Environmental Protection to the Antarctic Treaty (AIEP, 1998) identifies the German Environment Agency (Umweltbundesamt, UBA) as the competent authority for assessing and permitting German activities in the Antarctic. The AIEP and the Protocol protect native animals at individual and population levels. Activities that molest, handle, capture, injure or kill a native mammal or bird are prohibited (Annex II to the Protocol). However, exceptions can be granted for scientific or educational purposes. A permit cannot be issued if the activity is suspected to cause (a) harmful changes to the distribution, abundance or productivity of an animal species or its populations, (b) threats to endangered species or populations, or (c) significant detrimental effects on the environment and associated ecosystems. Any scientific research that is deemed by UBA to have the potential to create at least a minor or transitory impact is also evaluated by an independent committee of scientific experts (Sachverständigenkommission Antarktis, SV-KOM).

Underwater noise is part of almost all anthropogenic activities in the Antarctic, ranging from ship traffic to construction and scientific seismic surveys (**Figure 1**). Such noise can have profound effects on marine organisms and has been identified as a major stressor in the marine environment (see the collection of articles covering a diversity of species in Popper and Hawkins, 2016). Yet, no specific guidelines for noise production in the Antarctic have been established and noise has only once been considered at the Meetings of the Committee for Environmental Protection (CEP) since 2012.[1] The CEP normally meets once a year in conjunction with the Antarctic Treaty Consultative Meeting (ATCM) and (a) addresses matters relating to environmental protection and management, (b) provides advice to the ATCM, and (c) formulates measures or resolutions in furtherance of the principles and objectives of the Treaty for the adoption through the ATCM. The Scientific Committee on Antarctic Research (SCAR) is an inter-disciplinary committee of the International Science Council (ISC) and provides scientific advice to the Parties at the ATCM.

Arguably, amongst the species most vulnerable to noise are marine mammals since they specialize in using sound for communication, navigation and foraging, and therefore have evolved sensitive auditory systems (Au et al., 2000). The effects of ship noise on marine mammals have recently been reviewed (Erbe et al., 2019). Knowledge about the effects of noise on marine mammals is mostly based on studies from regions other than the Southern Ocean. Documented effects include potential

[1] ATCM 2019, WP 68, "Anthropogenic Noise in the Southern Ocean: an Update," submitted by SCAR.

FIGURE 1 | Sketch of sources of underwater noise in the Antarctic. All vessels (fishing vessels, cruise ships, research vessels, etc.) produce underwater noise in a nearly omni-directional pattern (indicated by circular sound wavefronts). Ships use echosounders that scan the sea floor with a narrow swatch of sound (indicated in yellow). Research station infrastructure and support includes construction activities, vessels as well as aircraft—all of which may be detected under water.

increases in stress (Rolland et al., 2012), behavioral changes such as short- and long-term avoidance of affected areas (Nowacek et al., 2007; Götz and Janik, 2013), auditory masking (Erbe et al., 2016), hearing threshold shifts (Finneran, 2015), and—in extreme cases—death (Schrope, 2002). Studies conducted outside of the Antarctic have shown that reactions to noise differ widely between marine mammal species (Ellison et al., 2012).

The Southern Ocean is in many ways not comparable to other ocean basins. In terms of biodiversity, the Antarctic is home to a range of marine species that cannot be found elsewhere on the globe. Some species are year-round residents of Antarctic waters, such as the ice-breeding pinniped species. Other species migrate to the Antarctic annually to forage. In fact, the Antarctic is of critical importance to migrating mysticete whales, which come here during the austral summer for feeding. During this time, they take in a large proportion (possibly up to 80%) of their annual energy requirements and store substantial amounts of lipids (some grow their body weight by 30–100%; Brodie, 1975; Lockyer, 1981; Reilly et al., 2004). In terms of acoustics, the marine soundscape of the Southern Ocean is a unique combination of sounds from Antarctic fauna, weather events and ice (plus anthropogenic sounds). Underwater sound propagation is strongly influenced by the low water temperature and ice cover around the Antarctic continent. Thus, we set out to determine the current state of knowledge on the effects of underwater noise on marine mammals in the Antarctic, to identify knowledge gaps, and to discuss research needs.

MARINE MAMMALS IN THE ANTARCTIC

Eight mysticete species (and subspecies), 10 odontocete species, and six pinniped species have been observed south of 60°S (**Table 1**). Out of these, the Antarctic blue whale is listed by the International Union for Conservation of Nature and Natural Resources (IUCN; iucnredlist.org) as "critically endangered," the pygmy blue whale, fin whale, and sei whale are listed as "endangered," the Antarctic minke whale is "near threatened," and the sperm whale is "vulnerable." Arnoux's, Gray's, and strap-toothed beaked whales, as well as the killer whale are data deficient; so their conservation status cannot be determined. Other Antarctic marine mammals are currently listed as "least concern."

With regard to the application and interpretation of the legal regulations relating to the Antarctic Treaty area, it is important to ascertain which marine species are relevant: The Environmental Protocol protects individual members of "native" mammal species and also protects the populations of all animal species, including sporadically occurring species. In this context, the word "native," which is used in the Environmental Protocol, has the same meaning as the notion of "true" Antarctic species, as defined in Boyd et al. (2002): "*those species whose populations rely on the Southern Ocean as a habitat, i.e., critical to a part of their life history, either through the provision of habitat for breeding or through the provision of the major source of food.*" For the Protocol, however, the "native" criterion is only applied to individual members of a species. With regard to populations, protection is extended to both native and non-native animal species, including those that occur only rarely, such as *Phocoena dioptrica*.

Information on distribution and abundance of Antarctic marine mammals is mostly scarce, although annual surveys were conducted as part of the International Whaling Commission (IWC) circumpolar IDCR/SOWER programs between 1978/79 and 2003/04. These programs surveyed a sector of roughly 60° longitude each year, from the ice edge to 60°S, generating abundance estimates for a number of species including the Antarctic blue whale (Branch, 2007), humpback whale (Branch, 2011), and Antarctic minke whale (IWC, 2013). These and most other visual surveys have been generally confined to ice-free areas and undertaken during the brief austral summer. Information on migrations, spatial distribution, and abundance in ice-covered areas (e.g., Herr et al., 2019) or during other times of the year is limited though growing—for example, as a result of autonomous passive acoustic monitoring, which can collect information on acoustic presence year-round (e.g., Van Opzeeland et al., 2008; Van Parijs et al., 2009). Field research in the Antarctic is expensive and limited in space and time, resulting in numerous data gaps (**Table 2**).

The available information indicates that blue, fin, humpback, and minke whales are found all the way to the ice edge throughout the austral summer season, with the peak of fin and humpback whale encounters tending to be further away from the ice edge than the highest densities of Antarctic blue and Antarctic minke whales (e.g., Tynan, 1998; Williams et al., 2014b). Passive acoustic observations have shown that Antarctic blue, Antarctic minke, and humpback whale distributions are, however, not limited by ice (van Opzeeland et al., 2013; Dominello and Širović, 2016; Thomisch et al., 2016). Observations of Antarctic minke whales show this species predominantly occurs in areas with dense ice cover (Williams et al., 2014b; Herr et al., 2019). Fin whales are acoustically present year-round in some areas (E. Burkhardt pers. comm.), although in other areas they seem to avoid ice cover (Sirovic et al., 2004; Herr et al., 2016). Sei and southern right whales are typically not encountered at the ice edge (Kasamatsu et al., 1996; Best, 2007). Killer whales occurring in Antarctic waters comprise four different ecotypes, which all occur beyond the ice edge in pack-ice areas (see de Bruyn et al., 2013 for a review). Southern bottlenose and Arnoux's beaked whales occur in open water south of 60°S up to the ice edge. Arnoux's beaked whales have furthermore during summer been observed to occur in pack-ice areas (Best, 2007). Sperm whales, other beaked whales, and the smaller odontocetes are found further away from the ice edge (Kasamatsu and Joyce, 1995). The Antarctic fur seal and the southern elephant seal are ice-tolerant, but open-water species, that generally depend on land for breeding (Boyd et al., 1998; Bornemann et al., 2004; Hindell et al., 2016). The crabeater, leopard, Ross, and Weddell seals also have pelagic phases, but are bound to the presence of sea-ice for breeding and molt, with each species exhibiting different sea-ice habitat requirements (Ropert-Coudert et al., 2014; Siniff, 2015). Distribution maps for marine mammals occurring around Antarctica are shown in the *Biogeographic Atlas of the Southern Ocean* (Ropert-Coudert et al., 2014).

TABLE 1 | Marine mammal species in the Antarctic, based on Ropert-Coudert et al. (2014).

Species Latin Name	Species Common Name	IUCN Conservation Status	Sightings <60°S
Mysticetes			
Balaenoptera acutorostrata	Dwarf minke whale	Least concern	common
Balaenoptera bonaerensis	Antarctic minke whale	Near threatened	regular
Balaenoptera borealis	Sei whale	Endangered	common
Balaenoptera musculus brevicauda	Pygmy blue whale	Endangered	rare
Balaenoptera musculus intermedia	Antarctic blue whale	Critically endangered	regular
Balaenoptera physalus	Fin whale	Endangered	regular
Eubalaena australis	Southern right whale	Least concern	common
Megaptera novaeangliae	Humpback whale	Least concern	regular
Odontocetes			
Berardius arnuxii	Arnoux's beaked whale	Data deficient	common
Globicephala melas	Long-finned pilot whale	Least concern	common
Hyperoodon planifrons	Southern bottlenose whale	Least concern	regular
Lagenorhynchus cruciger	Hourglass dolphin	Least concern	common
Mesoplodon grayi	Gray's beaked whale	Data deficient	rare
Mesoplodon layardii	Strap-toothed whale	Data deficient	rare
Orcinus orca	Killer whale	Data deficient	regular
Phocoena dioptrica	Spectacled porpoise	Least concern	rare
Physeter macrocephalus	Sperm whale	Vulnerable	regular
Ziphius cavirostris	Cuvier's beaked whale	Least concern	rare
Pinnipeds			
Arctocephalus gazella	Antarctic fur seal	Least concern	regular
Hydrurga leptonyx	Leopard seal	Least concern	regular
Leptonychotes weddellii	Weddell seal	Least concern	regular
Lobodon carcinophaga	Crabeater seal	Least concern	regular
Mirounga leonina	Southern elephant seal	Least concern	regular
Ommatophoca rossii	Ross seal	Least concern	regular

Reported encounter rates for mysticetes and odontocetes peak in January and February (Kasamatsu and Joyce, 1995; Kasamatsu et al., 1996). Many mysticetes migrate to the Antarctic in the austral summer to feed before they migrate to warmer waters where they breed in the austral winter (Lowther, 2018). There is increasing evidence of mysticete presence in the Antarctic throughout the austral winter from passive acoustic recordings (Sirovic et al., 2009; van Opzeeland et al., 2013; Thomisch et al., 2016). Of odontocetes, some killer whale ecotypes are resident in the Antarctic all year-round (Pitman and Ensor, 2003). Of sperm whales, only males venture this far south and stay over the winter (Kasamatsu and Joyce, 1995). Winter surveys indicate that 20 or more cetacean species have regular, potentially year-round presence in the Antarctic (Thiele and Gill, 1999; Thiele et al., 2004; van Waerebeek et al., 2010). Amongst pinnipeds, elephant seals and Antarctic fur seals forage in the Antarctic in the austral winter, but breed on subantarctic islands like the Kerguelen Islands, Macquarie Island, or South Georgia during the summer (Boyd et al., 2002; Rodríguez et al., 2017). All other Antarctic seal species are ice breeding and are resident in Antarctic waters south of the Antarctic Convergence year-round. However, some of these species, in particular leopard seals, can also be found on subantarctic islands (Lowther, 2018).

Information on the diet of some Antarctic marine mammals is scarce, though data are available for some whale species from whaling records, and for other species diet can be inferred from the same or related species in other geographic regions (see Pauly et al., 1998 for an overview). The mysticetes feed primarily on krill, but may also take small fish, zooplankton, and possibly squid. The odontocetes eat fish and squid, with certain killer whale ecotypes also hunting penguins and other marine mammals (both cetaceans and pinnipeds). Antarctic pinnipeds forage on krill, fish, zooplankton, and squid, with leopard seals also taking other seals and seabirds. Lowther (2018) provides a recent summary of the diets of Antarctic marine mammals.

Given the potential for anthropogenic activities occurring in Antarctic waters to affect critical life functions of marine mammals, it is imperative for environmental impact assessments to consider impacts on the acoustic habitat of marine mammals. Marine mammals actively and passively use sound in support of their various life functions, as do at least some of their prey species. Sound plays a role in marine mammal behavioral contexts, comprising social encounters, feeding, mother-offspring recognition, and mating (van Opzeeland et al., 2010; Janik, 2014; Reichmuth and Casey, 2014). Odontocetes use active biosonar for navigating and foraging (Au, 1993). All marine mammals likely listen to environmental sounds, as well as the sounds of predators and prey (Gannon et al., 2005; Janik, 2005). Interfering with sound usage and sensing while marine

TABLE 2 | Available information and knowledge gaps regarding Antarctic marine mammals.

		Knowledge on Antarctic abundance, trends, distribution, seasonality (0–5)	Knowledge on behavior while in Antarctica (0–5)	Under-water audio-gram	Data on TTS	Data on PTS	NOAA functional hearing group	Data on stress	Data on masking (e.g., CR)	Impacts of seismic surveying	Impacts of ships, noise and strikes	Impacts of sonar	Other responses to noise	Knowledge of prey species (0–5)	Info on PCoD
Mysticetes															
Balaenoptera acutorostrata	Dwarf minke whale	1 (Cooke, 2018a)	1				LF					Sivle et al., 2015; Kvadsheim et al., 2017	McGarry et al., 2017	2 (Skaug et al., 1997; Kato and Fujise, 2000; Secchi et al., 2003; Perrin and Brownell, 2009)	
Balaenoptera bonaerensis	Antarctic minke whale	3 (Kasamatsu et al., 1996; Leaper et al., 2008a; Bravington and Hedley, 2012; Herr et al., 2019)	2				LF							3 (Armstrong and Siegfried, 1991; Perrin and Brownell, 2009)	
Balaenoptera borealis	Sei whale	2 (Kasamatsu et al., 1996; Leaper et al., 2008a; Horwood, 2009)	1				LF							2 (Baumgartner and Fratantoni, 2008)	
Balaenoptera musculus brevicauda	Pygmy blue whale	2 (Branch, 2007; Branch et al., 2018; Cooke, 2018b)	2				LF							4 (Branch, 2007; Cooke, 2018b)	
Balaenoptera musculus intermedia	Antarctic blue whale	4 (Kasamatsu et al., 1996; Sirovic et al., 2004, 2009; Matsuoka et al., 2006; Branch, 2007; Leaper et al., 2008a; Thomisch et al., 2016; McCauley et al., 2018)	3				LF				Laist et al., 2001; Berman-Kowalewski et al., 2010	Goldbogen et al., 2013		4 (Branch, 2007; Cooke, 2018b)	
Balaenoptera physalus	Fin whale	3 (Gambell, 1985; Kasamatsu et al., 1996; Branch and Butterworth, 2001; Leaper et al., 2008a; Herr et al., 2016)	2				LF			Castellote et al., 2012	Laist et al., 2001; Castellote et al., 2012			2 (Aguilar, 2009)	
Eubalaena australis	Southern right whale	2 (International Whaling Commission, 2001; Leaper et al., 2008a)	2				LF	Rolland et al., 2012			Laist et al., 2001			2 (Kenney, 2009)	

(Continued)

TABLE 2 | Continued

		Knowledge on Antarctic abundance, trends, distribution, seasonality (0–5)	Knowledge on behavior while in Antarctica (0–5)	Under-water audio-gram	Data on TTS	Data on PTS	NOAA functional hearing group	Data on stress	Data on masking (e.g., CR)	Impacts of seismic surveying	Impacts of ships, noise and strikes	Impacts of sonar	Other responses to noise	Knowledge of prey species (0–5)	Info on PCoD
Megaptera novaeangliae	Humpback whale	4 (Kasamatsu et al., 1996; Branch, 2006, 2011; Pastene et al., 2006; Zerbini et al., 2006; Leaper et al., 2008a; Herr et al., 2016)	3				LF		(Dunlop et al., 2010) Lombard Effect	Dunlop et al., 2017	Laist et al., 2001; Blair et al., 2016	Sivle et al., 2015, 2016		3 (Baraff et al., 1991; Clapham, 2009)	
Odontocetes															
Berardius arnuxii	Arnoux's beaked whale	1 (Friedlaender et al., 2010; van Waerebeek et al., 2010)	1				MF							0	
Globicephala melas	Long-finned pilot whale	2 (Kasamatsu and Joyce, 1995; van Waerebeek et al., 2010)	2	Pacini et al., 2010			MF		(Visser et al., 2016) anti-masking behavior	Stone and Tasker, 2006		Antunes et al., 2014		2 (Gannon et al., 1997; Santos et al., 2014; Monteiro et al., 2015)	
Hyperoodon planifrons	Southern bottlenose whale	1 (Kasamatsu and Joyce, 1995; Branch and Butterworth, 2001)	2				MF							1 (MacLeod et al., 2003)	
Lagenorhynchus cruciger	Hourglass dolphin	1 (Kasamatsu and Joyce, 1995; Goodall, 1997)	1				HF							1 (Fernández et al., 2003; Braulik, 2018)	
Mesoplodon grayi	Gray's beaked whale	1 (van Waerebeek et al., 2010)	1				MF							1 (MacLeod et al., 2003)	
Mesoplodon layardii	Strap-toothed whale	1 (van Waerebeek et al., 2010)	1				MF							1 (Sekiguchi et al., 1996; MacLeod et al., 2003)	
Orcinus orca	Killer whale	3 (Kasamatsu and Joyce, 1995; Gill and Thiele, 1997; Branch and Butterworth, 2001; Leaper et al., 2008b; van Waerebeek et al., 2010)	3	Hall and Johnson, 1972; Szymanski et al., 1999; Branstetter et al., 2017)			MF	Ayres et al., 2012	(Bain and Dahlheim, 1994) spatial masking release, (Holt et al., 2011, 2009) Lombard Effect	Stone and Tasker, 2006	Ayres et al., 2012; Williams et al., 2014a	Miller et al., 2014		3 (Lauriano et al., 2007; Ford, 2009; Pitman and Durban, 2010, 2012)	

(Continued)

TABLE 2 | Continued

		Knowledge on Antarctic abundance, trends, distribution, seasonality (0–5)	Knowledge on behavior while in Antarctica (0–5)	Under-water audio-gram	Data on TTS	Data on PTS	NOAA functional hearing group	Data on stress	Data on masking (e.g., CR)	Impacts of seismic surveying	Impacts of ships, noise and strikes	Impacts of sonar	Other responses to noise	Knowledge of prey species (0–5)	Info on PCoD
Phocoena dioptrica	Spectacled porpoise	1 (Sekiguchi et al., 2006)	0				HF							1 (Natalie et al., 2018)	
Physeter macrocephalus	Sperm whale	3 (Kasamatsu and Joyce, 1995; Whitehead, 2002)	2				MF			Miller et al., 2009	Laist et al., 2001	Curé et al., 2016; Isojunno et al., 2016	Cure et al., 2013	3 (Whitehead, 2009)	
Ziphius cavirostris	Cuvier's beaked whale	1 (van Waerebeek et al., 2010)	1				MF				Aguilar Soto et al., 2006	Fernandez et al., 2005; Krysl et al., 2006; DeRuiter et al., 2013; Falcone et al., 2017		1 (MacLeod et al., 2003)	
Pinnipeds															
Arctocephalus gazella	Antarctic fur seal	3 (Hofmeyr, 2016)	3				OW							4 (Hofmeyr, 2016)	
Hydrurga leptonyx	Leopard seal	1 (Rounsevell and Eberhard, 1980; Borsa, 1990; Rogers, 2009; Southwell et al., 2012)	3				PW							2 (Rogers, 2009; Southwell et al., 2012)	
Leptonychotes weddellii	Weddell seal	1 (Hückstädt, 2015a)	3				PW							2 (Burns et al., 1998)	
Lobodon carcinophaga	Crabeater seal	1 (Hückstädt, 2015b)	3				PW							2 (Hückstädt et al., 2012)	
Mirounga leonina	Southern elephant seal	3 (Murray, 1981; Heimark and Heimark, 1986; Bester, 1988; McCann and Rothery, 1988; Bester and Hofmeyr, 2005; van den Hoff et al., 2007; Hofmeyr, 2015; Hindell et al., 2016)	3				PW							4 (Brown et al., 1999; Piatkowski et al., 2002; Bradshaw et al., 2003; Cherel et al., 2008; Field et al., 2011)	New et al., 2014
Ommatophoca rossii	Ross seal	1 (Hückstädt, 2015c)	2				PW							1 (Southwell et al., 2012)	

The columns relating to available knowledge show a ranking of 0 (none) to 5 (good) by the authors. TTS: Temporary threshold shift. PTS: Permanent threshold shift. NOAA's functional hearing groups are: low-frequency (LF), mid-frequency (MF), high-frequency (HF), otariids in water (OW), and phocids in water (PW). CR: Critical ratio. PCoD: Population Consequences of Disturbance.

mammals undergo critical life functions in the Antarctic can affect individuals and possibly populations in the Antarctic and beyond.

UNDERWATER ANTARCTIC NOISE

Ambient noise in the Antarctic can be of abiotic, biotic, and anthropogenic origin. Wind over open ocean leads to the entrainment of bubbles, which produce a broad spectrum of sound (Knudsen et al., 1948). Wind blowing over ice produces a different spectrum of sound. Wind and currents move ice flows and push icebergs together or against the seabed, resulting in distinct rubbing and cracking sounds, with the former being quite tonal in character (e.g., Gavrilov and Li, 2007). Temperature changes lead to ice cracking, which is typically impulsive and broadband.

Polar waters can be both noisier and quieter than the open ocean. The ice edge typically is an active acoustic zone with high sound levels due to ice breaking, colliding, and shearing (Haver et al., 2017). Conversely, it is quieter under the ice fields during stable conditions (Mikhalevsky, 2001). Marine mammals, fish, and crustaceans produce sound, often prolifically, resulting in continuous choruses in characteristic frequency bands. A multi-year recording at 0°E, 66°S was analyzed to present a statistical analysis of biotic and abiotic ambient noise, as a function of wind speed and ice cover, showing that whale and seal choruses generated distinct peaks in the ambient noise spectra (e.g., Antarctic blue whale chorus at 15–30 Hz, fin whales at 95–105 Hz, minke whales at 90–200 Hz, and leopard seals at 320–350 Hz; Menze et al., 2017).

Anthropogenic underwater noise in the Southern Ocean originates from ships—mostly research vessels, cruise ships, and fishing vessels. During the 2017/2018 austral summer, 98 research stations and 51 research or research support ships were registered with the Council of Managers of National Antarctic Programs (COMNAP)[2], 53 tourism ships were registered with the International Association of Antarctica Tour Operators (IAATO)[3], and 46 fishing vessels reported to the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)[4].

The latest season where data from COMNAP, IAATO, and CCAMLR were available for a more detailed analysis was the austral summer of 2016/2017. In terms of the cumulative amount of time spent by these types of vessels, IAATO tourist vessels contributed 3,200 ship-days,[5] CCAMLR fishing vessels contributed 1,400 ship-days[6], and COMNAP research vessels contributed 1,100 ship-days[7] in the 2016/2017 season. In terms of the total number of people carried into the Antarctic during the 2016/2017 season, cruise ships (73,400 people incl. staff and crew) surpassed research vessels (3,300 people incl. crew) and fishing vessels (2,100 people). In terms of person-days (i.e., the cumulative sum of the number of persons multiplied by the time each spends), research (797,000 person-days at fixed stations plus approximately 100,000 person-days on COMNAP ships) outweighed tourism (730,000 person-days) and fisheries (63,000 person-days).

Since 2015, CCAMLR has required an automated vessel monitoring system (VMS) for all fishing vessels (Commission for the Conservation of Antarctic Marine Living Resources, 2015). While detailed positions of the CCAMLR fishing fleet have been collected since 2015, they are treated as commercially confidential information by the CCAMLR secretariat; so positions for each nation's vessels are only disclosed to the appropriate authority for that contracting nation. Thus, only an aggregate list of vessels licensed by CCAMLR for fishing in the Antarctic is generally available, rather than their precise locations and tracks (Commission for the Conservation of Antarctic Marine Living Resources, 2018). Automatic Identification System (AIS) data are also available for some regions and times. Vessels from nations outside of the Antarctic Treaty, CCAMLR, and IAATO are missing from the corresponding databases; however, it is unlikely that there are a substantial number of such ships. Most private yachts do not report either. An extrapolated number of 95 non-IAATO yachts compared to 18 IAATO-yachts[8] entered Antarctic waters in the 2017/2018 austral summer.[9]

Antarctic tourism has increased since the 1950s (Enzenbacher, 1992). Cruise ships are present from October through March, peaking in January. While the number of operators, number of ships, number of voyages and number of passengers increased between 1992/1993 and 2018/2019, the number of operators and ships has leveled off; yet the number of voyages and passengers keeps rising (Bender et al., 2016; International Association of Antarctica Tour Operators [IAATO], 2018; **Figure 2**). Research vessels are present all year-round, peaking in January and February. The number of research vessels south of 60°S has doubled from about 12 in 2011/2012 to 25 in 2016/2017.[10] The number of licensed fishing vessels (46 in 2017/2018), the number of licensing periods (52 in 2017/2018), and the number of licensed areas (119 in 2017/2018) have remained fairly constant

[2]COMNAP Antarctic Facilities Master List v 2.0.0, dated 08.12.2017; https://github.com/PolarGeospatialCenter/comnap-antarctic-facilities/raw/73f28e19f7e93f9e9e8b2c4dfb620b510e5eb256/dist/COMNAP_Antarctic_Facilities_Master.xls

[3]ATCM XLI, IP 71: IAATO Overview of Antarctic Tourism: 2017–2018 Season and Preliminary Estimates for 2018–2019 Season. Data from Appendix 1; https://iaato.org/documents/10157/2398215/IAATO+overview/bc34db24-e1dc-4eab-997a-4401836b7033

[4]CCAMLR List of authorized vessel for season 2017/2018; https://www.ccamlr.org/en/compliance/list-authorised-vessels

[5]Based on IAATO 2017 statistics: 2016–2017 Summary of Seaborne, Airborne and Land-Based Antarctic Tourism.

[6]Assuming 30 days/ship; CCAMLR List of authorized vessel for season 2016/2017: https://www.ccamlr.org/en/compliance/list-authorised-vessels

[7]Data provided by COMNAP based on COMNAP's Ship Position Reporting System (SPRS). Ships are requested to report once per day.

[8]ATCM XLI, IP 71: IAATO Overview of Antarctic Tourism: 2017–2018 Season and Preliminary Estimates for 2018–2019 Season. Data from Appendix 1.

[9]ATCM XXXIX, IP 36: Antarctic Tourism Study: Analysis and Enhancement of the Legal Framework, submitted by Germany. The German Environment Agency commissioned a study that showed, that of the >200 known yachts that sailed in the Antarctic Treaty area between 1997 and 2013, only 16% were IAATO-members at the time of their Antarctic Voyage.

[10]Data provided by COMNAP based on COMNAP's Ship Position Reporting System (SPRS).

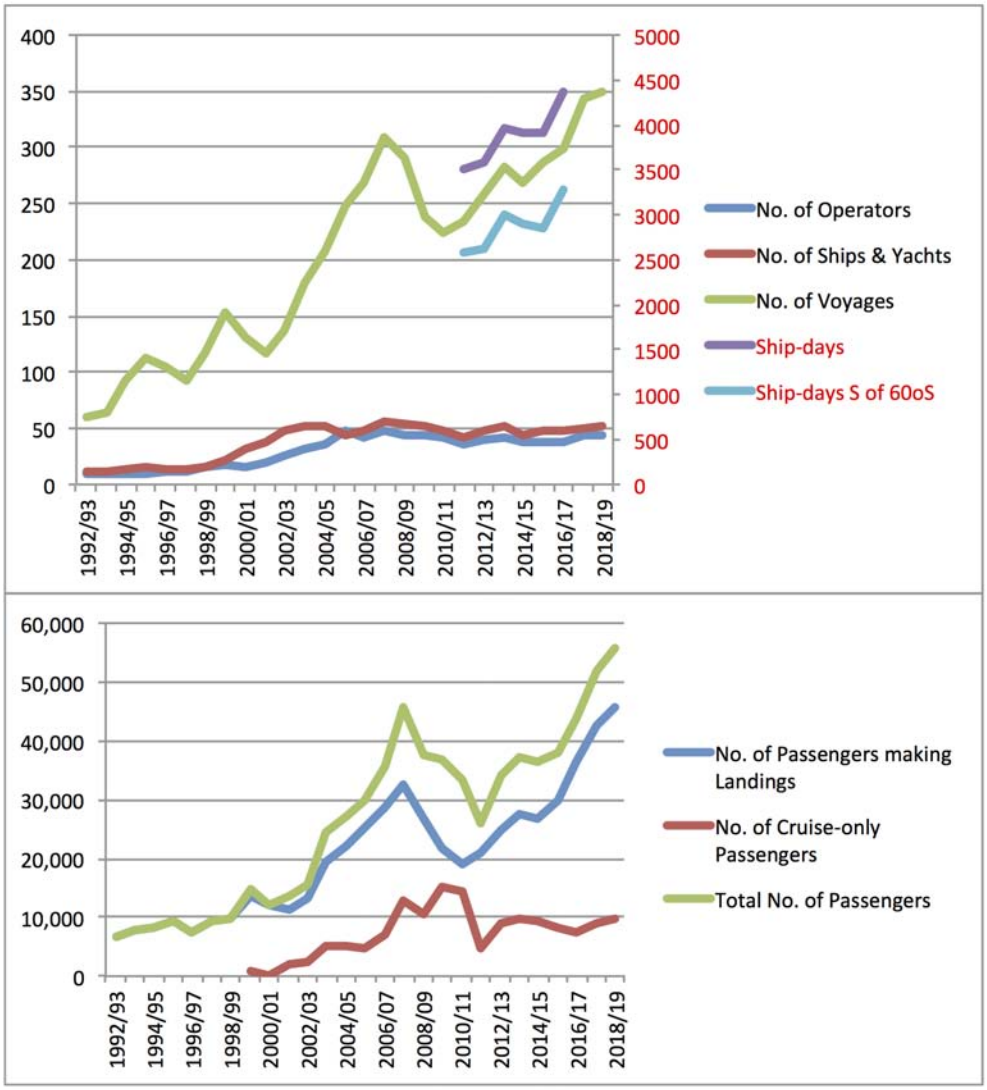

FIGURE 2 | Trends in Antarctic ship-based tourism. For more information on these data, see International Association of Antarctica Tour Operators [IAATO] (2018).

from 2011/2012 to 2017/2018[11]. Ships are not evenly distributed. Rather, the Antarctic Peninsula and the Ross Sea experience the most ship traffic of all types.

Ship noise is continuous and consists of a broadband (10 Hz–20 kHz) cavitation spectrum overlain with distinct propeller and engine tones and harmonics (5–200 Hz) (e.g., Arveson and Vendittis, 2000; Kipple, 2002; Wales and Heitmeyer, 2002). In addition, icebreakers produce sounds related to pushing and crushing ice (Erbe and Farmer, 2000; Roth et al., 2013). Broadband radiated noise levels of large ships including icebreakers can be as high as 200 dB re 1 μPa m (Allen et al., 2012; Roth et al., 2013). Ships typically run echosounders for depth-ranging, and the ATCM has produced Resolution H (2014) "*Strengthening Cooperation in Hydrographic Surveying and Charting of Antarctic Waters*," by which all ships of national Antarctic programs (and all other ships) are encouraged to collect hydrographic and bathymetric data using powerful echosounders while in the Antarctic Treaty area. Such echosounders repeatedly (every few seconds) emit pings at multiple frequencies (typically above 10 kHz) with source levels up to 240 dB re 1 μPa peak-to-peak (pk-pk) and 200 dB re 1 $\mu Pa^2 m^2 s$ (Crocker and Fratantonio, 2016; Crocker et al., 2018).

Research in the Antarctic is carried out from ships, land-based platforms, and air. Research station and wharf construction may involve geotechnical work, rock breaking, and pile driving—all of which generate noise underwater (e.g., Soloway and Dahl, 2014; Erbe and McPherson, 2017). Driving piles into the seafloor with a vibrator creates underwater noise at 10–1000 Hz with distinct tonal structure and levels up to 170 dB re 1 μPa rms at close range (Dahl et al., 2015). Percussive pile driving creates impulsive underwater noise of up to 227 dB re 1 μPa pk-pk and 201 dB re 1 $\mu Pa^2 s$ at close range (Hastings and Popper, 2005;

[11] data from https://www.ccamlr.org/en/compliance/list-authorised-vessels

Illinworth and Rodkin Inc, 2007). Aircraft produce noise in air, however, noise transmits into water directly below (e.g., Erbe et al., 2017b, 2018b). Additionally, some countries, such as Germany, Italy, Spain, Poland, China, South Korea, and Russia, have been undertaking marine seismic surveys for research in the Antarctic (Breitzke, 2014). Germany alone acquired 59,621 km of multichannel seismic survey lines between 1976 and 2011 (Boebel et al., 2009; Breitzke, 2014). Seismic airgun arrays emit broadband (5 Hz–20 kHz) pulses repeatedly (every 5–20 s) at levels up to 224 dB re 1 $\mu Pa^2 m^2 s$ (Ainslie et al., 2016; Li and Bayly, 2017).

Sound propagation in Antarctic waters differs from that at lower latitudes due to low surface temperatures and the possible presence of ice. In polar water, the sound speed increases with depth, which leads to upward refracting sound propagation paths and the establishment of a so-called surface duct. Sound trapped in the surface duct can travel over long ranges. Sound emitted near the surface will follow a refracted propagation path where it travels to some depth and then bends upward without interacting with the seafloor and thus without the associated reflection loss that occurs at the seafloor. Reflection occurs at the sea surface, and the associated loss depends on whether the surface is open or ice-covered, and on its roughness. First-year ice is typically smooth underwater and hence very little scattering loss occurs here, resulting in very effective sound propagation under such ice. Furthermore, given the deep bathymetry around Antarctica, there is no low-frequency mode stripping, meaning that low-frequency noise from ships or seismic airguns can travel over very long ranges (hundreds to thousands of kilometers; Siebert et al., 2014; Gavrilov, 2018). With such long-range propagation, the spectral and temporal features of sound change, because energy at different frequencies travels at slightly different speeds and along slightly different paths (termed "dispersion"; Horton Sr, 1974; Dushaw et al., 1993). Brief (100 ms), broadband (<20 kHz), high-amplitude pulses as emitted by seismic airguns turn into longer-duration (several seconds), narrower-band (<200 Hz), lower-amplitude, frequency-modulated sounds at distances of tens of kilometers (Yang, 1984; Siebert et al., 2014; Hastie et al., 2019). Such spectro-temporal changes in noise characteristics yield different types of noise impacts as a function of range.

NOISE IMPACTS

The effects of noise on marine mammals range from individual, short-term responses to population-level, long-term impacts (see Erbe et al., 2018a). In terms of severity, they also range from cases which might not result in any consequences, to those that prompt behavioral changes, mask communication, induce hearing loss, increase stress, or lead to death (e.g., in the case of tactical mid-frequency sonar affecting beaked whales; Fernández et al., 2013). Mortality can also occur in close proximity to underwater explosions (Ketten, 1995; Danil and St. Leger, 2011). These types of noise impacts have been reported not only for marine mammals but also for fishes and other taxa (e.g., Day et al., 2017, 2019; McCauley et al., 2017; Hawkins and Popper, 2018), which are preyed upon by marine mammals. Noise impacts on these taxa can thus indirectly affect marine mammals if noise leads to a physical reduction in prey availability or to a change in prey behavior that affects its availability to predators. Examples for each type of effect of noise on Antarctic marine mammals or their closely related northern species are summarized in **Table 3**.

While the above impacts are experienced by individual animals, they can lead to population-level impacts. Animals might be displaced from preferred habitats into areas with higher predation risk, lower prey abundance, or poorer prey quality. They might suffer reduced energy intake while expending more energy. Malnutrition, stress and hearing loss might compromise health and lead to shortened life span. If enough individuals in a population are affected, then population dynamics may change. The Population Consequences of Acoustic Disturbance (PCAD) and Population Consequences of Disturbance (PCoD) models provide a conceptual framework that link short-term individual impacts to population consequences (National Research Council, 2005; Harwood et al., 2014; National Academies of Sciences, Engineeering and Medicine, 2017). A well-studied example species is the elephant seal (both northern and southern), where disrupted foraging behavior due to noise leads to predictions of reduced foraging success in mothers; then a reduced maternal mass leads to reduced pup mass at weaning, which is predicted to negatively impact pup survival and lead to changes in population dynamics (New et al., 2014; Costa et al., 2016). Population consequences of disturbance can potentially be significant, in particular when noise-making activities occur in high-priority areas for a population.

Research on hearing abilities and the effects of noise on Antarctic marine mammals has been sparse and little data are available to assess the potential impacts of noise on their hearing. Out of the 23 marine mammal species that occur south of the Antarctic Convergence, a behavioral audiogram is only available for the killer whale (Branstetter et al., 2017), with some hearing information from auditory evoked potential measurements on a stranded long-finned pilot whale (Pacini et al., 2010). Behavioral audiograms remain the standard for hearing tests and provide a whole-animal response (including decision making by the animal); in contrast, auditory evoked potential audiograms reflect the averaged response of the auditory brainstem to acoustic stimuli only. The audiogram of the northern elephant seal could possibly be used as a surrogate for the southern elephant seal (Kastak and Schusterman, 1999). Although several anatomical predictions of the frequency range of hearing have been produced for mysticetes (Houser et al., 2001; Parks et al., 2007; Tubelli et al., 2012; Cranford and Krysl, 2015), no empirically measured audiogram exists for any mysticete species globally. Data on noise-induced hearing loss or impacts of stress in Antarctic marine mammals do not exist, although a fair amount of work has been performed on the endocrine response to stress in the southern elephant seal's close relative, the northern elephant seal (e.g., Ensminger et al., 2014; Jelincic et al., 2017). While the sounds made by Antarctic marine mammals have been documented (e.g., Erbe et al., 2017a), there is no information (such as critical ratios) to assess masking of those sounds by noise, except for four studies indicating anti-masking processes in humpback, killer, and long-finned pilot whales elsewhere (see Erbe et al., 2016). There have been no dedicated studies on

TABLE 3 | Examples of reported effects of noise on Antarctic marine mammals (* or closely related species).

Noise source	Species	Effect of exposure	References
Shipping	North Atlantic whale*	Increased stress levels	Rolland et al., 2012
Shipping	Humpback whales	Effects on foraging, including slower descent rates and fewer side-roll feeding events per dive	Blair et al., 2016
Shipping	Killer whales	Noise source avoidance, predicted masking of communication sounds	Erbe, 2002
Seismic airguns	Sperm whales	Disruption of foraging, decreased adult and infant mass, decreased infant survival, predicted population level effect	Farmer et al., 2018
Seismic airguns	Fin whales	Song modifications, sound source avoidance	Castellote et al., 2012
Pile driving	Harbor seals*	Noise source avoidance, predicted effect on hearing threshold	Hastie et al., 2015; Russell et al., 2016
Shipping, naval sonar	Blue whales	Changes in call rate	Melcon et al., 2012
Naval sonar	Blue whales	Disruption of foraging	Goldbogen et al., 2013; Southall et al., 2019a
Naval sonar	Northern bottlenose whales*, Baird's beaked whales*	Avoidance, interruption of foraging	Stimpert et al., 2014; Miller et al., 2015
Naval sonar	Minke whales	Avoidance responses over large distances.	Kvadsheim et al., 2017
Naval sonar	Sperm whales	Switch to a non-foraging, non-resting state. Reduction of time spent in foraging states and reduced probability of prey capture attempts.	Isojunno et al., 2016
Scientific echosounders	Beaked whales (*Ziphius* and *Mesoplodon* species)	Interruption of foraging or vessel avoidance. Reduced acoustic activity when echo-sounders were actively transmitting. When echo-sounders were in passive mode, the detection rate showed a 20-fold increase.	Cholewiak et al., 2017
Acoustic deterrent devices	Killer whales	Habitat exclusion	Morton and Symonds, 2002

the behavioral responses of marine mammals to noise in the Antarctic, though some studies have been undertaken on the same species in other regions (**Table 2**). It is uncertain how results from other regions (where animals potentially undergo different life functions; e.g., feeding in the Southern Ocean versus breeding at low latitudes) and other populations or species relate to Antarctic marine mammals, especially in light of the modulating influence that behavioral and exposure context can have on reactions to noise (Harris et al., 2018).

NOISE MANAGEMENT

Annex I to the Protocol and the Guidelines for Environmental Impact Assessment in Antarctica (Resolution 1, 2016) outline the Environmental Impact Assessment (EIA) process for activities in the Antarctic. The proponent prepares the EIA document, which describes the project and the environment, identifies potential interactions and consequences, determines the significance of predicted impacts, considers alternatives, and designs mitigation and monitoring programs (**Figure 3**). Monitoring is (a) required for activities expected to have more than a minor or transitory impact, (b) suggested for those of minor or transitory impacts, and (c) not required for those of less than minor or transitory impacts. The EIA is reviewed and assessed by national authorities. Projects with environmental impacts that are less than minor or transitory are allowed to proceed—potentially with conditions imposed. Projects with environmental impacts that are minor or transitory require that the proponent prepare an Initial Environmental Evaluation (IEE). Projects with environmental impacts that are more than minor or transitory require that the proponent prepare a Comprehensive Environmental Evaluation (CEE). The CEE is reviewed by all the Antarctic Treaty Parties, by the CEP, and at the ATCM. The final CEE addresses comments from this review process. The national authorities eventually make a decision to either reject the project or allow the project to proceed—likely with conditions imposed.

Since the EIA process is conducted at a national level, several countries and jurisdictions have developed guidelines and regulations for the management of underwater noise (Erbe, 2013; Lucke et al., 2016). Typically these involve exposure modeling and impact prediction, mitigation, and sometimes *in situ* monitoring related to intense sources such as seismic airguns or pile driving.

Impact prediction requires knowledge of sound characteristics and levels at which different types of impact occur. While the regulations in different countries often aim to protect the same or similar species from the same types of impact, the metrics, thresholds, and management procedures that are applied differ. One reason for these differences is that the impact of sound on marine mammals is an active field of science, and new knowledge is being delivered gradually. There is a general acceptance that hearing damage can result from either an instantaneous exposure to very high sound pressure levels or from the accumulated exposure to acoustic energy over an extended period of time. This requires management with dual criteria and thresholds to address the different types of sound sources in the ocean, one sound pressure based, the other energy

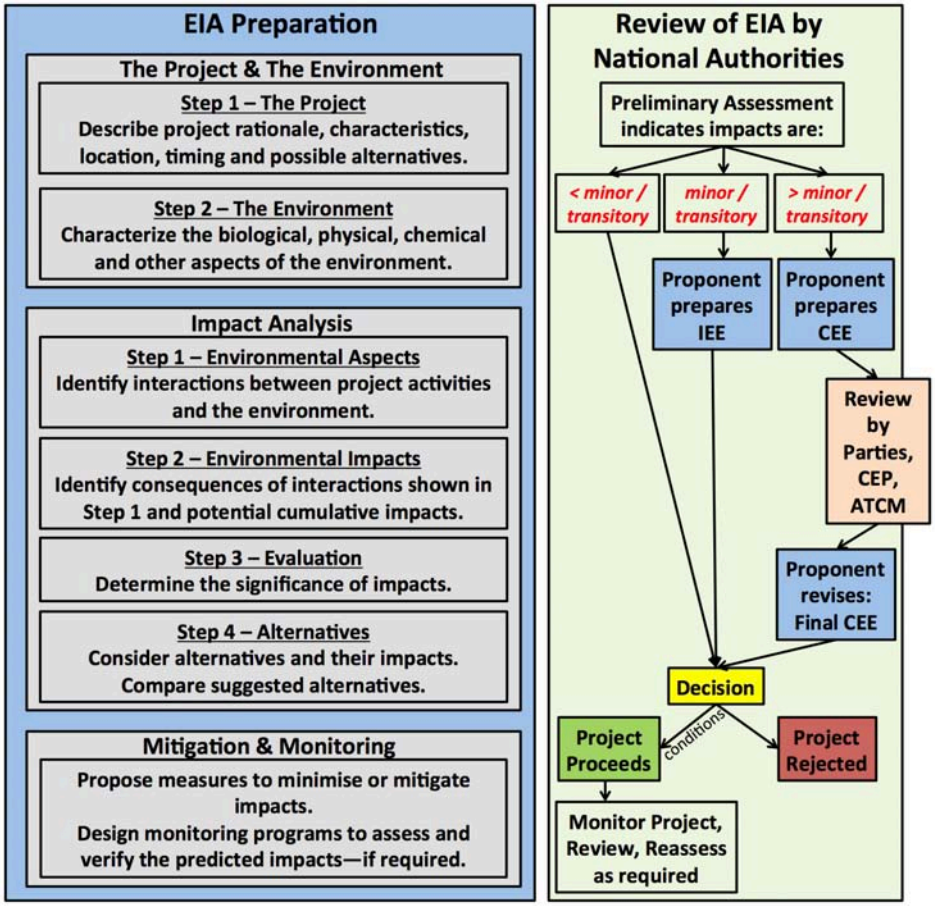

FIGURE 3 | A schematic of the EIA process outlined in Annex I of the Protocol. The Proponent prepares the EIA, which includes sections on the project, the environment, potential impacts, and mitigation and monitoring measures. The EIA is reviewed by national authorities. If impacts are expected to be greater than minor or transitory, the Parties of the Antarctic Treaty, the CEP and ATCM get to review the proposal as well. National authorities make the final decision whether a project goes ahead (potentially with imposed conditions) or not.

based (Southall et al., 2019b). Regulators aim to ensure that any exceedance of these thresholds does not have significant impacts to the noise-exposed populations. Energy-based criteria present particular practical challenges in that the animals' behavior, and in particular how they move in three dimensions with respect to a sound source, affects the received acoustic exposure. Often this is the least known and most uncertain component of a risk assessment.

For example, for high-frequency cetaceans such as porpoises and exposure from impulsive noise such as impact pile driving, the United States. uses a dual criterion (i.e., applies the one resulting in the largest effect distance) of 196 dB re 1 μPa zero-to-peak pressure and 140 dB re 1 $\mu Pa^2 s$ cumulative sound exposure (weighted and integrated over 24 h) as the onset of temporary threshold shift (TTS), and a dual criterion of 202 dB re 1 μPa zero-to-peak pressure and 155 dB re 1 $\mu Pa^2 s$ cumulative sound exposure (weighted and integrated over 24 h) as the onset of permanent threshold shift (PTS) (National Marine Fisheries Service, 2018). Germany applies a dual criterion of an unweighted single-impulse (i.e., not cumulative) sound exposure level of 160 dB re 1 $\mu Pa^2 s$ and 190 dB re 1 μPa peak-to-peak pressure at a range of 750 m from the pile in order to avoid TTS (Bundesministerium für Umwelt Naturschutz und Reaktorsicherheit , 2014). While the United States criteria are applied at the receiving animal, which can be anywhere around the source, the German criteria are referenced to the exact distance of 750 m from the source.

The criteria employed by the two countries are difficult to compare, and it is not possible to generalize which country uses stricter regulations, because the criteria apply at different ranges, and the site-specific sound propagation environment will affect at what range certain levels are exceeded. Furthermore, Germany uses unweighted sound exposure, while the United States weights exposures according to categorization to a defined functional hearing group. Germany uses single exposures, while the United States integrates over 24 h. Different regulators also vary in the degree of precaution they are minded to apply, given the high levels of uncertainty in so many aspects of this topic. Germany, for example, considers TTS as the beginning of injury, whereas the United States only considers the onset of

PTS auditory injury under the Marine Mammal Protection Act (MMPA) (National Marine Fisheries Service, 2016).

There are a number of mitigation procedures that can be applied to reduce noise exposures from various sources (Weir and Dolman, 2007; Dolman et al., 2009; Merck et al., 2014; Verfuss et al., 2016). Some methods can be applied at or close to the source. This might involve using a quieter source or one that produces a different type of signal that might reduce specific impacts (e.g., marine vibroseis versus seismic airguns; Duncan et al., 2017). Sound barriers (e.g., bubble curtains) may be installed near a fixed source to reduce sound propagation into the wider environment. Operations might be scheduled to occur at times when marine mammal abundance is expected to be lower or to avoid times of particular vulnerability, such as calving seasons (Van Opzeeland and Boebel, 2018). During operations, safety zones might be searched for marine mammals (e.g., using visual, infrared, and/or passive acoustic methods). A delay in initiating activities or a shut-down might result if animals are detected within mitigation zones. The effectiveness of this as a mitigation approach depends on the ability to detect animals and in many cases results in little reduction of risk (Leaper et al., 2015).

Mitigation effectiveness and practicality depend on the activities to be mitigated, the environment, the target species, and the approach taken. Multiple mitigation approaches are sometimes applied. Generally, mitigation can reduce the risks to marine mammals, but not eliminate them. Impacts such as behavioral disturbance and masking are particularly difficult to minimize except by reducing sound at the source. In the presence of knowledge gaps and uncertainty on noise impacts, regulators are expected to take a conservative approach, following the precautionary principle. What level of mitigation is reasonably practicable is debatable amongst proponents and regulators.

INTERNATIONAL RESPONSIBILITY

Concern about the effects of anthropogenic underwater noise on marine life is widespread and increasing (as evidenced by, e.g., publication patterns; Williams et al., 2015; Shannon et al., 2016). In some countries, underwater noise is considered a form of water pollution, alongside chemical pollution (e.g., in the European Union; van der Graaf et al., 2012). Sound underwater travels much faster and farther than it does in air. Depending on the sound propagation conditions, sound can travel hundreds of kilometers and traverse entire ocean basins. Noise therefore crosses legal boundaries and the noise received in one jurisdiction might originate in a region that is under different jurisdiction, making noise regulation and ultimately conservation management an international responsibility.

The Convention on Environmental Impact Assessment in a Transboundary Context (Espoo Convention, United Nations Economic Commission for Europe [UNECE], 1991) requires that member states undertake environmental impact assessments of planned activities, and then inform and consult other member states if the impacts are expected to occur in other states as well. There are several examples where countries that border the same body of water have reached international agreements to manage noise and other stressors. This is the case for some European seas (Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic Area - ACCOBAMS; Pavan, 2006; Authier et al., 2017), Baltic (Helsinki Commission - HELCOM; Backer et al., 2010), European Union (Marine Strategy Framework Directive - MSFD; van der Graaf et al., 2012), and other regions.

Other international organizations that recognize underwater noise as a threat to marine mammals (including in the Antarctic) are the IWC, the Convention on Biological Diversity (CBD), the Convention on Migratory Species (CMS), the International Maritime Organization (IMO), and the United Nations General Assembly. Anthropogenic underwater noise was the focus topic for the UN Informal Consultative Process on Oceans and the Law of the Sea in June 2018[12].

The IWC supported a workshop on global soundscape mapping in 2014 and a workshop on acoustic masking in 2016, and continues to discuss underwater noise at annual meetings of its Scientific Committee. In 2018, the IWC passed a Resolution on Anthropogenic Underwater Noise[13] by consensus, recognizing that chronic anthropogenic underwater noise is affecting the marine acoustic environment in many regions, and that there is emerging evidence that compromised acoustic habitat may adversely affect some cetacean populations.

In 2014, the Conference of the Parties to the CBD (Decision XII/23) encouraged parties to take appropriate measures to avoid, minimize, and mitigate the potential significant adverse impacts of anthropogenic underwater sound on marine and coastal biodiversity. It also encouraged governments to require environmental impact assessments (EIAs) for sound-generating offshore activities, and to combine mapping of the acoustic footprints of activities with habitat mapping to identify areas of risk.

The Conference of the Parties to the CMS adopted a Resolution on "Adverse impacts of anthropogenic noise on cetaceans and other migratory species" in 2017, which urges parties, whose flagged vessels travel beyond national jurisdictional limits, to undertake EIAs and manage the impact of anthropogenic noise on CMS-listed marine species and their prey. Guidelines for EIAs of underwater noise were also published in 2017 under this Convention.

The IMO stated that uncertainty as to the effects of noise should not preclude efforts toward developing quieting technologies for commercial ships (International Maritime Organization, 2009). The IMO developed guidelines on underwater noise reduction (MEPC.1/Circ.833) in 2014 acknowledging that noise from commercial ships may have both short- and long-term negative consequences on marine life. The International Convention for the Prevention of Pollution from Ships (MARPOL 1973/1978) and the International Code for Ships Operating in Polar Waters (Polar Code) were both developed by the IMO and have implications for ship noise in the

[12]http://www.un.org/depts/los/consultative_process/ICP-19_information_for_participants.pdf

[13]https://iwc.int/document_3685.download

Antarctic. MARPOL bans heavy fuel oil, both as fuel and cargo, from south of 60°S, thus limiting older vessels, which may emit more noise due to older, less efficient propulsion and design. The Polar Code requires that vessel masters consider marine mammal aggregation and migration areas when planning routes.

The International Association of Antarctic Tour Operators (IAATO) promotes environmentally responsible travel to Antarctica and could be an organization to also address underwater noise. However, some commercial operators are not members of the IAATO.

The Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) has the objective of conserving Antarctic marine life and focuses on commercial fisheries species (e.g., krill and toothfish). Impacts of anthropogenic noise on marine mammal prey species may thus need to be considered by CCAMLR.

RESEARCH AND MANAGEMENT NEEDS

In November 2018, we held an international workshop on the possible effects of noise on Antarctic marine mammals in Berlin, Germany. Twenty-nine workshop participants (15 biological scientists, 5 regulators, and 9 Antarctic seismic and ship noise producers) were asked what they saw as key research and management needs for this topic in the Antarctic. We then asked all participants to rate the importance of each topic on a scale from 1 (low importance) to 5 (high importance). The complete list of topics and their scores can be found in the **Supplementary Material**. The research needs that received the highest rankings by workshop participants were (1) better data on abundance and distribution of Antarctic marine mammals with particular urgency to identify areas of high abundance ("hotspots") and low abundance ("coldspots"), (2) hearing data for Antarctic marine mammals, in particular a mysticete audiogram, and (3) an assessment of the effectiveness of various noise mitigation options.

Management needs with the three highest scores overall were a refinement of noise exposure criteria, clear application guidance for environmental impact assessments, and transparency in regulatory decisions. Transparency was the highest-ranking need for proponents (4.67/5), though ranked lower by regulators (3.25/5). Regulators ranked the refinement of noise exposure criteria highest (4.5/5) and proponents agreed this was important (4.17/5). Biological scientists prioritized the need for agencies with an Antarctic interest (e.g., SCAR and CCAMLR) to join forces on noise management (4.4/5), establishing a public database on marine mammal distribution (4.2/5) and the sharing of research and ancillary data amongst users (e.g., of seismic data and echosounders data) (4.1/5). There were a few very specific research needs that directly relate to management and regulation requirements, such as (1) the allowance of hearing impairment recovery in cumulative exposure calculations, (2) justification and modification of the 24 h integration period for cumulative exposure calculations, and (3) choosing an appropriate metric and weighting to predict behavioral disturbance. These three needs and studies on responses to natural ambient noise were ranked of very high importance by potential noise producers, yet low by regulators.

Assessing and managing underwater noise and its potential impacts on marine mammals in the Antarctic is complex and difficult. Multiple countries operate in Antarctica, and many stakeholders and sectors have an interest in the Antarctic (tourism, fisheries, shipping, research, and conservation). In addition to the complicated management framework, there are significant scientific knowledge gaps. Antarctic species are understudied. Some undergo critical life functions while in the Antarctic (such as feeding by mysticetes before migration to breeding grounds at lower latitudes) and it can only be speculated how impacts potentially incurred in the Antarctic will affect the fitness of these animals when in other areas. Applying data on noise impacts from other areas or species should be avoided until similarities are proven. The unique aspects of the Southern Ocean (i.e., the species and their life functions in this environment, the ambient noise, and the sound propagation characteristics) constitute a critical need for research on the effects of anthropogenic noise.

Many of the research needs we present here have been internationally recognized and some are at least partially addressed by other entities. For example, the urgent need to identify marine mammal hotspots has also recently (October 2018) been recognized by the IUCN Marine Mammal Protected Areas Task Force, proposing 15 candidate Important Marine Mammal Areas (IMMAs) for the Southern Ocean and Sub-Antarctic Islands[14]. The Subcommittee on Ocean Science and Technology (SOST), which is a partnership between the United States Office of Naval Research, Chief of Naval Operations N45, the Bureau of Ocean Energy Management, the National Oceanic and Atmospheric Administration, and the Marine Mammal Commission, called for proposals for mysticete audiograms in mid-2018. The Joint Industry Program of Oil and Gas Producers currently include a study on masking in marine mammals. Some research needs, such as the effectiveness of certain mitigation methods, behavioral responses, and prey responses (e.g., availability of krill), could potentially be developed as proposals for future voyages to Antarctica.

Data sharing is one aspect of international collaboration and efforts are underway to make seismic and hydrographic data publicly available. There appear to be significant delays of several years in this process, but the complexity, effort, and costs of data preparation, warehousing, and support are considerable.

It is encouraging that the international scientific community is coming together to review current knowledge as reported here and that efforts are underway to fill some of the research gaps that we recognized. The Antarctic is unusual in its sound transmission characteristics, its species community, and the way in which humans use its waters. Not all findings from other areas of the world are necessarily applicable to this environment, and

[14]https://www.iucn.org/news/marine-and-polar/201810/fourth-important-marine-mammal-areas-workshop-adds-15-candidate-immas-southern-ocean-and-sub-antarctic-islands

studies addressing the effects of noise specifically in the Antarctic are often lacking. Furthermore, different nations operate around the continent without necessarily coordinating their efforts. The effects of multiple stressors and multiple sound sources have been recognized as a high research priority in the marine science community in general (Rudd, 2014). In the Arctic, an integrated approach in the management of noise sources has been called for Moore et al. (2012). A similar approach would be prudent in the Antarctic. In 2048, the Protocol on Environmental Protection of the Antarctic Treaty may be reviewed if one of the Parties requests it. Additional anthropogenic activities such as mining may be considered. Such activities would lead to an increase of noise in the Antarctic. We hope that our review here will contribute to identifying and steering where research and management actions are most needed to protect the Antarctic environment from anthropogenic noise as much as possible.

AUTHOR CONTRIBUTIONS

All of the authors participated in the workshop on Antarctic noise and effects on marine mammals, collaborated on the data gap table, and wrote sections of the report and/or manuscript. At the workshop, the following authors reviewed, summarized and presented the main topics. HH and AS-S: regulation. MM: human activities in Antarctica. MD: Antarctic marine mammal ecology. DH: hearing and hearing impairment. CE: acoustic masking. SK: behavior. JO: population consequences. RM: marine mammal prey impacts. JG: mitigation and monitoring. MS: ambient noise and propagation. MM, AM, and CE drew the figures. CE and VJ organized the workshop and produced the first draft of the overall manuscript.

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SUPPLEMENTARY MATERIAL

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A Meta-Analysis to Understand the Variability in Reported Source Levels of Noise Radiated by Ships From Opportunistic Studies

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Background: Commercial shipping is identified as a major source of anthropogenic underwater noise in several ecologically sensitive areas. Any development project likely to increase marine traffic can thus be required to assess environmental impacts of underwater noise. Therefore, project holders are increasingly engaging in underwater noise modeling relying on ships' underwater noise source levels published in the literature. However, a lack of apparent consensus emerges from the scientific literature as discrepancies up to 30 dB are reported for ships' broadband source levels belonging to the same vessel class and operating under similar conditions. We present a statistical meta-analysis of individual ships' broadband source levels available in the literature so far to identify which factors likely explain these discrepancies.

Methods: We collated ships' source levels from the published literature to construct our dataset. A Generalized Linear Mixed Model was applied to the dataset to statistically assess the contribution of *intrinsic* (i.e., related to ships' static and dynamic attributes) and *extrinsic* factors (i.e., related to both the protocol for hydroacoustic data acquisition and the noise data reduction procedure) to the reported broadband source levels.

Results: Amongst *intrinsic* factors, ships' speed-over-ground $\left(15.39\ \text{dB} \times \log_{10}\left[\frac{v}{1\ \text{knot}}\right], p\text{-value} < 0.001\right)$, ships' width $\left(12.03\ \text{dB} \times \log_{10}\left[\frac{b}{1\ \text{m}}\right]; p\text{-value} < 0.001\right)$, and ships' class (−6.07 to 2.08 dB; p-value $\in$ [< 0.001 to 0.036]) have shown the strongest correlations with broadband source levels. The hydrophone-to-source closest point of approach $\left(-4.83\ \text{dB} \times \left[\frac{\text{CPA}}{1\ \text{nmi}}\right]; p\text{-value} < 0.001\right)$ and the correction for surface-image reflections (21.73 dB; p-value $= 0.002$) contribute the most to explain the reported ships' broadband source levels' variability amongst *extrinsic* factors.

Conclusions: Our meta-analysis confirms a consensus that speed regulation can effectively reduce instantaneous ships' source levels. Neglecting Lloyd's mirror effects through the abuse of non-corrected spreading laws for propagation loss directly leads to a generalized under-estimation of the ships' source levels retrieved from the literature. This could eventually be addressed by a wider adoption of standardized methods of hydrophone-based sound recordings and of data processing to homogenize results and facilitate their interpretation to conduct environmental impact assessment.

Keywords: review of literature, ships' source levels, acoustic techniques, hydrophone-based observations, statistical methods

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1. INTRODUCTION

The exposure of marine life (e.g., fishes, invertebrates, mammals) to anthropogenic noise remains a worldwide environmental issue for marine ecosystems (Williams et al., 2015). The magnitude of the underwater radiated noise attributed to the merchant fleet has shown a monotonic increase over the past few decades (Nolet, 2017) forcing the international authorities to suggest recommendations and new laws of mitigation in order to maintain control of this trend (IMO, 2014). Recently, commercial shipping activities, regarded as the main contributor to the anthropogenic underwater noise budget, have shown constant year-to-year increases and are a global-wide phenomenon, hence adding to the concerns (Clark et al., 2009).

The impact of the shipping noise on marine mammals is of critical importance considering the vital role of acoustics for numerous species. Anthropogenic noise alters the social behavior of marine mammals (Gomez et al., 2016), masks their communications (Erbe et al., 2016), and impedes their ability to appropriately forage (Tyack et al., 2011). Instantaneous high-amplitude events or long-term exposition to continuous underwater noise can lead to temporary or permanent injuries to their auditory system (NOAA, 2015).

In Canada, for many species listed as endangered according to Canada's Species at Risk Act (2002), underwater noise of anthropogenic origin is already being regarded as a threat to their recovery and is identified as such in their recovery plans. In the St. Lawrence River (Québec, Canada), anthropogenic underwater noise is thus identified as a threat to the recovery of both the St. Lawrence Estuary beluga population and the Northwest Atlantic blue whale.

In this context and considering the expected increase of the maritime traffic (Kaplan and Solomon, 2016), especially in the St. Lawrence River (Gouvernement du Québec, 2015), multi-stakeholder processes are underway to identify options to mitigate merchant ships' underwater radiated noise (e.g., Audoly et al., 2014). However, a prerequisite to an underwater noise reduction campaign concerns the ability to properly measure sound pressure through hydrophone-based observations and to accurately estimate the ships' source levels, a challenge undertaken worldwide by several research groups (Audoly et al., 2014; MacGillivray et al., 2019).

In order to quantify the underwater radiated noise from merchant ships through opportunistic hydrophone-based observations, numerous steps must be carefully carried out including the choice of hydrophones, location, and an accurate hydrophone calibration (Robinson et al., 2014). This also involves a detailed understanding of the complex physical processes and their mathematical representation that describe the acoustic propagation in anisotropic underwater environments (Erbe et al., 2016).

Merchant ships' underwater noise have several origins, the principal being machinery, propellers, and cavitation (Audoly et al., 2017). It is well-known that variations in merchant ships' source levels exist inside a given vessel class and from one class to another (see e.g., Figure 2 of Veirs et al., 2016). Proper characteristics to each ship (e.g., architecture, type of engines, maintenance of the propellers and hull) and conditions of operation (e.g., speed, load) can have an impact on the source levels. For this work, these factors will be referred to as *intrinsic* in a sense that they originate from the ships' own static and dynamic characteristics.

Alternatively, large, and often intra-class, discrepancies on source levels are reported in the literature, hence suggesting a certain uncertainty on the measurements attributed to factors *extrinsic* to the ships themselves. These *extrinsic* factors refer to the data campaign protocol and the mathematical calculations required to convert received levels of sound at the hydrophones into source levels at the position of the ship's position. To list a few, we can think about the experimental design for data acquisition (e.g., hydrophones' locations) or how certain physical processes (e.g., surface-image reflections) are handled during the data processing phases. Although underwater radiated noise propagation is directly related to the medium's chemical and geophysical properties, measurements of source levels found in the literature are usually reported without corresponding error bars or uncertainties, hence making study-to-study comparison quite complicated.

In this context where regulators, natural resources and conservation managers are required to identify new ways to attenuate the underwater radiated noise attributed to the merchant fleet in a growing number of ecologically sensitive areas, the important variability in the results reported by acoustic experts in the literature needs further investigation. This motivated a meta-analysis to shed some light on the apparent discrepancies reported for source levels of merchant ships and to identify the contribution of quantifiable *intrinsic* and *extrinsic* factors responsible for those.

2. DEFINITIONS AND SCOPE

In this work, merchant ships include bulk carriers, unclassified cargo ships[1], container ships, passenger ships, tankers, and vehicles carriers. We followed the terminology regarding ships' source levels described in ISO 17208-1 (2016), ISO 18405 (2017), and ISO 17208-2 (2019).

1. Radiated Noise Level (RNL): Level of the product of the distance from a ship reference point of a sound source and the far field root-mean-square sound pressure at that distance for a specified reference value.
2. Monopole Source Level (MSL): Mean-square sound pressure level at a distance of 1 m from a hypothetical monopole source, placed in a (hypothetical) infinite uniform lossless medium.

[1] Simply referred as "cargo ships" by the different studies selected in section 4, these ships likely include a mixture of non-categorized bulk carriers, container ships, oil/chemical tankers, vehicles carriers and subcategories.

Source levels derived from underwater recordings that neglect surface-image reflections (i.e., Lloyd's mirror effects) are said to be RNL measurements. MSL values can be retrieved, in first approximation, using correction factors (listed in Appendix A of ISO 17208-2, 2019) applied to RNL measurements that were previously obtained using the standardized protocol described in ISO 17208-1 (2016). Higher precision can be obtained by using numerical algorithms (Collins, 1993; Porter and Liu, 1994) for the backpropagation of the received levels of noise instead of relying on the distance normalization of the standard spherical geometrical wave dilution. This latter method not only corrects for surface-image reflections but also compensates for the wave absorption of the seabottom sediments, bathymetric variations and channeling effects attributed to speed-of-sound gradients.

3. OBJECTIVES

The aims are to:

1.(a) Identify published studies that provide frequency-integrated (i.e., broadband) source levels for individual merchant ships;
(b) Characterize inter-study variability in source level measurements;
(c) Collate the data related to field campaigns and data processing.
2. Identify the contribution of quantifiable *intrinsic* and *extrinsic* factors that statistically explain the variability of the source level measurements. Emphasis will be accorded to *extrinsic* factors that can be objectively estimated from the information provided in each study. This excludes the precision of the hydrophone calibration as details of the pre-observations lab manipulations are rarely discussed in the selected studies.
3. Characterize the contribution of the ships' speed to the overall noise budget reported in the studies. Transiting speed is of particular interest as it appears to be a manageable factor to reduce the ships' radiated noise.

By achieving these objectives, we will provide key information and clarification about the interpretation of the ships' source levels reported in the literature. This will support ongoing management processes seeking to understand and mitigate the ships' radiated noise. This work will also be informative for the noise modeling endeavors carried out in the context of environmental impact assessment (e.g., Chion et al., 2017; Pennucci and Jiang, 2018).

4. METHOD

To identify studies that report opportunistic source level measurements of individual merchant ships and to quantitatively explore their variability, we first carried out a literature review using a keywords approach in databases of scholarly literature (Google Scholar, Scopus). The query used on these search engines is here listed as:

1. "Ship" AND
2.(a) "source levels" OR
(b) "sound signature" OR
(c) "acoustic signature" OR
(d) "noise signature" OR
(e) "radiated noise."

A close examination of the list of references of each returned hit was also carried out in order to identify articles and reports that have failed to be returned by the keywords combination mentioned above. Only studies displaying broadband source level measurements in units of dB re 1 μPa · m for individual merchant ships were selected. All source level measurements for single recordings were gathered in a unique datasheet in order to investigate agreements and discrepancies between studies and conduct subsequent analysis.

The fact that each selected study has its own protocol/methodology for data acquisition creates a non-independence of the datasheet's intra-study data i.e., a measurement taken from a specific study will likely be more similar to another measurement taken from the same study than a measurement taken arbitrarily from another study. This signifies that a standard generalized linear model (GLM) cannot be used in order to estimate how *intrinsic* and *extrinsic* factors contribute to explain the variability in reported ships' source levels. In our case, the data non-independence requires the use of a generalized linear mixed model (GLMM). The term "mixed" indicates that the model implies the use of at least one fixed effect (i.e., a variable for which we wish to quantify the effect on reported source levels) and at least one random effect (authors-specific in our case). Random effects are not calculated but they are used to indicate to the model that intra-study data are not independent which results in a proper estimation of the model's residual deviation and a non-biased quantification of the fixed variables' uncertainty.

GLMM analysis was conducted with the function *lmer* of the *lme4* package (Bates et al., 2015) using RStudio version 1.1.442 with R version 3.4.4. Confidence intervals and *p-values* (via Wald-statistics approximation) were calculated with the function *sjt.lmer* of the *sjPlot* package (Lüdecke, 2018). GLMMs were run using different combinations of fixed variables in order to minimize the Akaike information criterion (AIC) and explore the contribution of *intrinsic* and *extrinsic* factors to the variability in source level measurements. More specifically, *extrinsic* factors, in terms of methodological and technical parameters (see **Table 1**), will be regarded as possible sources for the inter-study variability.

Finally, we reviewed how the different studies characterized the relationship between ships' speed and source levels, either based on broadband measurements or from empirical models for ships' RNL/MSL predictions. This will deepen our understanding of the role played by speed on the ships' radiated noise.

5. RESULTS

5.1. Characterization of the Selected Studies

All in all, 2,275 single transits from 9 different studies are reported in this work. Technical details for each recording

TABLE 1 | Details of the Experimental Designs and Data Processing.

Selected articles	Date	Location	Protocol Water column (m)	Sample (Number of ships)	Standard	CPA (km)
Allen et al., 2012	2009–06 to 2009–09	Bar Harbor, ME USA	38.7–46.0	4	✗	$\mathcal{O}(1)$
Arveson and Vendittis, 2000	1980	Andros Island, Bahamas	1830	5	✗	$\mathcal{O}(<0.6)$
Bassett et al., 2012	2010–05 to 2011–05	Admiralty Inlet, WA USA	60	14[a]	✗	$\mathcal{O}(1)$
Kipple, 2002	2000–09 to 2001–06	Ketchikan, AK USA	360	12[b]	✗	$\mathcal{O}(0.1)$
Lesage et al., 2014	2004–2005	St. Lawrence Estuary, QC CANADA	20–250	11	✗	$\mathcal{O}(1)$
McKenna et al., 2012	2009–04	Canal de Santa Barbara, CA USA	580	29	✓	$\mathcal{O}(1)$
MCR International, 2011[c]	2011-06	English Channel, UK	87	28[d]	✓	$\mathcal{O}(0.1–1)$
...	2011–08	The Minch, Scotland	100	...	...	...
SMRU Canada, 2014	2013–06	Roberts Bank, Haro Strait, &	16–221	6	✗	$\mathcal{O}(1)$
...	...	Juan de Fuca Strait, BC CANADA	...	...	...	...
Veirs et al., 2016	2011–03 to 2013–10	Salish Sea, WA USA	10	2186	✗	$\mathcal{O}(1–<10)$

Selected articles	Manufacturer	Number of devices	Hydrophones Deployment depth (m)	Sensitivity (dB re 1 V μPa^{-1})	Bandwidth (kHz)	Type
Allen et al., 2012	C54XRS	3	5, 10, 25	−20	0.001–2.5	✗
Arveson and Vendittis, 2000	✗	5	60–460	✗	0.010–40	✗
Bassett et al., 2012	HTI 96 MIN	1	1 m above seabed	−166	0.020–30	Autonomous
Kipple, 2002	✗	3	60, 90, 120	✗	0.010–40	Autonomous
Lesage et al., 2014	ITC6050	1	15	−159	0.020–24	Monitored
McKenna et al., 2012	✗	1	570	✗	0.020–1	Autonomous
MCR International, 2011	Reson TC4032	1	∼29	✗	0.020–20	Autonomous
SMRU Canada, 2014	M8E Omni	5	3 m above seabed	−165 ± 3	0.010–64	Autonomous
Veirs et al., 2016	Reson TC4032	1	8	−164	0.012–40	Autonomous

Selected articles	Propagation loss[e]	Surface-image correction		Data processing Source approximation (m)	Source depth
Allen et al., 2012	CG, SG	✗	→	Dipole	✗
Arveson and Vendittis, 2000	SG	✗	→	Dipole	✗
Bassett et al., 2012	HG	✗	→	Dipole	10
Kipple, 2002	HG	✗	→	Dipole	✗
Lesage et al., 2014	HG	✗	→	Dipole	✗
McKenna et al., 2012	SG	✗	→	Dipole	7, 14
MCR International, 2011	SG	✗	→	Dipole	∝ Draft
SMRU Canada, 2014	RAM	N/A	→	Monopole	✗
Veirs et al., 2016	HG	✗	→	Dipole	✗

[a] *1364 transits were recorded by the authors although the individual results of only 14 merchant ships are provided.*
[b] *6 passenger ships constitute the dataset, each having been recorded twice in different operating modes.*
[c] *Two distinct observing missions were carried out in June and August of 2011.*
[d] *Includes both observing missions.*
[e] *CG: Cylindrical Geometry i.e., 10 $\log_{10}(r)$. SG: Spherical Geometry i.e., 20 $\log_{10}(r)$. HG: Hybrid Geometry i.e., [10..20] $\log_{10}(r)$. RAM: Range-dependent Acoustic Model (Collins, 1993).*
Upper panel: date and location where the recordings took place, height of the water column on deployment site, number of merchant ships' signature obtained, whether or not observations were carried using standard protocols, and the order of magnitude of the distances corresponding to the ships' closest points of approach (CPAs). Middle panel: hydrophones' technical details, number of devices used, and deployment depth. Bottom panel: backpropagation methods and corresponding source approximation.

TABLE 2 | Details of the Observational Results.

Selected articles	Data description	Vessel classes
Allen et al., 2012	• Individual broadband (0.001–2.5 kHz) source level measurements • Provides ships' individual physical properties • Provides ships' individual speed	• High-speed Crafts • *Passenger Ships* • Catamarans • Fishing Vessels
Arveson and Vendittis, 2000	• Individual broadband (0.010–40 kHz) source level measurements • Provides ship's physical properties • Provides ship's speed for each transit	• *Cargo Ships*
Bassett et al., 2012	• Mean broadband (0.020–30 kHz) source level measurements per vessel class • Provides means of the ships' physical properties per vessel class • Provides means of the ships' speed per vessel class • Individual broadband (0.020–30 kHz) source level measurements, physical properties, and speed for 24 ships of the authors' sample	• High-speed Crafts • Tugs • *Container Ships* • *Bulk Carriers* • *Vehicles Carriers* • *Cargo Ships*
Kipple, 2002	• Histograms of the individual broadband (0.010–40 kHz) source levels measurements as a function of the ships' speed • Provides ships' individual speed	• *Passenger Ships*
Lesage et al., 2014	• Individual broadband (0.010–24 kHz) source level measurements as a function of the ships' speed, length, gross tonnage, and year built • Provides ships' individual physical properties • Provides ships' individual speed	• *Oil/Chemical Tankers* • *Cargo Ships* • *Bulk Carriers* • *Container Ships*
McKenna et al., 2012	• Individual 1/3-octave frequency-dependent source level spectra • Individual broadband (0.020–1 kHz) source level measurements as a function of the ships' speed • Provides ships' individual physical properties • Provides ships' individual speed	• *Container Ships* • *Vehicles Carriers* • *Bulk Carriers* • *Cargo Ships* • *Chemical Products Tankers* • *Crude Oil Tankers* • *Product Tankers*
MCR International, 2011	• Individual broadband (0.020–2 kHz) source level measurements • Provides ships' individual physical properties • Provides ships' individual speed • Provides sea conditions and parameters for each recording	• *Tankers* • *Oil Tankers* • *Cargo Ships* • Fishing Vessels • *Bulk Carriers* • *Passenger Ships*
SMRU Canada, 2014	• Individual 1/3-octave frequency-dependent source level spectra as a function of the ships' speed • Individual broadband (0.010–70 kHz) source level measurements as a function of the ships' speed • Provides ships' individual physical properties • Provides ships' individual speed	• *Container Ships* • Tugs
Veirs et al., 2016	• Individual broadband (0.020–96 kHz) source level measurements • Provides ships' individual physical properties • Provides ships' individual speed	• Leisure Crafts • *Passenger Ships* • Tugs • Fishing Vessels • *Bulk Carriers* • *Tankers* • Military Vessels • *Container Ships* • *Cargo Ships* • Research Vessels • *Vehicles Carriers*

missions and a description of the data presented by each study are provided in **Tables 1**, **2**, respectively.

We emphasize, in **Table 1**, that all but one studies retained for this work backpropagated their received levels of noise to the sources' positions using variations of the geometrical spreading model while none of them reported having used corrections for surface-image reflections. From these specific studies, the retrieved RNLs (see section 2) are said to be surface-affected by the Lloyd's mirror effects (see e.g., Gassmann et al., 2017) and are hence referred to as dipole observations. SMRU Canada (2014) displays surface-corrected MSL measurements (see section 2) referred to as monopole observations in **Table 1**.

Vessel classes explored in this work are listed in italic in the right-hand column of **Table 2**. Whisker plots of each sample of merchant ships are plotted in **Figure 1**, hence illustrating the variability in broadband source level measurements between each study.

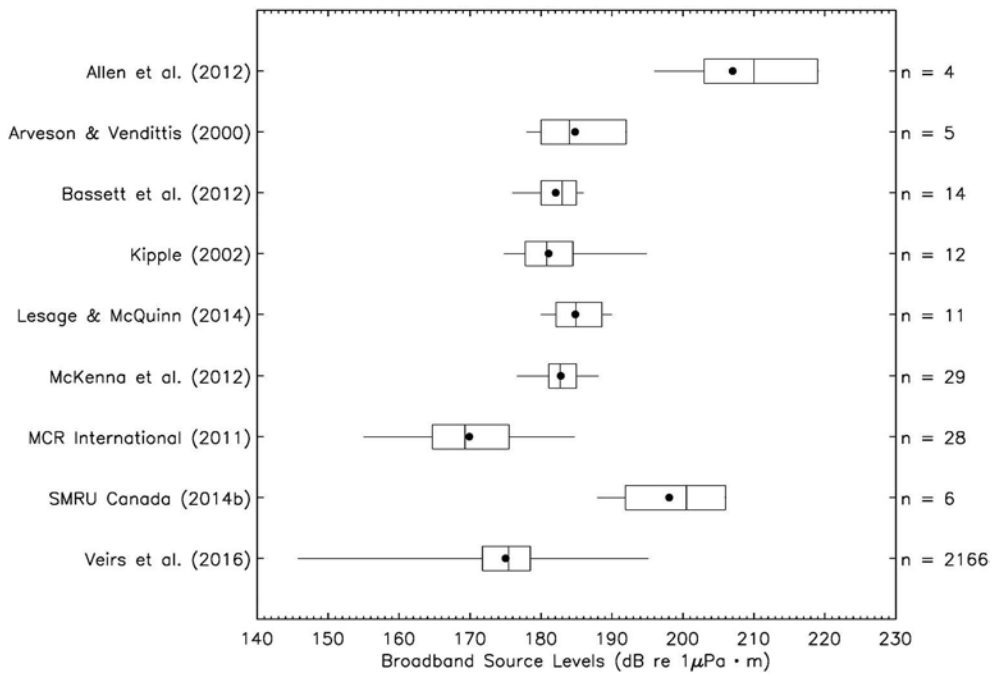

FIGURE 1 | Broadband source levels for merchant ships reported from studies listed in **Table 1**. The right-hand ordinate provides the number of vessels with available data published in each work. Whisker plots show the minimal value, the 25, 50, 75% quartiles, the maximal value, and the mean () of each sample.

5.2. Factors Explaining the Ships' Source Levels' Variability

Intrinsic factors available for the GLMM analysis are the ships' length (ℓ), width (b), speed (v), and classes (see section 2). Draft (d) and water displacement ($\propto \ell \times b \times d$) were not considered since the d parameter is missing from the Veirs et al.'s (2016) database (which accounts for the large majority of the 2,275 transits used in this work). The general consensus suggests that the logarithm of *intrinsic* factors, besides ships' classes, should be used to predict source level values (see **Table A1**).

Since we have no indications on how source level measurements behave with *extrinsic* factors that are linked to the methodological parameters and techniques of data processing, we chose to include them as linear predictors to the GLMM analysis. The *extrinsic* factors tested in the GLMM analysis are the lower and upper thresholds of the hydrophones' frequency bandwidth (f_0, f_1), the distance corresponding to the closest point of approach, the height of the water column at the hydrophones' position, the hydrophones' deployment depth, and the source approximation (i.e., whether RNL or MSL values were obtained). For the height of the water column and the deployment depth, we chose the largest value when an interval or more than one value are listed in **Table 1**. The source approximation was quantified as a standard Heaviside function that equals 1 when MSL values were gathered and 0 otherwise.

Different combinations of $\log_{10}$(*intrinsic* factors) + *extrinsic* factors were tested in an attempt to minimize the AIC using authors-specific random effects (see section 4). This was achieved using the v, b, class, CPA, and source approximation parameters while the 4 outliers provided by Allen et al. (2012) were ignored hereafter. The best-fitted model's coefficients are provided in **Table 3** and Equation (1) which indicate the correlation between *intrinsic*/*extrinsic* factors with the ships' source level values.

$$\begin{aligned}\text{Source Level} = {} & 147.94\,\text{dB} + 15.39\,\text{dB} \times \log_{10}\left[\frac{v}{v_0}\right] \\ & + 12.03\,\text{dB} \times \log_{10}\left[\frac{b}{b_0}\right] - 4.83\,\text{dB} \times \left[\frac{\text{CPA}}{r_0}\right] \\ & + 21.73\,\text{dB} \times H(\text{source}) + \phi(\text{class}), \end{aligned} \tag{1}$$

where v_0 = 1 knot, b_0 = 1 meter, r_0 = 1 nautical mile are reference values, and:

$$H(\text{source}) = \begin{cases} 1, & \text{if source approximation} = \text{MSL} \\ 0, & \text{if source approximation} = \text{RNL}, \end{cases} \tag{2}$$

$$\phi(\text{class}) = \begin{cases} (+)\,0.00\,\text{dB}, & \text{if class} = \text{Bulk Carrier} \\ (+)\,2.08\,\text{dB}, & \text{if class} = \text{Cargo Ship} \\ (+)\,1.66\,\text{dB}, & \text{if class} = \text{Container Ship} \\ (-)\,6.07\,\text{dB}, & \text{if class} = \text{Passenger Ship} \\ (+)\,1.31\,\text{dB}, & \text{if class} = \text{Tanker} \\ (+)\,0.81\,\text{dB}, & \text{if class} = \text{Vehicles Carrier}, \end{cases} \tag{3}$$

where the class reference was bulk carriers. Note that, in GLMM, qualitative parameters are always compared to the group's first element in alphabetical order.

TABLE 3 | Generalized Linear Mixed Model applied to our database.

	Source levels (dB)		
Predictor	**Estimate**	**Confidence interval**	***p*-value**
Intercept	147.94	141.20 to 154.67	<0.001
$\log_{10}(v/v_0)$	15.39	12.10 to 18.67	<0.001
$\log_{10}(b/b_0)$	12.03	9.78 to 14.28	<0.001
CPA/r_0	−4.83	−5.86 to −3.80	<0.001
H(source)	21.73	7.92 to 35.55	0.002
ϕ(class)	...	...	...
Bulk carrier (reference)	0.00	–	–
Cargo ship	2.08	1.49 to 2.68	<0.001
Container ship	1.66	0.98 to 2.34	<0.001
Passenger ship	−6.07	−7.35 to −4.79	<0.001
Tanker	1.31	0.55 to 2.08	0.001
Vehicles carrier	0.81	0.05 to 1.58	0.036

Authors-specific were used as random effects to handle the non-independence of the intra-study data. Results shown here for fixed effects are those that minimize the Akaike information criterion (AIC). Parameters v_0, b_0, and r_0 are reference values for the ships' speed, width, and closest point of approach and are respectively equal to 1 knot, 1 meter, and 1 nautical mile.

Equation (3) suggests that cargo and container ships may have the noisiest acoustical footprint which agrees with the results presented by Jalkanen et al. (2018).

5.3. Ships' Speed vs. Source Levels Relation

This subsection explores the linearity between $\log_{10}(v)$ and source levels according to (1) the observational data collected in this work (see section 5.3.1) and (2) the empirical models for source level predictions available in the literature (see section 5.3.2).

5.3.1. Observations

Speed is an *intrinsic* factors that can be regulated in order to favor an instantaneous noise attenuation of the merchant fleet. Our GLMM analysis revealed a positive correlation between a ship's source levels and speed. Therefore, a deeper analysis of the ships' speed *vs.* source levels relation is here developed by individually investigating each selected study. Results are provided in the upper panel of **Table 4** and highly suggest a positive correlation between the magnitude of the RNL/MSL measurements and the ships' speed. Slopes, in the $\log_{10}(v)$-space, range from 11.71 to 49.94 with a median of roughly 21 that agrees relatively well with the estimate found in **Table 3**. Authors typically point out however that this relation is subject to variations from one ship to another, some ships are even likely to produce more noise at speeds below their optimal cruising speed.

A study properly engineered to investigate the impact of a speed reduction on the ambient noise and the noise emitted by merchant ships was recently conducted under the Echo Program in the water basin of the Port of Vancouver (MacGillivray et al., 2019). This voluntary vessel slowdown trial provided source level measurements for merchant ships both inside and outside a speed reduction area in which the proposed speed limit was 11 knots. The source levels' variations of transiting ships were therefore solely attributed to a speed decrease since all other ship-related factors were kept constant between measurements. This approach makes the Port of Vancouver's vessel slowdown trial a valuable source of information in order to understand the impact of speed regulation on the levels of noise emitted by merchant ships. Both RNL and MSL noise-to-speed slopes were provided by the authors and are listed in **Table 4**'s middle panel.

MacGillivray et al. (2019) reveals that a 40% speed reduction in this sector results in a MSL decrease of about 10 dB. Slopes are typically a factor 2–4 steeper than what we found for data in **Table 3** (if GLMMs are processed with a source levels $\propto v$ model). Depending on the ship class, the noise-to-speed slopes vary from 1.4 to 2.8 dB knot^{-1} and appear to be slightly steeper for MSL measurements when compared to RNLs. A similar behavior can also be retrieved from **Table 4**'s upper panel with the SMRU Canada (2014) study showing the second steepest relation between source levels and $\log_{10}(v)$.

Even though a variability exists from one study to another, a large consensus seems established in the scientific community that speed reduction does indeed favors a decrease of the noise budget attributed to the merchant fleet (e.g., Audoly et al., 2017). However, this reduction of the instantaneous underwater noise radiated comes at the expense of an increase of the time spent by ships in a speed-restricted zone, hence potentially exposing nearby marine mammals to noise pollution for longer periods of time (McKenna et al., 2013; Chion et al., 2017).

5.3.2. Models

Empirical source level models listed in **Table 5** can also be used to estimate how broadband ships' source levels vary with speed changes. As in **Table 2**, vessel classes explored in this work are listed in italic in **Table 5**. Results of the source levels $\propto \log_{10}(v)$ regressions are provided in the bottom panel of **Table 4**. All models that numerically depend on the speed parameter predict an increase of the noise radiated with increasing speed. Models were tested for standard dimensions in terms of length, width, draft, and water displacement (see the right-hand column in **Table 4**'s bottom panel). Speed limits correspond to the minimal and maximal speeds retrieved, for each specific vessel class, from our 2,275 ships database (see **Supplementary Material**). The noise-to-speed slopes, in the $\log_{10}(v)$-space, range between 3.7 and 60, with a mean value of 48, roughly three times steeper than what was obtained from the GLMM approach in **Table 3**.

The mathematical formalism of each source level models listed in **Table 5** is provided in **Table A1**.

6. DISCUSSION

6.1. Impact of the Experimental Methodology

This work identifies two *extrinsic* factors, proper to the experimental design and the data processing approach, that may impact the post-processed value of ships' broadband source levels (see **Table 3**). Our GLMM analysis reveals that the values computed for broadband source levels will (1) decrease with the ships' closest point of approach increasing, and (2) increase

TABLE 4 | Source levels vs. ships' Speed Relation. Upper panel: Observational studies listed in **Tables 1, 2**.

Observations	***a*** **(SL = *a*log$_{10}$(*v*) + *k*)**	***r***	**Vessel classes**	**Number of ships**
Arveson and Vendittis, 2000	49.94	0.95	Cargo ships	5
Bassett et al., 2012	21.32	0.52	Container ships Bulk carriers Vehicles carriers Cargos	14
Kipple, 2002	21.34	0.37	Passenger ships	12
Lesage et al., 2014	18.22	0.28	Tankers Cargos Bulk carriers Container ships	11
McKenna et al., 2012	11.71	0.13	Container ships Vehicles carriers Bulk carriers Cargos Tankers	29
MCR International, 2011	19.15	0.04	Tankers Cargos Bulk carriers Passenger ships	28
SMRU Canada, 2014	27.36	0.58	Container ships	6
Veirs et al., 2016	25.04	0.16	Bulk Carriers Cargos Container ships Passenger ships Tankers Vehicles carriers	2186
Study	RNL (S = *av* + *k*)	MSL (SL = *av* + *k*)	Vessel classes	Number of ships
MacGillivray et al., 2019	2.66	2.83	Bulk carriers & Cargos	485
	1.46	1.50	Container ships	260
	1.75	1.71	Passenger ships	30
	2.52	2.65	Tankers	74
	1.57	1.57	Vehicles carriers	86
Models	*a* (SL = *a*log$_{10}$(*v*) + *k*)	*r*	Vessel classes	Details
Audoly and Rizzuto, 2015	45.46	0.99	Cargos	0.001 kHz $\leq f \leq$ 50 kHz 6.7 knots $\leq v \leq$ 22.1 knots ℓ = 200 m
	47.76	0.99	Container ships	0.001 kHz $\leq f \leq$ 50 kHz 9.0 knots $\leq v \leq$ 24.7 knots ℓ = 300 m
	42.01	0.98	Passenger ships	0.001 kHz $\leq f \leq$ 50 kHz 3.5 knots $\leq v \leq$ 25.3 knots ℓ = 225 m

(Continued)

TABLE 4 | Continued

Models	a (SL = $a\log_{10}(v) + k$)	r	Vessel classes	Details
	46.52	1.00	Tankers	0.001 kHz $\leq f \leq$ 50 kHz 8.5 knots $\leq v \leq$ 16.6 knots ℓ = 200 m
Breeding et al., 1996	60.00	1.00	Tankers	0.001 kHz $\leq f \leq$ 50 kHz 8.5 knots $\leq v \leq$ 16.6 knots ℓ = 200 m
Chion et al., 2017	5.78	0.95	Tankers Cargo ships Bulk carriers Container ships	0.001 kHz $\leq f \leq$ 50 kHz 6.7 knots $\leq v \leq$ 24.7 knots ℓ = 200 m
Luo and Yang, 2011	53.95	1.00	All 6 classes	0.001 kHz $\leq f \leq$ 50 kHz 3.5 knots $\leq v \leq$ 25.3 knots T = 20 kT
Ross and Alvarez, 1964	50.00	1.00	All 6 classes	0.001 kHz $\leq f \leq$ 50 kHz 5 knots $\leq v \leq$ 18 knots ℓ = 200 m
Simard et al., 2016	3.70	0.31	Cargo Ships Container ships Tankers	0.001 kHz $\leq f \leq$ 50 kHz 6.7 knots $\leq v \leq$ 24.7 knots ℓ = 200 m b = 32 m d = 8 m
Urick, 1983	60.00	1.00	Cargo Ships Tankers	0.001 kHz $\leq f \leq$ 50 kHz 6.7 knots $\leq v \leq$ 22.1 knots T = 20 kT m
Wales and Heitmeyer, 2002	0	–	–	Model speed-independent

Slopes a were computed using linear regressions on point-plot diagrams processed using the ($\log_{10}(v)$, Source Levels) data points provided in the authors' results. Pearson r correlation coefficient for each fit is provided. Middle panel: RNL and MSL speed-to-noise slopes provided by the ECHO Haro Strait slowdown project (MacGillivray et al., 2019). Bottom panel: Models from the literature providing source levels' mathematical formalisms. Slopes a were computed using linear regressions on point-plot diagrams processed using the ($\log_{10}(v)$, Source Levels) data points obtained by covering the parameter spaces given in the right-hand column.

when the methodological approach leads to MSL measurements (by opposition to range-independent RNLs). The following subsections detail these behaviors.

6.1.1. Closest Point of Approach

The closest point of approach between a ship and an array of hydrophones requires a compromise in order to increase the chances of good data quality. Distances below a few hundreds meters signify that the hydrophones could be located close to the ship's near field in which the approximation of a point source no longer holds. At such close range, noise is radiated from numbers of different points along the hull, each being characterized by its own source-to-receptor separation. Alternatively, very large CPAs require the use of numerical algorithms in order to properly estimate the transmission loss in complex environments that include variations of the geophysical properties of the underwater terrain and the physico-chemical characteristics of the body of water between the hydrophones and the source. In this work, 8 out of 9 studies retained present RNLs that were processed using geometrical spreading laws (see **Table 1**). Therefore, the impact attributed to complex underwater environments on the measurement of reliable source levels cannot be properly assessed. The data sample assembled in this work does show a decrease of the calculated source levels with greater CPAs to the source. This suggests that the error propagation caused by ignoring the underwater complexity in RNL measurements leads to an under-estimation of the true source level values.

6.1.2. RNL vs. MSL

Table 3 reveals that the use of spreading laws to backpropagate levels of sound received at the hydrophones to the sources' positions without adding corrective terms to compensate surface-image reflections may lead to an underestimation of the ships' source levels by as much as 35 dB (i.e., upper limit of the

TABLE 5 | Details of the theoretical models published in the literature that serve as RNL/MSL predictors.

Selected articles	Source	Parameters required	Vessel classes	Comments
Audoly and Rizzuto, 2015	Monopole	• Frequency (*independent variable*) • Length • Speed	• *Cargo Ships* • *Tankers* • Ferries • *Passenger Ships* • High-speed Crafts • Fishing Vessels • Research Vessels • Leisure Crafts • Tugs • Sailing Boats • *Container Ships*	• Provides individual models for each vessel class
Breeding et al., 1996	Dipole	• Frequency (*independent variable*) • Length • Speed	• *Merchant Ships* • *Tankers* • Fishing Vessels	• Known as the RANDI 3.1 model
Chion et al., 2017	Dipole	• Frequency (*independent variable*) • Length • Speed	• *Oil/Chemical Tankers* • *Cargo Ships* • *Bulk Carriers* • *Container Ships*	• Extension of the Breeding et al., 1996's model • Derived from the Lesage et al., 2014's sample
Luo and Yang, 2011	Dipole	• Frequency (*independent variable*) • Ship tonnage • Speed	• *Watercrafts*	• Extension of the original Ross model (see Ross, 2013) • Tonnage between 100 and 100 000 tons
Ross and Alvarez, 1964	Dipole	• Frequency (*independent variable*) • Length • Speed	• *Merchant Ships*	• Originally derived from the propellers' tip speed and the number of blades • Later, conveniently adapted to the ships' cruising speed
Simard et al., 2016	Monopole	• Frequency (*independent variable*) • Length • Breadth • Draft • Speed	• *Cargo Ships* • *Container Ships* • *Tankers*	• Referred as the LBDS model in the original paper • Notice the publication of a subsequent erratum
Urick, 1983	Dipole	• Frequency (*independent variable*) • Ship tonnage • Speed	• *Cargo Ships* • *Tankers* • Large Warships	• Originally derived from the propellers' tip speed • Later, conveniently adapted to the ships' cruising speed
Wales and Heitmeyer, 2002	Monopole	• Frequency (*independent variable*)	• *Merchant Ships*	• No statistically significant dependence between the source levels and the ships' physical dimensions and speed

Mathematical formalism is provided in ***Table A1****.*

confidence interval) which is clearly assessed in Panels (a) and (b) of Farcas et al.'s (2016) Figure 1[2]. This contributes to explain, for example, median source level values in the vicinity of 200 dB re 1 μPa $\cdot$ m reported by Simard et al. (2016), results that are similar to those listed in SMRU Canada (2014). Given that 8 out 9 observational studies reported in this work (see **Table 1**) and 5 out 8 source level models (see **Table 5**) are based on RNL measurements, our GLMM analysis supports the need to more rigorously assess what is the best-suited [to the studys needs] numerical algorithm regarding the backpropagation processing (see Table 1 of Farcas et al., 2016).

7. STUDY'S LIMITATIONS

This study would definitely benefit from the addition of more MSL measurements in order to properly assess the impact of the monopole *vs.* dipole approach on source levels' variability.

Other *extrinsic* factors (i.e., related to the field campaign) that likely play a role on the determination of the source level values cannot be easily quantified *a posteriori* and are beyond the scope of this paper.

7.1. Directionality and Recommended Hydrophone Angles

Usually treated as a point source in its far field, noise emitted by a ship is in fact directional and anisotropic. Hence, the alignment between a ship and an hydrophone will play a role in the sound levels recorded (Arveson and Vendittis, 2000; Gassmann et al., 2017).

The hydrophone angle, sustained between the source-to-hydrophone line and the sea surface, appears to lead to smaller source level measurements when small angles ($< 1^\circ$) are involved (e.g., see results from Veirs et al., 2016). Standard protocols (e.g., ANSI, 2009; ISO 17208-1, 2016) recommend to average three (3) simultaneous recordings of a source at hydrophone angles of 15°, 30°, and 45° (see Figure 1 of ISO 17208-1, 2016). For the sake of comparison, an hydrophone angle of 0.2° results in source level values 5 to 10 dB lower than what is obtained for a 10° angle in the 0.020−1 kHz bandwidth using a spreading law for backpropagation calculation (Gassmann et al., 2017). This difference is somewhat reduced to 3–7 dB when correcting for surface-image reflections (cf. using Equation 3 in Gassmann et al., 2017).

7.2. Estimation of the Ship Source Depth

For MSL calculations, uncertainty on the determination of the ships' source depth will have an impact on the transmission loss profiles predicted and, therefore, on the value computed for the source levels. Numerical algorithms for backpropagation such as BELLHOP (Porter and Liu, 1994) and RAM (Collins, 1993) have proven to be highly sensitive to the value chosen for the source's depth as an input parameter. Estimations off by few meters have shown variations in MSL measurements up to 10 dB at low frequencies (Arveson and Vendittis, 2000; Gassmann et al., 2017).

Although the importance of the source's depth on MSL predictions have been demonstrated (see e.g., Figure 3 of Gassmann et al., 2017), this parameter is rarely discussed in observational studies, hence making it difficult to quantitatively estimate its impact on the data presented in this current study.

7.3. Other Factors

Methods of calibration of the hydrophones and the conditions in which these are stocked before deployment will impact the electrical response of the hydrophones to sound (Dakin and Heise, 2015). Calibration in laboratory will have a precision of $\pm$ (0.5–2) dB while *in situ* underwater calibration will have a precision of $\pm$ (3–6) dB (Dakin and Heise, 2015).

The reader will also note that the Veirs et al.'s (2016) sample represents the large majority of the data available for this study (see **Figure 1**). This may be at the origin of certain statistical bias in the quantification of fixed effects (e.g., the closest point of approach) on the values calculated for source levels.

Environmental conditions will also impact the magnitude of the received levels of sound at the hydrophones (e.g., sea roughness, rain lapping, strong winds, waves, currents). The subtraction of this background noise is not trivial and makes it difficult to properly isolate ships' acoustic signatures. Finally, gradients in speed of sound, attributed to a stratification in water temperature, acidity and/or salinity, will induce sound refraction and create tunneling effects that can contaminate sound samples recorded by hydrophones located at very large distances.

8. CONCLUSION

This work constitutes a literature review and a meta-analysis of the studies aiming at opportunistically assess the levels of noise emitted by merchant ships at the source. It is particularly aimed at supporting the interpretation of the variability in ships' broadband source levels reported in the literature. We specifically focused on the apparent lack of consensus throughout the literature and identify the common ground between different studies aiming at opportunistically estimate of ships' source levels and their contributing factors.

The main results of our study are:

1. Our analysis revealed a positive correlation between source level measurements and ships' speed-above-ground i.e., source levels $\sim$ 15.39 dB $\times$ $\log_{10}\left[\frac{v}{1\,\text{knot}}\right]$. Limitations in transiting should definitely be considered as measure of mitigation in order to maintain underwater noise attributed to merchant ships within reasonable levels.
2. We have demonstrated that differences in methodological protocols for opportunistic measurements of ships' underwater noise contribute to the inter-study variability reported for source level values of merchant ships. This is reflected by the presence of both CPA and Source

[2]One can estimate, in Panel (a), a transmission loss of approximately 35 dB between the source and the diagram's lower-left corner. Applying this loss to the received levels illustrated in Panel (b)'s lower-left corner and backpropagating to the position of the source yields a RNL roughly 25 dB re 1 μPa $\cdot$ m short of the MSL value.

extrinsic factors as statistically significant explanatory variables in the best-fitted GLMM describing source levels; see Equation (1). That said, our results support the necessity to use standardized approaches to conduct hydrophone-based recordings of underwater noise sources. The backpropagation methods used to estimate ships' source levels from hydrophone measurements also needs to be adapted to both the experimental setup and environmental characteristics to control as much as possible for the biasing factors. In particular, the commonly used geometrical spreading laws are clearly unadapted to some complex underwater environments, leading to an under-estimation of the backpropagated source levels.

3. Error estimation and propagation need to be refined as source level measurements provided in the literature never include envelopes of uncertainty.

This study recommends that:

1. Narrowband or 1/3-octave band measurements of ships' source levels instead of broadband values should be made available to the scientific community. Our study demonstrated that the interpretation of broadband source levels is subject to confusion, hence making the comparison between studies that focus on ships' underwater radiated noise particularly difficult. The publication of narrowband measurements would definitely benefit our field of study and contribute to facilitate the data interpretation of secondary users of ships' source level measurements.
2. In order to properly quantify the impact of individual *intrinsic* factors (e.g., speed, load, draft, working engines) on the underwater noise radiated by merchant ships and ultimately mitigate them, control experiments could be designed in order to favor the simultaneous control and monitoring of the factors contributing to ships' noise. This way, the impact of a single factor (e.g., ships' speed, ships' load, number of engines in operation) can be quantified while others are kept constant. This approach, although more costly, will help to gain a better understanding of onboard noise sources and could serve as a baseline to improve the interpretation of the growing number of ships' source level data coming from opportunistic measurements.

DATA AVAILABILITY STATEMENT

The datasets for this manuscript are not publicly available because Data were provided during the submission process. Requests to access the datasets should be directed to clement.chion@uqo.ca.

AUTHOR CONTRIBUTIONS

Review of literature, data processing, and statistical analysis were conducted by DL. CC was responsible for reaching out to authors in order to obtain methodological details that were not *a priori* available in the original studies. Responsibilities for the redaction, figures, and tables processing of this manuscript were equally shared between DL and CC. JD contributed to the revision of the early versions of this paper and also provided funding to support this work.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00714/full#supplementary-material

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APPENDIX

Mathematical Formalism of Empirical Source Level Models

Equations for the empirical source level models mentioned in sections 5.2 and 5.3.2. are provided in **Table A1**.

TABLE A1 | Models' mathematical formalism.

Selected articles	Equations
Audoly and Rizzuto, 2015	Individual model for each vessel class. See the authors' section 5. Detailed example for cargo ships: $SL(f,v) = SL_{mach}(f,v) + SL_{prop}(f,v) + SL_{cav}(f,v)$ $SL_{mach}(f,v) = 136 + 15\ \log_{10}(v)$ if $f \leq 200$ Hz $SL_{mach}(f,v) = 186 - 22\ \log_{10}(f) + 15\ \log_{10}(v)$ if $f > 200$ Hz $SL_{prop}(f,v) = 109 - 5\ \log_{10}(f) + 50\ \log_{10}(v)$ if $f \leq 80$ Hz $SL_{prop}(f,v) = 156 - 30\ \log_{10}(f) + 50\ \log_{10}(v)$ if $f > 80$ Hz $SL_{cav}(f,v) = 79 + 10\ \log_{10}(f) + 60\ \log_{10}(v)$ if $f \leq 50$ Hz and $v \geq v_{cav}$ $SL_{cav}(f,v) = 129 - 20\ \log_{10}(f) + 60\ \log_{10}(v)$ if $f > 50$ Hz and $v \geq v_{cav}$ $SL_{cav}(f,v) = 0$ if $v < v_{cav}$ where f and v are in units of Hertz and knots, and $v_{cav} = 10$ knots.
Breeding et al., 1996	$SL(f,v,\ell) = SL_0(f) + 60\log_{10}\left(\frac{v}{12}\right) + 20\log_{10}\left(\frac{\ell}{300}\right) + df \times d\ell + 3.0$ $SL_0(f) = -10\ \log_{10}(10^{-1.06\ \log_{10}(f) - 14.34} + 10^{3.32\ \log_{10}(f) - 21.425})$ if $f \leq 500$ Hz $SL_0(f) = 173.2 - 18\ \log_{10}(f)$ if $f > 500$ Hz $df = 8.1$ if $f \leq 28.4$ Hz $df = 22.3 - 9.77\ \log_{10}(f)$ if $f > 28.4$ Hz $d\ell = \left[\frac{\ell^{1.15}}{3643.0}\right]$ where f, v and ℓ are respectively in units of Hertz, knots, and feet.
Chion et al., 2017	$SL(f,v,\ell) = SL_0(f) - \left[\frac{144.7}{v-12}\right]\log_{10}\left(\frac{v}{12}\right) + 20\log_{10}\left(\frac{\ell}{300}\right) + df \times d\ell$ $SL_0(f) = -10\ \log_{10}(10^{-1.06\ \log_{10}(f) - 14.34} + 10^{3.32\ \log_{10}(f) - 21.425})$ if $f \leq 500$ Hz $SL_0(f) = 173.2 - 18\ \log_{10}(f)$ if $f > 500$ Hz $df = 8.1$ if $f \leq 28.4$ Hz $df = 22.3 - 9.77\ \log_{10}(f)$ if $f > 28.4$ Hz $d\ell = \left[\frac{\ell^{1.15}}{3643.0}\right]$ where f, v and ℓ are respectively in units of Hertz, knots, and feet.
Luo and Yang, 2011	$SL(f,v,T) = SL_0(v,T) + 20 - 20\ \log_{10}(f_0(v))$ if $f \leq f_0(v)$ $= SL_0(v,T) + 20 - 20\ \log_{10}(f)$ if $f > f_0(v)$ $SL_0(v,T) = 112 + 50\ \log_{10}\left(\frac{v}{10}\right) + 15\log_{10}(T)$ $f_0(v) = 1000 - 900\left[\frac{v}{40}\right]$ where f, v and T are respectively in units of Hertz, knots, and tons.
Ross and Alvarez, 1964	$SL(f,v,\ell) = 190.5 + 50\ \log_{10}\left(\frac{v}{10}\right) + 20\ \log_{10}\left(\frac{\ell}{150}\right) - 20\ \log_{10}(f)$ where f, v and ℓ are respectively in units of Hertz, knots, and meters.
Simard et al., 2016	$SL(f,\ell,b,d,v) = 285.40 + 0.0496\ f - 4.8 \times 10^{-7}\ (f - 2108.26)^2 - 69.33\ \log_{10}(f) -$ $49.29\ (\log_{10}(f) - 2.70016)^2 - 58.50\ (\log_{10}(f) - 2.70016)^3 -$ $41.54\ (\log_{10}(f) - 2.70016)^4 - 7.62\ (\log_{10}(f) - 2.70016)^5 +$ $13.47\ \log_{10}(\ell) - 0.55\ b + 0.0008\ (b - 26.8854)^3 + 0.706\ d +$ $20.164\ \log_{10}(v) - 505.1\ (\log_{10}(v) - 1.12024)^3 + 2891.9\ (\log_{10}(v) - 1.12024)^5$ where f, ℓ, b, d, and v are respectively in units of Hertz, meters, meters, meters, and knots.
Urick, 1983	$SL(f,v,T) = 95 + 60\ \log_{10}(v) + 9\ \log_{10}(T) - 20\ \log_{10}(f)$ where f, v and T are respectively in units of Hertz, knots, and tons.
Wales and Heitmeyer, 2002	$SL(f) = 230 - 45.94\ \log_{10}(f) + 9.17\ \log_{10}\left[1 + \left(\frac{f}{340}\right)^2\right]$ where f is in units of Hertz.

Intrinsic factors v, ℓ, b, d, and T are respectively the ship's speed, length, breadth, draft, and tonnage. Frequency is f in Hz units.

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Night and Day: Diel Differences in Ship Strike Risk for Fin Whales (*Balaenoptera physalus*) in the California Current System

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Collisions with ships (ship strikes) are a pressing conservation concern for fin whales (*Balaenoptera physalus*) along western North America. Fin whales exhibit strong diel patterns in dive behavior, remaining near the surface for most of the night, but how this behavior affects ship-strike risk is unknown. We combined diel patterns of surface use, habitat suitability predictions, and ship traffic data to evaluate spatial and temporal trends in ship-strike risk to fin whales of the California Current System (CCS). We tested a range of surface-use scenarios and found that both increased use of the upper water column and increased ship traffic contribute to elevated ship-strike risk at night. Lengthening nights elevate risk during winter throughout the CCS, though the Southern California Bight experienced consistently high risk both day and night year-round. Within designated shipping lanes, total annual nighttime strike risk was twice daytime risk. Avoidance probability models based on ship speed were used to compare the potential efficacy of speed restrictions at various scales. Speed reductions within lanes may be an efficient remediation, but they would address only a small fraction (13%) of overall ship-strike risk. Additional speed restrictions in the approaches to lanes would more effectively reduce overall risk.

Keywords: ship strike, fin whale, *Balaenoptera physalus*, California Current, diel dive behavior, behavioral ecology

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INTRODUCTION

Of the great whales, fin whales (*Balaenoptera physalus*) were hunted in the highest numbers (Aguilar, 2009) and today they are among the most often struck by ships (Laist et al., 2001). Collision with ships (ship strike) is currently considered the most pressing conservation issue for fin whales in the eastern North Pacific (National Marine Fisheries Service [NMFS], 2010; Carretta et al., 2018), where productive coastal ecosystems overlap with busy shipping areas. While the connectivity and structure of fin whale subpopulations in this ocean basin remain poorly understood (Carretta et al., 2018), fin whales in the California–Oregon–Washington stock are listed as Endangered under the Endangered Species Act (Carretta et al., 2018), and those in the adjacent

Pacific Canada population are listed as Threatened under Canada's Species At Risk Act (Committee on the Status of Endangered Wildlife in Canada [COSEWIC], 2019).

The fin whale accounts for a high proportion of documented ship-strike mortalities in U.S. waters (Jensen and Silber, 2003; Douglas et al., 2008; Neilson et al., 2012; Carretta et al., 2018) and elsewhere (Panigada et al., 2006). From 2009 to 2015, there were 10 documented fin whale mortalities attributed to ship strike along the coast of California, eight of which occurred in the Southern California Bight (NOAA, unpublished data). Ship-whale encounter models for the U.S. west coast Exclusive Economic Zone indicated that the ship-strike mortality rate for fin whales is twice that of blue whales (*Balaenoptera musculus*) and 2.4× that of humpback whales (*Megaptera novaeangliae*), and is estimated to be 2.7× above the Potential Biological Removal limit for non-natural mortality currently set by the National Marine Fisheries Service (Rockwood et al., 2017).

Off the west coast of Vancouver Island, Canada, fin whale ship-strike mortality rates are estimated to be nearly equal to those of the far more abundant humpback whale due to their specific distributions in relation to the busy shipping area of Juan de Fuca Strait (Nichol et al., 2017). In the waters between Vancouver Island and continental North America, an area where sighting rates of fin whales are low, 12 fin whales have been found dead with evidence of ship strike since 1986, though the causes of these mortalities are unconfirmed (Towers et al., 2018).

A necessary first step in mitigating this problem is identifying the areas where the risk of ship strike is greatest. To do this, spatially explicit risk models are typically developed based on the co-occurrence of ships and whales (e.g., Fonnesbeck et al., 2008; Williams and O'Hara, 2010; Redfern et al., 2013; Nichol et al., 2017; Rockwood et al., 2017). In these models, ship traffic distributions are derived from publicly available or previously published data archives, and whale distributions are inferred from field surveys or habitat models that include field data, either surveys (e.g., Nichol et al., 2017; Rockwood et al., 2017) or tag deployments (e.g., Scales et al., 2017). A common result of this approach is that strike risk is effectively defined as the spatially explicit feasibility of an interaction between a whale and a ship. If an interaction occurs, it may or may not involve detection (by the whale and/or the ship's crew), attempted avoidance (also by the whale and/or the ship), or collision (lethal or non-lethal).

Some studies have incorporated further complexity into these basic risk assessments by considering factors such as likelihood of whale avoidance (e.g., Kite-Powell et al., 2007; McKenna et al., 2015), expected rates of collision (e.g., van der Hoop et al., 2012; Nichol et al., 2017; Rockwood et al., 2017), and the lethality of collision (e.g., Vanderlaan and Taggart, 2007; Wiley et al., 2011; Conn and Silber, 2013; Nichol et al., 2017). These factors often include details about ships, such as type, size class, noise characteristics, such as source levels and frequency bands, and hull draft, as well as the behavioral response of whales such as the ability to detect and successfully avoid ships. The speed of oncoming traffic is often found to be a primary determinant of both the probability of a collision and its lethality, such that mortality increases significantly at higher speeds (Kite-Powell et al., 2007; Wiley et al., 2011; McKenna et al., 2012, 2015; Nichol et al., 2017; Rockwood et al., 2017). All of these factors must be considered in order to accurately estimate whale mortalities based upon models of ship-strike risk, as well as to weigh mitigation options within ship Traffic Separation Schemes (TSS, hereafter referred to as shipping lanes), such as speed reductions, lane shifts, and designated "Areas to Be Avoided" (Rockwood et al., 2017).

Whale behaviors have been incorporated into ship-strike studies in terms of detection and avoidance (e.g., Kite-Powell et al., 2007; McKenna et al., 2015; Rockwood et al., 2017), but rarely in terms that *precede* interaction with a ship. The mortality model in Rockwood et al. (2017) was the first to incorporate one such *a priori* behavior, i.e., the time whales spend at various depths, which was modeled from whale-borne time-depth recording tags. This is a critical parameter, since putative strike risk is only real when whales are within near-surface waters in which collision is feasible. In principle, patterns in a species' vertical habitat use can be an important factor in ship-strike risk, but this remains to be studied for fin whales.

An emerging understanding is that some baleen whales exhibit strong diel patterns in their use of vertical habitat (Panigada et al., 1999; Panigada et al., 2003; Calambokidis et al., 2007; Friedlaender et al., 2009, 2013, 2015, 2016; Burrows et al., 2016; Tyson et al., 2016; Keen et al., 2019). In the case of fin whales of the California Current System (CCS), these patterns include increased use of near-surface waters (within 20 m) at night (Keen et al., 2019). This diel vertical shift in habitat use would logically result in greater spatial overlap between fin whales and transiting ships at night, when the abilities of whale and ship to detect and avoid one another based on visual cues are likely impaired. Other factors, such as the alertness and responsiveness of whales to ships, may shift along with this increased surface use at night and affect the balance of strike risk factors (e.g., for other baleen whale species: Nowacek et al., 2004; Kite-Powell et al., 2007; McKenna et al., 2015). Such diel modes of habitat use, and their associated behavioral contexts, could affect strike risk in predictable ways, but these interactions have not yet been explored.

Our primary objective in the present study, therefore, was to incorporate diel patterns of fin whale habitat use and ship traffic into spatially explicit models of ship-strike risk within the CCS at various spatial and temporal scales, with particular focus on the most heavily trafficked areas in and near shipping lanes. We then used these models to evaluate the potential efficacy of several ship speed reduction strategies to mitigate ship-strike risk.

MATERIALS AND METHODS

General Study Approach

Our approach had two main stages. First, we used ship positional data from Automatic Identification System (AIS), fin whale diel ratios of surface use, and habitat suitability models to assess monthly and diel patterns of strike risk across distinct regions in our study, as well as in and around shipping lanes. We calculated this strike risk as the product of ship traffic volume and the proportion of time fin whales spend in the upper 20 m of the water column, scaled by habitat suitability. Second, we used

avoidance probabilities as permuted functions of vessel speed to explore ship-strike expectations under various speed reduction scenarios in and around shipping lanes and at various times. We calculated strike expectations as the product of ship-strike risk and the probability of a ship and whale failing to avoid each other.

Table 1 details the conceptual framework we used to invoke these factors. We carried out our study at the same spatial resolution (0.05 × 0.05° grid; 27,360 grid cells) used in habitat suitability models from Scales et al. (2017). We did so for years 2009, 2011, and 2013, which correspond to the same period of whale tag data collection as Scales et al. (2017). Due to gaps in 2009 shipping data (the section "Ship Traffic"), we focus most results upon 2011 and 2013 (the sections "General Findings" and "Strike Expectation and Speed Reductions"). Speed reduction analyses were carried out using 2013 as a case study since it was the most recent year of data we analyzed.

Study Area

We defined our study area as the portions of the west coast of North America used in Scales et al. (2017) for which ship traffic data were also available (**Figure 1**). Ship-strike risk factors were examined on three nested scales: (i) the entire study area (126–116°W, 30–50°N); (ii) three subregions known to have relatively high levels of ship traffic and to contain a shipping lane, which we will refer to as the Southern California Bight (121–116.5°W, 31.5–34.5°N), the San Francisco Bay Area (125–121.75°W, 36.5–39°N), and the Pacific Northwest (126–122°W, 46–49.75°N), which included southwest Vancouver Island, Juan de Fuca Strait, Strait of Georgia, Salish Sea, Puget Sound, and the coastal waters of Washington, and Oregon; and (iii) the three shipping lanes within those regions, referred to here as the Los Angeles, San Francisco, and Juan de Fuca lanes. Note that the Pacific Northwest subregion included both U.S. and Canadian waters, and the Juan de Fuca lane straddles the international border. Shipping lanes were delineated using polygons retrieved from data.gov[1]. For each grid cell in the study area, the hours of darkness and daylight in each month of the study were determined based on its centroid's coordinates using the package "oce 1.0-1" in R 3.5.1 (R Core Team, 2016).

Note that we included the Pacific Northwest in our study area despite relatively few sightings in this region, particularly from the interior waters of Juan de Fuca Strait, the Strait of Georgia, the Salish Sea, and Puget Sound (Ford, 2014; Towers et al., 2018), and despite the fact that habitat suitability models for this region were based on a limited number of tag deployments off the Washington coast (Scales et al., 2017) and thus may be less accurate than models from areas with more tag data. We included the Pacific Northwest for the following reasons: First, this is a heavily trafficked marine area used by fin whales and other large cetaceans, and is therefore of general interest (Nichol et al., 2017; Towers et al., 2018). Second, waters off southwestern Vancouver Island, including the approaches and western portion of the Juan de Fuca shipping lane, have recently been identified as a region of high ship-strike risk for fin whales (Nichol et al., 2017). Third, encounters with both live fin whales and dead fin whales with evidence of ship strike have occurred within the interior waters between Vancouver Island and the mainland throughout the last 20 years (Towers et al., 2018, and references therein). Fourth, the region represents potential habitat into which the recovering fin whale populations might expand. Finally, its high-latitude waters demonstrate the potential effects of seasonal changes in daytime length on patterns of ship-strike risk.

Ship Traffic

Automatic Information System (AIS) data were downloaded from Marine Cadastre[2] (Bureau of Ocean Energy Management and National Oceanic and Atmospheric Administration, 2016). AIS transmissions collected by coastal stations and satellites include the position and characteristics for all ships greater than 300 gross tons as required by the International Maritime Organization, for most ships in U.S. waters greater than 19.8 m (65 ft) as required by the U.S. Coast Guard, and for voluntary use by smaller vessels. The decimated AIS data provided as ArcGIS geodatabases by Marine Cadastre included a position update roughly every minute for all AIS-transmitting vessels and had been processed by the Coast Guard for quality control. Each update included a unique ship identifier (MMSI), voyage identifier, vessel dimensions including hull draft (decimeters), timestamp, latitude, longitude, speed over ground (SOG; kn), and course over ground. Geodatabases were provided for each year-month within each Universal Transverse Mercator (UTM) zone. The U.S. west coast and southernmost waters of British Columbia, Canada, falls within UTM zones 10 (Pacific Northwest and north-central California) and 11 (Southern California Bight region). Shipping data were processed at 0.05° resolution to characterize traffic during day and night. AIS data were analyzed in R after data conversion to ASCII format in ArcGIS v10.5. Within each year-month of each UTM zone, the following procedure was carried out for every unique vessel.

(1) AIS data were filtered to include only ships underway (>1 kn) and with a reported length of at least 19.8 m and hull draft of >0 m (to remove faulty data; following Rockwood et al., 2017). Smaller vessels, which certainly pose a strike risk to fin whales, are excluded here due to two factors; (i) small vessels are not required to have AIS, so there is no way to fully explore strike risk, and (ii) collisions with smaller vessels are less likely to lead to mortality (Laist et al., 2001).
(2) Solar elevation was calculated for every AIS transmission according to its timestamp, latitude, and longitude (using R package "oce"). Data with negative solar elevations (i.e., sun below the horizon) were classified as nighttime; those with positive solar elevations were classified as daytime).
(3) Data were then converted into spatial lines and grouped into daytime and nighttime datasets. Care was taken to ensure that breaks in AIS updates did not contribute to track distances. That is, track distances are based only on the data that are available for underway ships with

[1]https://catalog.data.gov/dataset/shipping-fairways-lanes-and-zones-for-us-waters44831

[2]http://marinecadastre.gov/ais/

TABLE 1 | Metrics and concepts used in this study.

Metric	Definition	Source	Details
Traffic volume	Number of transit hours for all underway AIS-reporting ships >19.8 m length within a given area and time period.	Publicly archived AIS data (Marine Cadastre)	Unit is transit hours. Calculated for each month.
Surface use	Proportion of time fin whale spends near the surface (<20 m).	Both theoretical and empirical (Keen et al., 2019)	A fraction between 0 and 1.
Potential strike exposure	The number of transit hours during which a fin whale, if present, would be near the surface and therefore exposed and vulnerable to passing ships.	Calculated based on metrics above	Exposure = Traffic volume × Surface use Unit is transit hours. Calculated for each month.
Habitat suitability	A relative scale of fin whale habitat suitability.	Predicted based on models in Scales et al. (2017) and publicly archived data.	A fraction between 0 and 1. The unit for this metric is arbitrary and therefore indicates only relative differences in risk across space and time. Calculated for each month.
Strike risk	The danger presented to fin whales by ship traffic *before* accounting for avoidance capabilities. The greatest risk will occur where opportunities for whale-ship interactions are maximized, i.e., where the most suitable habitats overlap with the highest traffic volume and when fin whale surface use is greatest.	Calculated based on above metrics	Potential strike exposure scaled by habitat suitability index. The unit for this risk metric is arbitrary and therefore indicates only relative differences in risk across space and time. Calculated for each month.
P(Avoidance)	The probability of a fin whale and ship on collision course avoiding each other.	Modeled	In this simplified framework, avoidance probability is a threshold function of vessel speed (see the section "Materials and Methods").
P(Strike)	The probability of a fin whale and ship on collision course failing to avoid each other.	Modeled	*P*(Strike) = 1−*P*(Avoidance)
Strike threshold	The ship speed at which *P*(Avoidance) = 0.50 = *P*(Strike)	Modeled	The inflection point of the threshold function that determines *P*(Avoidance) (see the section "Materials and Methods"). Unit is kn.
Strike expectation	The expected outcome of dangers posed by ship traffic, approximated by scaling ship-strike risk by the probability of avoidance.	Modeled	Expectation = Risk * *P*(Strike) No units; a relative metric across space and time. Calculated for each month.
Diel ratio	The ratio of nighttime to daytime values for any of the above metrics.	Calculated	Ratios greater than 1 indicate that the metric is greater at night. Data are assigned to nighttime and daytime datasets according to geospatial coordinates and timestamps.

functioning AIS receivers, and are therefore a conservative estimate of actual ship activity.

(4) Daytime and nighttime grids of the study area (0.05 × 0.05°) were populated with a suite of metrics: distance (km) covered in transit, number of transits, mean SOG (kn), and hours in transit, which was calculated by dividing distance by mean SOG. These metrics were calculated by clipping track lines to each grid cell and calculating line length (km) using functions in R packages "rgeos" and "raster." If a valid SOG is not reported during a ship's transit of a grid cell, the mean underway SOG reported for all of its entries in the UTM zone was used to approximate transit time within the cell.

Previous ship-strike risk studies of fin whales have parsed AIS data by ship type, size class, and/or speed class under the logical assumption that these distinctions are important factors in the avoidance and lethality of strikes (e.g., Nichol et al., 2017; Rockwood et al., 2017). We did not parse AIS data this way under the assumption that any interaction with an underway ship longer than 19.8 m (as well as smaller vessels) poses serious risks to a whale, and because it remains unknown how avoidance probabilities differ by ship type or size. Further, monthly and geographic variation in ship traffic composition likely impacts the nature of avoidance and lethality, but in ways and to extremes that are currently unknown. Incorporating such factors into our strike risk framework would require another series of theoretical models. Finally, our goals of (i) analyzing diel patterns in strike risk at large scales in space and time, then (ii) exploring how those broad patterns influence potential mitigation measures required us to remove complexity wherever possible in order to reduce the computational load.

For each month, traffic was characterized by "volume" (combined hours underway from all reporting ships throughout a given area of the study grid) and "rate" during day and night (volume per duration of the diel period; **Table 1**). "Overall" activity was calculated by adding daytime and nighttime metrics together. Differences in nighttime and daytime activities were quantified using a "diel ratio" (night/day, **Table 1**), such that relatively high nighttime activity is indicated by a ratio greater than 1.0. Similar metrics were computed for the following other factors in our strike risk analysis (the sections "Potential Strike Exposure," "Habitat Suitability," Ship-Strike Risk," "Shipping Lanes," and "Strike Expectation and Speed Reductions").

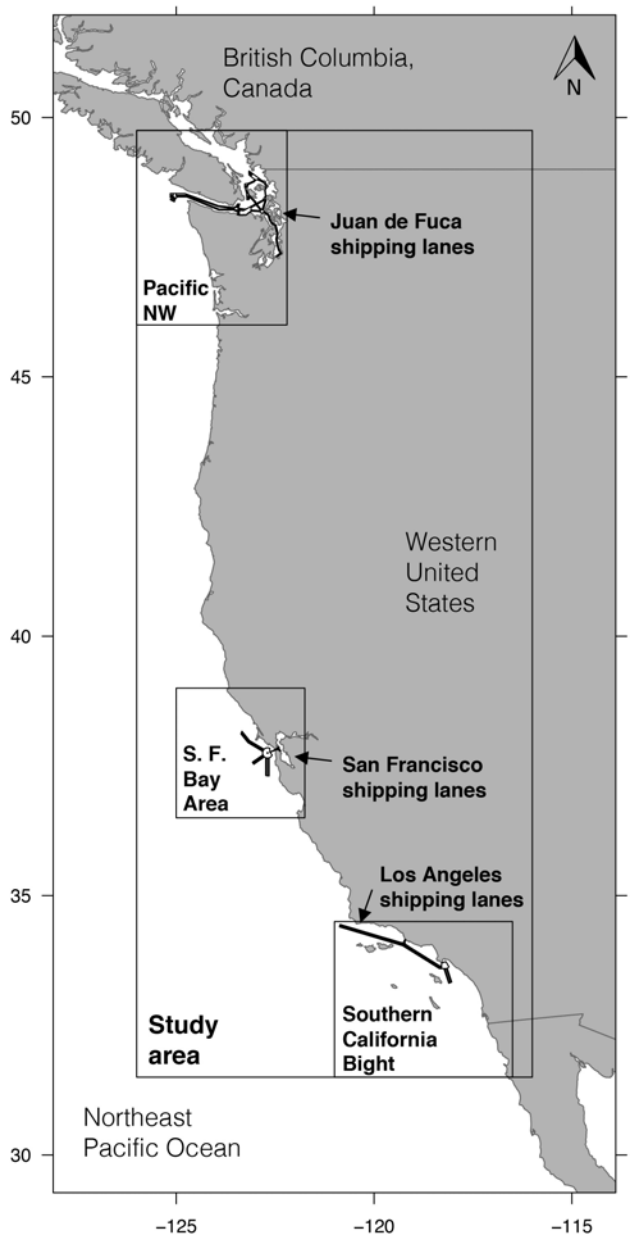

FIGURE 1 | Map of the North American west coast indicating study area, subregions (Southern California Bight, San Francisco Bay Area, and Pacific Northwest), and their shipping lanes (Los Angeles, San Francisco, and Juan De Fuca, respectively).

We examined the effects of ship speed on an annual timescale by calculating mean transit speed in each grid cell, defined as the total kilometers traveled by all ships divided by the total transit time for all ships. Maps were generated to examine geographic patterns in the means and diel ratios of ship speed.

The AIS data available on MarineCadastre.gov were incomplete for UTM zone 10 (Pacific Northwest and north-central California) in June and July 2009, leading us to focus on 2011 and 2013 for most of our results.

Potential Strike Exposure

Diel modes of surface use were incorporated into our risk analysis using the concept of potential strike exposure (hereafter, exposure) defined as the number of transit hours during which a fin whale, if present, would be near the surface and therefore exposed and vulnerable to passing ships (**Table 1**). Exposure analyses were conducted by scaling traffic volume by the proportion of time fin whales spent at the surface. For all months in 2011 and 2013, this was done using empirical surface use data (i.e., the proportion of time spent above 20 m) of 0.57 at night and 0.42 during the day (diel ratio = 1.36:1), which were the median values derived by Keen et al. (2019) from 12 tag deployments on fin whales in southern California (8,753 sets of dives + post-dive surface time over 264.3 days; mean deployment duration of 22.1 days). Their study found some evidence of geographic and seasonal variation in diel surface use. However, given the complexity of our analysis, we used a diel ratio of surface use based on the published values, then augmented this empirical diel ratio with alternative surface use scenarios, using 2013 as a case study. In generating these alternatives, we simplified matters by permuting nighttime surface use while holding daytime surface use at 0.4, and assuming that nighttime use would always be greater than daytime use based on previously published observations. Therefore, in addition to the empirical ratio of 1.36:1 (0.57:0.42), the following five diel ratios of surface use were also investigated: 1:1 (0.4:0.4), 1.25:1 (0.5:0.4), 1.5:1 (0.6:0.4), 1.75:1 (0.7:0.4), and 2:1 (0.8:0.4).

Habitat Suitability

Monthly fin whale habitat suitability in each grid cell, scored from 0 to 1, was calculated using the prediction model presented in Scales et al. (2017), which is a high-resolution, multi-parameter Generalized Additive Mixed Model based on tracks of 67 tagged fin whales between 2008 and 2015 (n = 58 in Southern California, n = 9 off Washington State) and publicly available datasets of physiographic and dynamic environmental variables (**Table 1**). This model used the same grid cell resolution and study area as the present study (0.05 × 0.05° grid; 27,360 grid cells). Physiographic variables included seafloor depth and bathymetric rugosity. Dynamic oceanographic variables included seasonal thermal front frequency and monthly composites of sea surface temperature and chlorophyll-*a*. These predictors were included in the model on the basis of AIC corrected for small sample size. In this model, fin whale habitat use was established using filtered tag data weighted according to tag duration, to reduce bias associated with the location of deployment and uneven tracking durations, and low weights were applied to the first 10 days of tracking. The data sources, diagnostics, results, and discussion of this model are detailed in Scales et al. (2017).

Ship-Strike Risk

Strike risk was calculated as the product of habitat suitability and potential strike exposure, which itself is a product of ship traffic volume and fin whale surface use (**Table 1**). Because habitat suitability is a relative index without absolute units, ship-strike risk is also a relative measure without units. For each month of

each year, this calculation was carried out within every grid cell for its location-specific periods of day and night.

Shipping Lanes

To understand the relative contribution of ship traffic within heavily trafficked shipping lanes to overall CCS ship-strike risk, we compared strike risk factors (i.e., ship traffic, habitat suitability, potential strike exposure, ship speed) within lanes to their overall values for the entire study area on monthly and annual time scales. The approaches to shipping lanes, in which port-bound traffic is funneled into an incoming lane and outbound traffic disperses, also present areas of concentrated strike risk. To understand the diel patterns in ship traffic and strike risk within these approaches with better resolution, we analyzed grid cells as a function of their great-circle distance to the nearest edge of lane polygons.

Strike Expectation and Speed Reductions

We have defined strike risk as the inherent danger of collision posed to fin whales before their interaction with a nearby ship (**Table 1**). To evaluate the *expectation* of strike based on this risk, we must incorporate the probability of a fin whale and/or ship successfully avoiding collision, which is generally treated as a partial function of ship speed (Laist et al., 2001; Vanderlaan and Taggart, 2007; Conn and Silber, 2013). We examined the effects of ship speed on an annual timescale by calculating mean transit speed in each grid cell, defined as the total kilometers traveled by all ships divided by the total transit time for all ships. Individual ship speeds varied about this annual mean. While speed and many other variables would determine the exact avoidance probability of a specific whale–ship interaction, our purpose here is to assess the *relative* change in expected strike rate thanks to various reductions in the mean transit speed within a given area.

Fin whales have been observed responding to the presence of smaller vessels (Jahoda et al., 2003), but no further data exists on fin whale avoidance capability. To explore ship-strike expectation and the potential effects of speed reductions within our study area, we explored fin whale avoidance using a theoretical threshold probability function (**Figure 2A**), to mirror what was found in North Atlantic right whales in Kite-Powell et al. (2007) and used in Gende et al. (2011) and Rockwood et al. (2017) for fin whales:

$$P(Avoidance) = \frac{1}{1 + e^{-0.5(V-T)}}$$

In this framework the upper asymptote, which describes the maximum probability of avoidance, regardless of ship speed, is set to 1.0, and the maximum slope of the threshold response is set to −0.5 (the negative indicates the probability of avoidance declines with increasing ship speed). V is the ship speed, and T is the inflection point of the function, which we will refer to as the strike threshold: the ship speed at which a whale and a ship on collision course have a 50% chance of avoiding each other [i.e., T is where P(Avoidance) = P(Strike) = 0.50]. A lower strike threshold means that fin whales are less responsive to an oncoming ship; even if the ship is traveling slowly, it is still likely to strike the whale. In this framework, the probability of ship strike, P(Strike), is 1−P(Avoidance). Because no data exists regarding fin whale strike thresholds, we incorporated avoidance probabilities into strike expectation analyses using a set of avoidance models with various permutations of the strike threshold, ranging from 0 to 20 kn at intervals of 0.5 kn.

Annual Changes in Strike Expectation

We applied this set of theoretical avoidance models to compare mean strike expectation in 2011 and 2013. For each avoidance model, we calculated P(Avoidance) for each grid cell according to its annual mean ship speed. To do so, we calculated a set of P(Avoidance) for all avoidance models given the mean ship speed of each grid cell in the study area. We then used the P(Avoidance) in each cell to scale its mean strike risk, which for this exercise was calculated using the empirical surface use diel ratio of 1.36:1. The result for each year was a mean strike expectation for all grid cells for each avoidance model.

Nighttime Speed Reductions

A similar routine was implemented to assess the efficacy of speed reductions applied throughout the study area on a 24-h basis versus during nighttime hours only. In this model, we calculated the strike expectations across strike thresholds using the 24-h and nighttime means of ship speed, respectively, using 2013 as a case study since it is the most recent year of data we analyzed. We then reduced each cell's mean speed by 1 kn and re-calculated the overall strike expectation for the study area. We measured the efficacy of the speed reduction as the fraction of the original strike expectation represented by the new prediction. We repeated this for speed reductions of 2, 3, 4, and 5 kn.

Speed Reduction Buffers Around Shipping Lanes

We next used theoretical strike thresholds to estimate the effect of speed reductions within increasing radii from the edges of shipping lanes, again using 2013 as a case study year. To do so, we calculated the study area-wide strike expectation across strike thresholds. We then isolated the grid cells within various radii from the edges of the three lanes. We tested radii ranging from 0 km from lane edges to 150 km at intervals of 25 km. We applied speed reductions (1–5 kn) within those cells and re-calculated the area-wide strike expectation. We measured the efficacy of the speed reduction similar to above, as the fraction of the original strike expectation represented by the new prediction.

RESULTS

General Findings

We found that ship-strike risk is mediated by several factors, including the seasonal oscillation in nighttime duration, the relatively stable spatial distribution of ship traffic volume, higher rates of ship traffic at night (**Figure 3**), seasonal dynamics in fin whale habitat suitability, and diel patterns in fin whale surface use. The majority of strike risk occurs at night, even in summer when

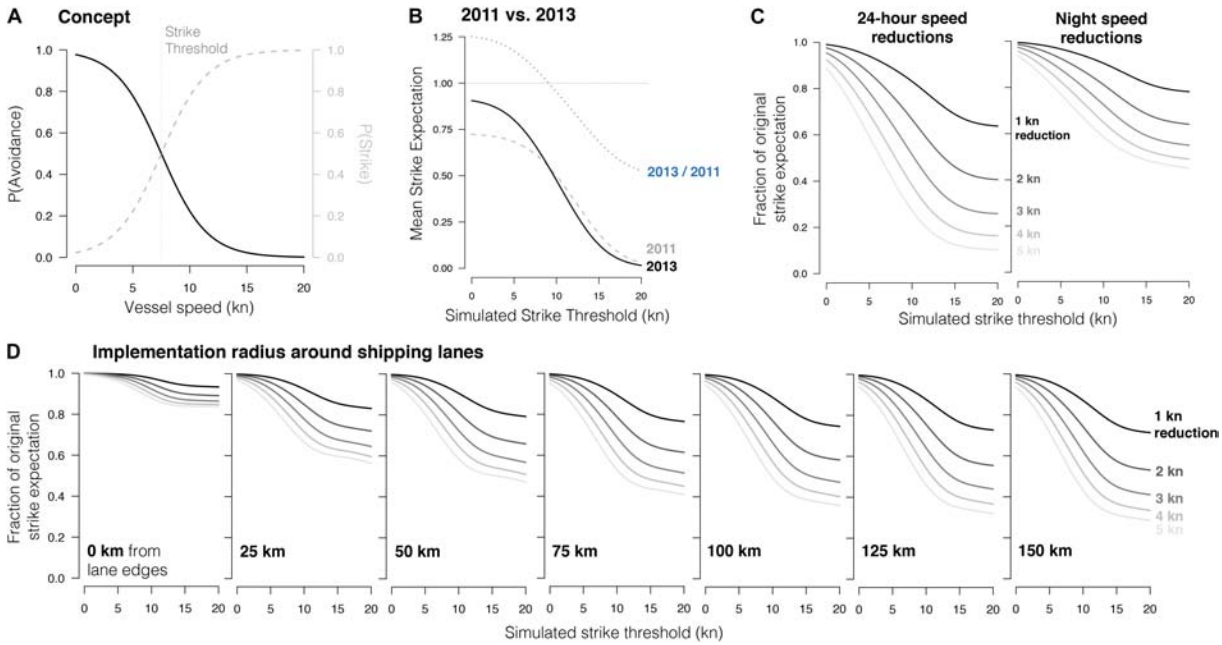

FIGURE 2 | Conceptual framework **(A)** and model results **(B–D)** of ship-strike avoidance probability as a threshold function of ship speed. **(A)** Concept demonstrated in which the strike threshold, i.e., the speed at which a fin whale has a 50% chance of either avoiding a ship [*P*(Avoidance), black line] or being struck [*P*(Strike), dashed line], is 7.5 kn. **(B)** Impact of strike threshold on mean strike expectation in 2011 (gray, dashed) and 2013 (black, solid). The ratio of those two curves (2013/2011) is provided in dashed blue. **(C)** Comparison of the effects of speed reductions when applied to all hours **(left)** vs. nighttime hours only **(right)**. The effect of the speed reduction (*y*-axis) is a function of the fin whale strike threshold (*x*-axis) and the mean reduction in ship speed (lines). The reduction effect is measured as the fraction of the original strike expectation presented by the new strike expectation after the reduction has been applied. **(D)** Expected reduction in strikes with the implementation of ship speed reductions within various buffers about shipping lanes, from 0 (far left, i.e., speed reductions are implemented within lanes only) to 150 km from the edges of lanes (far right). For all models shown here, potential strike exposure and risk were calculated using a 1.36:1 ratio of night:day use of the upper 20 m.

nights are short (**Figure 4**). This pattern is most pronounced in high-latitude winter when nights are longest, despite relatively low wintertime volumes of ship traffic. Periods of maximum strike risk varied seasonally across subregions (**Figure 5**). In shipping lanes, strike risk doubled at night and was highest in the winter. However, lanes comprised only 0.87% of the study area and 13% of total ship-strike risk (**Figure 6**).

Below we present further details, organized in decreasing order of geographic scope. All findings from all years are visualized in the **Supplementary Material**, including an atlas with maps of results for all stages of analysis for all months in all three years.

California Current System

Ships in the AIS dataset had an average reported length of 183.44 m, beam of 27.60 m, and draft of 8.39 m (**Supplementary Table S1**). Mean transit speed decreased between 2009 (12.79 kn) and 2011 (12.60 kn, 1% decrease), and again from 2011 to 2013 (11.03 kn, 12% decrease; two sample, one-sided *t*-test, df = 58,334, $p < 0.0001$; also verified with two-sample, one-sided Kolmogorov–Smirnov test, $p < 0.0001$) (**Supplementary Table S2** and **Supplementary Figure S1**). No strong patterns in the diel ratio of ship speed were evident in study area maps (**Supplementary Figure S1**), regional maps, or the approaches to shipping lanes (**Supplementary Figure S2**, showing 2013 only).

Total distance covered by ship traffic, in kilometers, increased by 2% from 2011 (27.3 million) to 2013 (27.9 million). Due to the decrease in transit speed between these years, traffic volume (transit hours) increased by 21% from 2011 (1.2 million) to 2013 (1.4 million) (**Supplementary Table S2** and **Figure 3**). The highest overall traffic volume occurred in late summer (July–September; **Figure 3**). Daytime traffic volume peaked in July, while nighttime traffic volume peaked in October (**Supplementary Figure S3**). The highest monthly diel ratio of traffic volume (1.9:1) occurred in December, while the lowest occurred in June (∼0.7, **Figure 3**). The diel ratio was greater than 1.0 for approximately half the year, from the end of summer to the beginning of spring. Both daytime and nighttime traffic rates, which scale traffic volume by the duration of diel periods (**Table 1**), were highest in August and lowest in January, for both day and night (**Supplementary Figure S4**). Nighttime rates were higher than daytime rates throughout the year, especially in winter months (diel ratio of ∼1.2).

Overall, potential strike exposure to ship traffic, which adjusts for diel modes of surface use (**Table 1**), was greatest from July to October and lowest in January and February (**Figure 4**). Exposure was greater at night in most months, and highest in October–November. Permutations of the diel ratio of surface use indicated that the highest diel ratios yielded the highest overall exposure, but the shape of the seasonal pattern was not changed. For diel

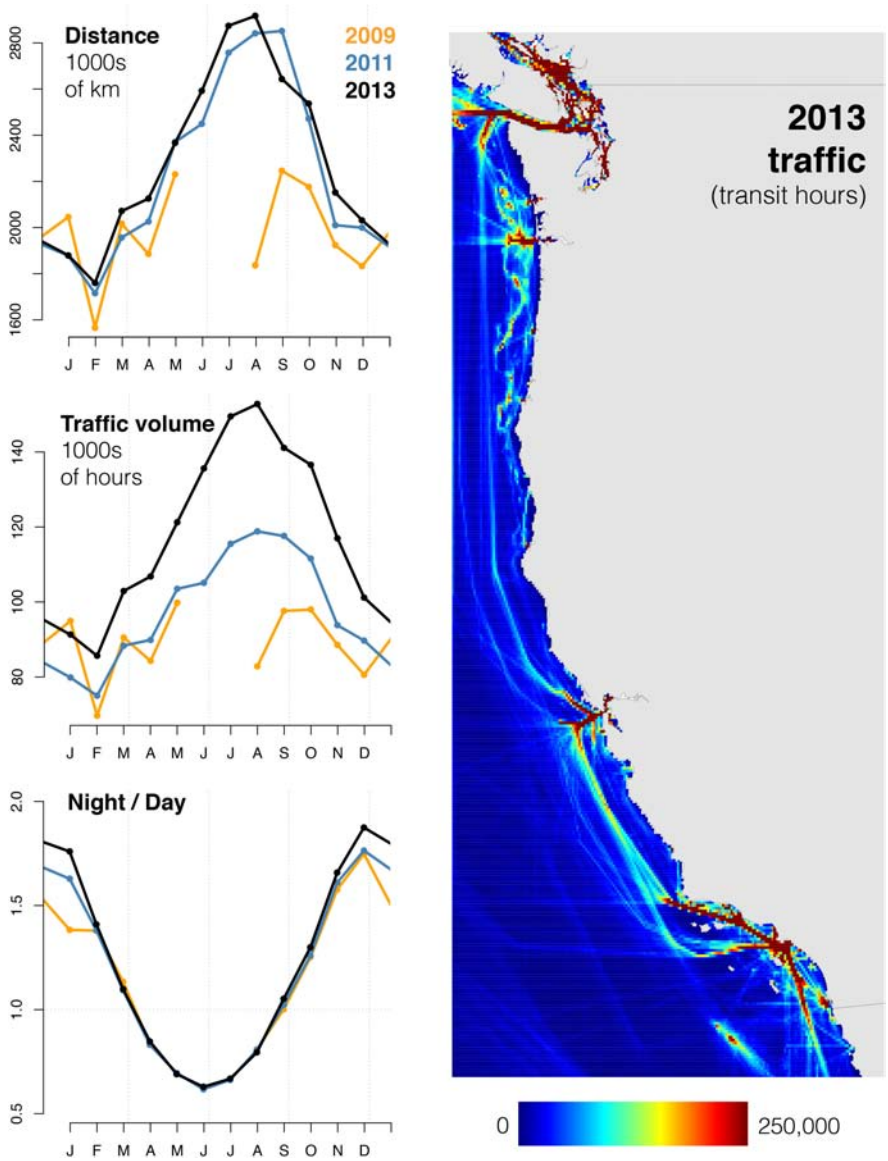

FIGURE 3 | Overall traffic volume across months in 2009 (orange), 2011 (blue), and 2013 (black), using three different metrics: distance traveled by all transmitting ships **(top left)**, traffic volume, i.e., transit hours logged by all transmitting ships **(center left)**, and the ratio of nighttime and daytime transit hours **(right)**. Map **(right)** displays total traffic volume for 2013.

ratios greater than 1.38, nighttime exposure was higher in every month of the year.

Using the empirical surface use diel ratio of 1.36:1, daytime exposure was highest in July and lowest in January. Nighttime exposure was highest in October and lowest in April–June. The diel ratio of overall exposure was highest in December–January (∼2.5) and lowest in May–July (∼0.95), indicating that exposure was greater at night for 9 months of the year.

In all 3 years examined, mean habitat suitability was greatest in July–September. The Southern California Bight and the Pacific Northwest consistently yielded the most suitable habitat in the study area (**Supplementary Figure S5**; see **Supplementary Atlas** for results from 2009 and monthly results for all years). Central Californian waters between Cape Conception and Cape Mendocino yielded low suitability in most months except July, August, and September. Complete results of the habitat suitability model are provided and discussed in Scales et al. (2017). Patterns in ship traffic, habitat suitability, and therefore ship-strike risk patterns were similar in 2011 and 2013. Mean ship-strike risk throughout the CCS was substantially higher in July–August, with local maxima in November and March (**Figure 4**; 2013 only). The highest diel ratios of surface use yielded the highest overall ship-strike risk. For all diel ratios of surface use that we tested, the majority of ship-strike risk throughout the year occurred at night. Ratios greater than 1.5:1 resulted in greater strike risk at night than during the day for every month of the year.

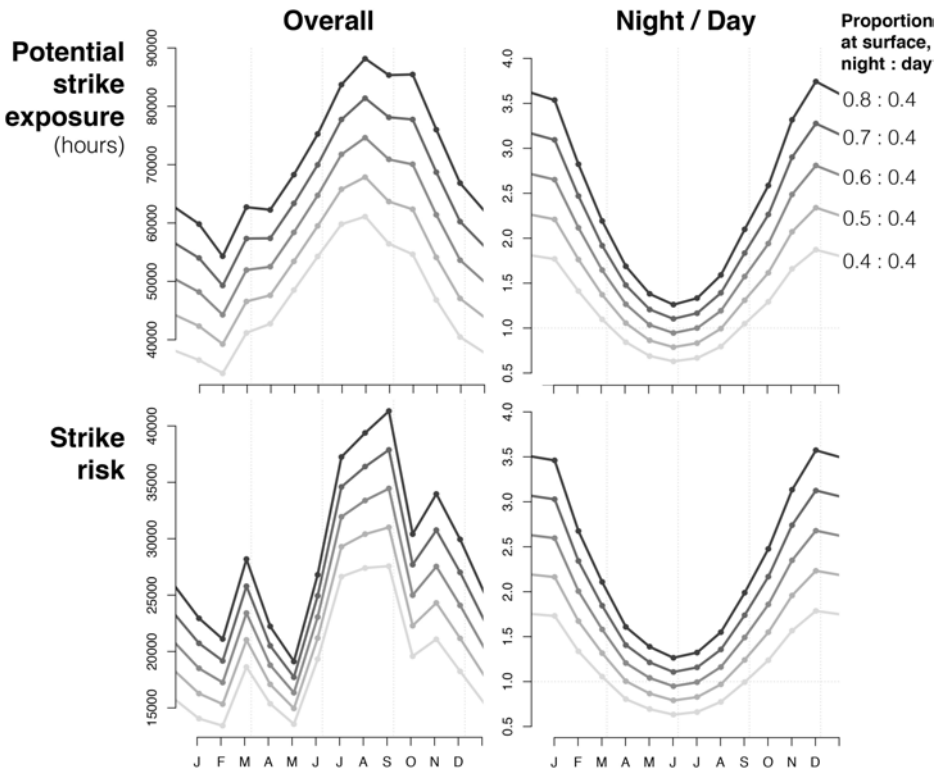

FIGURE 4 | Monthly patterns of 2013 potential ship-strike exposure (i.e., traffic volume scaled by night/day proportion of time whales spend at surface) and strike risk (i.e., strike exposure scaled by habitat suitability) in 2013, with various scenarios for the proportion of time that fin whales spend near the surface during night and day. "Overall" is the cumulative amount of exposure (in hours) and risk (no units) for each month; "Night/Day" is the ratio of cumulative exposure/risk at night to that during day (values above 1.0 = exposure/risk is greater at night).

Subregional Patterns

A closer look at the subregions with the most traffic (the Southern California Bight, San Francisco Bay Area, and Pacific Northwest) demonstrated the influence of latitude and habitat suitability on seasonal patterns in strike risk (**Figure 5** and **Supplementary Figure S6**). In all subregions, traffic rates (transit hours per hour of diel period) were higher at night all year long. The Southern California Bight experienced heavier traffic volume from July to November, while the San Francisco Bay Area experienced less seasonal variability (**Figure 5**), but was lower all year than both the Southern California Bight and the Pacific Northwest. Due to its higher latitudes, Juan de Fuca traffic underwent dramatic seasonal changes in diel ratios for traffic volume (>2.0:1 in winter, nearly 0.5:1 in summer).

Potential strike exposure was higher at night nearly year-round in the Southern California Bight and San Francisco Bay Area (**Figure 5**). In the Pacific Northwest, exposure was higher during day for May, June, and July, the longest daytime periods of the year within the study area. In all 3 years, mean habitat suitability in the Southern California Bight was highest in July, October, and November and lowest in February and March (**Figure 5**). In the San Francisco Bay Area, habitat suitability was relatively low in most months but increased abruptly in July–October. In the Pacific Northwest, habitat suitability was highest in December–March and August–September, with consistent drops in May and October. These patterns interacted to yield the greatest strike risk from July to November in the Southern California Bight, July to September in the Bay Area, and December–March and July–September in the Pacific Northwest.

Shipping Lanes

From 2011 to 2013, within the three shipping lanes (Los Angeles, San Francisco, and Juan de Fuca), ship traffic increased by 12% in terms of distance traveled (3.8−4.3 million km) and by 22% in terms of traffic volume (0.17−0.20 million transit hours) (**Supplementary Table S2**).

The three shipping lanes were found to constitute 0.87% of the study area (Los Angeles: 0.27%; San Francisco: 0.18%; Juan de Fuca: 0.42%; **Supplementary Table S3**). Given the similarities in traffic patterns for 2011 and 2013 the following shipping lane analyses were conducted using 2013 as a case study. Mean habitat suitability within Los Angeles and Juan de Fuca lanes was approximately 50% higher than the San Francisco lane. Habitat suitability within the Los Angeles lane was highest in late fall and early winter (**Supplementary Figure S7**). In the San Francisco lane, suitability was high during July–September only. In the Juan de Fuca lane, suitability was highest in winter.

Ships in the shipping lanes contributed 14% of overall traffic volume (Los Angeles: 3%; San Francisco: 3%; Juan de Fuca: 7%) (**Supplementary Table S3**). When data from all lanes were

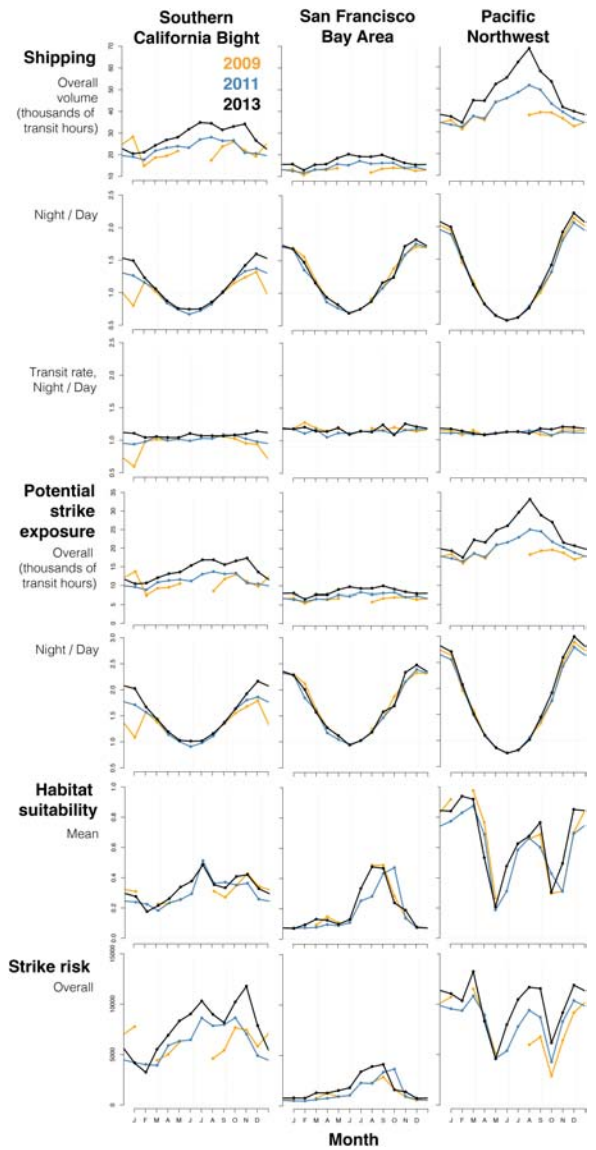

FIGURE 5 | Monthly patterns in ship traffic, potential ship-strike exposure, habitat suitability, and strike risk in three of the busiest subregions of the California Current System. Note that results have not been scaled by the area of the subregion; only differences in seasonal patterns should be compared. Here, exposure and risk were calculated using a 1.36:1 diel ratio of night:day use of the upper 20 m.

combined, these percentages remain roughly the same for day and night. The San Francisco lanes host a higher density of traffic (18.89 transit h km^{-1}) than Los Angeles (11.66 transit h km^{-1}) and Juan de Fuca (17.89 transit h km^{-1}). Traffic density increased at night in San Francisco (to 21.63 transit h km^{-1}), while it decreased in the other lanes. Although the diel ratio of traffic volume was close to 1.0 overall (0.98 in Los Angeles and 0.96 in Juan de Fuca), it was much higher in the San Francisco lane (1.37). Monthly patterns in traffic volume within lanes (**Supplementary Figure S7**) mirrored the pattern described above for the entire study area.

When using the empirical diel ratio of surface use of 1:36:1, shipping lanes contributed 14% of overall ship-strike exposure. The diel ratio of potential strike exposure was a mean of 1.43 across all lanes, highest in San Francisco (1.87), and lowest in Juan de Fuca (1.31). Using this diel ratio, traffic within lanes contributed 13% of overall ship-strike risk (Los Angeles: 5%; San Francisco: 2%; Juan de Fuca: 6%) (**Supplementary Table S4**). The highest 24-h density of strike risk occurred in the Los Angeles lanes, but the highest nighttime strike-risk density occurred in the San Francisco lane. Across all lanes, the risk of ship strike was nearly doubled at night (diel ratio of 1.94:1; Los Angeles = 1.80:1; San Francisco = 2.53:1; Juan de Fuca = 1.78:1). The seasonal pattern in the diel ratio of strike risk (highest in winter and lowest in summer) was present in all three lanes (**Supplementary Figure S7**).

Approximately 50% of traffic volume and strike risk in the study area occurred within 50 km of shipping lane boundaries, where ships are either queuing up to enter the lanes or fanning out as they exit (**Figure 6**). More than 65% of overall traffic volume and strike risk occurred within 100 km of shipping lanes (<25% of the study area).

Patterns in strike risk factors differed within the approaches to the three shipping lanes (0–200 km away) (**Supplementary Figure S8**). Overall speed declined in the approaches to the San Francisco and Los Angeles lanes, but there was no such change in the Juan de Fuca approach (**Supplementary Figure S2**). Habitat suitability decreased with increasing distance from the three lanes, indicating that lanes were located in potential high-use areas for fin whales. In all three approaches, traffic volume and strike risk increased dramatically within 25 km of the lanes. In the Los Angeles approach, most traffic volume occurred at night; the diel ratio was particularly high from 60 to 150 km out from the lane, with a prominent peak in nighttime traffic volume at approximately 110 km. Within 150 km of the San Francisco lane, most traffic volume occurred during day with the exception of the final 25 km. In the Juan de Fuca approach, traffic from 60 to 140 km occurred mostly during day.

When data from all months in 2013 were combined, strike risk was considerably higher at night throughout the approaches to all three lanes (diel ratio > 1.5) (**Supplementary Figure S8**). In the Los Angeles approach, strike risk was highest from 75 to 125 km away and peaks 110 km away (diel ratio = 2.6). In the San Francisco approach, strike risk was highest in the immediate vicinity of the lane (diel ratio = 2.1) but increased again beyond 150 km away. In the Juan de Fuca approach, strike risk was highest within 50 km of the lane (diel ratio varied between 1.7 and 2.1) with no clear pattern further out.

Strike Expectation and Speed Reductions

Our overall finding was that, given patterns of ship transit speed in the CCS, an avoidance response would only substantially impact overall ship strike rates if the probability of strike changed dramatically at speeds between 7 and 13 kn. The effectiveness of speed reductions therefore depends upon this strike threshold. Despite the diel patterns in strike risk reported

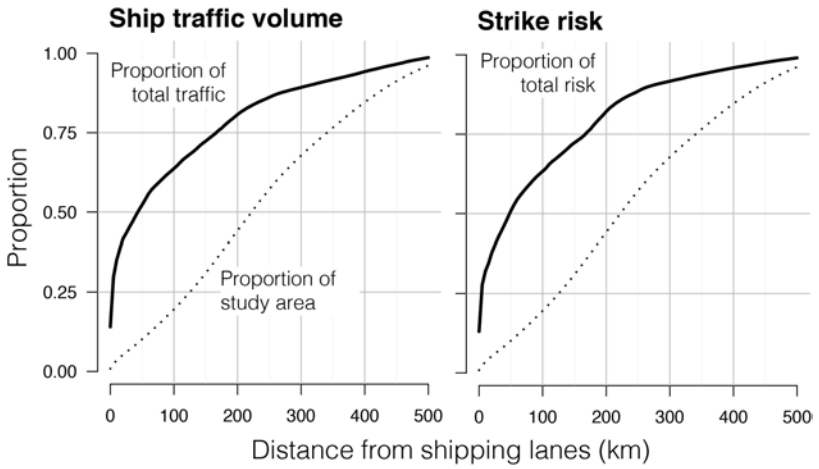

FIGURE 6 | The proportion of overall traffic volume and ship-strike risk occurring within increasing radii of shipping lanes in the California Current System. The dotted line indicates the proportion of the study area occurring within a certain distance of lanes. The position of the black line at that same distance indicates the proportion of the study area-wide sum total of the variable occurring within that area. For example: ∼70% of ship traffic volume occurs within 100 km of lanes, despite representing only ∼25% of the study area. Risk was calculated using the empirical diel ratio (1.36:1) of night:day use of the upper 20 m.

above, 24-h reductions in speed would still be twice as effective as nighttime-only speed reductions (**Figure 2C**). If such reductions can only occur within a limited area, they would reduce ship-strike expectation most effectively when applied around shipping lanes in a 25–50 km buffer (**Figure 2D**). Further details are provided below.

Annual Changes in Strike Expectation

In both 2011 and 2013, avoidance models indicated that ship-strike expectation increased when strike thresholds were lower (i.e., when fin whales were less likely to avoid slower ships; **Figure 2B**). The steepest increases in strike expectation occurred between strike thresholds of 7 and 13 kn, indicating that patterns of fin whale response to ships in this speed range will have the greatest impact on overall strike rates. Changes in mean ship speed within this range would also affect strike rates. Due to the decline in mean speed between years (12% decrease from 12.6 kn in 2011; **Supplementary Table S2** and **Supplementary Figure S1**), the exact location of the strike threshold determined whether ship-strike expectation increased or decreased from 2011 to 2013. If the fin whale strike threshold is below 9 kn, then strike expectation was much higher in 2013 in both absolute and proportional terms (**Figure 2B**). If the threshold is above 9 kn, strike expectation was higher in 2011. However, since overall risk was low in high strike threshold scenarios, the difference between the 2 years was negligible in absolute terms but proportionally substantial.

Nighttime Speed Reductions

Strike threshold models in our 2013 case study demonstrated the intuitive results that (i) the impact of speed reductions on strike expectation depended on the strike threshold of fin whales, (ii) that strike expectation lowered with greater speed reductions, and (iii) 24-h speed reductions yielded lower strike expectation than nighttime reductions (**Figure 2C**). For example, if the fin whale strike threshold was at 10 kn, a 24-h speed reduction of 1 kn would reduce strike expectation by approximately 15%. Doubling the speed reduction to 2 kn would also double the effect (∼30% reduction in strike expectation), but reductions of 3 kn or greater yielded diminishing returns. These threshold avoidance models demonstrated that speed reductions would have a substantial impact only if avoidance probability was a steep function of ship speed within only a certain and limited range of ship speed, 7–13 kn (**Figure 2C**).

In our simplified framework in which the strike threshold was the same both day and night, the efficacy of speed reductions at night was approximately half that of 24-h reductions (**Figure 2C**). At a strike threshold of 10 kn, for example, a nighttime-only reduction of 1 kn reduced overall strike expectation by <10%. Looking across all strike thresholds, the same drop in strike expectation could be achieved with a 24-h speed reduction of 1 kn or a nighttime-only reduction of 2.5−3.0 kn. In order to match the same effect of a 24-h speed reduction of 2 kn, nighttime reductions would have to be >5 kn.

Speed Reduction Buffers Around Shipping Lanes

Since shipping lanes contained only 14% of west coast traffic volume in our 2013 case study (**Supplementary Table S3**) and surrounding waters contained high volumes as well (for example, waters within 50 km of lanes contained 50% of overall traffic volume; **Figure 6**), the greatest reduction in strike expectation might be achieved with the strategic application of speed reductions within a spatial buffer surrounding lanes. Strike threshold models were used to explore what the most effective spatial buffer might be under various avoidance scenarios (**Figure 2D**).

Buffers of 25 and 50 km yielded the most dramatic reductions in strike expectation across strike threshold and speed reduction scenarios, with diminishing returns using greater buffer radii. For example, assuming a strike threshold of 10 kn, a 2-kn speed

reduction applied within a 50 km buffer would yield a 20% drop in area-wide strike expectation; the same speed reduction applied strictly within lanes would reduce strike expectation by <5%.

DISCUSSION

Collision with ships is considered the most pressing conservation issue for the endangered fin whales in U.S. waters (National Marine Fisheries Service [NMFS], 2010; Carretta et al., 2018). To understand the scope of this issue and address it strategically, the behavioral ecology of fin whales must be considered when assessing strike risk and evaluating potential mitigation measures. Current estimates suggest fin whale strike mortality rates alone are 2.7× the Potential Biological Removal limit for non-natural mortality (Rockwood et al., 2017), and these estimates did not account for increased surface use at night, which our findings suggest is when collision risk is highest. The improved strike risk estimates presented here enable us to highlight priority management areas, compare possible mitigation strategies within those areas, and identify future research priorities.

Our findings highlight the interacting factors governing strike risk, which influence the efficacy of management strategies. At night, when fin whales tend to remain near the surface, ship traffic rates are slightly higher than during the day. Traffic volume also exhibits strong diel patterns in the approaches to shipping lanes, perhaps due to the availability of on-shore labor. These patterns exacerbate risk during winter when nights are longest, particularly in higher latitudes. The distribution and extent of suitable habitat shift on a monthly basis underneath the relatively stable geography of shipping, and the alignment of all of these patterns within a given subregion determines the seasonality of ship-strike risk for fin whales. Throughout the CCS, however, total strike risk was generally highest at night all year long, even during long summer days. Within coastal shipping lanes, total annual nighttime strike risk was twice the daytime risk. In reality, this nighttime risk is likely compounded by reduced likelihoods of visual detection and avoidance on the part of the ship, and perhaps also the whale.

Comparable Studies and Future Directions

The geography of strike risk we found differs from the predicted mortality distributions in Rockwood et al. (2017), particularly in the Pacific Northwest and the waters of central California from Pt. Conception to Cape Mendocino, where our assessment of annual overall risk is comparatively low. The differences can be attributed primarily to our different data sources for fin whale distribution. We selected a habitat suitability model developed in part from tag data, which is susceptible to bias due to deployment location and inherently small sample sizes (Scales et al., 2017), because it was the only year-round dataset available for the entire west coast. Rockwood et al. (2017) used data from line-transect surveys conducted by the National Marine Fisheries Service Southwest Fisheries Science Center (NMFS SWFSC), which are systematically distributed in space but not in time; they are conducted between July and December and their spatial effort in each month depends on logistical constraints and weather (Becker et al., 2016). Interestingly, our monthly analyses indicate high levels of habitat suitability and strike risk for waters off central California in July through September, but low levels for other months. If we had based habitat suitability models only upon tag and environmental data from late summer and early fall (Scales et al., 2017), our findings would align better with Rockwood et al. (2017). The discrepancy in the Pacific Northwest may also be a result of our habitat suitability model's extrapolation for this data-poor area (see the section "Materials and Methods" as well as discussion in the section "Limitations").

The same NMFS SWFSC data were used in a ship-strike risk study for fin whales in the Southern California Bight (Redfern et al., 2013). Again, the fin whale distributions used in that model, which indicated higher densities offshore in the northern Southern California Bight, were similar to our habitat suitability distributions for summer and fall. In all other months, however, we found consistently high habitat suitability throughout the Southern California Bight, particularly inshore and in southern portions. Ultimately, ensemble approaches that examine multiple datasets, multiple modeling techniques, and ideally multiple species (e.g., Redfern et al., 2013) will be critical to understand variability in the system and uncertainty in model predictions.

Our strike risk estimates for the offshore waters of the Pacific Northwest subregion concurred in some respects with those in Nichol et al. (2017), in which fin whale distribution was based upon four-season aerial line-transect surveys off southwest Vancouver Island. In that study, the highest fin whale densities were observed offshore and above the continental slope, further out than the most suitable habitat predicted by Scales et al. (2017) and the present study (**Supplementary Figure S5**). Fin whales were found within the easternmost extent of their surveys in the Strait of Juan de Fuca, where heavy shipping traffic yielded high strike risk despite the low fin whale densities observed. Nichol et al. (2017) also modeled strike lethality for this region based on ship type and speed. They observed that the offshore fin whale distribution overlapped a region with faster ship traffic, which resulted in another high-risk area west of the shelf break.

The area in which our habitat suitability models diverge most dramatically from current knowledge, however, is within the interior waters between Vancouver Island and the mainland, where fin whale encounters are rare (Ford, 2014; Towers et al., 2018). At a minimum, the habitat suitability and strike risk we predicted highlight this area's potential viability as future habitat for the recovering fin whale population. Given the density of ship traffic, the long hours of darkness during winter, and the present number of ship strikes reported in the region (Towers et al., 2018), our results further suggest that ship-strike risk here may be considerable, especially in winter. We recommend increased survey and tagging effort in these waters, particularly in winter months.

Our findings suggest that fin whale surface use and ship avoidance are key determinants of strike probabilities, but our knowledge of both is limited. Collecting data to inform these parameters, for both fin whales and other recovering baleen whale species, is a clear next step, particularly in areas where elevated ship traffic and whale habitat suitability overlap. Additional

winter and spring occurrence data from throughout the CCS would also improve risk estimates for these months, when surface zone use may be highest (Keen et al., 2019). Research focused on the traffic patterns of, and whale responses to, smaller vessels would allow us to assess another potential source of strike risk. The National Marine Fisheries Service ship strike database, which we did access for this study, could be used to assess current knowledge of strikes involving small vessels.

Limitations

Our analyses were based on several simplifications and assumptions, which were necessary due to (i) the computational burden of processing AIS ship traffic data, and (ii) the paucity of data available on fin whale behavioral responses to ships. We assumed that AIS data are accurate and adequately represent the maritime traffic that poses the greatest risks to fin whales, but military traffic and many small watercraft are often excluded from AIS, so strike risk is likely greater than we determined here.

As stated previously, our models were also simplified to assume that all AIS-transmitting ships, whether 65 or 1,000 ft in length, represent the same risk to a fin whale. We did not categorize ship traffic or parse strike risk according to ship type, hull draft, or speed. Instead, we simplified models by treating the upper 20 m of the water column as the zone of risk (2.4× the mean hull draft in our AIS dataset) because that was the surface use boundary used in the most relevant diel dive behavior studies (Keen et al., 2019). In reality, the radius of a ship's hydrodynamic draw, which could pull whales toward the ship's hull as it passes close by, increases with both hull draft and transit speed (Silber et al., 2010). McKenna et al. (2015) assumed a zone of hydrodynamic risk of 2× the hull draft. Rockwood et al. (2017) compared whale mortality rates based on strike zones of 1× and 2× ship draft; doubling the strike zone increased whale mortality rates by 17–37%.

We also assumed that diel patterns in fin whale surface use were constant throughout the study area, across all months, and regardless of local habitat conditions, when in actuality behavior was likely more nuanced. Currently fin whale behavior while at the surface at night is not well understood. Depth-sensor tags in Keen et al. (2019) demonstrated that deep dives essentially ceased after nightfall, but behaviors within the surface zone could not be resolved with the tag technology in use. A multi-sensor tag deployed on a fin whale off California in Friedlaender et al. (2015) also recorded that deep dives ceased after dark, and the tag accelerometer provided no indication of feeding behavior at the surface. Analysis of the diel horizontal movements of the animals in the present study are forthcoming and may provide additional insight into behavior at night.

In their analysis of tag data from central and southern California, Keen et al. (2019) suggest that the diel ratio of surface use is highest in winter and spring, which would further compound the seasonal risk patterns we present here. We recommend further study into the seasonal and geographic variation of surface use, and the underlying drivers thereof, particularly in year-round high-use areas such as the Southern California Bight (Scales et al., 2017). The seasonal migratory movements of California–Oregon–Washington and Pacific Canada fin whales are not well understood, but are thought to diverge from the canonical migratory behaviors of other sympatric baleen whales (i.e., breeding in low latitude regions in winter, feeding in high-latitude regions in summer) (Ford, 2014; Scales et al., 2017). Extended occupancy within high-risk sub-regions would also increase ship-strike risk for some fin whales in and around the CCS.

Within our avoidance models we assumed that the strike probability function was the same both day and night, when in reality differences are likely created by altered behavioral states as well as compromised detectability for both whale and ship. Additionally, in predicting expected strike rates, our primary concern was the eventuality of collisions, not their lethality. The number of uncertainties in play make it difficult to extrapolate beyond an assessment of strike risk to a prediction of strike rates or mortality rates, which would be necessary in order to assess the population-level effects of the diel patterns presented here. We used the above assumptions to simplify our analysis, reserving theoretical computations for the stages involving the most uncertainty regarding fin whale behavior: the proportion of time spent at the surface and the avoidance response to ships given their transit speed. Avoidance response models allowed us to move beyond strike risk to scenarios of strike expectation and the potential efficacy of various ship speed reduction measures. In our analysis we prioritized avoidance modeling over predictions of lethality, which is also expected to be a function of ship speed (Conn and Silber, 2013), for three reasons: first, speed-dependent avoidance rates would have been needed in order to then estimate mortality rates; second, we consider it probable that avoidance rates likely reflect lethality patterns in most cases, since any ship that a fin whale fails to avoid is likely moving quickly enough to cause serious injury, if not mortality; and third, the uncertainties and assumptions involved in a lethality analysis would outweigh its usefulness in a management context. Data pertaining to the relationship between ship type, draft, speed, and strike lethality would be invaluable in these matters, and further encouragement of voluntary reporting by ships could be one means of addressing these knowledge gaps.

Several studies have treated whale avoidance response as a non-linear function of ship speed (e.g., Kite-Powell et al., 2007; Gende et al., 2011), while elsewhere it has been treated as linear (Conn and Silber, 2013). In the non-linear framework we used, speed reductions would measurably ameliorate strike rates only if the baseline speed in question is within a certain range of the whale's avoidance threshold. For example, a 5-kn speed reduction from 25 to 20 kn may not improve a fin whale's chances of avoidance, but the same reduction from 12 to 7 kn may help greatly. In fact, the former scenario may lead to an *increase* in strike expectation, since the drop in speed to 20 kn increases the temporal overlap of whales and ships in an area without reducing their probability of avoidance. A compounding factor here is the lethality of strikes, which is expected to diminish once low speeds are reached (Gende et al., 2011). Our models demonstrated (1) that the actual location of the fin whale strike threshold determines whether strike expectation increased or decreased from 2011 to 2013 (**Figure 2B**), and (2) that the efficacy of speed reductions hinges upon the location of the strike threshold for fin

whales (**Figures 2C,D**). It should be noted that while we treated strike thresholds as a stable value in our models, the ability of fin whales to detect and avoid ships is likely dependent upon their behavioral state, light conditions, acoustic cues, previous experience with ships, and possibly many other dynamic factors.

In the Pacific Northwest, the high strike-risk prediction must be interpreted within the context of conflicting considerations. The habitat suitability models used were based on tags deployed primarily in California waters, with only a few tag deployments in the Pacific Northwest (Scales et al., 2017). Boat-based line-transect surveys in the U.S. waters of Washington and Oregon have yielded low fin whale densities (Becker et al., 2016); however, effort was limited in winter, when habitat suitability was predicted by the Scales et al. (2017) model to be highest. Acoustic detections of fin whales peak during winter and spring (Oleson and Hildebrand, 2012), lending support to the habitat predictions our results are based upon.

Ship-Strike Mitigation

Our analyses revealed that the coast's shipping lanes contained 14% of traffic volume and contributed 13% of all strike risk, which confirm the conclusion in Rockwood et al. (2017) that mitigation measures enforced only within lanes would address just a small fraction of the CCS ship-strike problem. Modifications to lane placement, which may locally reduce fin whale strikes (although additional species with different distributions also need to be considered; Redfern et al., 2013), would also only partially alleviate a portion of the strike risk.

Given that the implementation of speed reductions is unlikely to be feasible at the scale of the entire CCS, we explored alternative solutions by scaling the application of speed reductions in space and time. Our avoidance models suggested that 24-h speed restrictions applied around and within lanes would be more effective and feasible than nighttime restrictions implemented everywhere. For example, a 2-kn speed reduction within 50 km of lanes would reduce CCS strike expectation by 20–30%, depending on strike threshold (8–13 kn, respectively; **Figure 2D**). To achieve the same strike reduction using nighttime-only restrictions, mean nighttime ship speed throughout the CCS would have to decrease by a minimum of 3 kn (**Figure 2C**). Our speed reduction buffer models could be helpful in future considerations of adjustments/extensions to shipping lanes off the California coast, and an additional measure of conservation and ship-strike risk reduction or impact could be achieved by correlating lane adjustments with ship speed reductions.

Monthly variation in strike risk suggests seasonal mitigation within some subregions of the CCS might be an effective strategy. These monthly changes were driven by shifting habitat suitability, latitudinal and seasonal variation in hours of darkness, and thus variation in total surface use by fin whales (**Figure 5** and **Supplementary Figure S6**). Our models suggest the San Francisco Bay Area harbors highly suitable fin whale habitat, and correspondingly high strike risk, for only a few months of the year (August–October). However, it is important to note that while seasonal mitigation may be effective in the case of fin whales, other at-risk species that use the area must also be considered. In Southern California, habitat suitability and ship-strike risk are relatively high for the majority of the year (**Figure 5** and **Supplementary Figure S6**), indicating that four-season mitigation policies would be most appropriate for this region.

Our avoidance models indicated that even if ship speed reduction policies are extreme, strike expectation could be reduced but not eliminated. Based on a reasonable strike avoidance threshold of 9–12 kn, a 3-kn reduction applied throughout the area would reduce the ship-strike expectation by roughly half. This would be a considerable step forward, but additional management would still be necessary to reduce mortality to within maximum sustainable levels (Rockwood et al., 2017). Speed reductions should therefore be considered an effective measure, one that also improves fuel efficiency, and reduces anthropogenic noise and emissions (Corbett and Fischbeck, 1997). However, a viable mitigation plan will also necessitate (i) considering mitigation measures beyond our study area, (ii) considering spatial management schemes such as the expansion of shipping lanes and Areas to Be Avoided, and (iii) reckoning with the economic infrastructure and the mentality of the consumer base that drive shipping levels to such great heights. However, the scale of a problem should never be used to warrant inaction, but rather to elevate urgency and resolve. Our hope is that the findings presented here will support effective solutions to the ship-strike problem in the CCS and elsewhere.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. These data can be found here: http://marinecadastre.gov/ais/.

AUTHOR CONTRIBUTIONS

All authors consulted on designing the analyses and edited the manuscript. KS conducted the habitat suitability modeling. BR and EK analyzed the AIS data. EK conducted all other analyses and wrote the first draft of the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00730/full#supplementary-material

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A Case Study of a Near Vessel Strike of a Blue Whale: Perceptual Cues and Fine-Scale Aspects of Behavioral Avoidance

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Despite efforts to aid recovery, Eastern North Pacific blue whales faces numerous anthropogenic threats. These include behavioral disturbances and noise interference with communication, but also direct physical harm – notably injury and mortality from ship strikes. Factors leading to ship strikes are poorly understood, with virtually nothing known about the cues available to blue whales from nearby vessels, behavioral responses during close encounters, or how these events may contribute to subsequent responses. At what distance and received levels (RLs) of noise whales respond to potential collisions is difficult to observe. A unique case study of a close passage between a commercial vessel and a blue whale off Southern California is presented here. This whale was being closely monitored as part of another experiment after two suction-cup archival tags providing acoustic, depth, kinematic, and location data were attached to the whale. The calibrated, high-resolution data provided an opportunity to examine the sensory information available to the whale and its response during the close encounter. Complementary data streams from the whale and ship enabled a precise calculation of the distance and acoustic cues recorded on the tag when the whale initiated a behavioral response and shortly after at the closest point of approach (CPA). Immediately before the CPA, the whale aborted its ascent and remained at a depth sufficient to avoid being struck for ∼3 min until the ship passed. In this encounter, the whale may have responded to a combination of cues associated with the close proximity of the vessel to avoid a collision. Long-term photo-identification records indicate that this whale has a long sighting history in the region, with evidence of previous ship encounters. Therefore, experiential factors may have facilitated the avoidance of a collision. In some instances these factors may not be available, which may make some blue whales particularly susceptible to deadly collisions, rendering efforts for ship-strike reduction even more challenging. The fine-scale information made available by the

integration of these methods and technologies demonstrates the capacity for detailed behavioral studies of blue whales and other highly mobile marine megafauna, which will contribute to more informed evaluation and mitigation strategies.

Keywords: ship strike, blue whale, near collision, active avoidance, behavioral response, perceptual cues

INTRODUCTION

Like most baleen whales, blue whales (*Balaenoptera musculus*) were greatly depleted by commercial whaling (Monnahan et al., 2014). Abundance estimates from mark-recapture data suggest no evidence of an increase in this population since the early 1990s (Calambokidis, 2013), with the population currently estimated at 1,647 individuals. With pre-whaling abundance estimates modeled at between 1,823 and 3,721 individuals, this has led some to the conclusion that blue whales had returned to carrying capacity (Monnahan et al., 2014). However, the coastal habitats where blue whales feed on euphausiid aggregations (Rice, 1974; Croll et al., 1998; Fiedler et al., 1998; Calambokidis et al., 2009, 2015) overlap with human activities. As a result, these whales are vulnerable to many anthropogenic threats, including ship strikes.

Ship-strikes off California have resulted in the death of at least nine blue whales from 2007 to 2011 (Berman-Kowalewski et al., 2010; Carretta et al., 2013), though this is an underestimate of the true number due to the small proportion of large whale mortality that is documented (Heyning and Dahlheim, 1990; Kraus et al., 2005; Williams et al., 2011). A recent model estimated a true mortality of 18 blue whales per year off the United States West Coast (Rockwood et al., 2017). That is nearly eight times greater than the potential biological removal limit (Carretta et al., 2011), defined under the United States Marine Mammal Protection Act of 1972 as the maximum number of animals, not including natural mortalities, that may be removed while allowing that stock to reach or maintain its optimum sustainable population. The factors leading to a ship strike are poorly understood, difficult to predict, and subsequently difficult to prevent. Despite mitigation efforts, including ship speed limits and adjustments to the size and location of the major shipping lanes (DeAngelis et al., 2010; McKenna et al., 2012a; Redfern et al., 2013), ship strikes continue, and questions remain about the role the behavioral response of the animal plays in ship-strike risk.

Previous research found that during nine close encounters with large commercial ships, blue whales did not respond by moving horizontally, but may have altered their diving behavior. These dives were only observed when ships were within a few hundreds of meters of the whales, a range that might not allow for much avoidance time (McKenna et al., 2015). Their constrained response time may result from external cues that are only detectable – or interpreted as a threat – at limited distances, making them vulnerable to ship strikes. The detectable perceptual cues (e.g., visual and acoustic) corresponding to the presence of close-range vessels that provoke these types of avoidance responses are unknown. It is hypothesized that blue whales use visual cues to identify prey patches on the surface (Goldbogen et al., 2013a; Friedlaender et al., 2017) and could conceivably use vision to identify a large ship. Although whales may be able to visually detect ships at or below the surface over short ranges and under ideal ambient light conditions, sound propagates much further in water than light, likely making sound the primary sensory cue for whales orienting to their surroundings. Blue whales are acoustically active animals (Oleson et al., 2007) and noise from commercial ships directly overlaps with their vocalization frequency range. These ships emit a significant amount of low-frequency underwater noise (<1,000 Hz), which poses additional threats to this endangered population (e.g., masking whale communication, increasing stress, and resulting in habituation to ship presence, potentially limiting avoidance responses and times) (McKenna et al., 2012b).

A unique incident involving a well-documented close passage between a large ship and a tagged blue whale arose during an experimental study of blue whale behavioral response to military sonar (see: Southall et al., 2019). Fine scale movement and acoustic data were collected, including estimated distances between the whale and ship, vessel noise received levels (RLs) on the tag, and three-dimensional fine-scale kinematic behavioral response. We use this unique event to gain insights into the various perceptual cues that may be used by whales to avoid ships, and to evaluate implications for ship strike risk.

MATERIALS AND METHODS

Data Collection

On September 13, 2014, a blue whale was dual tagged with a TDR10 tag (Wildlife Computers, Redmond, WA, United States) and a digital acoustic recording tag (DTAG-3; Woods Hole Oceanographic Institution, Woods Hole, MA, United States; Johnson and Tyack, 2003), in the Santa Barbara Channel (SBC) (33.66°N, 118.30°W). Both tags were simultaneously attached *via* suction cups in a single tagging approach at 0848 (local time henceforth). The animal was tagged as part of ongoing studies of whale behavior in shipping lanes (McKenna et al., 2015) and the Southern California Behavioral Response Study (SOCAL-BRS), a multi-year study of the response of different cetaceans to exposure of Navy sonar sounds conducted in the Southern California Bight (see Southall et al., 2019). As part of the SOCAL-BRS experiment, the animal was exposed to a 30-min experiment involving simulated mid-frequency (3–4 kHz) active sonar (MFAS), which ended 62 min prior to the close encounter with a large commercial ship.

A tagging boat (5.9 m rigid-hull inflatable boat; RHIB) was used to deploy the tags with a ∼5-m carbon fiber pole. The whale exhibited no visible reaction during tagging and resumed the behavior observed prior to tagging (i.e., consistent traveling). The animal was photographed and compared with known individuals in the Cascadia Research photograph identification

catalog database (Calambokidis et al., 2009, 2015). While a skin sample was collected *via* biopsy, the sex of the animal was identified as female from a previous biopsy of this individual. The tagged animal's positions were recorded during a focal follow in order to provide georeferenced positions for the pseudotrack generated from tag data (see section "Distance Calculations"). In the focal follow two vessels were involved in observing the tagged whale. The RHIB stayed 100–200 m away until the whale made its terminal dive, then slowly approached the location to record the exact dive position from the whale's footprint. A larger (22 m) vessel remained at distances of 362 to 2,750 m (on average 500–1,500 m) from the whale when it was at the surface and provided visual tracking support. Both vessels followed the methodology developed for the SOCAL-BRS experiment to ensure the presence of small boats would not impact behavior (see: Southall et al., 2012, 2016).

The DTAG-3 recorded dual-channel acoustics at a 240-kHz sampling rate, while pressure, temperature, and a tri-axial accelerometer and magnetometer were sampled at 250 Hz. The TDR10's pressure sensor recorded at 1 Hz and the FastGPS sensor took sub-second instantaneous satellite position snapshots when the tag emerged from the water during surfacings of the whale. Both tags were deployed with VHF transmitters used for locating the tagged whale and for tag recovery. The DTAG-3 remained attached to the animal for 5.7 h while the TDR10 remained attached for 15 hr. The data from the two tags were synchronized based on the timestamps.

Kinematic Analysis

The three-axis accelerometer and magnetometer data from the DTAG-3 were down-sampled to 5 Hz and corrected in MATLAB (Mathworks, Natick, MA, United States) so the axes aligned with the "whale frame" using periods of known orientation (Johnson and Tyack, 2003). Animal orientation (i.e., pitch, roll, and heading) was calculated using custom-written MATLAB scripts (Johnson and Tyack, 2003; Cade et al., 2016). Animal speed was determined from the root-mean-square (RMS) amplitude of flow noise from tag acoustics (Goldbogen et al., 2006; Simon et al., 2009). Lunges indicative of feeding were detected from the DTAG-3 data using a custom-developed lunge detection algorithm [similar to Allen et al. (2016)]. Depths recorded by the TDR10 pressure sensor were assessed in R (R Core Team, 2019) using the package "diveMove" (Luque, 2007) to determine the number of dives and maximum depth per dive performed by the tagged whale. Dives recorded only on the TDR10 were manually audited for the presence of vertical lunges as a coarse determination of presumed feeding. Dives were classified as lunge-feeding or non-lunge feeding based on the presence or absence of lunges during each dive. This gave us four generalized behavioral states for each dive.

Distance Calculations

Ship positions from the Automatic Identification System (AIS), the global ship tracking system used by vessel traffic services, were obtained for the period when the whale was tagged from an AIS receiver on Santa Cruz Island (33.995°N, 119.632°W). Whale surface locations were resolved from satellite position snapshots for surfacings detected on the TDR10's FastGPS sensor during which an adequate number of satellites (>4) were identified. We generated a georeferenced pseudotrack at 1 Hz sampling rate using the depth, pitch, speed, and known geographic reference points of the tagged animal (GPS positions from the TDR10 and focal follow positions) (Wilson et al., 2007). Ship positions were interpolated to 1-s intervals with the "ST_Line_Interpolate_Point" function in PostGIS assuming a constant speed and course over ground. The PostGIS "ST_Distance_Sphere" function was used to calculate horizontal distances from the tagged whale to every ship present in the AIS data. Three-dimensional straight-line distances were calculated as the hypotenuse of the horizontal and vertical distance between the ship and the whale and rounded to 10-m intervals. Horizontal distances were calculated as distance between the whale and the closest point to the ship after accounting for the location of the AIS transmitter on the ship and orientation relative to the whale. Vertical distances were calculated as the distance between the ship's reported draft and the whale's depth (determined from the TDR10's pressure sensor).

Acoustic Analysis

The acoustic data from the DTAG-3 were initially viewed as 60-s spectrograms calculated from 10 Hz to 120 kHz in MATLAB using Triton, custom-written software (Wiggins, 2003), to identify ship noise. To extract sound levels from the DTAG-3, the acoustic data were first decimated to 48 kHz, and the broadband (0 Hz–48 kHz) RMS received sound pressure levels (dB re 1 μPa) were calculated in 1-s intervals. Additionally, the power spectral density was calculated at a 1s-resolution and then summed over 1/3-octave band sound pressure levels (dB re 1 μPa) for bands with center frequencies ranging from 160 Hz to 20 kHz, using methods described in Merchant et al. (2015).

Noise generated from water flowing over the tag hydrophone (flow noise) can contribute to acoustic measurements of actual noise in the environment at frequencies up to 1 kHz. Flow noise highly correlates with whale swim speed and fluking (Goldbogen et al., 2006; Simon et al., 2009) and the noise tends to predominate at frequencies below 100 Hz (Fletcher et al., 1996). Therefore, this study excluded 1/3-octave bands below 140 Hz from the calculations of noise levels associated with the vessel. Flow noise above 140 Hz, to the extent it was present, was considered to be a relatively constant element of overall noise and included as part of the noise level calculations.

Controlled Exposure Experiment

As part of the SOCAL-BRS project, the animal was exposed to simulated MFAS from 1045 to 1115 PDT (local), during which a stationary experimental sound source (deployed from the M/V Truth) was positioned at ranges from ∼800 m to >2 km from the whale. Prior to the controlled exposure experiment (CEE), prey mapping with a calibrated multi-beam echosounder occurred from 0910 to 1008 [as in Friedlaender et al. (2016)]. From tag deployment, until the CEE began (117 min), the animal's baseline behavior was recorded during focal follow. After 30 min of MFAS exposure, post-exposure focal follow and prey mapping began, which ended at 1238. The animal was

feeding before, during, and after the CEE and while behavioral changes were identified as a result of the experiment CEEs (Southall et al., 2019), these were ephemeral in nature. The animal exhibited typical deep feeding dives for the 62 min-period following the CEE and prior to the vessel encounter. The DTAG-3 detached from the whale at 1416 and the TDR10 detached at 2346.

Photograph Identification

Based on the identification of the whale from matches in Cascadia Research's catalog and database, the animal was a known female that had been seen previously 23 times off the California coast in eight different years beginning in 1987. Most of the sightings were in the Southern California Bight in the vicinity of Palos Verdes Peninsula, a region near the shipping lanes leading to the port of Los Angeles/Long Beach and near where the animal was tagged in this study. The animal was also sighted off Pt. Reyes, California, a region near the northbound shipping lanes leaving San Francisco Bay. The animal was previously tagged during the 2011 SOCAL-BRS on August 3, 2011, however the tag remained attached for only 1 hr and therefore no playback experiment occurred. This whale was also sighted when the tagged whale and another whale were involved in the capsizing of a 23-foot private vessel off San Diego on July 2, 2014 (∼2 months prior to the encounter described here), after the boat approached the whales to take photographs. There were no reports of injury to the whales following the incident.

RESULTS

The R package "diveMove" detected 118 dives from the TDR10 pressure sensor data (**Figure 1**). The DTAG-3 pressure sensor captured the first 33 of these dives. Of the 118 dives detected, 12 were deep lunge-feeding dives, 35 were deep non-lunge feeding dives, 4 were shallow lunge-feeding dives, and 67 were shallow non-lunge feeding dives. At the onset of tagging, the whale was making a series of deep non-lunge feeding dives interspersed with lunge-feeding dives as she traveled southeast along the 200-m contour line (**Figure 1**). Two lunge-feeding periods were identified, one from 0910 to 1057, which occurred during the CEE and included 1 deep and 3 shallow lunge-feeding dives, and one from 1613 to 1930, which included 8 deep and 4 shallow lunge-feeding dives. Sunset occurred at 1854. From 1930, the onset of civil twilight, until the TDR10 tag detached at 2346, the dive record suggested a resting bout of 4 h and 15 min during which the whale stayed shallower than 35 m and no lunges were detected.

The TDR10 collected 122 resolvable GPS locations. Distance calculations between the ship and whale tracks revealed three instances where an underway ship was within 2 km of the tagged whale. The closest point of approach (CPA) between the Mokihana, a 263-m container ship traveling at 11.3 knots, and the tagged whale occurred at a horizontal distance of 93 m while the whale was at a depth of 67.5 m (**Figures 1**, **2**). The corrected horizontal distance from the AIS transmitter on the boat at the starboard side closest to the whale was 77 m and the corrected vertical distance between the whale and the reported draft of the ship (10 m) was 57.5 m. The 3D straight-line distance between the Mokihana and the tagged female blue whale was approximately 100 m. The other two ships passed at horizontal distances greater than 1.5 km from the whale and occurred after the MFAS CEE during the post-exposure focal follow and prey mapping.

Behavioral Response During CPA With Mokihana

Prior to the CPA with the Mokihana, the tagged whale was ascending from a deep non-lunge feeding dive (max depth = 277.5 m). The whale began to slow its ascent ∼90 s before

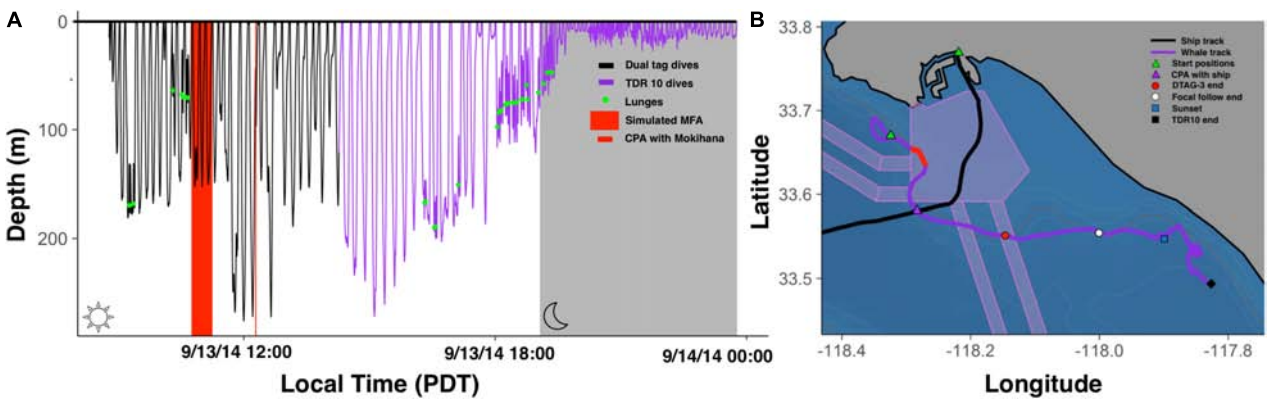

FIGURE 1 | (A) Dive record from DTAG-3 data (black) and TDR10 data (purple). Detected lunges are indicated by green circles. Red shading indicates simulated mid-frequency (3–4 kHz) active sonar playback as part of the SOCAL-BRS CEE. Red line indicates closest CPA with container ship Mokihana. **(B)** Horizontal tracks of Mokihana (black) and tagged whale (purple). Green triangles represent start positions for ship and whale. Red shading indicates period of the CEE. Purple triangle indicates the whale location during the CPA. Red circle indicates end of DTAG-3 recording, white circle indicates conclusion of focal follow, the blue square indicates sunset, and black square indicates end of TDR10 attachment. Shipping lanes are pink polygons and contour lines are represented in light gray from 50 to 500 m (in 50 m increments), with the 200-m contour in dark gray.

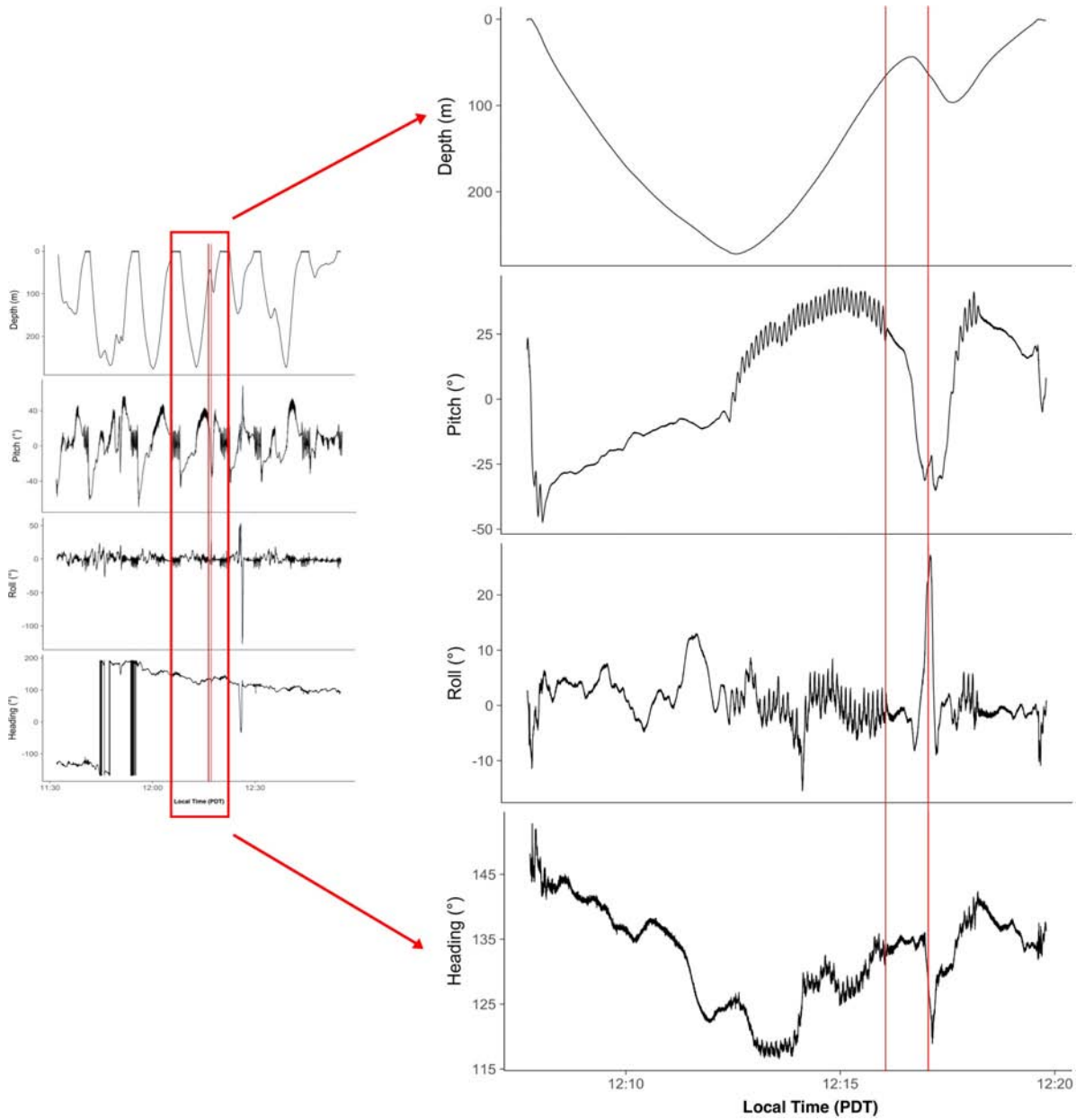

FIGURE 2 | Close up of fine scale kinematic data recorded during the CPA with the Mokihana. The first panel illustrates the depth (meters), the second panel shows pitch (degrees), the third panel illustrates roll (degrees), and the fourth panel illustrates heading (degrees). The first solid red line indicates the onset of a behavioral response by the whale. The second solid red line indicates the CPA with the Mokihana.

the CPA. Forty-seconds before the CPA, while the ship was at an approximate 3D straight line distance (hypotenuse between the ship and the whale) of 300 m from the whale, the tagged whale reversed into a descent. Kinematic data from the DTAG-3 shows a change in pitch, which corresponds to the switch to descent. The CPA occurred as the whale was at a depth of 57.5 m from the ship's draft. By this time the ship was approximately 100 m away from the whale at a 3D straight line distance. The data also indicate that the whale rolled to the left and changed its heading quickly at the CPA. The tagged whale resumed its ascent and surfaced after a ∼3-min delay from the previous projected surfacing time (**Figure 2**).

Before the close approach of the vessel, the broadband (RMS) ambient noise was generally ∼125–130 dB re 1 μPa (**Figures 3**, **4**). The overall ambient conditions in this environment were likely strongly influenced by aggregate vessel noise in the general area, including the Mokihana. However, as the ship approached, there was a rapid increase in the acoustic energy at higher frequencies (>1 kHz) with a typical spectral and temporal pattern associated with large vessels (McKenna et al., 2012b). The lower frequency

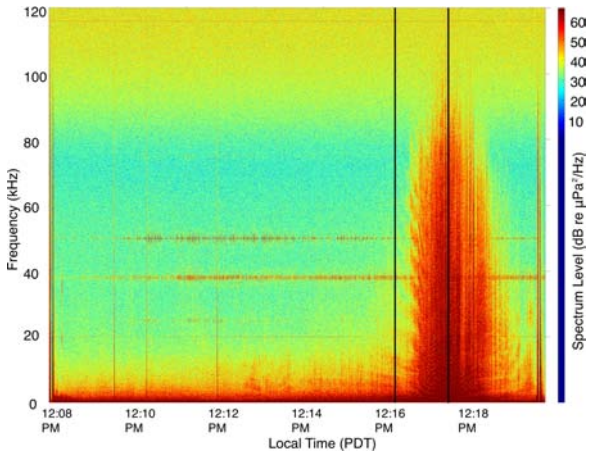

FIGURE 3 | Spectrogram showing the acoustic signal of the Mokihana during the CPA as recorded on the DTAG-3 hydrophone. The first black vertical line indicates the onset of a behavioral response by the whale. The second black vertical line indicates the CPA with the Mokihana. Spectrogram parameters: NFFT = 240000, 90% overlap, Hanning window.

bands (<1 kHz) exhibited an initial drop, associated with the cessation of fluking by the whale. These lower frequency bands then exhibited a rapid increase in levels, with no concurrent increase in fluking activity. The increase was instead associated with the passing of the ship within 100 m of the whale. The broadband sound level at CPA peaked at 135 dB re 1 μPa compared to ∼125 dB re 1 μPa at the last approximate point with similarly no fluking activity (**Figure 4**), representing a 10-dB increase over ambient broadband levels. Higher frequency (>1 kHz) 1/3-octave levels increased by up to 40 dB over pre-ship ambient levels. Additionally, as indicated in the noise spectra (**Figure 3**) and the broadband RMS RLs (**Figure 4**), there was a relatively abrupt change in the received sound levels around the point at which the whale initiated a change in behavior. There was a subsequent peak in the noise in all frequencies corresponding to the CPA of the Mokihana. The 1/3-octave band sound levels (**Figure 4**) indicate that the whale initiated a response dive when higher frequency (>1 kHz) RLs were only a few dB above ambient levels, just prior to reaching their maximum values. The whale only resurfaced after the vessel passed and was moving away, at which point RLs and the prevalence of higher frequency noise energy from the vessel were decreasing. The broadband RMS sound levels indicate a second peak after the passage of the ship, which corresponds to the resumption of fluking (evident in pitch, **Figure 2**) as the whale ascends. This peak in acoustic energy is only evident in the low frequency components of the 1/3-octave band levels, further indicating the second peak in broadband sound levels is due to increased flow noise associated with fluking.

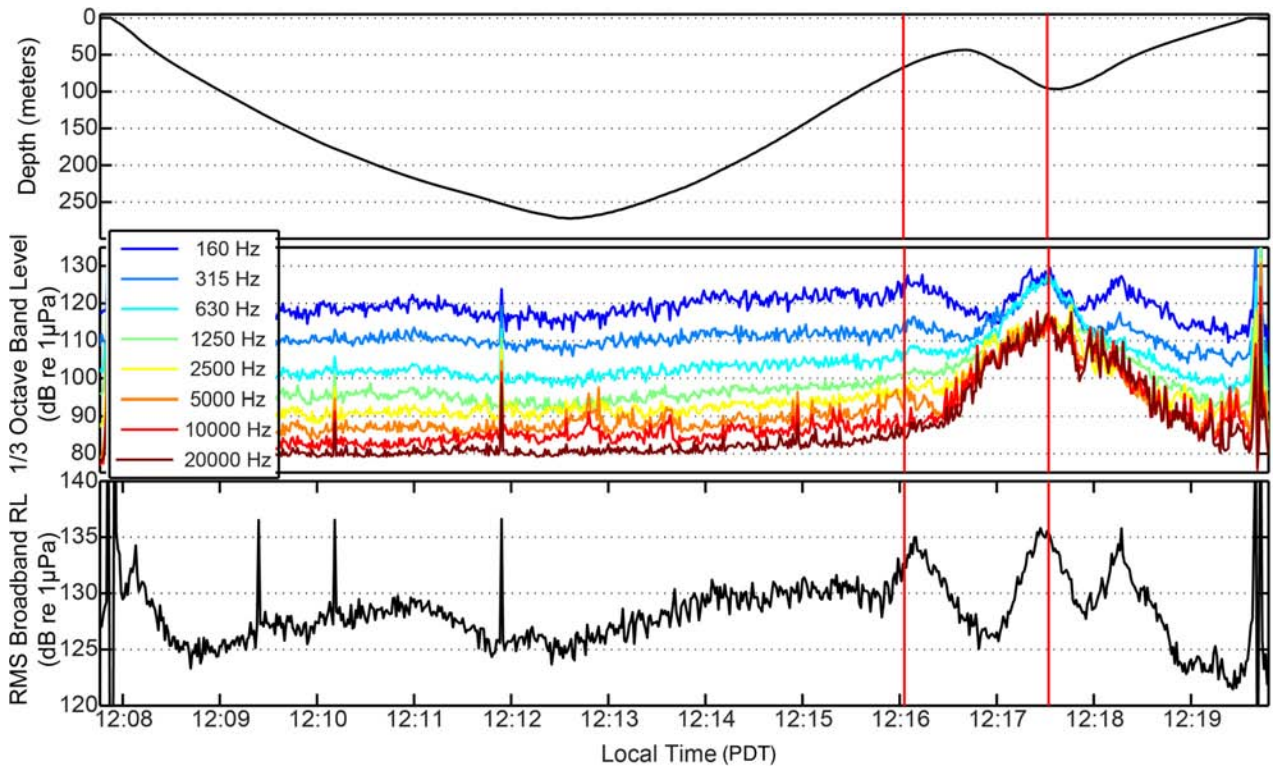

FIGURE 4 | Received levels (RLs) recorded on the DTAG-3 during the passage and CPA of Mokihana. Both 1/3-octave band sound levels **(middle panel)** and broadband (RMS) measurements **(bottom panel)** are shown. The **upper panel** shows the depth of the tagged whale during the same time period. The first red vertical line indicates the onset of a behavioral response by the whale. The second red vertical line indicates the CPA with the Mokihana.

DISCUSSION

The unique dataset from this case study provides a detailed account of the closest documented encounter between a large commercial vessel and a blue whale. The exact cues used to facilitate the successful avoidance in this close encounter case were unknown. However, contemporaneous data from multiple platforms (i.e., fine-scale kinematic, acoustic, movement, position, demographic, and long-term sighting history data) available in this study provided a comprehensive picture of the interaction, allowing us to explore the potential visual and acoustic cues available to the whale. It is likely that the observed behavioral response to the close ship passage resulted from some integration of these multi-modal indicators of close presence rather than any single parameter (e.g., maximum RL) driving the avoidance response.

The observed response behavior of the whale in this study occurred during an ascent from a deep non-lunge feeding dive when the whale aborted its ascent to the surface in order to descend back down to a deeper, and potentially safer depth, until the ship had passed overhead. There appeared to be no change in the direction of the whale as it traveled along the shelf edge perpendicular to the course of the ship. This mirrored the behavioral response previously described by a blue whale in McKenna et al. (2015). The whale also performed a 25-degree left-hand roll as the ship passed overhead.

The focal follow of the whale was a consistent part of the observation/tracking of this whale and lasted from 0736 (nearly 1 h prior to tagging) through 1436 (with a small number of follow-up observations through 1756). The only exceptions from this routine involving other types of approaches were well before or after the ship close approach and included approaches by the RHIB to deploy tags at 0848, an approach to conduct a unmanned aircraft system flight over the whale around 1000 (and ending by 1010), and two approaches to collect biopsy and fecal samples between 1340 and 1436. No obvious strong reactions were noted to these approaches (a potential acceleration was noted as a reaction to the biopsy collection at 1340). There were no sudden changes or close approaches to the tagged whale immediately before, during, and after the close approach with the Mokihana, allowing us to reliably detect changes during close encounters. Given that these approaches were not within an hour of the ship close approach and did not elicit a response, we are confident the specific and unusual observed response documented around the time of the ship CPA and described here is primarily related to the encounter with the Mokihana.

The cues whales use to detect the presence of a ship will likely influence how they respond and the amount of time they may have to react before a potential collision. Although cetacean vision is monochromatic, they do have adaptations for better underwater vision, including large, flattened eyeballs; enlarged pupils; and a tapetum lucidum, which translates to increased light intake and clearer images (Dawson, 1980; Mass and Supin, 2007). Deep-diving whales also have higher rhodopsin, a light-sensitive protein in the rod cells that confer greater sensitivity toward blue-shifted underwater light (Jacobs, 1993; Southall et al., 2002; Dungan et al., 2016). This suggests that in a clear ocean, whales could make use of any available light within the euphotic zone. In turbid waters, reduced visibility may increase the risk of ship strike; however, in our study, the Beaufort Sea State was reported as a 4, and the whale was 67.5 m from the surface. The whale may have been close enough to the surface to see the downwelling light blocked by the nearly 300-m cargo ship, similar to how they would assess prey distribution. Additionally, rolling 25 degrees, an uncommon response for blue whales near the surface (Segre et al., 2018), is suggestive of deliberate behavior, and would enhance panoramic vision (120–130° visual field) in multiple dimensions (Goldbogen et al., 2013a), allowing the whale to watch the ship pass overhead. Because cetacean vision functions in air and water (Supin et al., 2001), this whale also may have seen the ship approaching when the whale was at the surface.

At the time the whale initiated its response, there was only a minimal increase in the overall ship noise level above background levels (as detected on the tag) although there was a rapid increase in relative levels of high-frequency noise. This indicates that the whale may have reacted to the these changes in acoustic cues of the vessel's proximity soon after they were available. However, the ship was only audible on the tag above background levels once it was within extremely close range (∼300 m). Additionally, the main source of noise – the propeller – is located at the stern of the ship, so at the maximum received sound level, hundreds of meters of ship had already passed overhead. This suggests that a whale ahead of a ship may have very little acoustic information to indicate its approach and therefore only extremely limited time to initiate an appropriate behavioral response. Several factors can affect the ability of whales to detect and locate the sounds of approaching ships, including acoustical shadowing if the propellers are located shallower than keel depth, masking of ship noise by ambient sound from other ships, and the Lloyd's Mirror Effect whereby refraction of lower frequency sounds from the surface leads to extreme sound attenuation at shallow depths (Gerstein et al., 2005).

Additionally, the maximum RMS broadband received sound levels exceeded pre-ship sound levels by ∼10 dB, a value well below those associated with avoidance and diving behavioral responses of shallow-diving blue whales to active sonar sounds (see: Southall et al., 2019). While these have different contexts than continuous noise associated with vessels, the data are consistent with the observation that the response was not necessarily driven by an aversive reaction to a perceived loud sound. Rather, the increase in ship noise above ambient conditions, and other factors we were unable to measure (e.g., Doppler shifts indicating relative motion), were potentially integrated with visual information to indicate the close proximity of the ship to the whale that resulted in the observed response. However, as background ocean noise levels increase, particularly driven by greater shipping traffic (Ross, 1993; Andrew et al., 2002; Chapman and Price, 2011; Southall et al., 2018), it may prove to be even more difficult for a blue whale to detect acoustic cues in order to locate and avoid passing ships. If blue whales are not detecting acoustic cues, or the acoustic cues are below individual hearing thresholds, they must rely solely on visual detection, which greatly reduces the range that they can detect an oncoming ship.

The whale's behavioral state at the time of the close encounter may have played a role in its behavioral response. Lunge feeding was not detected in the dive recorded by the DTAG-3 during the CPA. However, lunges were detected in dives before and after the CPA. The dive occurring during the CPA may have been part of a larger foraging bout or constituted traveling in search of a new prey patch. Behavioral state has been shown to influence the context-dependent behavioral response of tagged blue whales, including during playback experiments with ship noises and navy sonar (Goldbogen et al., 2013b; Southall et al., 2018, 2019). Feeding whales may be distracted (Chatterton, 1926; Horwood, 1981; Watkins, 1986) and thus be less capable of detecting – and, therefore, avoiding – approaching vessels. They may also ignore ships in favor of their current behavior (e.g., feeding, socializing, migrating) or due to habituation (Laist et al., 2001; Nowacek et al., 2004; Silber et al., 2010).

The avoidance of a collision between the tagged whale and large vessel may not have been solely due to the animal's behavior. Specifically, the ship's speed may have played a role by giving the whale enough time to respond. At the time of the close passage and onset of the observed behavioral response by the whale, the ship was going 11.3 knots. This ship had recently left the Precautionary Area of the SBC Traffic Separation Scheme. Matson, Inc., which owns the Mokihana, was participating in a vessel speed reduction trial incentive program, which aimed to slow ships in the SBC from 14–18 knots to 12 knots. In addition to reducing air pollution, slowing ships to 12 knots has been shown to greatly reduce the chances of a lethal ship strike (Vanderlaan and Taggart, 2007; Gende et al., 2011; Wiley et al., 2011; Conn and Silber, 2013; McKenna et al., 2015). The Mokihana had not yet picked up speed, which may have allowed the additional reaction time for the animal to arrest its ascent and avoid a potential collision. The behavioral action may not have been as effective if the vessel was traveling at greater speeds (McKenna et al., 2015), and the whale could have been struck at the surface or gotten close enough to the ship's draft that the propeller suction effect created by the ship's hydrodynamic flow could pull the whale toward the hull (Silber et al., 2010) resulting in a ship strike.

One of the hypotheses to arise from the research of McKenna et al. (2015) is that because the evolutionary history of blue whales did not include threats at the surface, whales have not developed an effective behavioral response strategy for this surface hazard. Our study confirms that there are some sensory cues available to the whale, but only at relatively close ranges (<300 m) and under certain oceanographic conditions. This may mean that even experienced individuals cannot always effectively adapt to the threat of shipping traffic. However, this may be further compounded by potential habituation to the presence of ships in important habitats. We know from the long sighting history of the tagged whale that it spent large amounts of time in high ship traffic areas, was exposed to military sonar, and was even involved in the capsizing of a small boat. The whale in this study was able to make last minute behavioral changes in response to the ship when it was already extremely close. However, this response may not be effective in all situations, making blue whales particularly vulnerable to ship strikes. The two key data points from our study – distance and acoustic cues (including RLs and frequency content) – will aid future models in determining when animals would need to respond to avoid being hit by a ship.

The combination of the distinct methodologies and technologies presented in this case study allowed for the collection of high-resolution behavioral information to examine a blue whale's response during a close encounter with a large vessels. Not only has this filled in gaps in our current understanding of blue whale exposure to anthropogenic threats, which will contribute to more informed evaluation and mitigation strategies, but this study provides an example of how multiple methodologies can be combined to conduct behavioral studies in other highly mobile marine megafauna. Future work will examine close encounters from multiple whales to determine if certain contextual factors lead to a higher rate of behavioral response. This information can be used by managers to reduce the risk of exposure to ships or increase the chances of a successful evasion during a ship encounter.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

This study was carried out in accordance with the recommendations of Institutional Animal Care and Use Committee protocols (#AUP-6) and National Marine Fisheries Service Authorizations and Permits for Protected Species (#14534-2). The protocol was approved by the Institutional Animal Care and Use Committee protocols (#AUP-6) and National Marine Fisheries Service Authorizations and Permits for Protected Species (#14534-2).

AUTHOR CONTRIBUTIONS

JC and BS conducted the field work and collected the data. AS and JC conceived of the presented idea. AS processed and analyzed the kinematic dive data, conducted the distance calculations, and wrote the draft manuscript. AA and BS conducted the acoustic analysis. JF created the whale pseudotrack track file. AS, AA, JC, JF, MM, and BS discussed the results and contributed to the final manuscript.

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frontiers in Marine Science

ORIGINAL RESEARCH
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Effects of Variability in Ship Traffic and Whale Distributions on the Risk of Ships Striking Whales

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Assessments of ship-strike risk for large whales typically use a single year of ship traffic data and averaged predictions of species distributions. Consequently, they do not account for variability in ship traffic or species distributions. Variability could reduce the effectiveness of static management measures designed to mitigate ship-strike risk. We explore the consequences of interannual variability on ship-strike risk using multiple years of both ship traffic data and predicted fin, humpback, and blue whale distributions off California. Specifically, risk was estimated in four regions that are important for ship-strike risk management. We estimated risk by multiplying the predicted number of whales by the distance traveled by ships. To overcome the temporal mismatch between the available ship traffic and whale data, we classified the ship traffic data into nearshore and offshore traffic scenarios using the percentage of ship traffic traveling more than 24 nmi from the mainland coast, which was the boundary of a clean fuel rule implemented in 2009 that altered ship traffic patterns. We found that risk for fin and humpback whale populations off California increased as these species recovered from whaling. We also found that broad-scale, northward shifts in blue whale distributions throughout the North Pacific, likely in response to changes in oceanographic conditions, were associated with increased ship-strike risk off northern California. The magnitude of ship-strike risk for fin, humpback, and blue whales was influenced by the ship traffic scenarios. Interannual variability in predicted whale distributions also influenced the magnitude of ship-strike risk, but generally did not change whether the nearshore or offshore traffic scenario had higher risk. The consistency in the highest risk from the traffic scenarios likely occurred because areas containing the highest predicted number of whales were generally the same across years. The consistency in risk from the traffic scenarios suggests that static spatial management measures (e.g., changing shipping lanes, creating areas to be avoided, and seasonal speed reductions) can provide an effective means of mitigating risk resulting from ship traffic variability off California.

Keywords: species distribution modeling, interannual variability in species distributions, commercial shipping, variability in ship traffic, spatially explicit risk assessment, ship-strike risk

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INTRODUCTION

Ship strikes are one of the largest sources of human-caused mortality for fin (*Balaenoptera physalus*), humpback (*Megaptera novaeangliae*), and blue (*B. musculus*) whales on the United States West Coast (Carretta et al., 2017). The risk of ships striking whales (hereafter, ship-strike risk) has been assessed for these species in several regions: for all three species off the entire United States West Coast (Rockwood et al., 2017) and southern California (Redfern et al., 2013, 2019), for blue whales off the entire United States West Coast (Hazen et al., 2017; Abrahms et al., 2019b), and for humpback whales off San Francisco (Dransfield et al., 2014). Many of these risk assessments and those conducted for various locations around the world (e.g., Sri Lanka and Canada; Williams and O'Hara, 2010; Priyadarshana et al., 2016) use a single year of ship traffic data and predictions of whale distributions that are averaged over several years. Redfern et al. (2013) and Redfern et al. (2019) are exceptions because they used more than one year of ship traffic data; Hazen et al. (2017) and Abrahms et al. (2019b) developed near real-time tools to predict blue whale distributions. However, most ship-strike risk assessments do not account for variability in ship traffic, species distributions, or both.

Recent studies suggest variability in both ship traffic and whale distributions off California. An analysis of ship traffic off California between 2008 and 2015 suggested that air pollution regulations implemented at both state and international levels changed the primary routes used by ships (Moore et al., 2018). Specifically, the California Air Resources Board implemented a rule on July 1, 2009 that required ships to use cleaner burning fuels when traveling within 24 nmi of the mainland coast. After implementation of the rule, ships began traveling farther offshore (**Figure 1** and **Table 1**) to reduce the amount of time spent using the cleaner fuels, which were more expensive. Use of offshore routes was found for the entire coast of California (Moore et al., 2018) and for the major California ports: San Francisco Bay (Jensen et al., 2015) and Los Angeles/Long Beach (McKenna et al., 2012).

The boundary for clean fuel use was extended to include the area around the Channel Islands in the Southern California Bight on December 1, 2011. The International Maritime Organization (IMO) also increased the international standards for clean fuels in the U.S. Exclusive Economic Zone on August 1, 2012, which brought the international standards closer to the California standards. Ship traffic off southern California showed the strongest change in association with these changes in fuel regulations (Moore et al., 2018). From 2012 to 2014, some traffic began to return to the original nearshore routes and offshore traffic shifted beyond the new clean fuel boundary (**Figure 1** and **Table 1**). Ship traffic off San Francisco showed similar, but smaller changes, while traffic in the remaining areas primarily stayed offshore (**Figure 1** and **Table 1**). The IMO and California clean fuel standards were similar by 2015 and ship traffic in all regions was similar to the nearshore patterns observed in 2008 (**Figure 1** and **Table 1**).

Species abundances and distributions have also changed off California. There is strong evidence that fin and humpback whale abundance has increased at broad scales in the North Pacific (Barlow et al., 2011; Moore and Barlow, 2011), suggesting that current levels of ship strikes do not negatively affect these species at these broad scales. However, ship strikes may be an issue at regional scales. Specifically, populations of humpback whales that breed off Mexico and Central America are listed as Threatened and Endangered, respectively, on the United States Endangered Species List. Both populations feed off California and it is possible that ship strikes could have negative population-level consequences. It is also possible that a unique population of fin whales remains year-round in the southern California Bight (Forney and Barlow, 1998; Calambokidis et al., 2015) and that ship strikes may impact this population.

There is no evidence that the abundance of blue whales in the North Pacific is increasing and it has been suggested that this population may have reached carrying capacity (Monnahan et al., 2015). However, it has also been suggested that blue whales are shifting farther north, potentially in response to changing ocean conditions (Calambokidis et al., 2009). Consequently, abundance may be increasing off British Columbia and in the Gulf of Alaska, but decreasing off California. Interannual variability in species distributions was also identified as an important source of uncertainty in habitat models for fin, blue, and humpback whales built using line-transect survey data (Forney et al., 2012) and telemetry data (Hazen et al., 2017).

Static management measures, which include changing shipping lanes and establishing areas to be avoided, are commonly used to mitigate ship-strike risk. For example, the International Maritime Organization (IMO) adopted five measures between 2002 and 2009 that relocate ship traffic in waters off eastern North America to minimize the co-occurrence of right whales and ships (Silber et al., 2012). Traffic separation schemes were established by the IMO for the major California ports and are reflected in the dominant ship traffic patterns (i.e., the darkest blue color off Southern California and San Francisco in the 2008 map in **Figure 1**). These lanes were modified in 2013 to reduce ship-strike risk. Voluntary and incentivized speed reductions have also been implemented in these traffic separation schemes to mitigate ship-strike risk (e.g., Freedman et al., 2017) because studies (e.g., Conn and Silber, 2013) have shown that the probability of a fatal ship strike increases at higher ship speeds. Voluntary shipping lanes have also been used to manage ship traffic in National Marine Sanctuaries (e.g., Monterey Bay National Marine Sanctuary) and off southern California. Interannual variability in species distributions could reduce the effectiveness of static management measures designed to reduce ship-strike risk. We used ship traffic data and whale distributions off California to explore the consequences of interannual variability on ship-strike risk.

MATERIALS AND METHODS

Ship Traffic Data

Automatic Identification Systems (AIS) are maritime tracking systems that were adopted by the IMO and were initially required (December 31, 2004) on international voyages for

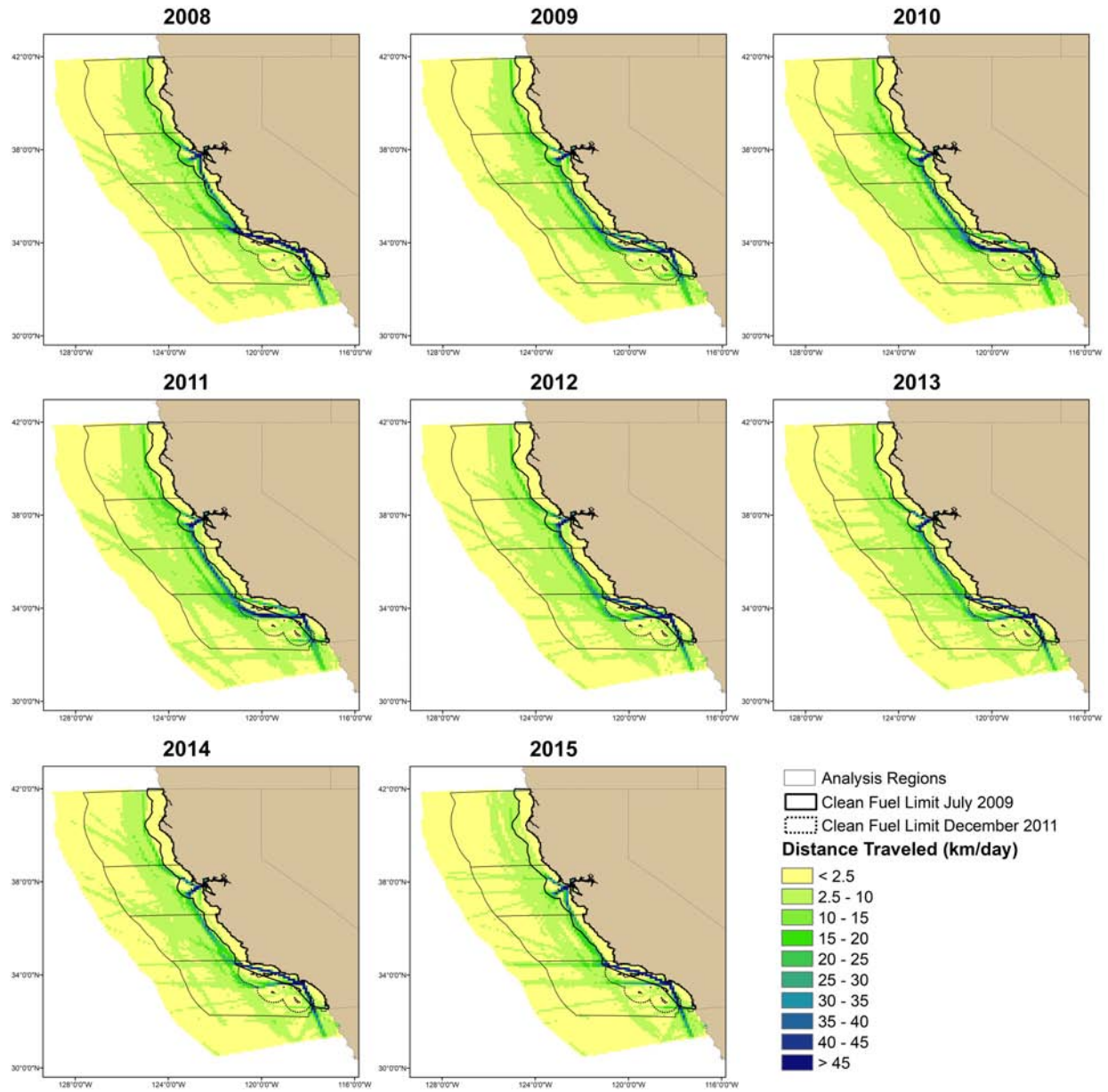

FIGURE 1 | Maps of distance traveled (km/day) by ships from July–December in a 10 × 10 km grid. Analyses were conducted in four regions that are named from north to south as north, San Francisco, central, and south. Traffic was classified into nearshore (2008 and 2015; defined as having more traffic within the July 2009 clean fuel boundary compared to offshore traffic) and offshore (2009–2011; defined as a majority of traffic outside the July 2009 clean fuel boundary) scenarios. Classification of the 2012–2014 traffic was done on a region-by-region basis.

all ships ≥300 gross tons, domestic voyages for cargo vessels ≥500 gross tons, and on all passenger ships (International Maritime Organization [IMO], 2014). The type and range of ships required to use AIS has expanded since this time (for example, see[1]). Data include dynamic information, such as ship position, speed, and course, and static information, such as ship identifier, type, and dimensions. Data obtained from the US Coast Guard's terrestrial Nationwide Automatic Identification System extend throughout the U.S. exclusive economic zone (i.e., out to 200 nmi) because the system was designed to improve navigational safety, search and rescue, and maritime security. However, the amount of data received may decrease farther from shore due to transmission loss.

Moore et al. (2018) analyzed an 8-year time series of terrestrial AIS data (2008–2015). We followed the methodology of Moore et al. (2018) to summarize the cumulative distance traveled by ships between July and December each year in a 10 × 10 km grid. We selected ships greater than 80 m in length and used only

[1]https://www.navcen.uscg.gov/?pageName=AISRequirementsRev

TABLE 1 | The percentage of traffic (i.e., the sum of the distance traveled per day in each grid cell) that occurs more than 24 nmi from the mainland coast (i.e., the boundary established by the California Air Resources Board as part of the clean fuel rule implemented on July 1, 2009) in each region.

	2008	2009	2010	2011	2012	2013	2014	2015
North	84	95	96	96	96	97	96	87
San Francisco	54	64	66	67	64	63	62	52
Central	84	96	98	98	98	97	97	78
South	38	62	69	69	56	56	52	48

ships traveling at a speed over ground ≥2.5 knots to ensure that only underway ships were included in the analyses. A radius of approximately 5.6 km from the center of a grid cell was used to calculate the cumulative distance traveled because the area of the resulting circle is the same as the area of the grid cells. We divided the July–December cumulative distance traveled in each grid cell by the number of days of AIS data collection to account for data gaps (e.g., missing data in 2008 and 2010).

Whale Distributions

We used predictions of fin, humpback, and blue whale distributions from models produced by Becker et al. (2016). Becker et al. (2016) developed whale-habitat models using 7 years of line-transect survey data collected by NOAA Fisheries' Southwest Fisheries Science Center between July and December (i.e., 1991, 1993, 1996, 2001, 2005, 2008, and 2009). Habitat variables were derived from *in situ* and remotely sensed oceanographic data for humpback whales and output from a Regional Ocean Modeling System for fin and blue whales. Data from the 2009 survey could not be used in the humpback whale models because the survey and, concomitantly, the *in situ* oceanographic sampling did not cover the entire study area.

Becker et al. (2016) used generalized additive models (GAMs) (Wood, 2006) to relate habitat variables to the number of whales in transect segments that were approximately 5 km. They fit GAMs in the R (version 3.1.1; R Core Team, 2014) package "mgcv" (version 1.8–3; Wood, 2011). The models were used to predict the number of whales in an approximately 10 × 10 km grid for distinct 8-day composites covering the entire survey period. We use the annual average of these predictions and the average across all years of survey data in our analyses (hereafter, annual and mean predictions). The ship-traffic grid was different from the whale distribution grid. Consequently, the predicted whale densities were overlaid on the ship traffic grid and used to derive the predicted number of whales in each ship traffic grid cell. This approach preserved the total abundance in the study area.

Ship-Strike Risk

We assess risk in four regions (**Figure 1**) that are important for ship-strike risk management. In particular, the southern and San Francisco regions contain the major California ports and efforts have been made in both regions to mitigate ship-strike risk. The central region contains traffic traveling between California ports, while the northern region contains traffic traveling to ports in the Pacific Northwest. The regions extend from the shore to 250 km beyond the shelf edge, which is an important topographic feature for many species of baleen whales because of its role in concentrating their prey (e.g., Fiedler et al., 1998; Croll et al., 2005). The shelf edge was derived from a global, seafloor geomorphic features map (Harris et al., 2014). The offshore boundary of the regions encompasses a majority of California ship traffic (approximately 90–95%) and reduces the potential for bias from AIS signal decay farther offshore.

Latitudinal breaks for each region were selected using biogeographical boundaries (i.e., Point Conception for the boundary between the southern and central regions) and biologically important areas (BIAs) for blue and humpback whales (BIAs have not yet been defined for fin whales). These BIAs were defined by Calambokidis et al. (2015) and represent areas where concentrations of feeding animals were observed in multiple years of non-systematic, coastal surveys designed to maximize encounters with blue and humpback whales for photo-identification and tagging studies.

We estimate ship-strike risk by multiplying the predicted number of whales by the mean daily kilometers of ship traffic within a grid cell. Consequently, we are defining risk as the co-occurrence between whales and ships, as has been done for multiple species (e.g., Vanderlaan et al., 2009; Williams and O'Hara, 2010; Redfern et al., 2013). There is a temporal mismatch between the ship traffic and whale data available off California: ship traffic data are available from 2008 to 2015, while whale distribution data are available from 1991 to 2009. To overcome this temporal mismatch, we classified the time series of ship traffic data into traffic scenarios using the percentage of ships traveling more than 24 nmi from the mainland coast (i.e., the 2009 clean fuel boundary).

The percentage of traffic traveling more than 24 nmi from the mainland coast was the lowest in 2008 and 2015 in all four regions (**Table 1**). We used the distance traveled in each grid cell for these 2 years to define a nearshore traffic scenario. The percentage of traffic traveling more than 24 nmi from the mainland coast generally increased from 2009 to 2011 (**Table 1**). Consequently, we used the ship traffic data from 2009 to 2011 to define an offshore traffic scenario. Traffic patterns vary by region in 2012–2014 and cannot be assigned to a single scenario (**Table 1**).

The total distance traveled by ships decreased throughout the time series of ship traffic data. Consequently, we had to correct for this decrease in distance traveled to meaningfully combine years of ship traffic data. Specifically, we calculated a correction factor by dividing the kilometers traveled in each region by the mean kilometers traveled in each region over all years. A correction factor greater than one implied that the distance traveled in that year was higher than the mean for that region. We divided the distance traveled by ships in each grid cell by this correction factor. Ship speeds also decreased through the time series of ship traffic data (Moore et al., 2018). Consequently, we cannot meaningfully include ship speed in our definition of ship-strike risk.

We initially assessed the risk associated with the ship traffic scenarios using mean predicted whale distributions (i.e., the average of the predictions across all years of survey data). In

particular, risk associated with the nearshore traffic scenario was defined by the mean and standard error of the risk in 2008 and 2015 (i.e., risk in each year was calculated as the mean predicted number of whales multiplied by the distance traveled by ships). The risk associated with the offshore traffic scenario was defined by the mean and standard error of the risk in 2009, 2010, and 2011. We also calculated the mean and standard error of the risk in 2012, 2013, and 2014, but classify its association with the traffic scenarios on a region-by-region basis. To compare risk among regions, we summed the risk for all grid cells in each region and divided by the area of the region. We compared risk in the traffic scenarios and calculated the percent change in risk for the nearshore versus offshore scenarios and nearshore versus 2012–2014 traffic scenarios. Finally, we mapped risk for the nearshore and offshore traffic scenarios using mean predicted whale distributions and the 2008 (representative of the nearshore scenario) and 2010 (representative of the offshore scenario) ship traffic data.

Risk calculated using the mean whale distributions was compared to risk derived from the annual predictions from the Becker et al. (2016) models to understand how interannual variability in species distributions affects risk. Interannual variability in locations containing the highest predicted number of whales (**Figure 2**) was assessed using the number of years (7 years for fin and blue whales; 6 years for humpback whales) the prediction in each grid cell was among the highest 5% of all predictions. These analyses were conducted using all whale predictions (i.e., from the coast of California out to 300 nmi). Changes in predicted abundance are not included in this metric because the highest 5% of predictions are calculated for each year. This metric identifies areas that consistently contained the highest predicted number of whales, areas that never contained the highest predicted numbers, and areas of variability.

To assess the effect of variability in whale distributions on risk, we calculated the percent change in risk for the nearshore versus offshore traffic scenarios for each species, region, and survey year (i.e., a total of 80 calculations). Finally, we estimated the change in the total risk experienced by each whale population since the start of the survey period. Specifically, we looked at the ratio of the risk in each survey year to the risk at the start of the survey period (i.e., 1991) for the traffic scenario (i.e., nearshore or offshore) resulting in the highest risk for each species in each region.

RESULTS

The region that had the highest density for each species also had the highest mean ship-strike risk (i.e., risk assessed using the mean of the predicted whale distributions; **Figure 3**). In particular, density and mean risk were highest for fin whales in the central region, for humpback whales off San Francisco, and for blue whales in the south. However, the magnitude of the mean risk within a region was influenced by the ship traffic scenarios (**Figures 3**, **4**, and **Table 2**). Interannual variability in predicted whale distributions also influenced the magnitude of ship-strike risk, but generally did not change the effect of the traffic scenarios on risk (**Figure 5**). Specifically, interannual variability did not change whether the nearshore or offshore traffic scenario had higher risk.

In the south, ship-strike risk for fin whales was lowest for the nearshore traffic scenario (**Figures 3**, **4**). There was a 16% increase in the mean risk associated with the offshore traffic scenario for fin whales and a 5% increase in risk when traffic occurred both nearshore and offshore (i.e., the traffic scenario for 2012–2014; **Table 2**). Interannual variability in predicted fin whale distributions resulted in a 9–23% increase in risk for the offshore traffic scenario (**Figure 5**). The opposite pattern was seen for humpback and blue whales in the south (**Table 2** and **Figures 3**–**5**). In particular, mean risk decreased by 20% and annual risk by 6–27% for humpback whales for the offshore versus nearshore traffic scenario. Mean risk decreased 18% when traffic occurred in both locations. Mean risk decreased approximately 6% for blue whales for the offshore versus nearshore traffic scenario. Risk generally decreased for the offshore versus nearshore traffic scenario (range 4–10%) for annual predicted blue whale distributions. However, offshore traffic had a higher risk than nearshore traffic for the blue whale distributions predicted in 1993. Among all regions, years, and species, predicted blue whale distributions in 1993 represented the only reversal in the traffic scenario (i.e., nearshore versus offshore) having the highest risk.

In the central region, risk for all three species was highest for the nearshore traffic scenario and lower for the offshore traffic scenario, which is represented by traffic in both 2009–2011 and 2012–2014 for this region (**Table 2** and **Figures 3**–**5**). In particular, mean risk decreased by 9% for fin whales (range in annual predictions = 2–14%), 34% for humpback whales (range in annual predictions = 30–37%), and 18% for blue whales (range in annual predictions = 14–23%) when traffic occurred offshore (2009–2011) compared to nearshore.

The shift between nearshore and offshore traffic in the central region corresponded to a change from ships using primarily the northern and southern approaches off San Francisco (2008 and 2015) to increasing use of the western approach (2009–2011) or the western and northern approaches (2012–2014; **Figure 1**). Use of the western approach allowed ships to minimize travel nearshore (i.e., within 24 nmi of the mainland coast). Risk increased 6% for fin whales (range in annual predictions = 1–11%) off San Francisco for the offshore versus nearshore traffic scenario (i.e., in association with increased use of the western approach) and increased 5% for the 2012–2014 versus nearshore traffic scenario (i.e., in association with increased use of the western and northern approaches) (**Table 2** and **Figures 3**–**5**). The opposite pattern was seen for humpback and blue whales off San Francisco (**Table 2** and **Figures 3**–**5**): risk decreased 16% for humpback whales (range in annual predictions = 15–17%) and 13% for blue whales (range in annual predictions = 10–19%) for the offshore compared to nearshore traffic scenarios. Risk also decreased for the 2012–2014 traffic scenario compared to nearshore traffic scenario, although the change was generally smaller (**Table 2**). Risk in the northern region for all species followed similar patterns as off San Francisco. However, risk was generally lower in the north (**Figure 3**) and the percent changes

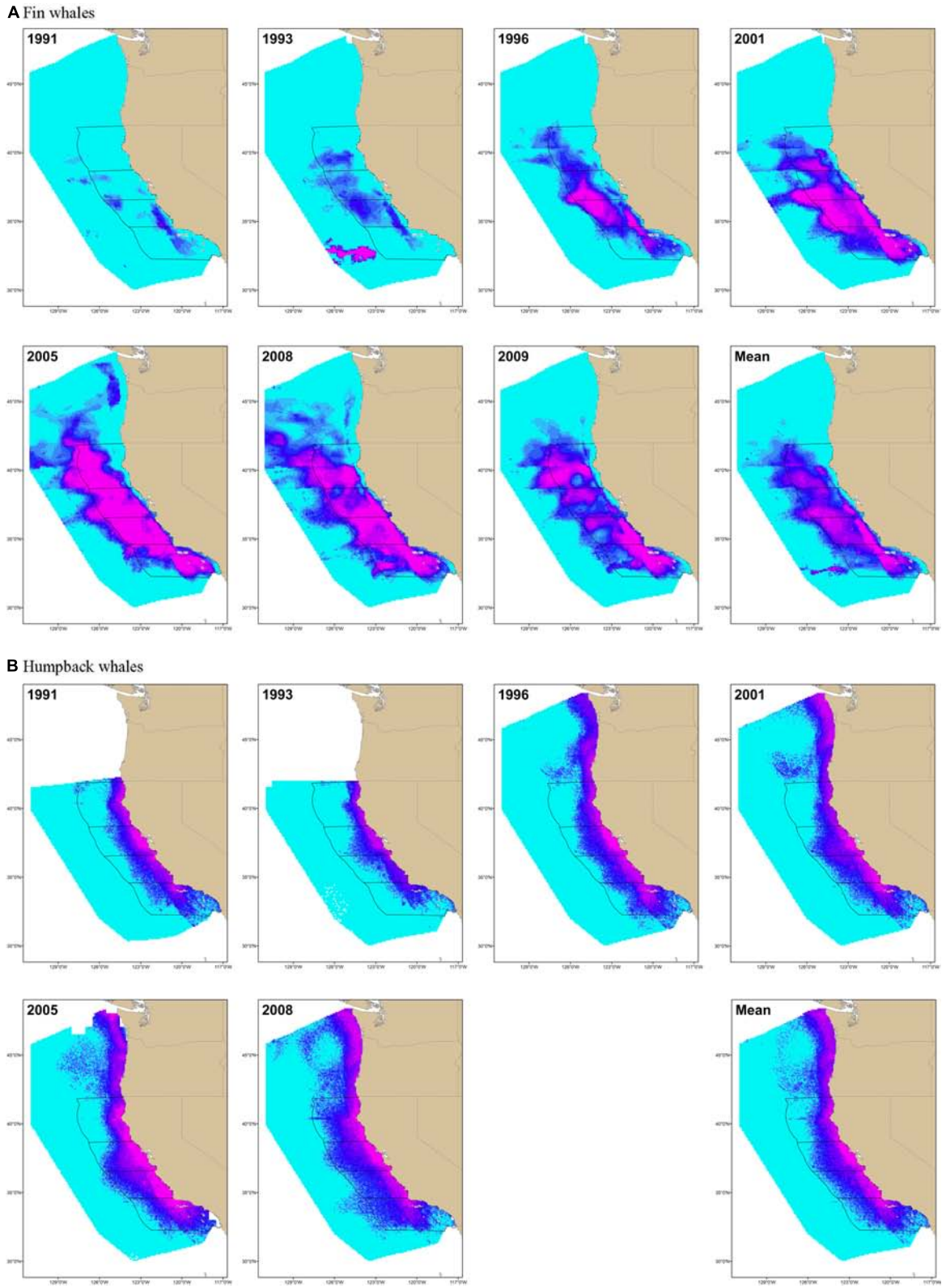

FIGURE 2 | Continued

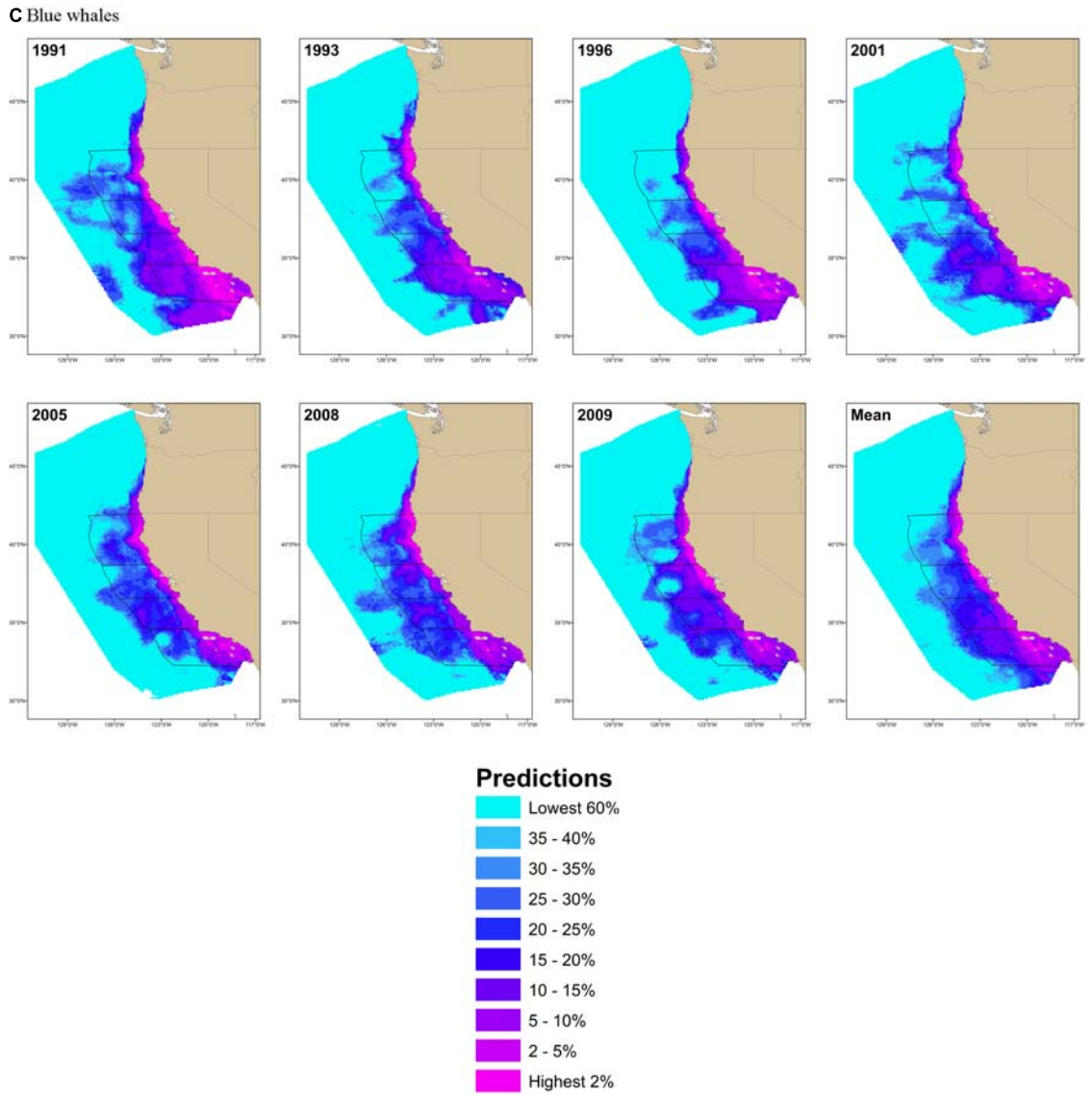

FIGURE 2 | Maps of annual and mean predicted species distributions for **(A)** fin, **(B)** humpback, and **(C)** blue whales. Habitat variables were derived from *in situ* and remotely sensed oceanographic data for humpback whales and output from a Regional Ocean Modeling System for fin and blue whales. Data from the 2009 survey could not be used in the humpback whale models because the survey and, concomitantly, the *in situ* oceanographic sampling did not cover the entire study area. Surveys were only conducted off California in 1991 and 1993; consequently, predictions of humpback whale distributions could not be made off Oregon and Washington in these 2 years. Other differences in the study area boundary for humpback whales and missing predictions within the study area (white grid cells) arise from gaps in oceanographic sampling. The highest 2, 5, 10, . . ., and 40% of predicted values were calculated using the mean predicted densities. Data from all years were categorized using the values from the mean to show changes in density among years.

in risk associated with the traffic scenarios were larger (**Table 2** and **Figure 5**).

The previous results show that interannual variability in predicted whale distributions influenced the magnitude of ship-strike risk, but that the difference in risk from the nearshore versus offshore traffic scenario was consistent across all years of predicted whale distributions. To understand these results, we identified areas that consistently contained the highest predicted number of whales, areas that never contained the highest predicted numbers, and areas of variability (**Figure 6**). In each region, areas containing the highest predicted number of whales were generally the same across years, as were areas that never contained the highest predicted numbers.

Ship-strike risk was also influenced by large-scale increases in whale abundance and large-scale shifts in distributions (**Figure 7**). In particular, the total risk experienced by both fin and humpback whale populations generally increased between the 1990s and the 2000s in all regions, consistent with previously documented increases in their abundance throughout the North Pacific. The total risk experienced by the blue whale population

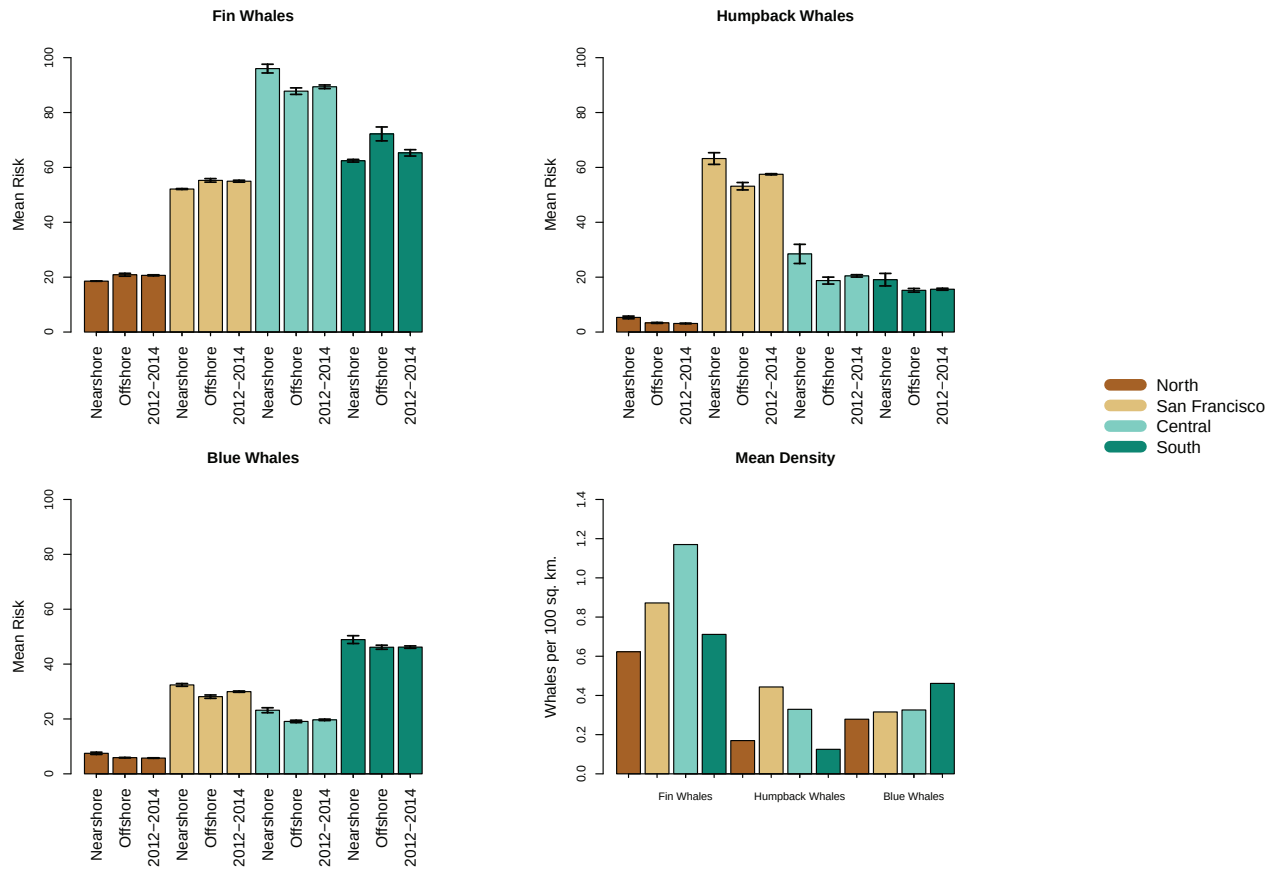

FIGURE 3 | The scenarios associated with the 2008–2015 ship traffic (i.e., nearshore or offshore; 2012–2014 was classified as nearshore or offshore on region-by-region basis) were defined using a boundary that was 24 nmi from the mainland coast (see text for details). The ship traffic scenarios influenced ship-strike risk (predicted number of whales multiplied by the km/day traveled by ships) for fin, humpback, and blue whales in each region. The risk shown here was calculated using the mean predicted whale distributions. The standard error of the mean risk was calculated using the years of ship traffic data in each scenario. The mean predicted whale density (whales per 100 km^2) in each region is also shown.

increased in the northern region in the 2000s, consistent with previously documented northward shifts in their distribution throughout the North Pacific.

DISCUSSION

Most ship-strike risk assessments do not account for variability in species distributions, ship traffic, or both. We used whale distributions and ship traffic data off California to explore the consequences of interannual variability on ship-strike risk. We found that areas containing the highest predicted number of humpback and blue whales were the same among all years of predictions (**Figure 6**). Predicted fin whale distributions varied more than predicted humpback and blue whale distributions. However, the highest fin whale predictions were always found far from the coast and were never found close to the coast (**Figure 6**). All three whale species feed off California during the time period associated with the predictions from the Becker et al. (2016) models (i.e., July–December). The stability of the presence and absence of the highest whale predictions observed at the scale of our study suggests spatially persistent feeding areas or that these areas are large enough to encompass ephemeral features associated with feeding (Becker et al., 2019). These results are consistent with the findings of Abrahms et al. (2019a) that blue whale migrations more closely tracked long-term averages of productivity than contemporaneous measurements of productivity. Abrahms et al. (2019a) also found that blue whales foraged in areas that had higher and more stable long-term productivity.

The stability of the whale predictions resulted in specific ship traffic scenarios consistently having higher ship-strike risk. In particular, either the nearshore or offshore traffic scenario (defined as within or more than 24 nmi from the coast, respectively) consistently had the highest risk for each species and region. Changes in ship traffic scenarios off California (i.e., nearshore versus offshore) were initiated by the shipping industry in response to air pollution regulations. The consistency in risk suggests that static spatial management measures (e.g., changing shipping lanes, creating areas to be avoided, and seasonal speed reductions) can provide an effective means of mitigating risk

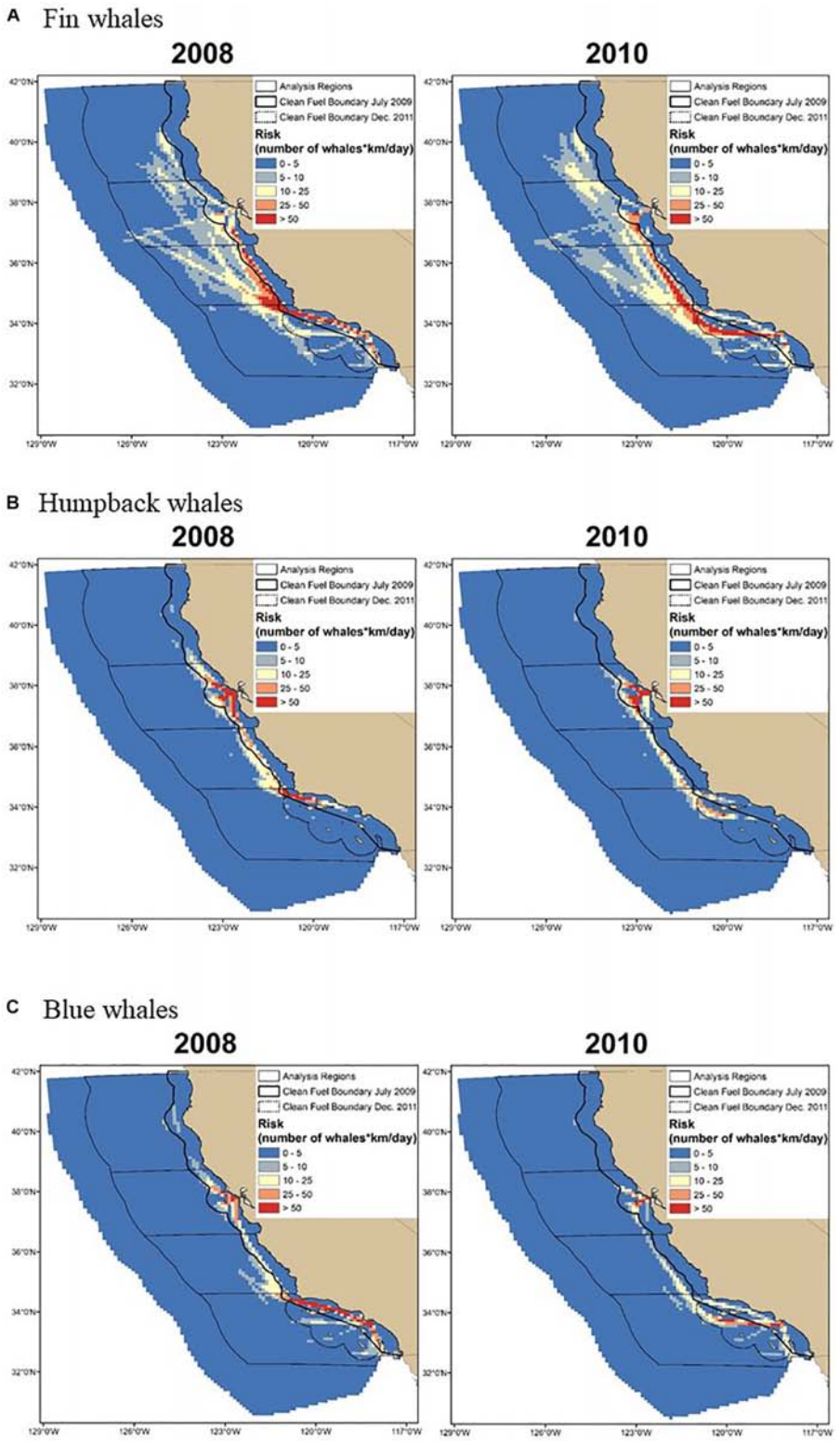

FIGURE 4 | Ship-strike risk for **(A)** fin, **(B)** humpback, and **(C)** blue whales calculated using mean predicted whale distributions is shown for a year with nearshore (2008) and offshore (2010) traffic. Ship-strike risk was defined as the predicted number of whales multiplied by the km/day traveled by ships, which is a measure of co-occurrence. Consequently, the numbers used to define the break points in the maps provide a relative measure of increasing risk. The same break points were used for all species and years to allow for comparisons among maps. The percentage of ships traveling more than 24 nmi from the mainland coast (i.e., the 2009 clean fuel boundary) was used to define nearshore versus offshore traffic (see text for details).

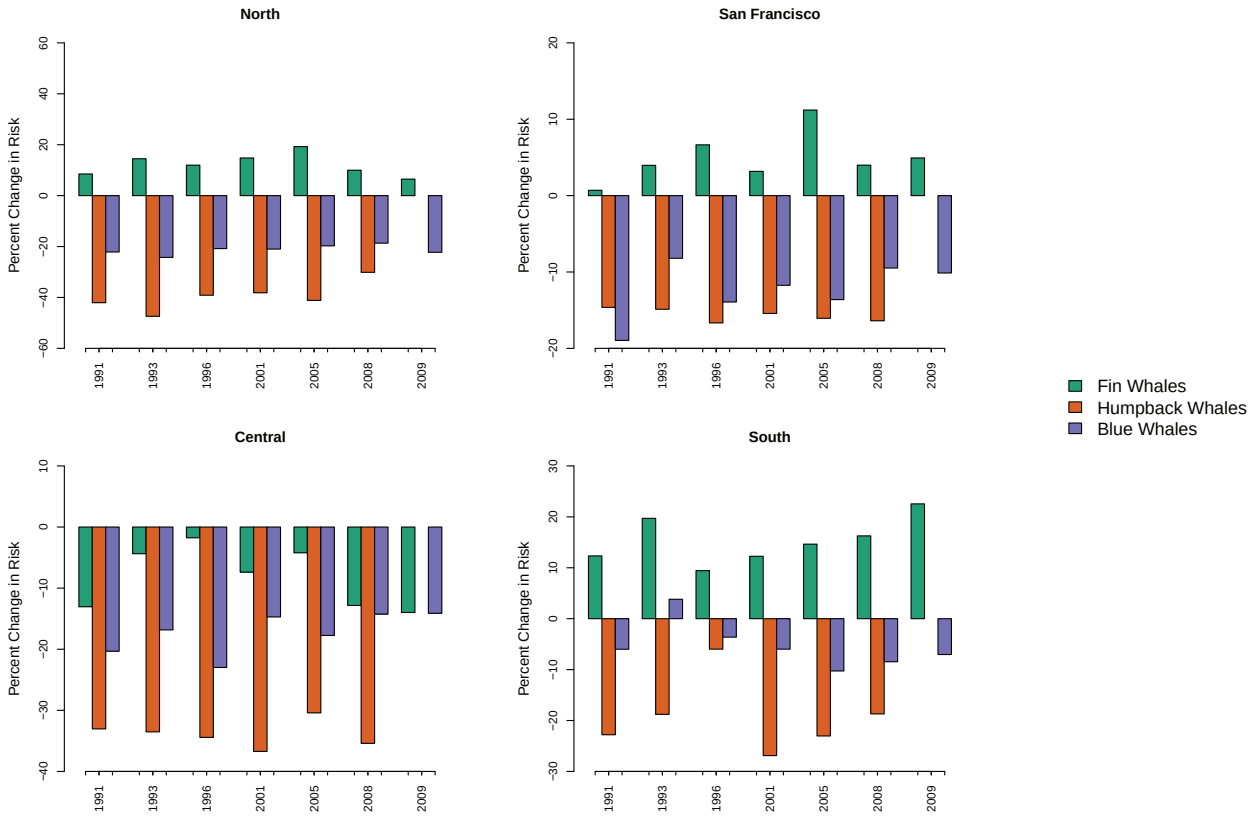

FIGURE 5 | We assessed the effect of interannual variability in whale distributions on ship-strike risk using the percent change between risk for the nearshore versus offshore traffic scenarios; the *x*-axes represent years in which marine mammal surveys were conducted. Negative values indicate a decrease in mean risk from the offshore, compared to the nearshore, traffic scenario. Whether the nearshore traffic scenario poses a higher ship-strike risk than the offshore traffic scenario in each region is consistent across all years for each species, except the 1993 blue whale predictions in the southern region.

resulting from ship traffic variability off California. For example, risk was highest for all three species in the central region when traffic occurred nearshore, rather than offshore (**Figures 3–5**). Consequently, mean risk for all three species can be reduced by up to 35% if traffic follows an offshore route similar to the routes followed by ships in 2009–2011. This reduction in risk does not mean that ship strikes will be eliminated, but that the number of strikes will be minimized over long time periods. There are several possible reasons the nearshore traffic scenario represented a greater overlap with fin whale distributions in this region. Fin whale abundance is higher close to shore in this region, compared to the other regions. Additionally, while the percentage of traffic within 24 nmi of the coast follows the nearshore and offshore traffic scenario definitions in the central region, traffic was generally shifted farther from the coast and was more diffuse in the nearshore traffic scenario.

Ship-strike risk was different for fin whales versus blue and humpback whales in the other regions. In particular, risk

TABLE 2 | The percent change between mean risk from nearshore versus offshore traffic and nearshore versus 2012–2014 traffic (which occurred offshore in some regions and both nearshore and offshore in other regions; **Table 1**), where nearshore and offshore traffic were defined using a boundary that was 24 nmi from the mainland coast (see text for details).

	Fin		Humpback		Blue	
	Offshore	**2012–2014**	**Offshore**	**2012–2014**	**Offshore**	**2012–2014**
North	13	11	−37	−41	−21	−23
San Francisco	6	5	−16	−9	−13	−8
Central	−9	−7	−34	−28	−18	−15
South	16	5	−20	−18	−6	−6

Negative values indicate a decrease in mean risk from the offshore traffic scenario or 2012–2104 traffic.

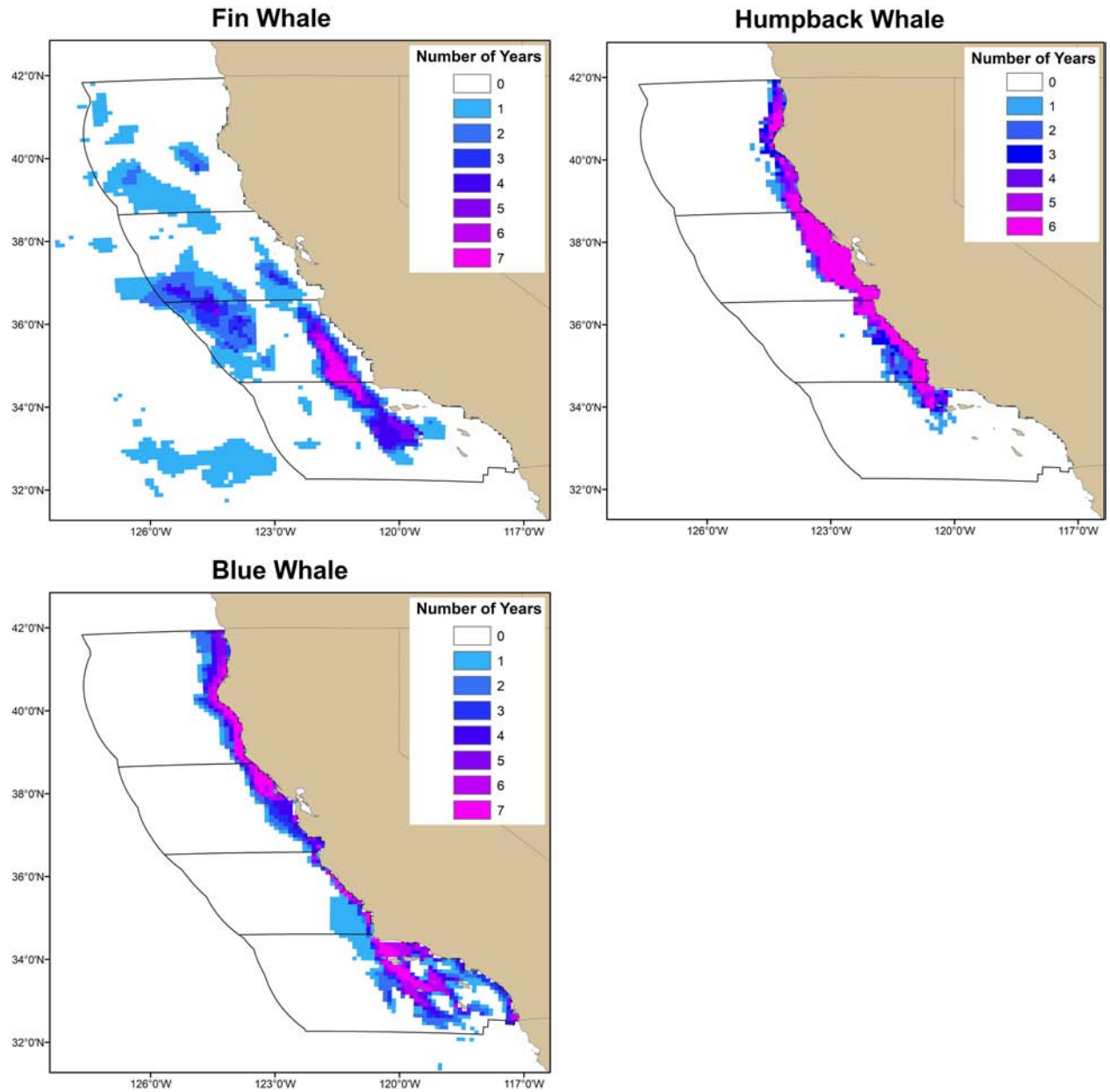

FIGURE 6 | Interannual variability in locations containing the highest predicted number of whales was assessed using the number of years (7 years for fin and blue whales; 6 years for humpback whales) the prediction in each grid cell was among the highest 5% of all predictions. Although these analyses were conducted using all whale predictions (i.e., from the coast of California out to 300 nmi), the regions used to assess ship-strike risk (black polygons) are also shown. Areas with the highest predicted number of whales were consistent among years as were areas that never contained the highest predictions.

for blue and humpback whales was highest in the northern, San Francisco, and southern regions for the nearshore traffic scenario. Risk for fin whales was highest for the offshore traffic scenario in these regions. Consequently, more detailed and fine-scale analyses are needed to design strategies that can mitigate risk for all species in these regions. For example, Redfern et al. (2019) developed methods to estimate ship-strike risk in strategies proposed by stakeholders to reduce risk in the Southern California Bight and found that speed reductions and expanding the existing area to be avoided may provide an optimal solution for addressing stakeholder needs and reducing ship strikes. Analyses are also needed to address risk from January-June because studies have found seasonal changes in fin (Scales et al., 2017) and humpback (Becker et al., 2017) whale distributions off California. Finally, we used a measure of whale-ship co-occurrence (i.e., predicted number of whales multiplied by the cumulative distance traveled by ships) to estimate risk and it is possible to estimate risk using encounter rate theory, which

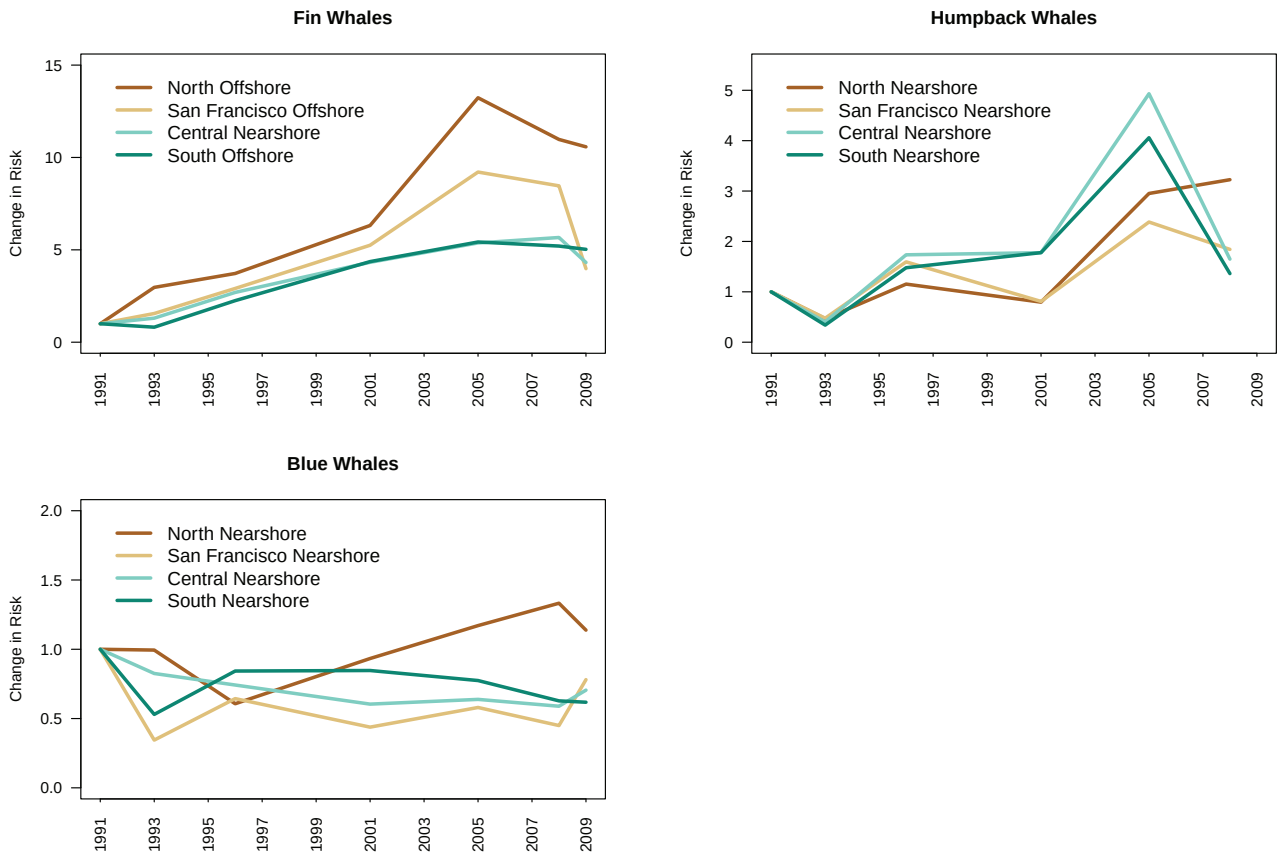

FIGURE 7 | We estimated the change in the total ship-strike risk experienced by each whale population using the ratio of the risk in each survey year to the risk at the start of the survey period (i.e., 1991). We used the risk from the traffic scenario (i.e., nearshore or offshore; see text for details and **Figure 3**) resulting in the highest risk for each species in each region. For humpback and blue whales, the nearshore traffic scenario had the highest risk in all regions. For fin whales, the offshore traffic scenario had the highest risk for all regions except the central region. Risk was highest from the nearshore traffic scenario in the central region for fin whales. The *x*-axes represent years in which marine mammal surveys were conducted.

can incorporate ship speed and whale behavior (Martin et al., 2016). Moore et al. (2018) used Conn and Silber's (2013) equation relating ship speed to the probability that a ship strike is fatal to estimate that reductions in ship speeds in the Santa Barbara Channel (i.e., the 2008 traffic pattern in the southern California region; **Figure 1**) represent a 20% reduction in the probability of a fatal strike. Consequently, ship speeds should be considered when designing strategies that can mitigate risk (e.g., Redfern et al., 2019).

Our analyses show that static, spatial management strategies can be used to mitigate ship-strike risk from nearshore versus offshore traffic off California. Strategies could include routing ships through areas that consistently had lower predicted whale densities and establishing areas to be avoided or requiring reduced ship speeds in areas with consistently higher predicted whale densities. At finer scales, it is important to consider where ships transition from offshore to nearshore travel. It is also important to consider how ships travel when they are nearshore. For example, nearshore traffic off southern California occurs in shipping lanes in the Santa Barbara Channel. In 2007, four blue whales were struck and killed, most likely in these shipping lanes. Seasonal voluntary and incentivized speed reductions have been used in this area to reduce ship-strike risk (Freedman et al., 2017). However, there is little compliance with voluntary speed reductions and incentivized speed reductions only reach a small percentage of ships traveling in this region and require continued financial support (Freedman et al., 2017).

Our study suggests that the magnitude of ship-strike risk may increase as whales recover from whaling. For example, increases in the abundance of fin and humpback whale populations in the North Pacific were associated with increased ship-strike risk off California (**Figure 7**). The magnitude of risk may also be affected by shifts in whale distributions in response to climate change. For example, a broad-scale, northward shift in blue whale distributions throughout the North Pacific was associated with increased ship-strike risk for blue whales off northern California (**Figure 7**). The magnitude of ship-strike risk observed for fin, humpback, and blue whales was also influenced by the location of ship traffic. The spatial variability in ship traffic patterns observed off California in response to air pollution regulations may also occur in other regions. Air pollution regulations are

being considered for many regions, including the Mediterranean Sea and off Japan, Australia, Singapore, and China (Moore et al., 2018). It is important to understand how potential shifts in ship traffic in response to these regulations will affect ship-strike risk for large whales. Our study suggests that static management strategies may effectively mitigate risk from variability in ship traffic patterns, if whales congregate in consistent locations for feeding and breeding.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

JR, EB, and TM conceived of the work and contributed to the analyses and writing the manuscript.

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Quantifying Ship Strike Risk to Breeding Whales in a Multiple-Use Marine Park: The Great Barrier Reef

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Spatial risk assessments are an effective management tool used in multiple-use marine parks to balance the needs for conservation of natural properties and to provide for varying socio-economic demands for development. The multiple-use Great Barrier Reef Marine Park (GBRMP) has recently experienced substantial increases in current and proposed port expansions and subsequent shipping. Globally, large whale populations are recovering from commercial whaling and ship strike is a significant threat to some populations and a potential welfare issue for others. Within the GBRMP, there is spatial conflict between the main breeding ground of the east Australian humpback whale population and the main inner shipping route that services several large natural resource export ports. The east coast humpback whale population is one of the largest humpback whale populations globally, exponentially increasing (11% per annum) close to the maximum potential rate and estimated to reach pre-exploitation population numbers in the next 4–5 years. We quantify the relative risk of ship strike to calving and mating humpback whales, with areas of highest relative risk coinciding with areas offshore of two major natural resource export ports. We found females with a dependent calf had a higher risk of ship strike compared to groups without a calf when standardized for group size and their inshore movement and coastal dependence later in the breeding season increases their overlap with shipping, although their lower relative abundance decreases risk. The formalization of a two-way shipping route has provided little change to risk and projected risk estimates indicate a three- to five-fold increase in risk to humpback whales from ship strike over the next 10 years. Currently, the whale Protection Area in the GBRMP does not cover the main mating and calving areas, whereas provisions within the legislation for establishment of a Special Management Area during the peak breeding season in high-risk areas could occur. A common mitigation strategy of re-routing shipping lanes to reduce risk is not a viable option for the GBRMP due to physical spatial limitations imposed by the reef, whereas speed restrictions could be the most feasible based on current ship speeds.

Keywords: spatial risk assessment, ship strike, great barrier reef, humpback whale, shipping, AIS, breeding ground

INTRODUCTION

Shipping is one of the world's largest industries and extremely important to world economic trade, accounting for 80% of global trade by volume and more than 70% of its value (UNCTAD, 2019). The world shipping fleet has been continuously growing since the 1990s and has doubled in number over the last 12 years, with ships increasing in both size and designed speed capacity to accommodate this trade growth (UNCTAD, 2018). Globally, seaports and other restricted waterways (like canals) are expanding and adapting to meet changes in the industry, resulting in infrastructure expansion projects tied to evolving development plans to take advantage of regional and global opportunities. Shipping is one of the most extensive and pervasive uses of the marine environment, which is exacerbated in coastal areas due to increased interaction with other human uses (i.e., fishing) and protected marine species (Tournadre, 2014). Marine protected areas (MPA's) are recognized as one of the best ways to conserve and protect marine habitats and species in our oceans (Kelleher, 1999). The management of multiple-use marine parks though, particularly in World Heritage Areas, requires a balance between conserving the natural properties of the area and providing for increasing or shifting socio-economic demands for development. Marine spatial planning and spatially explicit risk assessments are important management tools to balance these interests and manage multiple users.

While there are a range of potential impacts associated with shipping activity (e.g., groundings, collisions, oil and chemical spills and introduction of invasive species), ship strike and noise pollution have the greatest impact on marine mammals. Ship strike and ship noise are the main, current anthropogenic threats to whales worldwide (Cates et al., 2017; Erbe et al., 2019) due largely to the global increase in shipping. While increases in shipping traffic have resulted in the rise in ambient noise at low frequencies (10–100 Hz) in many ocean regions, ships also emit significant energy at higher frequencies (10 of kHz) and can therefore have potential impacts on low frequency specialist (e.g., baleen whales) as well as higher frequency specialists (e.g., odontocetes) (Erbe et al., 2019). However, the impacts from ship noise are less tangible than that of ship strike. Ship strikes represent a conservation concern for some whale species in their recovery from 20th century commercial whaling, and a welfare issue for other species exhibiting significant population recovery and increasing interactions with vessels. Quantifying the population-level extent of ship strike mortality on whales, however, is notoriously difficult due to inherent reporting biases and because collisions with large vessels are frequently unnoticed and consequently go unreported (Laist et al., 2001; Panigada et al., 2006; Vanderlaan and Taggart, 2006; Peel et al., 2018). The most well-documented example of ship strike having a detrimental population-level effect on whale recovery is that of the North Atlantic right whale (*Eubalena glacialis*), with the major cause of population decline directly linked to ship strike (Laist et al., 2001; Laist et al., 2014). In contrast, for other whale species (e.g., humpback whale, *Megaptera novaeangliae*) that show strong recovery toward pre-exploitation population levels, ship strike is less of an impact at the population-level and more of a potential welfare issue at the individual-level as a result of non-fatal injuries (Bejder et al., 2016). Analysis of records worldwide (Vanderlaan and Taggart, 2006) and within Australia (Peel et al., 2018), demonstrate that humpback and right whales are the most frequently reported species involved in ship strikes. In Australia, despite the lack of reported incidents involving large ships (one reported case), there are indications that collisions between large ships and humpback whales occur and that the number of reports do not reflect the number of incidents. This is demonstrated by photographs of live humpback whales showing significant wounds consistent with propeller cuts from large ships and stranding events resulting in mortality of humpback whales with wounds suggestive of large ships given the nature and severity of the wound (Peel et al., 2018).

In Australia, the Great Barrier Reef (GBR) is a UNESCO World Heritage Area (GBRWHA) covering approximately 348,000 km^2, within which the Great Barrier Reef Marine Park (GBRMP) comprises 99% of this area. The Marine Park is managed by the Great Barrier Reef Marine Park Authority (GBRMPA) as a multiple-use marine park, which supports a wide range of activities such as tourism, defense, fishing, boating and shipping. The GBRMP is recognized as one of the world's best managed marine protected areas (UNESCO, 2012), although management of it is complex due to overlapping State and Federal jurisdictions and that sometimes the two levels of government are politically ideologically opposed. The GBR is designated a Particularly Sensitive Sea Area (PSSA) by the International Maritime Organization (IMO) because of its potential risk of damage from international shipping activities. Consequently, shipping is well regulated through an established Vessel Traffic Service (REEFVTS) monitoring system and mandatory vessel reporting for vessels >50 m in length. In 2014, a two-way shipping route through the GBR was formalized by the IMO, which predominantly follows previous ship traffic patterns and now provides well-defined lanes to enhance the safety and efficiency of shipping (**Figure 1**). However, there are current and projected increases in shipping throughout the GBRWHA, predominantly due to the export of natural resources. Australia is one of the world's largest exporters of natural resources, with approximately 87% of Australia's total cargo in 2014–15 attributed to international exports predominantly of coal and liquefied natural gas (LNG) (Bureau of Infrastructure, Transport and Regional Economics [BITRE], 2017). Due to substantial coastal development and port expansions related to the mining industry, UNESCO is closely monitoring Australia's commitment to the sustainability of the GBR as a World Heritage Area.

Concurrent to Australia's growth in shipping, the eastern Australian population of humpback whales is one of the world's fastest growing population of humpback whales. The population has been undergoing an exponential rate of recovery (approx. 11% increase per annum) over the last couple of decades after facing near extinction from commercial whaling (Noad et al., 2016) and their breeding ground also occurs in the GBRMP (Smith et al., 2012). In 2015, the estimated population size was 25,000 whales and projected to be ∼41,000 whales in 2020. There is little evidence of slowing, with estimates of recovery ranging between 58–98% due to uncertainty of the historical abundance

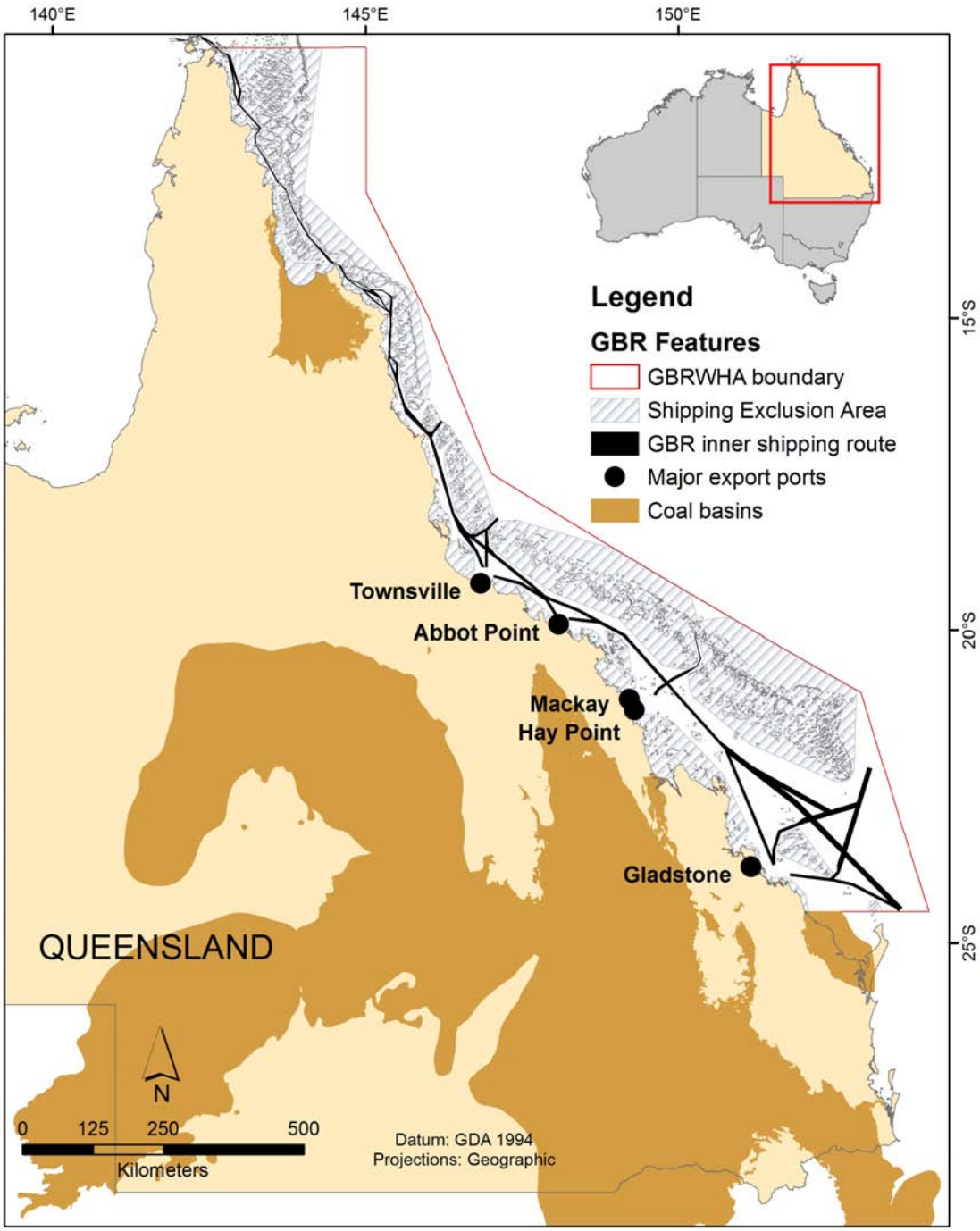

FIGURE 1 | Spatial extents of coal basins, locations of major coal export ports in the State of Queensland, and two-way inner shipping route within the Great Barrier Reef World Heritage Area.

(Noad et al., 2008; Bejder et al., 2016; Noad et al., 2016). Their core breeding aggregation overlaps with the inner shipping route that services all ports on the Queensland coast (Smith et al., 2012). Ship strikes involving large vessels and whales can result in death or serious injury to individuals with the level of risk depending on whale density, behavior, the time of year, vessel density and vessel speed (Cates et al., 2017). With increased shipping activity and whale population size, there is concern for an increased risk in whale fatalities from vessel strikes and increases in non-fatal injuries. While ship strike is unlikely to have a population-level effect on humpback whales in the GBR given the increasing population size, their increased interaction with ships on their

breeding ground is likely to be an emerging management issue that could result in welfare issues to the whales from non-fatal injuries (Peel et al., 2018).

To understand the risk of ship strike to whales, it is necessary to understand both distribution and densities of whales and shipping. We provide a spatially explicit ship strike risk framework using humpback whales on their breeding grounds in the multiple-use GBRMP. We modeled relative ship strike risk for whales involved in two different reproductive behaviors of calving (groups containing the presence of a calf) and mating (groups without a calf present), to determine whether there were spatial differences in risk related to reproductive behavior. We compare ship strike risk before and after the formalization of the two-way inner shipping route, to evaluate the effect that defined lanes of ship traffic has on risk to the whales. Finally, we modeled future projected risk of ship strike to humpback whales based on an annual rate of whale population increase for several growth rates in ship traffic. The quantitative risk assessment of ship strike to whales allowed an evaluation of current measures of protection for humpback whales in the multiple-use marine park and potential mitigation measures available to reduce risk.

METHODOLOGY

Aerial Surveys

The GBRWHA is a large area (348,000 km^2), which makes systematic surveys of the entire area prohibitively costly. Based on a predictive spatial habitat model that was developed using opportunistic presence-only whale sighting data (Smith et al., 2012), line transect aerial surveys were undertaken in 2012 and 2014. The aerial surveys sub-sampled specific regions of the GBRWHA according to their own specific objectives. The 2012 aerial survey was designed to validate the predictive spatial habitat model by surveying three main areas predicted to have low, medium and high habitat suitability, at a time representing peak whale abundance during the breeding season. The aerial survey was undertaken over 8 days (3rd to 10th August) with a total areal coverage of 63,723 km^2; Mackay (34,626 km^2), Townsville (17,126 km^2) and Port Douglas (11,971 km^2). On-effort flight time was 15.75 h, of which 97.1% of time conditions were in Beaufort sea-state $\leq$3. The objective of the 2014 aerial survey was to determine the coastal distribution of humpback whales around major coastal/port areas within a region in the GBRWHA of high whale density later in the breeding season, past peak whale abundance when there are more females with newborn calves. The survey was undertaken offshore of Gladstone and Mackay over 11 days (26th August – 5th September), with a total areal coverage of 72,752 km^2. On-effort flight time was 18.3 h, of which 98.8% of time was undertaken in Beaufort sea-state $\leq$3. The aerial surveys were undertaken using a Partenavia Observer P-68B 6-seater, twin engine, high-wing aircraft and a double platform observer configuration. Rear observers were acoustically and visually (using curtains) isolated from the front observers to allow perception bias to be calculated. Whale sightings included species identification, declination (using a Suunto PM-5/360PC clinometer) and horizontal (protractor) angles to the group, total number of animals' visible, number of calves and sighting cue.

Species Distribution Model

The distribution and densities of humpback whales in the GBR from the 2012 and 2014 aerial surveys were modeled using the method described in Hedley and Buckland (2004). This requires a detection function fitted to the sighting data to estimate the "effective strip width," to create a detection-adjusted density surface model using generalized additive models (GAM's). Detection probabilities, and corrections for perception bias, were estimated using Mark-Recapture Distance Sampling models as described in Laake and Borchers (2004) and Burt et al. (2014) using the MRDS package (Laake et al., 2015) in R (R Development Core Team, 2015). To improve detection function fit, perpendicular sighting distances were left truncated at 0.2 km and right truncated at 4 km, and sightings of uncertain species identification were excluded from the analyses. A final detection function was selected using Akaike Information Criterion (AIC) and examining model diagnostics. A density surface model was then developed using a GAM model by segmenting track lines into pre-defined lengths of approximately 10 km to capture adequate environmental variability using functions for spherical geometry from the R "geosphere" library (Hijmans, 2016). Values of each environmental covariate were converted into rasters in ArcMap 10.1 (ESRI) and matched to the midpoints of each along-track segment. The numbers of whale groups and total animals (including the presence and number of calves) were then summed and a total effective strip area estimated for each segment. These models use a smooth over geographical space, informative environmental covariates and an offset term provided by the effective strip area of each segment. A Tweedie distribution was used to account for over-dispersion in the counts of groups per segment. Collinearity in the various spatial/environmental covariates were assessed using multi-panel scatterplots and Pearson correlation coefficients. All sightings that were included in the distance analyses were used to fit the spatial models. Uncertainty in the estimation of the detection function was incorporated into the variance of the spatial model using a method described in Williams et al. (2011) and Miller et al. (2013). This procedure involves fitting the density surface model with an additional random effect term that characterizes the uncertainty in the estimation of the detection function, via the derivatives of the probability of detection with respect to their parameters.

Physiographic variables of water depth, seabed slope, and (geodesic) distances to the nearest coastline and reef features were estimated for the midpoints of each along-track segment. Monthly mean values of dynamic remotely sensed environmental predictor variables were interpolated to the midpoint of each along-track segment. Daily sea surface temperature (Integrated Marine Observing System [IMOS], 2015a; in °C, gridded at 0.02°), sea surface height anomaly (IMOS, 2015b; in meters, gridded at 0.58° >0.51°), and sea surface chlorophyll *a* (IMOS, 2015c; mg m-3, gridded at 0.01°) values for the GBR region were averaged at each grid point for the month of August in 2012 and August and September in 2014. Predictions of whale densities

across the GBR were undertaken at a 1 × 1 km grid cell resolution to produce density models for three different whale groups: (1) all whales, (2) groups that contained a calf (hereafter, calf groups) and (3) groups in which a calf was not present (hereafter, non-calf groups). Sightings and modeled distributions of calf groups are used as a proxy to identify likely calving areas and non-calf groups to identify potential mating areas. A 1 × 1 km grid cell size was chosen to provide enough spatial resolution to distinguish a specific shipping lane and to avoid the issue of vessels and animals not in close proximity being classed as co-occurring and contributing to risk within the spatial risk assessment.

Shipping Data

All large vessels transiting through the GBRWHA are monitored with AIS by the REEFVTS and ships are only permitted to transit through Designated Shipping Areas. In December 2014, the IMO formalized a two-way shipping route in the GBR that extends from the Torres Strait in the north and terminates at the southern boundary of the GBRMP (**Figure 1**). The two-way shipping route follows pre-existing traffic patterns through the GBR and now encourages shipping to follow well-defined northbound and southbound lanes, although it is not mandatory to travel within these lanes.

AIS data were obtained from AMSA in the form of their craft tracking system (CTS) product, which provides processed ship locational data sampled to a 5 min frequency. AIS data were analyzed for each year between 2013 and 2016, which covered the time period when systematic aerial surveys for humpback whales were undertaken and the formalization of the two-way shipping route. Shipping data were restricted to Class "A" cargo, tanker and passenger vessels ≥80 m in length for 3 months of the humpback whale breeding season (July, August, and September). Only vessels ≥80 m were included for the following reasons: vessels of this size and larger predominantly inflict fatal or severe injuries (Laist et al., 2001), larger vessels traverse predictable routes, AIS data provides relatively accurate ship positional data and previous risk assessments of ship strike to whales (e.g., Redfern et al., 2013) have focused on larger vessel size classes. The AIS data did not have navigational status of the vessel available, which can be used to filter out vessels not underway (e.g., anchored). Consequently, we applied a filter of >0.4 knots to the data to remove stationary/anchored vessels that will have limited risk for ship strike.

To use the AIS data in the risk assessment framework, we created trackline data from the point data representing 5-min AIS positions of each individual vessel based on a unique ship-related identifier, the Maritime Mobile Service Identity (MMSI). This converts the data in each cell from time to distance data. Trackline data were created by joining contiguous unique point positions of ship locations less than 60 min apart, with the exception of positions separated in time between 30 and 60 min with a change in ship's course over ground greater than 5 degrees (due to uncertainty of the ship's path of travel). Positions greater than 60 min apart were excluded. The 1 × 1 km whale density grid over the entire GBRWHA region was used to summarize the distance traveled by ships within each grid cell from the trackline data.

Risk Modeling Framework

To quantify relative risk of ship strike we calculated the Relative Expected Fatality (REF) of a whale from the risk of a ship strike. This incorporates a measure of co-occurrence of a whale and ship in a given grid cell (Redfern et al., 2013), and uses vessel beam as an exposure factor and the equation from Conn and Silber (2013) to estimate the probability of a lethal whale strike given vessel speed. This approximates the risk of a fatal ship strike more accurately than co-occurrence alone, because the severity of a ship strike is related to the speed of a vessel. A whale risk index was calculated by multiplying the ship and whale density with the mean vessel beam and the probability of a lethal whale strike given the mean vessel speed for each grid cell for each of the years 2013 to 2016. We summarized the risk for each year and three whale group categories (all whale groups, calf and non-calf groups). The cumulative total, mean, minimum, and maximum risk observed were calculated and the estimates were then standardized to account for differences in the number of vessels between years by dividing the risk estimates by the total km's traveled by all vessels in the GBR. Relative risk was also summarized at a decreased resolution of 50 × 50 km grid cells to identify risk patterns at the broader regional scale. To investigate whether there was a change in the risk of ship strike to humpback whales due to the IMO formalization of the inner GBR two-way shipping route, relative risk was compared before (2013/2014 ship data) and after (2015/2016 ship data) the formalization.

Projected Future Risk of Ship Strike

Predicting future relative risk based on projected growth rates can be difficult because it is uncertain how increases in shipping and whale population size will change temporally or spatially. To predict future relative risk we assumed that there are no changes to the spatial distribution of ships or whales, which is likely to be more uncertain for whales due to an increasing population size (e.g., through range expansion) than for shipping that follow formalized shipping lanes. We calculated future risk for each grid cell by multiplying an annual proportional increase of whale abundance (11%) and five ship traffic growth rates around a 3.5% expected mean growth rate, from 1.5 to 5.5%. The expected growth rate of the whale population can be considered robust due to surveys since the 1990s producing consistent estimates of approximately 11% per year (Noad et al., 2016). The Australian Bureau of Resources and Energy Economics (BREE) predicted the average annual growth in coal ship traffic between 2011 and 2025 in the GBR, according to a range of likely scenarios; optimistic was 6.31, 5.12% was moderate, 3.71% low and 3.06% the most conservative case (Braemar Seascope, 2013).

RESULTS

Humpback Whale Distribution Model

There were a total of 637 sightings of humpback whale groups from the combined aerial surveys, 365 group sightings (589 individuals) in 2012 and 272 (461 individuals) in 2014 (**Figure 2**). The breakdown of calf groups between years and mean whale encounter rates are in **Table 1**. There was a lower

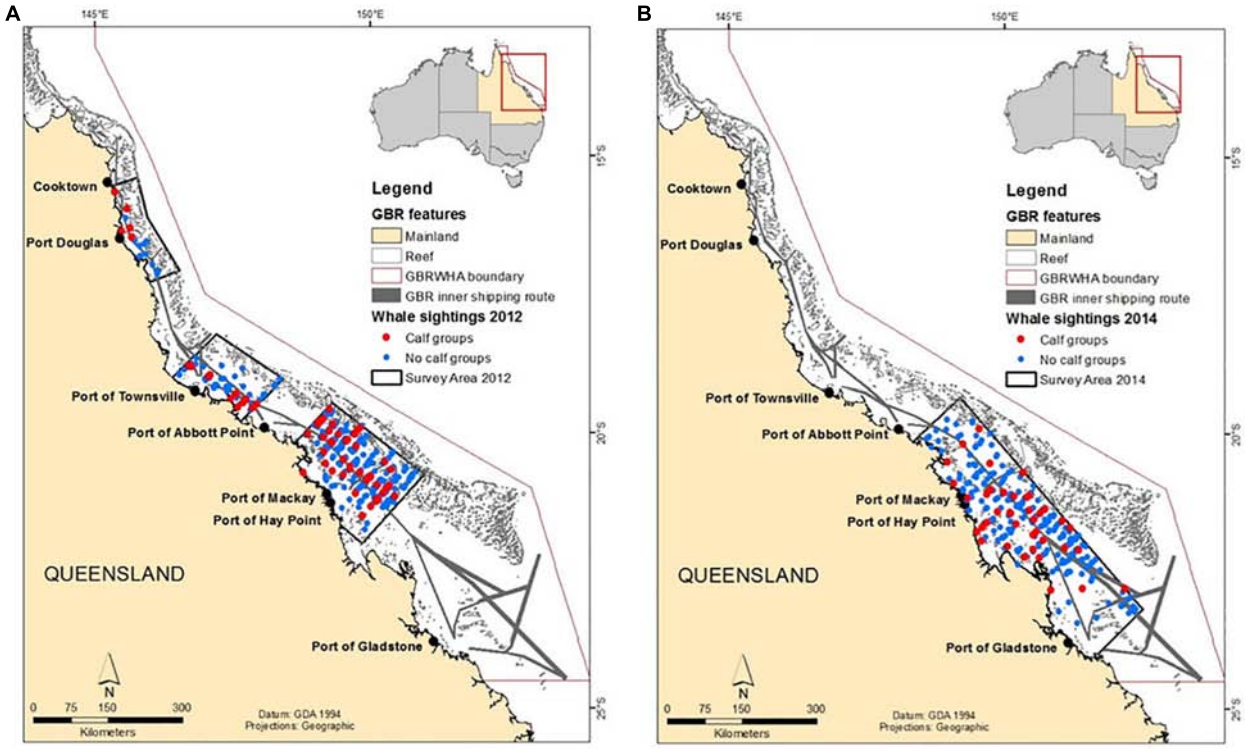

FIGURE 2 | Distribution of humpback whale sightings of calf and non-calf groups during the **(A)** 2012 and **(B)** 2014 aerial surveys.

relative abundance of whales in 2014 compared to 2012 due to undertaking the aerial survey later in the breeding season, past the expected peak of whale abundance. The detection function was fit using sighting data pooled across both survey years and a total 561 sightings remained after truncation of the data were used for density surface modeling.

The prediction from the best density surface model of humpback whale density for each of the three reproductive categories are in **Supplementary Figure S2**. Due to only a small amount of survey effort in bathymetric values of 90 m and deeper (only 122 km of a total of 6650 km across both survey years), no density predictions were made for waters deeper than 90 m. The most significant parameters in describing humpback whale distribution and density were depth and SST. The models predicted higher densities of humpbacks in shallow water (e.g., 20–60 m deep) and within a sea surface temperature range between 21 and 23°C.

The predicted distribution of whale densities for all whale groups combined in the GBR followed a similar pattern for both 2012 and 2014 (**Supplementary Figure S1**). Two main areas of higher whale density during peak whale abundance are located approximately 120 km to the north and southeast of Mackay (**Figure 3**). The modeled distribution of calving areas (sightings of groups with a calf present) in 2012, and to a lesser extent in 2014, occurred throughout the length of the GBR whereas mating areas (groups without a calf) were predominantly restricted to the southern GBR (**Figure 3**). However, given calf groups were sighted among non-calf groups in inshore and offshore waters, there does not appear to be any distinct separation of calving versus mating areas. The highest number of whale sightings was in the southern GBR region, although the northern GBR region offshore of Cairns had a proportionally higher calf-to-adult ratio (1:4) compared to the southern GBR offshore of Mackay (1:7.9) (**Figure 2**). In 2014, there was a significant change in the distribution of calf groups closer to the coast compared to non-calf groups (**Figures 2**, **3**). If we compare the distribution of calf groups in 2012 to those in 2014 (**Figures 3B,D**) and assume little inter-annual variation in whale distribution, the predicted distribution suggests that groups with a calf move closer to the coast later in the breeding season.

Shipping Data

There was a slight increase in the number of ships per year between 2013 (N = 1466) to 2016 (N = 1687) and no detectable within year variation, such that the 3 months within the year were comparable. The majority of class A vessels (≥80 m in length) used in the analysis over the four years were cargo vessels (87%), followed by Tankers (12%) and a small number of passenger vessels (1%). There was a consistent pattern in the length of the vessels across all years that ranged from 80 (the minimum cut-off) to 300 m, with a higher frequency of vessel length closer to larger sized ships (mean = 205 m and median = 222 m). There was also a consistent pattern in vessel beam with a mean and median of 32 m (range = 10–50 m). The average vessel speed was 12.6 knots (median = 12.4 knots, max = 25 knots) with 85% of vessel transits faster than 10 knots

TABLE 1 | The number of sightings, relative abundance and mean encounter rates for groups of all whales and groups containing a calf during the 2012 and 2014 aerial surveys.

	2012		2014	
	All whales	Calf groups	All whales	Calf groups
Total number of sightings	365	100	272	59
Total number of individuals	589	121	461	218
Relative abundance	7487	2440	3627	648
Mean encounter rate (SE)	0.143 (SE 0.003)	0.033 (SE 0.0009)	0.122 (SE 0.004)	0.016 (SE 0.0007)

and 11% faster than 15 knots (74% >10 knots ≤15 knots). Vessel speeds greater than 15 knots occurred in specific areas of the GBRMP, specifically the Whitsunday Islands offshore Abbott Point port and Gladstone port.

Relative Ship Strike Risk Maps

Formalization of IMO Two-Way Shipping Route

Given there was little change in shipping numbers between years and a higher relative abundance of whales in 2012 compared to 2014, there was a higher average relative risk of ship strike (two-fold increase) at the peak abundance of the season in 2012 compared to later in the breeding season in 2014 (**Supplementary Figure S3**). A comparison of ship strike risk to whales pre- and post-formalization of the inner shipping route shows minimal difference in the risk to whales (**Supplementary Figure S3**). Fundamentally, the two-way route formalized existing traffic patterns into well-defined shipping lanes, such that there was little variation in shipping traffic distributions pre and post the IMO formalization.

Spatial Risk of Ship Strike to Humpback Whales

At the time of peak whale abundance on the breeding grounds (2012 whale model), the areas of higher relative risk of ship strike to humpback whales occur in areas where shipping traverses two areas of higher predicted whale density (**Figures 3A,B**, **4**) in the southern GBR. All patterns of risk were consistent across all years of shipping data, due to negligible differences in ship numbers between years. At the finer spatial resolution (1 × 1 km) several areas of high ship strike risk (>80%) were identified, including offshore of the Port of Abbott Point and Port of Mackay/Hay Point (**Figure 4**). At the coarser spatial resolution (50 × 50 km), the areas of high risk (>80%) were restricted to the one location offshore of the Port of Mackay/Hay Point thus corresponding to a greater area of high risk (**Figure 4**).

Overall, cumulative risk of ship strike for humpback whales at the group level is higher for non-calf groups compared to groups with a calf, due to there being significantly more sightings of non-calf groups (75%) compared to calf-groups (25%). However, when standardized for the total number of whales in each group, the risk was consistently higher for groups containing females with a dependent calf in both 2012 and 2014 (**Supplementary Figure S4**). During peak abundance within the breeding season (using the 2012 whale model), there was consistency in the areas of high risk of ship strike for both calf and non-calf groups in areas located in the southern GBR lagoon, offshore of the Port of Hay Point and Mackay (**Figures 5A,B**). However, as the breeding season progresses there was a change in the spatial distribution of groups with a calf from offshore to inshore waters (**Figures 3B,D**). This resulted in a spatial change in risk to a greater area of overlap with the shipping lane and coastal waters. While the area of ship strike risk for groups with calves increased later in the breeding season (**Figures 5B,D**), there was a reduction in the level of risk due to a decrease in relative whale abundance.

Projected Ship Strike Risk

The population of whales is increasing at an exponential rate and concurrently there are current and projected increases in shipping in the GBR. We calculated a projected risk of ship strike to humpback whales over a 10-year period based on an 11% annual increase in whale population size (Noad et al., 2016). We used four different shipping traffic growth rates based on a conservative (1.5%) to optimistic (5.5%) estimate and projected forward to 2028. Based on the different shipping traffic growth rates, there is predicted to be between a three (conservative ship growth) and five-fold increase (optimistic ship growth) in the risk of ship strike to humpback whales in the GBR within the next 10 years (**Figure 6**).

DISCUSSION

Management of multiple-use marine parks and World Heritage Areas requires a balance between conservation and socio-economic demands for development. Spatially explicit risk assessments provide the ability to manage multiple users of the marine environment, reduce environmental impacts and reduce conflict among users (Hope, 2006). Within the GBRWHA, there is considerable overlap between shipping lanes and the breeding aggregation of humpback whales for which shipping traffic and whale population size are both increasing. Furthermore, the expansive physical structure of the GBR limits the ability to segregate these two uses of the Marine Park and implement a common mitigation measure of re-routing shipping channels away from Biologically Important Areas. It is unlikely that ship strikes will have a population-level effect on the whales given the population is increasing close to its maximum potential rate (∼11% per annum). However, there is concern

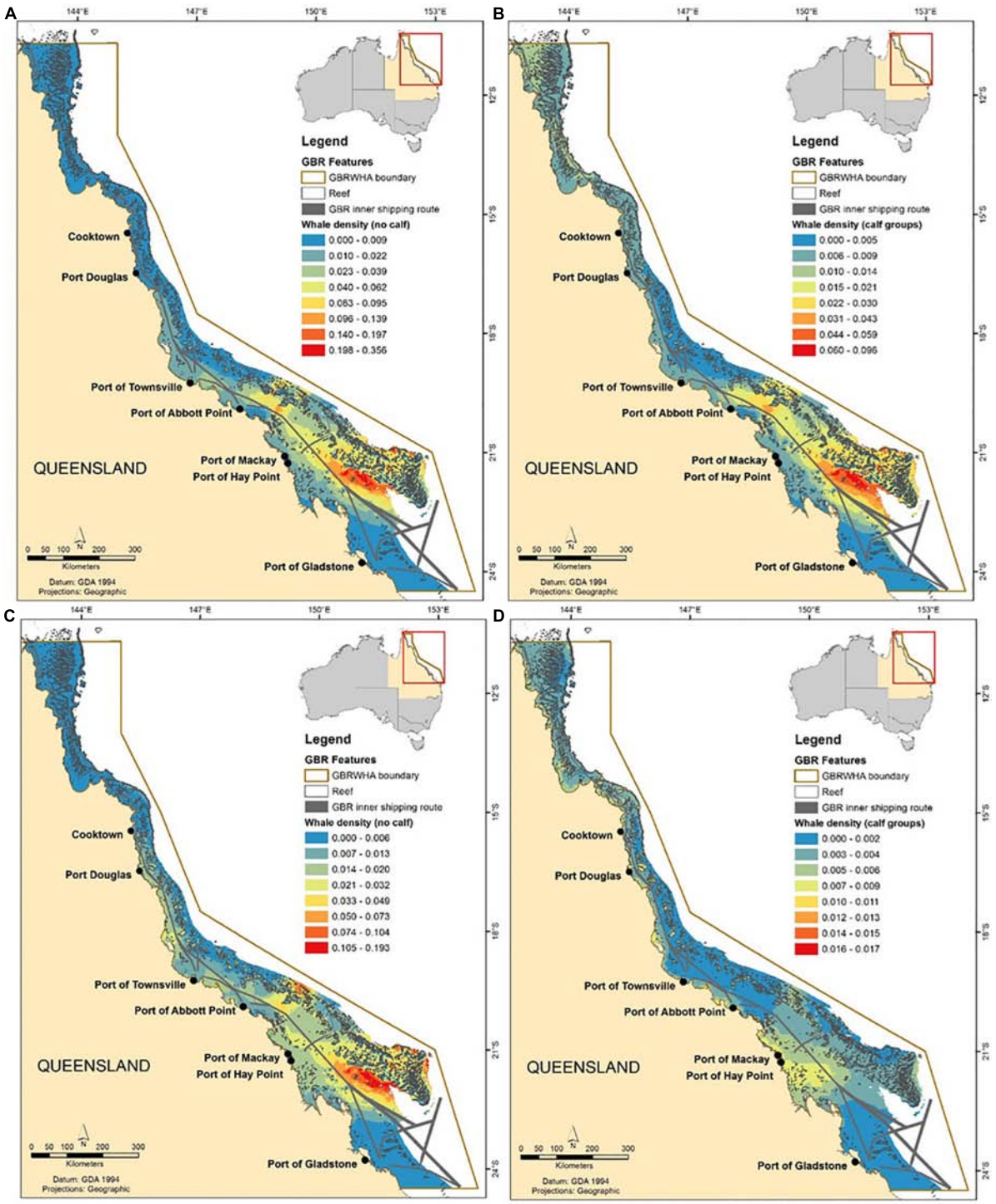

FIGURE 3 | Modeled distribution and densities of humpback whale **(A)** non-calf groups and **(B)** calf groups in August 2012 and **(C)** non-calf groups and **(D)** calf groups in September 2014 in the Great Barrier Reef Marine Park with the overlayed inner shipping route.

for a potential increase in whale fatalities from ship strikes, and welfare concerns arising from non-fatal injuries, due to greater interaction between breeding whales and ships. The spatially explicit risk assessment has identified specific areas within the GBRMP of higher relative risk of ship strike to whales from large commercial ships. This should provide the basis to evaluate the level of threat to whales from ship strike and focus future research areas to aid

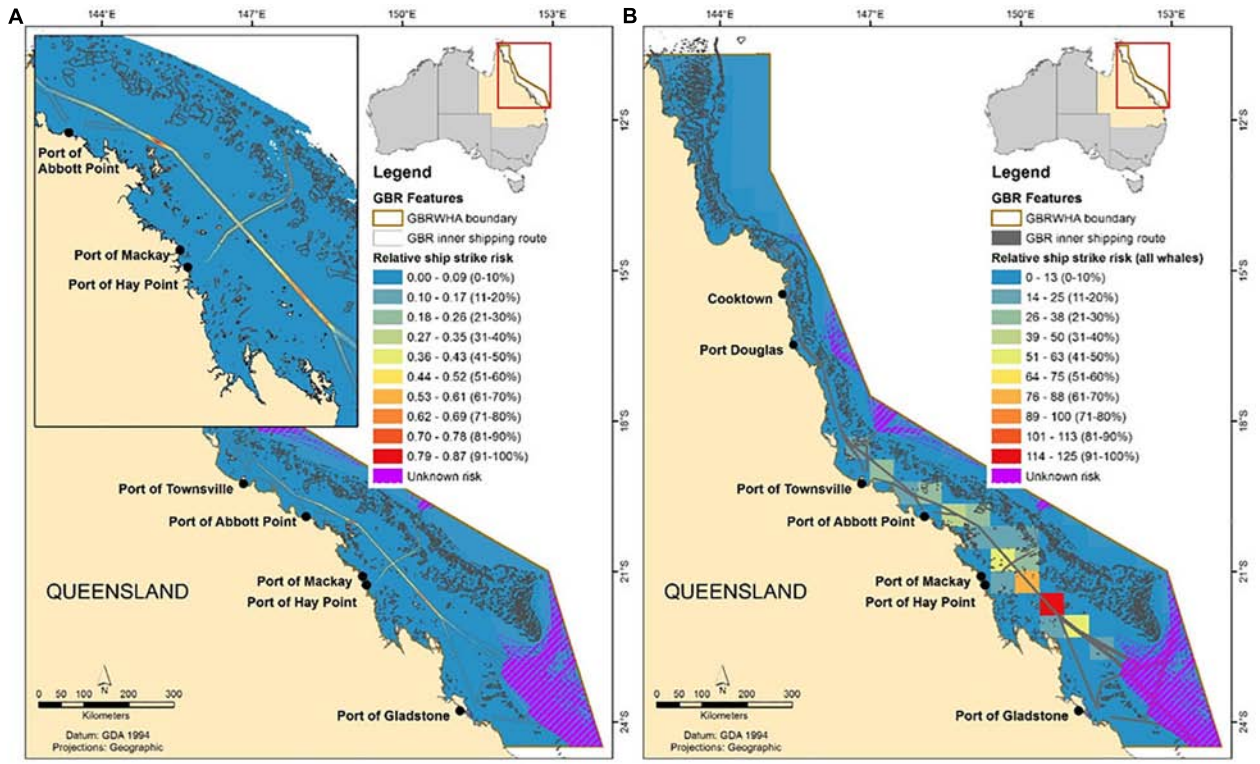

FIGURE 4 | Relative ship strike risk for all humpback whale groups in the GBRWHA based on the 2012 whale model and 2016 shipping data at **(A)** 1 × 1 km and **(B)** 50 × 50 km grid cell resolution.

informed management decisions on the types of mitigation measures necessary.

The Importance of Spatial Resolution on Risk Estimates

To understand the risk of ship strike to whales, it is necessary to understand both the distribution and densities of whales and shipping. Generally, there will be a degree of uncertainty when quantifying both of these. Accurately identifying whale distribution and density within and between years for a mobile marine species' is difficult without considerable sampling effort, and shipping traffic can vary based on specific port activities and global economic factors. This is an important consideration when undertaking spatially explicit risk assessments and identifying an appropriate spatial resolution for the data. Often shipping lanes are only several km's in width and necessitate high-resolution data (e.g., 1 × 1 km). However, uncertainty in whale distribution data may not support such high resolution. In our study we sub-sampled whale distribution within the GBRMP then modeled, and extrapolated on 2 years of survey data, which incorporates a certain degree of uncertainty in whale distribution. The spatial resolution also depends on the spatial scale over which management decisions are being conducted (e.g., tens, hundreds or thousands of km). In the case of the GBRMP, we advocate the large area of the Marine Park (344,000 km^2), covering 14 degrees of latitude, requires undertaking the spatial risk assessment at a resolution coarser (e.g., 50 × 50 km) than what AIS shipping data necessitates (e.g., 1 km). Finer spatial resolution in the data might be required for localized, small-scale applications such as port developments, whereas regional management planning and zoning would necessitate coarser resolution.

We quantified risk at a fine scale (1 × 1 km) and coarse scale (50 × 50 km) and **Figure 4** demonstrates the effect that spatial resolution can have on identifying areas of risk. At the finer spatial resolution, several areas of high ship strike risk (>80%) were identified, including the area offshore of Abbott Point. While this is an area of high risk, the extent of it covered approximately 20 km of the shipping lane and consequently at the coarser spatial resolution it did not comprise an adequate proportion of the area as high risk. Consequently, the areas of high risk were restricted to the one location offshore of the Port of Mackay/Hay Point.

Higher Ship Strike Risk to Females With Calves

The relative risk of ship strike differed for whales of different reproductive class, with groups without calves having a higher overall cumulative risk of ship strike. This was due to there being significantly more non-calf groups (75% in total) than groups with calves (25%). However, when risk is standardized for the total number of whales in each group type, the risk is higher throughout the GBRMP for groups with a female and

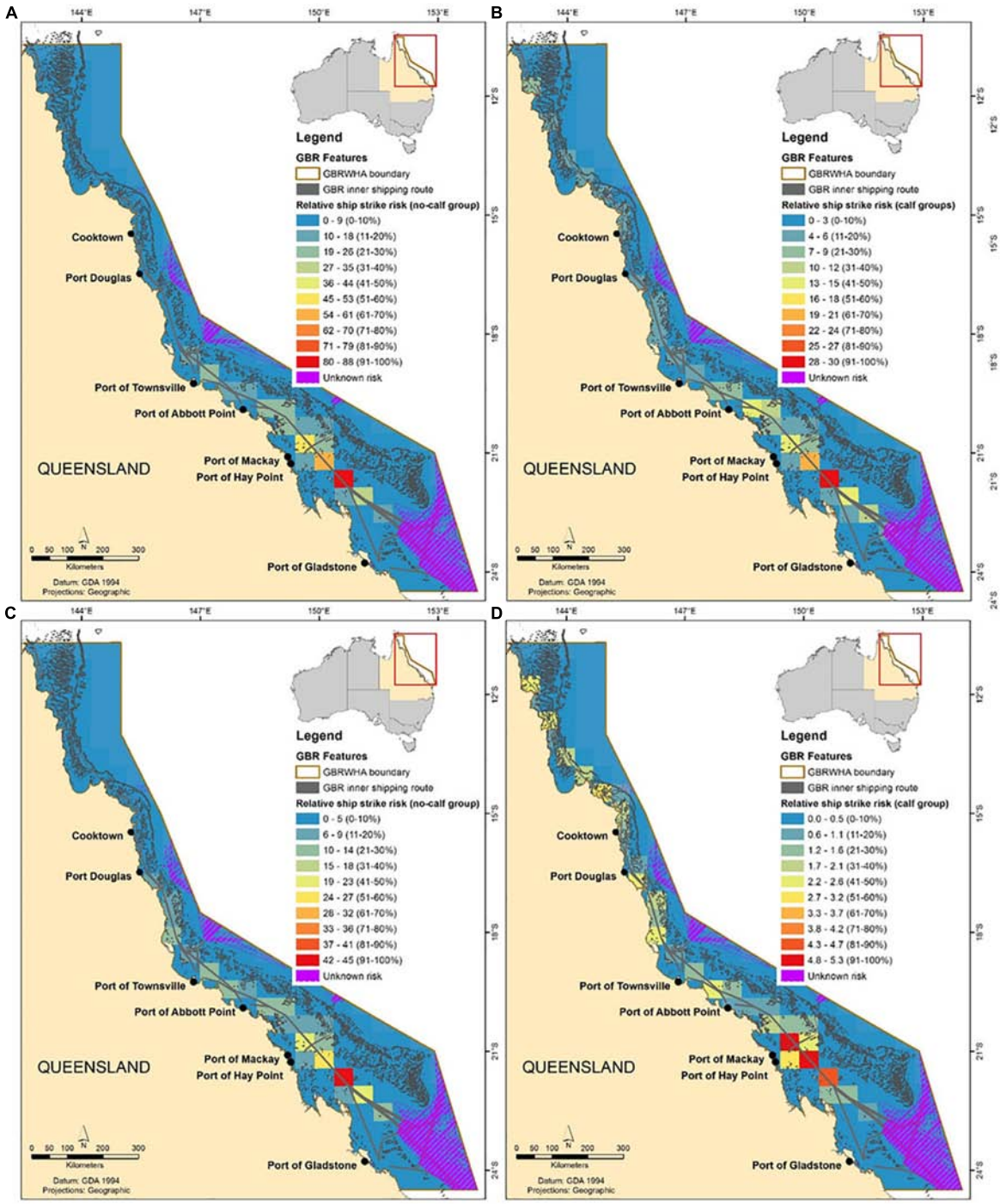

FIGURE 5 | Relative ship strike risk for humpback whale **(A)** non-calf groups and **(B)** calf groups in August 2012 and **(C)** non-calf groups and **(D)** calf groups in September 2014 based on 2016 ship data at a 50 × 50 km grid cell resolution.

dependent calf compared to non-calf groups. While there was no distinct separation of calving versus mating areas, given calf groups were sighted among non-calf groups in both inshore and offshore waters, calving areas (based on sightings of groups with calves) occurred throughout the length of the GBR and mating areas (groups without a calf) were predominantly restricted to the southern GBR (**Figure 2**). This northern GBR had a higher calf-to-adult ratio (1:4) compared to the southern GBR (1:8) offshore of Mackay.

During peak whale abundance within the breeding season, areas of relative high ship strike risk to calf and non-calf whale groups were consistently identified in the southern GBR lagoon

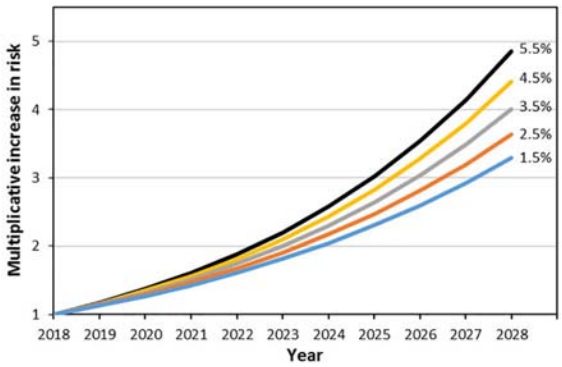

FIGURE 6 | Projected increase in relative expected fatality based on an annual 11% increase in whale population size and various projected annual shipping traffic increases (colored lines).

offshore of the Port of Mackay and Hay Point (**Figures 4A,B**). Later in the breeding season when more females have given birth and the southward migration away from the breeding ground has started, the relative abundance of whales decreases and there is a shift in the distribution by groups with females and a dependent calf from offshore to coastal, inshore waters (**Figures 2**, **3D**). This assumes minimal inter-annual variation in whale distribution, which seems plausible given non-calf groups in 2014 occurred in a similar area to non-calf and calf groups in 2012. Females with calves from several other populations of humpback whales (e.g., Ecuador, Hawaii, Brazil) also display a preference for coastal, shallow water habitat on their breeding grounds (Félix and Botero-Acosta, 2011; Craig et al., 2014; Guidino et al., 2014; Gonçalves et al., 2018; Pack et al., 2018). The shift in distribution of calf groups to inshore waters resulted in greater overlap with the shipping lane, and an increase in the area of higher (>80%) ship strike risk. However, the level of risk was considerably lower in September compared to peak whale abundance in July/August (**Figure 5**) due to a lower relative abundance. Currently, the ship strike risk framework does not incorporate whale behavioral data that could differ among age and social classes (e.g., vessel avoidance). Estimates of risk will be affected if certain classes of whales exhibit behavioral attributes that make them more or less susceptible to ship strike. For example, calf groups could be more at risk of ship strike compared to non-calf groups if they have a higher level of exposure to a ships' strike zone as a consequence of dive behavior e.g., frequent shallow dives. The risk framework does not incorporate time spent at the surface due to insufficient behavioral data.

Formalization of Two-Way Shipping Route and Projected Ship Strike Risk

From December 2014, the IMO formalized the inner two-way shipping route through the GBR. While the route in the northern GBR was existing, a new section was added to the southern GBR that corresponded to existing shipping traffic patterns. Therefore, patterns in shipping traffic through the GBR predominantly remained unchanged. Consequently, there has been little difference in ship strike risk to whales resulting from the formalization of the two-way route. Implementation of the formalized shipping lane had the greatest change to the distribution of shipping traffic offshore of Gladstone, in the area of unknown risk. This area could not be modeled due to insufficient whale data, although is likely a high risk area for ship strike given the large export volume of LNG from Gladstone port and the multiple shipping routes crossing a high density of whales undergoing a constrained migration movement in this region (Smith et al., 2012).

Future estimates of shipping volume in the GBR suggests a potential increase of 4–5% annual growth rate, based on Qld port industry forecasts over the period 2012–2032 for all vessels and ports (PGM Environment, 2012). Projected ship strike risk based on conservative (1.5%) to optimistic (5.5%) ship traffic growth rates show a three to five-fold increase in risk to whales over the next 10-year period (**Figure 6**). Over this time period, there is a potential doubling of the humpback whale population size from the current estimate of 25,000 whales (Noad et al., 2008; Bejder et al., 2016; Noad et al., 2016). The current population size is estimated to have reached over 50% of pre-exploitation levels, with estimates of recovery ranging between 58–98% due to uncertainty of the historical abundance (Noad et al., 2008; Bejder et al., 2016; Noad et al., 2016). Of particular interest over the next 10 years for the East Australian population of humpback whales is the population recovery trajectory, with the possibility of a population leveling to an uncertain carrying capacity between 26,000 to 42,000 whales (Ross-Gillespie et al., 2015). This highlights the necessity for understanding natural versus anthropogenic impacts on the recovery of whale populations.

The relative ship strike risk maps identified the southern half of the GBRMP from Townsville to north of Gladstone (approx. 19°S–22°S), including the east-west Hydrographers Passage route offshore of Mackay, to have the largest relative risk within all of the Marine Park (**Figure 4**). The whale density models show these areas correspond to where shipping traverses two higher predicted whale density areas (**Figure 3**). This encompasses four major trading ports and likely to also include Gladstone. These five ports make up the majority of export trade, particularly of natural resources such as coal, along the GBR coast representing 78% ($51.75 billion) the total throughput of all Qld. ports (Queensland Government Statistician's Office, 2017). The Qld. commodity market is currently, and into the future, dominated by the trade of coal and liquefied natural gas (LNG), with Australia currently the second largest global exporter of LNG. Consequently, the risk to whales is only likely to increase unless there is significant downturn in coal and LNG exports. This highlights the importance of an informed understanding of the threats to the population (e.g., ship strike).

Conservation Implications and Current Protective Measures

Ship strike of whales is a global issue that has resulted in various management measures aimed at reducing the risk to whales. A spatial management approach commonly implemented involves the establishment of time and area specific modifications, for example Seasonal Management Areas

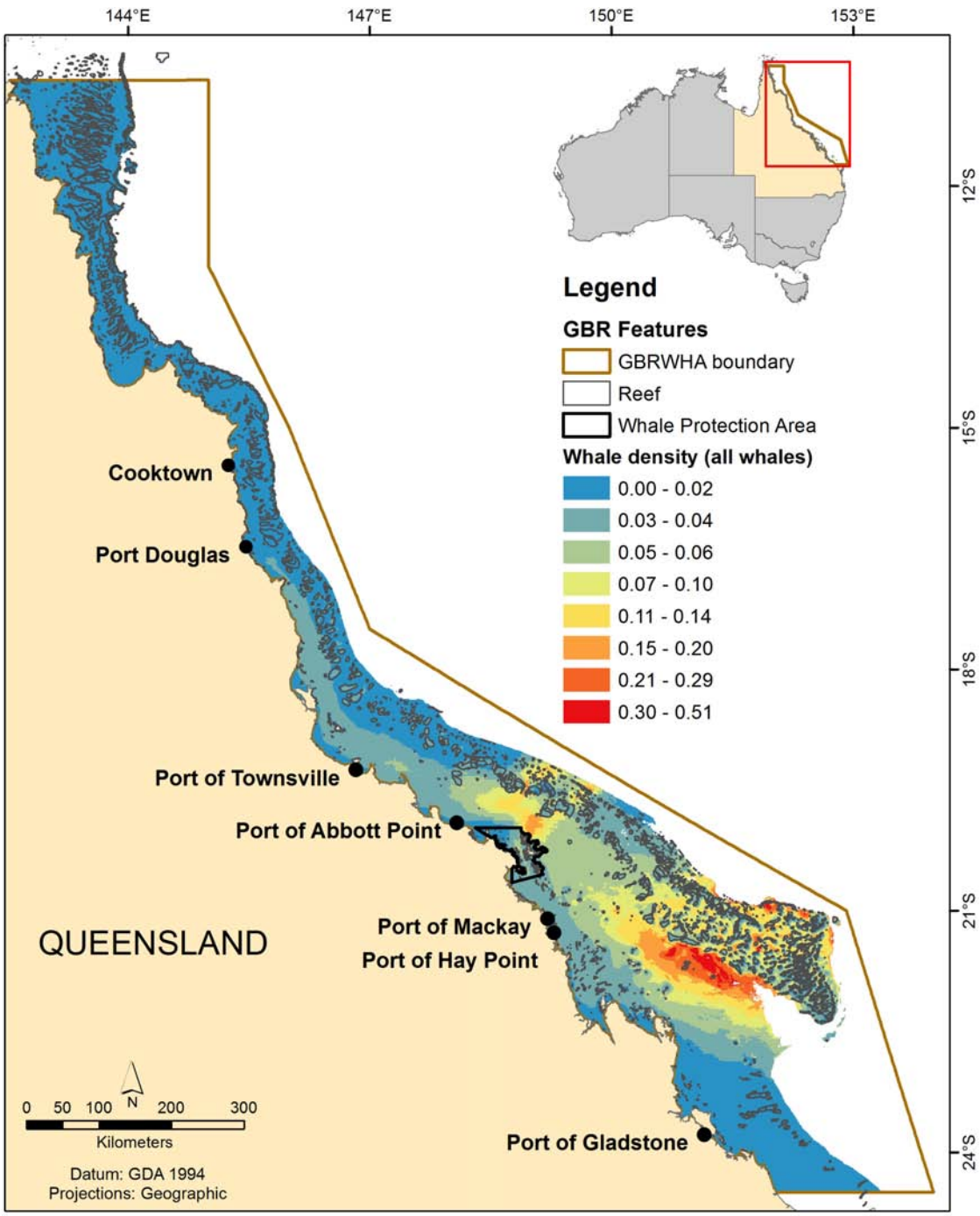

FIGURE 7 | Map of the whale protection area and modelled distribution and density of 'all whale' humpback whale groups during August 2012, with red areas indicating high whale density.

and Traffic Separation Schemes (TSS) (Silber et al., 2012). Vessel routing and speed restrictions have both been shown to reduce the probability and severity of ship strikes (Vanderlaan and Taggart, 2006; Vanderlaan and Taggart, 2009; Wiley et al., 2011; Conn and Silber, 2013; Laist et al., 2014). Within the GBRMP, mitigation options are more limited because the extensive reef structure of the GBR constrains ship traffic movement between the reef and the coastline. This significantly limits the viability of re-routing measures due to the limited space within the Designated Shipping Area (**Figure 1**). Furthermore, there is a dynamic temporal component to the distribution of whales throughout the

breeding season, with movement of calf groups from offshore to inshore waters.

A feasible management option within the GBRMP is the designation of a Special Management Area (SMA) for which there is provision under the *Great Barrier Reef Marine Park Regulations 2019* for purposes outlined in the *Great Barrier Reef Zoning Plan 2003*, which include the conservation of a particular species or resource e.g., aggregation sites. Species Conservation areas are a type of SMA that have been implemented for dugongs in the GBRMP to restrict human activities and minimize disturbance. Currently, a Whale Protection Area (WPA) (which is not an SMA) exists for whales in the Whitsunday area to restrict the distance that vessels can approach breeding whales and minimize disturbance. However, this area was primarily established to manage tourism vessels involved in whale watching and other tourism activities. The WPA clearly does not cover the areas of highest density of whales and greatest risk of ship strike to breeding whales in the GBR (**Figure 7**). The relative risk maps have identified areas that represent sufficient risk to breeding humpback whales and warrants consideration of suitable mitigation to reduce the risk. Current legislation provides the opportunity to establish a Species Conservation area as part of a SMA that could help focus management effort. Specific mitigation options could be the focus of further research into understanding the magnitude of the threat of ship strike to humpback whales and could range from voluntary reporting of whale sightings by onboard observers to mandatory speed restrictions. AMSA in partnership with the IMO could impose seasonal speed restrictions in targeted areas to reduce ship strike risk. Speed restrictions could be a viable and cost-effective management option given the evidence that vessel speed reductions of large vessels to $\leq$10 knots significantly reduces the risk of ship strike (Vanderlaan and Taggart, 2006) and many vessels already travel close to that speed in the GBRMP (74% of vessel transits are between 10 and 15 knots and only 11% >15 knots).

Currently, the ship strike framework provides a relative metric across the study area useful for comparing relative risk, although cannot be inferred as an estimation of actual mortality. Calculating absolute risk is currently problematic due to insufficient knowledge on many parameters associated with ship strike (e.g., response/avoidance behavior of whales to vessels), large uncertainty/variance related to species spatial distributions (e.g., intra- and inter-annual variability) and unknown parameters that have not been modeled (e.g., survivorship from blunt force trauma). The framework does enable managers to assess different scenarios of speed restrictions and its effect on risk, due to incorporating vessel speed as a factor in the risk calculation. To improve the ship strike framework, incorporation of different vessel characteristics (e.g., vessel draught and potential depth of strike zones including hydrodynamic effects) and whale behavior (e.g., time at surface and avoidance) are required, if and when, data are available. While the true relationship between relative and absolute risk remains unknown, these data provide the best source of information to aid in the identification of potential hotspots of high interactions between whales and shipping in the GBR.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

The animal study was reviewed and approved by the Murdoch University Animal Ethics Committee permits ((O2487/12 and O2680/14).

AUTHOR CONTRIBUTIONS

JS conceived the fieldwork design, collected the data, and led the writing of the manuscript. JS, NK, SC, DP, JR, and TM analyzed the data. All authors contributed critically to the drafts and gave final approval for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00067/full#supplementary-material

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Using Satellite AIS to Analyze Vessel Speeds Off the Coast of Washington State, U.S., as a Risk Analysis for Cetacean-Vessel Collisions

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Most species of whales are vulnerable to vessel collisions, and the probability of lethality increases logistically with vessel speed. An Automatic Identification System (AIS) can provide valuable vessel activity data, but terrestrial-based AIS has a limited spatial range. As the need for open ocean monitoring increases, AIS broadcasts relayed over earth-orbiting satellites, satellite AIS (SAIS), provides a method for expanding the range of AIS broadcast reception. We used SAIS data from 2013 and 2014 to calculate vessel density and speed over ground around the coast of Washington state in the northwestern United States. Nearby shipping lanes connecting the Ports of Seattle, Tacoma, Portland, and in Canada, Vancouver, have the greatest density of vessel traffic arriving and departing. Knowledge of shipping activity is important in this area due to the nearby presence of NOAA designated Cetacean Density and Distribution Working Group's Biologically Important Areas (BIA) for large whale species vulnerable to vessel collisions. We quantified density and speed for each vessel type that transits through BIA's. We found that cargo and tanker vessels traveled the farthest distance at the greatest speeds. As ship-strike risk assessments have traditionally relied on terrestrial AIS, we explored issues in the application of SAIS data. Temporal gaps in SAIS data led to a resulting systematic underestimation of vessel speed in calculated speed over ground. However, SAIS can be helpful in documenting minimum vessel speeds across large geographic areas and across national boundaries, especially beyond the reach of terrestrial AIS receivers. SAIS data can also be useful in examining vessel density at broad scales and could be used to assess basin-wide open ocean routes. Future use of additional satellite platforms with AIS receivers and technological advances will help rectify this issue and improve data coverage and quality.

Keywords: satellite automatic identification system, cetacean ship strikes, geographic information systems, olympic coast national marine sanctuary, baleen whales, biologically important areas, shipping speed

INTRODUCTION

Shipping, a highly globalized industry, is an important component of international trade, and is correlated with global economic patterns (Schwehr and McGillivary, 2007; Rodrigue, 2010; Frisk, 2012). While the world's vessel fleet is growing over time (Silber et al., 2010), so does the need to effectively track vessel movements. The Automatic Identification System (AIS) is a non-proprietary tracking technology standardized by the International Telecommunications Union and required for most vessels by the International Maritime Organization's Safety of Life at Sea Convention (Tetreault, 2005; United States Coast Guard [USCG], 2008). Originally conceived to improve ships' navigational safety, terrestrial AIS broadcasts are limited roughly to line-of-sight (Calder and Schwehr, 2009), so coverage does not extend well into the open ocean (Silber et al., 2014). Satellite AIS (SAIS), since 2008, can help overcome the terrestrial line-of-sight limitation by collecting AIS broadcasts from a constellation of earth-orbiting satellites (Ball, 2013; Robards et al., 2016).

Despite the initial intent of AIS as a navigational aid, the data provide valuable insight on human use of marine areas in the context of environmental conservation. Events like the establishment of large off-shore protected areas clarify the need to extend the range of traditional AIS broadcasts to better monitor vessel activity far from shore (McCauley et al., 2016). SAIS data have been employed to monitor fishing activity and protected area regulation compliance (Natale et al., 2015; de Souza et al., 2016; Rowlands et al., 2019). Terrestrial AIS has provided valuable data to assess the risk of collision between ships and whales near established shipping lanes in coastal areas (Williams and O'Hara, 2010; Wiley et al., 2011; Redfern et al., 2013; Jensen et al., 2015), but lethal interactions may also occur outside of these highly monitored areas (Rockwood et al., 2017). For off-shore areas, SAIS has the potential to contribute vessel activity data necessary for collision risk assessments (Williams and O'Hara, 2010; van der Hoop et al., 2012).

Since 2009, the International Whaling Commission has reported more than 1,200 confirmed incidents of vessel collisions with whales (Cates et al., 2017). Vessel speed is an important component of shipping's potential impact on cetaceans (Gende et al., 2011; Conn and Silber, 2013; Currie et al., 2017). The conservation benefit to whales by reducing vessel speed is well established and is generally expressed as a simple logistic relationship between vessel speed and probability of lethality (Vanderlaan and Taggart, 2007; Gende et al., 2011; Wiley et al., 2011; Conn and Silber, 2013). There is a significant positive relationship, and the greatest rate of change generally occurs between 9 and 15 knots, corresponding roughly to an increase in probability of lethality from 20 to 80% (see Vanderlaan and Taggart, 2007). Vessel speed limits help to reduce the anthropogenic mortality risk and possibly collision probability (Gende et al., 2011; Conn and Silber, 2013). AIS provides data on vessel speed which has been combined with density to examine threats of collision (Felski et al., 2015; Rockwood et al., 2017).

Our objective was to use available SAIS data to delineate spatial locations where vessel traffic density and speed were high in known areas of cetacean concentrations off the coast of Washington state. The use of SAIS data was necessary because only a little more than half of the study area water was within the potential range of terrestrial AIS.

In Washington state waters, there were 19 of 130 (15%) strandings from 1980 to 2006 that showed evidence of collisions with vessels. Numerous biases in collision detection lead to underestimates in true numbers of mortalities (Douglas et al., 2008), so the actual number is likely higher (Williams et al., 2011). There are Biologically Important Areas (BIAs) in this research area for gray (*Eschrichtius robustus*) and humpback whales (*Megaptera novaeangliae*) (Calambokidis et al., 2015; **Figures 1**, **2**). BIAs are species, region, and time specific areas which the National Oceanic and Atmospheric Administration (NOAA) Fisheries, Cetacean Density and Distribution Working Group has identified as important for reproduction, feeding, migrating, or small and residential populations (Ferguson et al., 2015).

At the time of this writing, NOAA has declared an active Unusual Mortality Event (UME) for gray whales along the United States west coast from California to Alaska indicating above-average mortality rates: 34 strandings in Washington state waters and 10 in Canadian waters between January and September of 2019 (National Oceanic and Atmospheric Administration [NOAA], 2019). The gray whale BIAs in our study area were based on migratory corridors between annual feeding and reproductive areas, from numerous survey methods and expert opinion (Calambokidis et al., 2015; **Figure 2**). Assessing patterns of vessel behavior in these areas can help managers evaluate potential impacts to this vulnerable population suffering from elevated mortality events. While the United States Fish and Wildlife Service removed the species-level listing of humpback whales on the Endangered Species List, some unique migratory populations are still listed as Threatened or Endangered (United States Fish and Wildlife Service [USFWS], 2016). The humpback whale BIAs were based on surveys and opportunistic sources about highly concentrated feeding animals (Calambokidis et al., 2015; **Figure 2**). We used these BIAs as a proxy for areas of high cetacean concentration for each species.

SAIS can now facilitate a complete global picture of vessel activity (Skauen, 2019). The growing number of satellites in orbit capable of transmitting SAIS data suggests that temporal gaps between satellite passes over an area will be minimized (McCauley et al., 2016). If this trend continues, it may become increasingly more attractive to use SAIS data instead of terrestrial AIS data due to its spatial coverage. We took this opportunity to investigate and evaluate potential issues related to monitoring vessel density and speed using SAIS data that may need to be rectified before more pervasive use of the technology can occur. AIS is an imperfect data source as it is limited by human input, data corruption, signal noise, GPS faults, and gyrocompass or other instrument failure onboard the target vessel (Aarsæther and Moan, 2009; McGillivary et al., 2009; Silber and Bettridge, 2010; Robards et al., 2016). Further, gaps in SAIS data are also evident (Allen et al., 2018; Eriksen et al., 2018), especially in areas of high activity due to satellite congestion (Jia et al., 2019). Despite these issues, both AIS and SAIS provide insights on vessel activity otherwise unavailable.

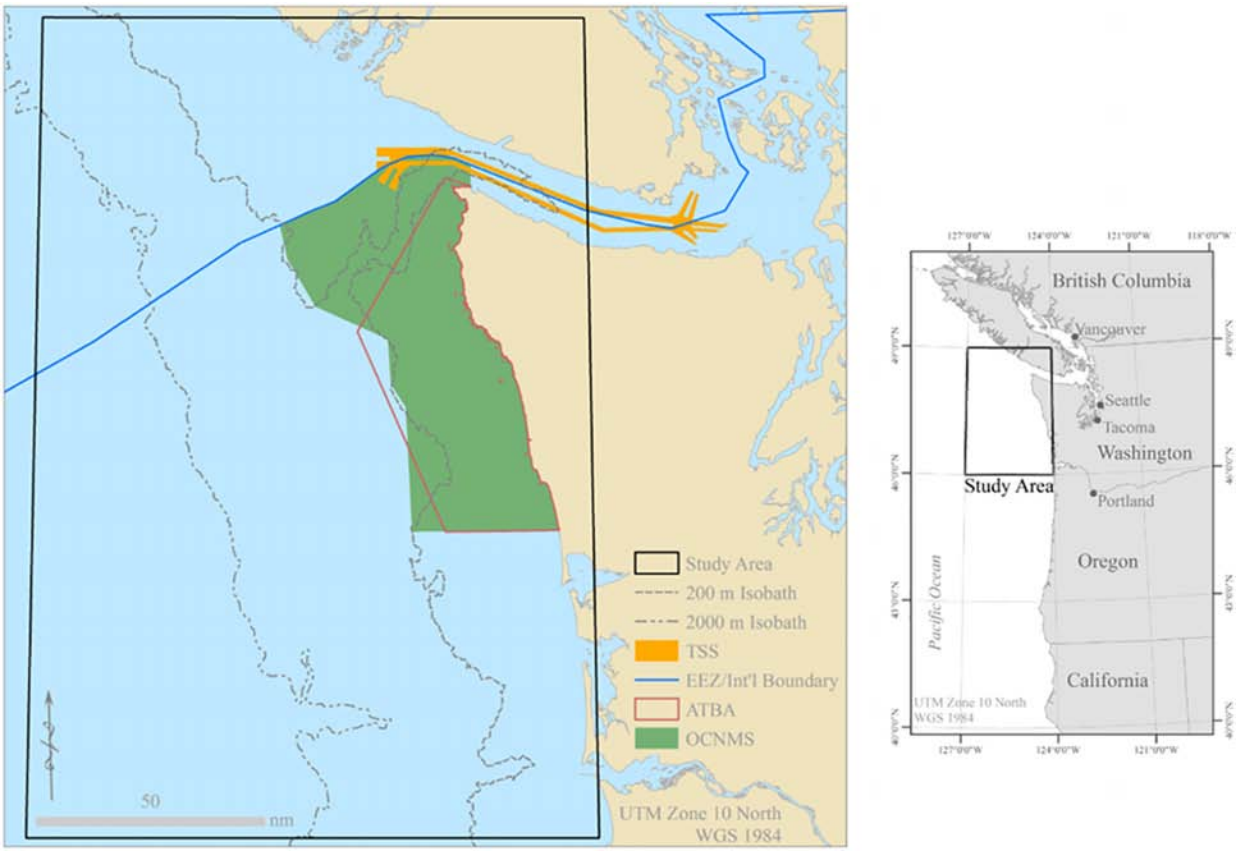

FIGURE 1 | Bathymetry and administrative areas in the study area along the Pacific Northwest coast including Washington state in the United States and southwest Vancouver Island in Canada. Note: Traffic Separation Scheme (TSS), Exclusive Economic Zone (EEZ), Area To Be Avoided (ATBA), Olympic Coast National Marine Sanctuary (OCNMS).

In this research, we provide a baseline understanding of vessel activity off the Washington coast as a foundation for future risk assessments. Extensive research on risk to North Atlantic right whales on the United States east coast has shown the effectiveness of reduced speed and routing alternatives in reducing whale mortality from vessel collisions (Laist et al., 2014). However, the vessel patterns within United States waters outside the Strait of Juan de Fuca has still not been studied. Multiple species of large whales off the coast of Washington state are vulnerable to collisions (Douglas et al., 2008; Silber et al., 2010). This geographic focus area is important due to its connections to several primary ports and the presence of multiple species of slow-reproducing whales recovering from past population declines.

MATERIALS AND METHODS

Study Area

Our research site was offshore from the important North American west-coast ports of Vancouver, Seattle, Tacoma, and Portland (**Figure 1**). The specific study area was between 46–49°n and 124–127°W, and defined the extent of SAIS data collection. Reaching 90 to 125 nm offshore of the state of Washington in the northwestern United States, the study area extended from roughly the mouth of the Columbia River in the south to the Strait of Juan de Fuca and southern Vancouver Island in the north.

Inside the Strait of Juan de Fuca are the Ports of Tacoma and Seattle, major United States shipping ports, and Port Metro Vancouver (**Figure 1**). Port Metro Vancouver is the largest Canadian port, handling roughly the same amount of total tonnage as the Port of New York and more than Seattle and Tacoma combined (United States Army Corps of Engineers [USACE], 2016; Port Metro Vancouver, 2017). Tacoma and Seattle ranked 7th and 10th, respectively, in the United States for total container ship traffic in 2013 and 29th and 31st, respectively, for total tonnage in the United States in 2014 (United States Army Corps of Engineers [USACE], 2015, 2016). These ports connect internationally to East and Southeast Asia, and domestically to Alaska, Hawaii, and the West Coast of the United States. Additionally, the Port of Portland is located inland from the mouth of the Columbia River. Portland ranked 25th among United States ports for total container ship traffic in 2013 and 28th for total tonnage in 2014 (United States Army Corps of Engineers [USACE], 2015, 2016).

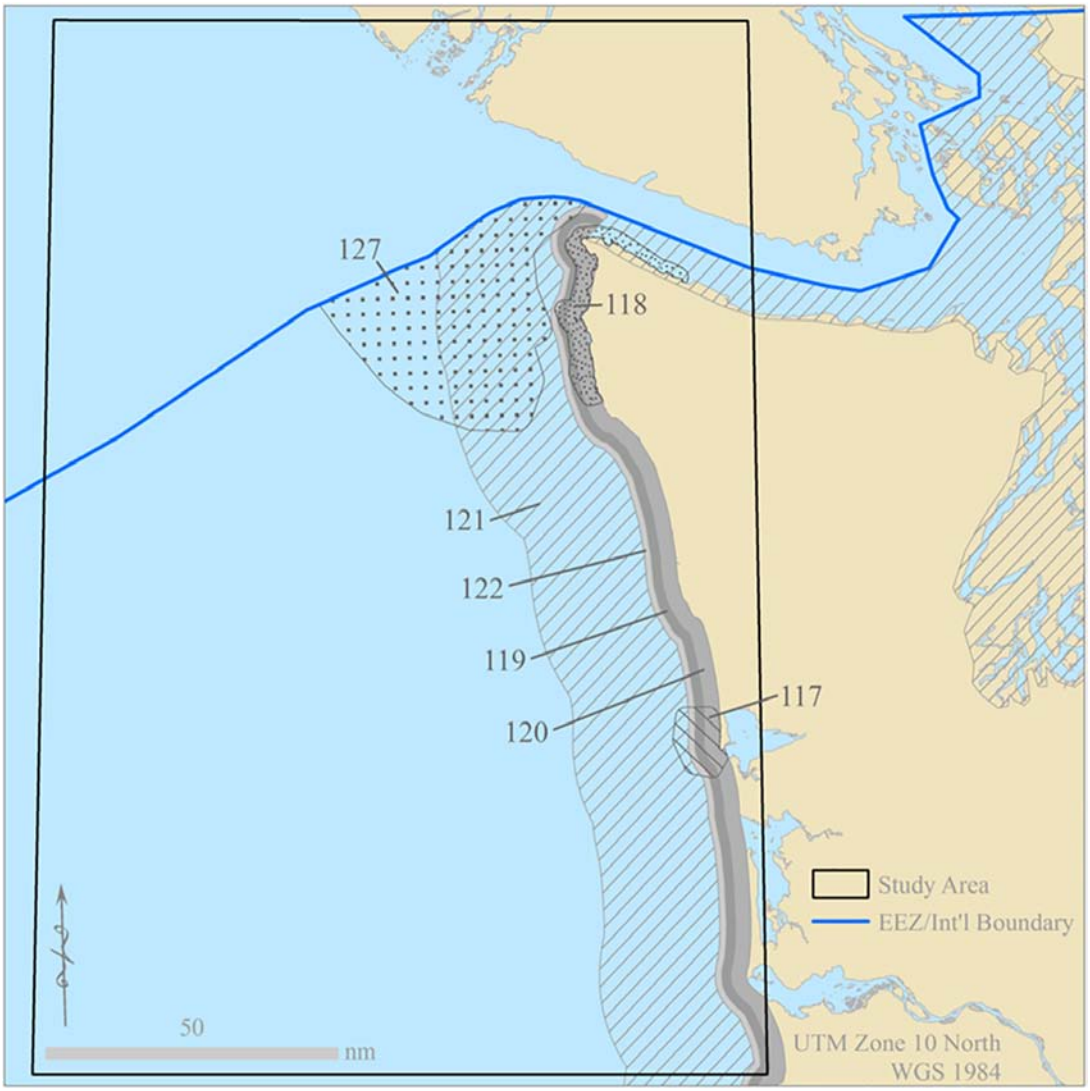

FIGURE 2 | Biologically Important Areas (BIA) (see **Table 1** for species and type) for gray (*Eschrichtius robustus*) and humpback whales (*Megaptera novaengliae*) along the coast of the North American Pacific Northwest (Calambokidis et al., 2015). All BIAs are for gray whales with the exception of BIA 127 for humpback whales.

Within the full 75,367 km^2 study area, 88% is open water and 12% is land. Administrative areas in the study area include the NOAA administered Olympic Coast National Marine Sanctuary (OCNMS), an International Maritime Organization (IMO) designated Area to be Avoided (ATBA), and part of the United States Coast Guard (USCG) controlled Juan de Fuca Traffic Separation Scheme (TSS) (**Figure 1**). Within the study area, there are seven BIAs, six for gray whales and one for humpback whales (**Table 1**; and **Figure 2**).

SAIS Information

Satellite AIS data were collected by exactEarth Ltd. (Cambridge, ON, Canada) for the calendar years 2013 and 2014, and received from the OCNMS. Each record in the tables corresponded to an individual AIS broadcast. AIS information is comprised of static information that does not change over the course of a voyage, and dynamic information that can change as frequently as every AIS broadcast.

The original SAIS data that were received had 3,045,407 records for the year 2013 and 2,941,900 for 2014. The years 2013 and 2014 had consistent fuel sulfur regulations (10,000 ppm or 1.0%) for vessels operating within the North America Emissions Control Area, 200 nm from the coast (Environmental Protection Agency [EPA], 2010). Thus, there were no temporal changes in traffic patterns based on emissions controls standards during our study period (Jensen et al., 2015). A further reduction in sulfur regulations to 0.1% occurred in 2015 (International Maritime Organization [IMO], 2019). Although evidence from California suggests that ships may alter their speed based on new regulations (Moore et al., 2018), differences in shipping patterns are seen to principally reflect longer term economic changes (Jensen et al., 2015). The results of this study are meant to inform management

TABLE 1 | Summary of Biologically Important Areas (BIA) within the North American Pacific Northwest (Calambokidis et al., 2015).

Number	Species	Name	Type	Months	Are a within study area (nm)^2	Description
117	Gray whale	Grays Harbor	Feeding	April–November	86.8	Outside Grays Harbor
118	Gray whale	Northwest Washington	Feeding	May–November	150.1	N o rthwest tip o f Olympic Peninsula
119	Gray whale	Northbound Phase A	Migration	January–July	652.3	Within 8 km of coast
120	Gray whale	Northbound Phase B	Migration	March–July	393.8	Within 5 km of coast
121	Gray whale	Potential Presence	Migration	Jan.–July, Oct.–Dec.	4111.8	Within 47 km of coast
122	Gray whale	Southbound	Migration	October–March	828.7	Within 10 km of coast
127	Humpback whale	Northern Washington	Feeding	May–November	989.2	Northern part of United States EEZ

of past trends and provide insight on the use of SAIS data for evaluating vessel activity.

Data Preparation

We conducted data quality control, starting with removing duplicate SAIS records, defined as records having the same Maritime Mobile Service Identity (MMSI), latitude, longitude, and time. The MMSI is a unique, regulated, and coded identifier for a ship. The second quality control step was to remove all records with a missing or null MMSI. Next, we created a tabular relationship between dynamic SAIS information and static vessel information, using the MMSI as a primary key. We used ArcGIS (Environmental Systems Research Institute [ESRI], 1999–2018) geoprocessing and the programming language Python (Python Software Foundation, 1990–2019) running in PyScripter (Vlahos, 2005–2015) to write or modify numerous Python scripts.

The first script reprojected the point data into the Universal Transverse Mercator (UTM) Zone 10 North projection based on the Geographic Coordinate System (GCS) World Geodetic System (WGS) 1984, which allows distance to be measured in meters. AIS latitude and longitude are collected from a GPS receiver, which is based on GCS WGS 1984. The second Python script ran a spatial selection of only the SAIS points that were broadcasted from the water, and eliminated random error points located on land (Jensen et al., 2015). We added and calculated new fields for season and day/night based on the time stamp. Seasons were defined as Winter (January–March), Spring (April–June), Summer (July–September), and Autumn (October–December) (Forney and Barlow, 1998; Becker et al., 2014; Jensen et al., 2015). Day and night were defined by using published nautical twilight times from the United States Naval Observatory, Astronomical Applications Department for Forks, WA and Ocean Shores, WA for the years 2013 and 2014. The nautical twilight was defined per month by using the average time from the 15th of each month over the study area (Jensen et al., 2015).

We created ship transit line segments and evaluated transit contiguity (Jensen et al., 2015) by joining sequential SAIS data points from the same vessel to create straight line segments between these points. The time difference and distance between sequential points defined the calculated speed over ground (SOG). While reported SOG from terrestrial AIS systems has previously been used to evaluate threat of collision to cetaceans in coastal areas (van der Hoop et al., 2012; Conn and Silber, 2013), we chose to use a calculated SOG due to frequent gaps in SAIS data. Eriksen et al. (2018) used calculated SOG to identify SAIS records with large temporal or positional gaps, therefore unusable in analysis, by removing vessel transits with implausible speeds. We employed a similar approach (see below). The final part of the script evaluated transit contiguity, a single vessel on a continuous transit, based on MMSI, Trip ID, time between broadcasts, and heading difference.

We merged all 24 months' of line segments and the 22 OCNMS-specified vessel types (see **Table 2**) together to create complete transit lines. We analyzed the years 2013 and 2014 together to simplify calculations, despite differences in overall mean vessel speeds. In 2013, 42.8% of the data were removed due to duplicate records, whereas this was only 5.8% for 2014. We conducted analyses at the vessel type level to minimize inherent inter-type vessel differences.

Calculated Speed Truncation

Truncation of calculated SOG values, an attempt to enumerate the highest possible legitimate speed per vessel type, was necessary because of errors in the data or processing that led to implausible SOG values and means, common in SAIS data (Eriksen et al., 2018). The truncation threshold was an estimate of the maximum plausible and attainable speed for each vessel type. The maximum calculated speed of liquefied gas carriers, cable layers, and pollution control vessels were less than one knot under the truncation threshold. Furthermore, the three vessel types in the tanker category had identical truncation thresholds. Combined, these relative measures of accuracy and precision indicate that the truncation threshold method was a workable approximation for maximum speed.

Truncation meant removing any record where the calculated speed was greater than a given threshold from the following equation:

$$TT_\alpha = \bar{x}_\alpha + (3 \times \sigma_\alpha)$$

where TT_α is the truncation threshold for vessel type α, $\bar{x}_\alpha$ is the mean non-0 broadcast SOG value for type α, and σ_α is the standard deviation of non-0 broadcast SOG values for type α. This equation was derived from examining histograms and statistics of broadcast SOG values and validated by expert opinion (G. Galasso, Deputy Superintendent, Olympic Coast National Marine Sanctuary, pers. comm.). The use of non-zero broadcast SOG values was necessary because 36.7% of all

TABLE 2 | Overall calculated and broadcast speed over ground (SOG) by vessel type for 2013 and 2014 combined.

	Calculated speed over ground			Broadcast speed over ground		
Vessel type	**Mean (Knots)**	**SD (Knots)**	**Count**	**Mean (Knots)**	**SD (Knots)**	**Count**
Cargo						
Bulk carrier	10.5*	4.1	963279	8.5	5.4	976195
Cargo ship	11.2*	3.8	77292	9.5	5.2	78431
Container ship	16.1*	4.7	382776	13.4	7.3	392957
Refrigerated cargo	11.7*	2.6	2851	8.4	5.3	3144
RORO Cargo Ship	16.9*	5.7	45424	14.0	8.1	46857
Vehicle carrier	14.0*	4.9	194421	11.5	6.8	197485
Tanker						
Chemical carrier	11.8*	3.7	51393	9.8	5.5	52312
Liquefied gas carrier	13.5*	3.8	2627	12.8	4.5	2660
Oil tanker	12.4*	3.4	198504	10.3	5.5	201838
Tug						
Articulated tug barge	9.1*	2.7	85510	7.5	4.3	86458
Tug	6.7*	3.2	317997	5.4	3.9	322224
Fishing						
Fishing vessel	3.0*	3.6	1400609	2.5	3.4	1408449
Passenger						
Passenger ship	18.2*	4.7	50239	15.8	7.8	51086
Miscellaneous						
Cable layer	8.1	5.1	2056	8.3	5.1	2061
Dredger	3.6*	3.7	50877	2.5	3.4	51693
Drill ship	3.0	4.4	750	3.2	4.4	752
Pollution control	0.6*	1.9	55869	0.4	1.8	55941
Private vessel	7.5*	5.3	24783	6.2	5.6	25025
Public vessel	6.5*	8.1	276096	6.2	8.4	282194
Research ship	5.0*	4.4	37174	4.4	4.3	37404
Supply ship	10.4	6.6	4306	10.9*	6.7	4531
Unknown	3.8*	3.9	833	3.1	3.9	842

Vessel types are as specified by the Olympic Coast National Marine Sanctuary. The asterisk (*) indicates a statistically significantly higher mean SOG ($\alpha = 0.05$) when comparing calculated versus broadcast SOG within a vessel type.

broadcast SOG values were zero, skewing the mean and inflating the standard deviation.

We overlaid vessel transit lines with truncated average speed values across a range of areas of interest. This included the entire space and time of the study area, and spatial subsets that included the OCNMS, ATBA, and the BIAs during active months. We examined temporal subsets that included day versus night and the four seasons. The overall truncation rate of 1.28% allowed the vast majority of data to be retained for further analysis. However, truncation was necessitated in almost 55,000 records, due to either anomalous location or time stamp broadcasts.

Several vessel types with the highest truncation thresholds (supply ship, container ship, roll-on/roll-off (RORO) cargo ship, and public vessel) also had some of the highest rates of data truncation (**Supplementary Table S1**). The source of these high rates of truncation and any possible correlation within SAIS is unclear.

Hexagon Average Speed and Density

The Olympic Coast NMS has used hexagons, one square statute mile or 2.6 square kilometers in area, as a unit of measurement or observation as part of their spatial planning process (N. Wright, Marine Geographer, Olympic Coast NMS, pers. comm.). There are 29,542 homogenous hexagons for the entire study area, including the ONMS. We calculated mean SOG and number of vessel transits per month for vessel transits across each hexagon that intersected the BIAs.

RESULTS

Total Transits and Distance Traveled

The commercial ports of Vancouver, Seattle, Tacoma, and Portland drove much of the vessel traffic in the study area. Bulk carriers had the most cargo transits. Miscellaneous category vessels without private and public vessels accounted for a very small proportion of vessel transits and distance traveled.

The 42,629 sum total transits of all vessels in the years 2013 and 2014 covered 2,694,197 nm. Fishing vessels account for the most total vessel transits at 26.9%, followed by bulk carriers at 23.5% (**Supplementary Table S2**). The only other vessel type to have greater than 10% of the total transits is container ships (10.4%). Seasonally, as found in Jensen et al. (2015) outside San Francisco Bay, vessel speeds did not change overall, with the exception of passenger ships that traveled more slowly in the autumn and winter, although still at speeds greater than 15 knots (**Supplementary Table S3**).

The cargo category made up 41.7% of total transits, the most of any category. With the exception of public vessels and private vessels, most vessel types in the miscellaneous category registered very few transits. The remaining seven vessel types in the miscellaneous category only account for 2.2%. Vessels in the cargo and tanker categories averaged 27.7 transits per day.

Bulk carriers accounted for the most distance traveled by any one vessel type (32.0%). Only fishing vessels (16.8%) and container ships (13.2%) accounted for more than 10% of total distance traveled. Cargo category vessels traveled more than half (56.0%) of all distance traveled.

Overall Average Calculated SOG

Passenger ships, including ferries, showed the greatest average calculated SOG, 18.2 knots (**Table 2**). These were followed by RORO cargo ships (16.9 knots), container ships (16.1 knots), and vehicle carriers (14.0 knots). The five vessel types with the greatest average calculated SOG also had the five greatest average broadcast SOG. Supply ships were the only miscellaneous category vessel type to average greater than 10 knots (10.4 knots). Public vessels had the greatest variability of speeds, with a standard deviation of 8.1 knots, followed by supply ships (6.6 knots) and RORO cargo ships (5.7 knots).

With the exceptions of three vessel types (cable layer, drill ship, and supply ship), all vessel types had a greater calculated SOG than broadcast SOG. We tested if the calculated and broadcast speeds came from the same statistical population using both the parametric Welch t-Test and the non-parametric

Mann-Whitney-Wilcoxon Test. Further, with the exception of four vessel types (fishing vessels, dredgers, pollution control, and research ships), all vessel types had an equal or greater broadcast SOG standard deviation than calculated SOG standard deviation (**Table 2**).

Biologically Important Areas

There are three feeding BIAs in the study area, each of which was transited by most vessel types (**Figure 3**). The areas of the BIAs most frequented by vessels were the western and northern regions of the humpback whale feeding BIA. The southeastern part of this BIA is inside the IMO-designated ATBA, which specifies an area that all ships greater than 400 gross tonnage should avoid for safety and environmental concerns. The shipping lanes entering and exiting the Strait of Juan de Fuca had the highest density of vessels. Vessels in the Grays Harbor and Northwest Washington feeding BIAs for gray whales were not as common, with the exception of the Strait of Juan de Fuca. Commercial vessels infrequently transited the Grays Harbor BIA.

There were four gray whale migration BIAs within the study area. The Northbound Phase A, Northbound Phase B, and Southbound migration BIAs are located within eight-, five-, and ten-kilometer buffers of the coast, respectively. Due to this coastal proximity and the ATBA, there were relatively few vessels in any of these BIAs. However, the potential presence BIA extends 47 km from the coast and was transited by all vessel types (**Figure 4**). Fishing vessels, tugs, private vessels, public vessels, and research ships utilized the entire area, while most vessels in the cargo and tanker categories avoided the ATBA, which overlaps with a large portion of the central part of this BIA.

Container ships had the greatest combined average SOG and density, particularly in the northern region of the BIA. The northern portion of the BIA at the mouth of or inside the Strait of Juan de Fuca had the highest densities of most vessel types. Drill ships and unknown vessel types were very uncommon in the potential presence BIA. Tugs and ATBs showed very different movement patterns. Tugs traversed the entire BIA, but ATBs followed the pattern of commercial vessels and avoided the ATBA.

A full tabular statistical summary for calculated SOG in the four BIAs analyzed can be found in **Supplementary Table S4**. Most transits for most vessel types across the BIAs occurred at less than 15 knots (**Table 3**). Notable exceptions were the fastest vessel types (container ship, RORO cargo ship, vehicle carrier, and passenger ship). Grays Harbor had the greatest proportion of vessel speeds below 15 knots. This BIA is just offshore, so vessels were likely approaching or leaving port at slower speeds.

DISCUSSION

Satellite AIS data are valuable for assessing broad-scale patterns of human activity in off-shore waters (Rowlands et al., 2019) and will likely grow in value and accessibility over time (McCauley et al., 2016), as the use of SAIS-derived data allows for vessel tracking much further from the coast than is possible with terrestrial receivers. By analyzing vessel density and speed using SAIS data, our contribution to the current understanding of cetacean collision risk assessment was two-fold. First, we broadly assessed vessel density and speed by vessel type and further narrowed the assessment within the active months of BIAs for local cetaceans. Second, we investigated issues related to the nature of SAIS data and their potential impact on its successful use for informing cetacean-vessel collision risk assessments in the future.

Overall Average Calculated Speed Over Ground

Most vessel types had a greater average calculated SOG than average broadcast SOG. It is important to use calculated SOG in any analysis so that the risk from potential vessel collisions is not underestimated by using broadcast SOG. Bulk carriers, cargo ships, refrigerated cargo, and chemical carriers had an average broadcast SOG less than 10 knots, but an average calculated SOG greater than 10 knots, the speed limit for North Atlantic right whale seasonal management areas (Laist et al., 2014). Similarly, container ships and RORO cargo ships crossed the 15 knot threshold when average calculated SOG was considered instead of average broadcast SOG. Exceeding these thresholds could have important management implications. More research is needed to document the difference between broadcast and calculated SOG when using SAIS data.

Container ships were one of the fastest and most common vessel types. Although passenger ships did not comprise a large proportion of transits or total distance, they were the fastest vessel type, and thus warrant special consideration in any potential future risk assessment. Fishing vessels and bulk carriers, the most common vessel types, had average calculated SOG less than or near the 10 knot speed restrictions that are commonly used in whale management areas (Laist et al., 2014). All other cargo and tanker ships had average calculated SOG values above 10 knots. Vessel categories that transited across the study area (cargo, tanker, and passenger) tended to have greater average speeds than those working within the study area (fishing, miscellaneous). These categories should warrant potentially differing policy and analysis considerations. Tugs and ATBs had characteristically different patterns, with more common tugs behaving like small vessels and less common ATBs behaving like larger cargo ships. ATBs are generally much larger than tugs, and are likely subject to ATBA restrictions.

Biologically Important Areas

The northern-most portion of the BIAs had the highest concentration of vessel traffic and the fastest average vessel speeds. The Northern Washington feeding BIA for humpback whales had the greatest number of transits among the feeding BIAs, due to its location just offshore of the TSS. While this BIA does not extend north of the United States EEZ to cover the shipping lanes between the Strait of Juan de Fuca and Alaska and Asia, it does overlap the shipping lanes toward the United States West Coast and Hawaii. The Northwest Washington feeding BIA for gray whales was infrequently transited, as it is located close to shore. The Grays Harbor feeding BIA for gray whales was

FIGURE 3 | Average calculated speed over ground (SOG) and vessel transits per month in the feeding Biologically Important Areas (BIA) for gray and humpback whales for 2013 and 2014. Darker colors represent faster average speeds and more transits per month for each vessel type. The unit of observation is the square statute mile hexagon.

FIGURE 4 | Average calculated speed over ground (SOG) and vessel transits per month in the gray whale potential presence Biologically Important Area (BIA) for 2013 and 2014. Darker colors represent faster average speeds and more transits per month for each vessel type. The unit of observation is the square statute mile hexagon.

transited most commonly by tugs, fishing, public, and private vessels. Commercial traffic in this area was uncommon.

The potential presence migration BIA for gray whales is spatially extensive and located from the mouth of the Strait of Juan de Fuca south along the coast, and was transited by all vessel types. The majority of large, commercial vessels abided by the ATBA restrictions. Notable vessel types inside the ATBA were tug, public, private, research, and fishing vessels. Each of these vessel types had a low average calculated SOG. The other gray whale migration BIAs are located within several kilometers of

TABLE 3 | Percent of calculated speed over ground (SOG) values less than 15 knots for the entire research area and within each of the Biologically Important Areas (BIA) for 2013 and 2014.

		Biologically important areas			
Vessel type	**Overall**	**Grays Harbor**	**NW Washington**	**Potential presence**	**N Washington**
Cargo					
Bulk carrier	95.2	99.4	98.3	94.7	93.4
Cargo ship	92.2	100.0	100.0	91.3	86.2
Container ship	33.8	55.2	30.7	40.6	32.8
Refrigerated cargo	89.7		100.0	89.7	79.5
RORO cargo ship	33.7	77.8	26.9	47.4	34.8
Vehicle carrier	44.8	90.2	67.4	53.1	32.5
Tanker					
Chemical carrier	90.2	98.8	92.0	91.7	90.7
Liquefied gas carrier	61.1		0.0	82.2	75.7
Oil tanker	83.6	100.0	97.0	86.5	87.1
Tug					
Articulated tug barge	99.7		100.0	99.7	99.8
Tug	100.0	100.0	100.0	100.0	100.0
Fishing					
Fishing vessel	100.0	100.0	100.0	100.0	99.9
Passenger					
Passenger ship	17.7	100.0	34.8	25.5	17.0
Miscellaneous					
Cable layer	98.6			98.1	99.3
Dredger	100.0	100.0	100.0	99.9	100.0
Drill ship	100.0			100.0	100.0
Pollution control	99.9	100.0	99.2	99.5	100.0
Private vessel	93.2	99.1	93.7	91.2	85.4
Public vessel	84.0	93.1	58.9	68.3	83.7
Research ship	99.3	100.0	99.8	99.5	99.7
Supply ship	83.3	100.0	98.2	71.8	100.0
Unknown	100	100.0	100.0	100.0	100.0

the coast (**Figure 2**). Exploratory analyses showed relatively few vessels transits in these areas, so they were not considered for further analysis.

The BIAs were designated solely within the United States EEZ and do not cross international boundaries (Calambokidis et al., 2015). However, the shipping routes between the Strait of Juan de Fuca and Asia and Alaska continue north and west of the United States EEZ, in an area with frequent humpback whale sightings (Calambokidis et al., 2015). The study area extended to cover Canadian waters, and vessel speed and density remain high in the shipping routes extending toward Asia and Alaska (dark red in **Figure 5**). We recommend that future analyses or management planning concerning whales or vessel traffic, including risk analyses, should be considered a transboundary effort.

Marine Spatial Planning

Using AIS in the risk analysis process is one potential tool in marine management (Wiley et al., 2013). Marine Spatial Planning (MSP) is designed as an adaptive spatial planning process to help manage current and future human activities in the marine environment to meet a variety of objectives and minimize user-user and user-environment interactions by engaging multiple stakeholders (Ehler and Douvere, 2007; Ehler, 2008; Foley et al., 2010; Redfern et al., 2013). Recent examples of successful MSP for cetacean protection are the shifts in the TSS outside Boston, Massachusetts, and San Francisco Bay, California (Wiley et al., 2013; United States Coast Guard [USCG], 2013). The scientific processes used stakeholder involvement throughout, created numerous alternatives, showed how challenges can help the process, and used AIS to evaluate and monitor results (Wiley et al., 2013).

Since AIS is an international standard (Tetreault, 2005), the SAIS data that were provided crossed jurisdictional boundaries along the border with Canada. Off the coast of British Columbia, risk to several species of cetaceans has been investigated (Williams and O'Hara, 2010). The findings of the present research, including calculated SOG and SAIS data, should be important considerations in any MSP processes off the coast of the state of Washington and British Columbia in Canada. The

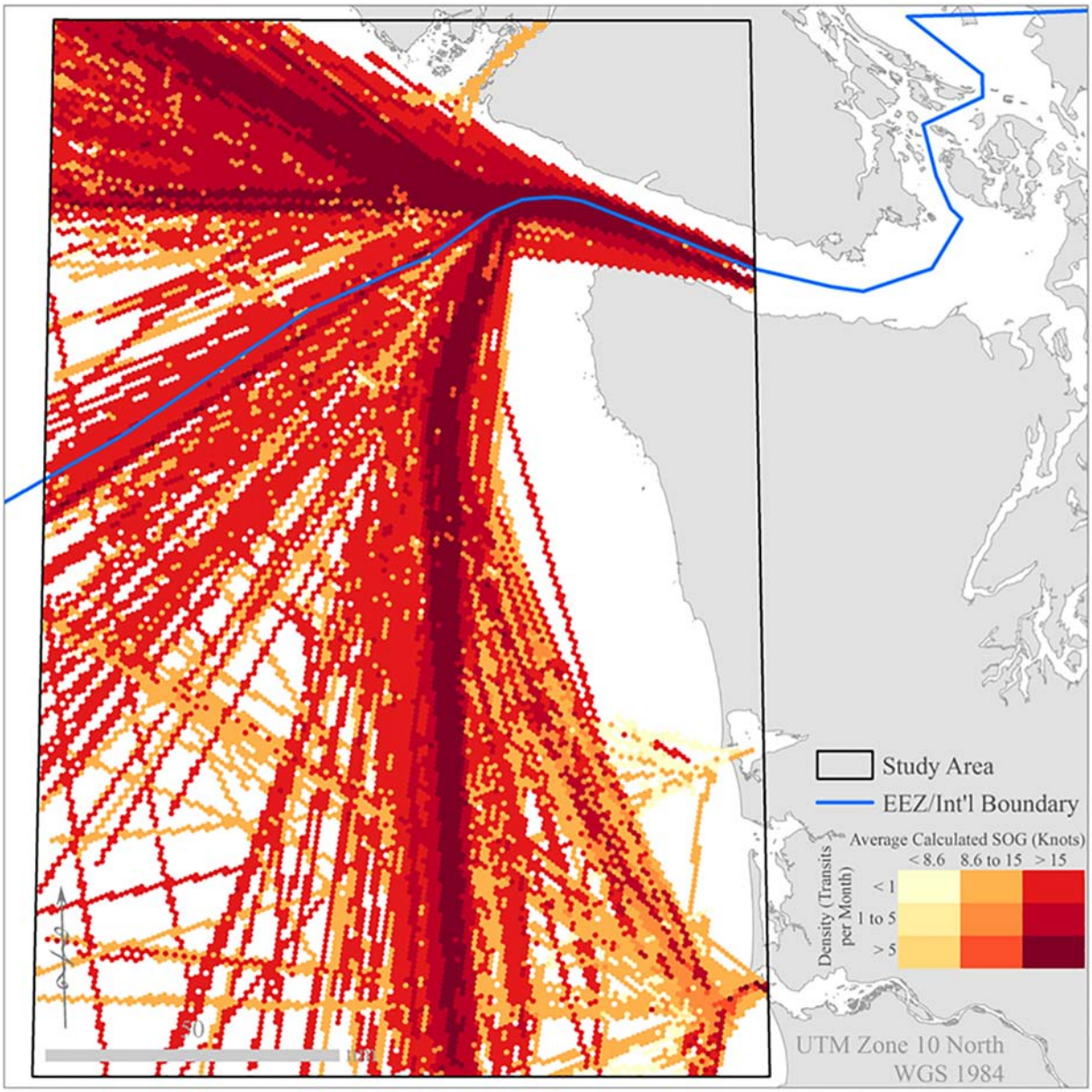

FIGURE 5 | Container ship calculated speed over ground (SOG) and transits per month for 2013 and 2014. The area northwest of the blue Exclusive Economic Zone (EEZ) exhibits high vessel speed and density, although it was not included in the Biologically Important Area (BIA) designations.

SAIS data, within its limitations and at small scale, are an effective means to delineate areas of high use for vessel traffic, even across international boundaries. Calculating vessel speed is critical to avoid underestimating vessel speed and the probability of a lethal vessel and cetacean collision.

Limitations

One of the main limitations of this research is the temporal resolution of the available SAIS data. The ability to precisely track vessel movement decreases with increasing time between sequential points. Vessels can potentially transit around corners, but large amounts of time between SAIS broadcasts can make transit lines appear to cut those corners. This is evident in many vessel types on the northwestern tip of the Olympic Peninsula and the near-shore BIAs (see **Figure 6** as an example). This introduces the potential for a vessel to appear to cross a BIA when in fact it did not. The TSS, controlled by the USCG and used by vessels for insurance and accident coverage purposes, never intersects the Northwest Washington BIA. However, the TSS circumnavigates this BIA, and it is therefore possible that a vessel remaining in the TSS could appear to transit through the BIA. This results in calculated transits across the BIA, sometimes at high SOG, that never actually occurred. The other BIAs, the TSS, ATBA, and OCNMS are also susceptible to this limitation.

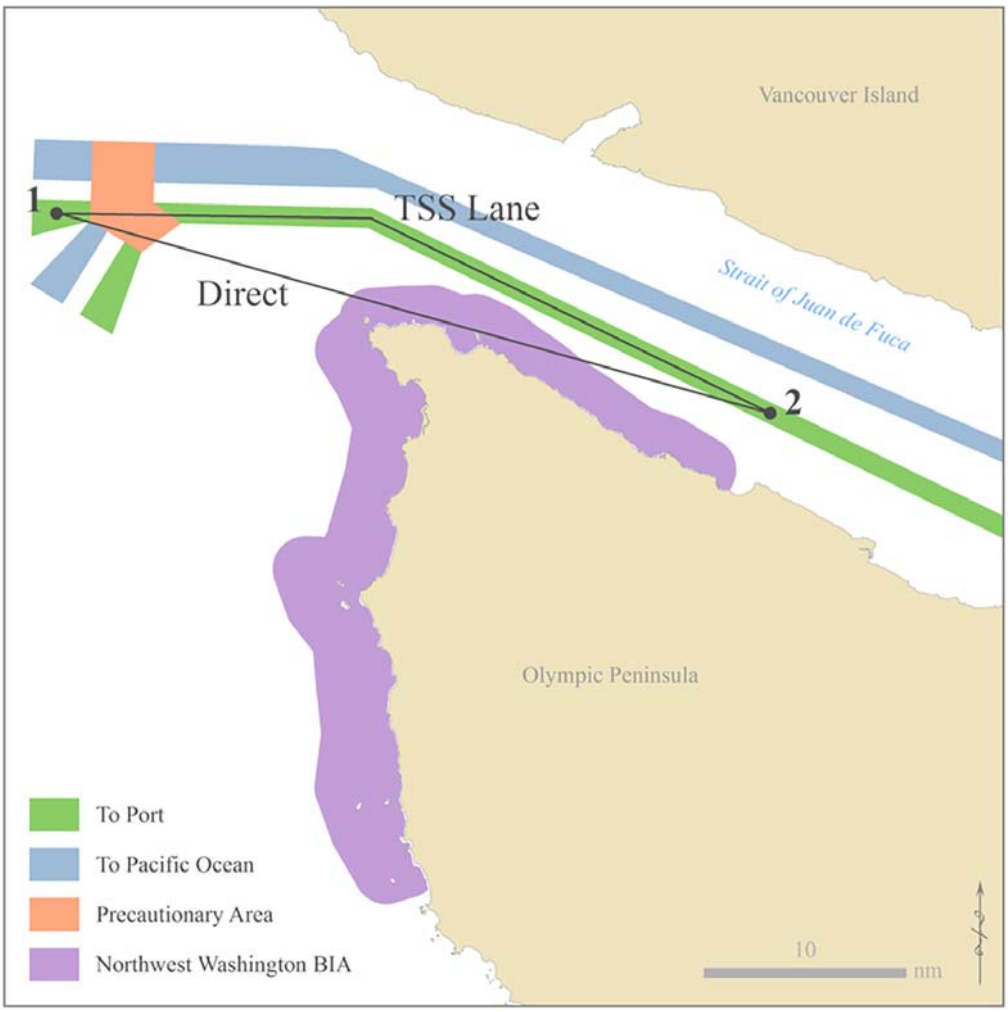

FIGURE 6 | Potential vessel transit lines cutting across the Northwest Washington Biologically Important Area (BIA) due to temporal gaps between sequential points in the SAIS data, (1) and (2). The vessel could actually remain inside the Traffic Separation Scheme (TSS), but could appear to cut across the BIA, leading to false transits.

Since the shortest distance between two points is a line and the time between points remains identical despite the actual path taken, the calculated SOG, while faster than broadcast SOG, is still a systematic underestimation of true SOG, assuming random GPS error. As distance increases for a vessel to traverse around a corner and time remains the same, speed also must increase. Using the example in **Figure 6**, the TSS lane distance and speed are greater than the direct distance and speed between points 1 and 2. Although we may be overestimating the number of transits through some administrative areas, this would be associated with an underestimation of SOG. Most vessels are required to broadcast every few seconds while under way using engine, but the time gap between SAIS records is frequently on the order of minutes or hours. The uncertainty in vessel path is unknown, but will increase with path sinuosity.

As an emerging technology, current SAIS presents tremendous opportunities for research, but caution should be used and uncertainty addressed when using this technology for large scale applications. Temporal gaps in vessel transits add uncertainty to transit path and calculated SOG that was not quantified in this research. Duplicate records accounted for 42.8% of total SAIS records for the year 2013. Broadcast SOG values were zero in 36.7% of all records. There were numerous time, location, and missing value errors that had to be addressed prior to data analysis. Units of measurement for vessel length were also not consistent. These factors cast into question the reliability of individual values and the present quality of SAIS data as a whole, and add uncertainty to automated aggregate calculations. SAIS data in its current state should not be used in a policy enforcement context or for documenting individual presence or absence in an administrative area at large scale. However, SAIS data can be helpful to assess general or overall compliance within an area of interest. As calculated SOG is an underestimation of true vessel speed, SAIS can be helpful in documenting minimum vessel speeds across large geographic areas, especially beyond the reach of terrestrial AIS receivers.

SAIS is useful in examining vessel density at broad scales, and could be used to assess basin-wide open ocean routes. Future additional satellite platforms with AIS receivers will only increase the quality of SAIS data and decrease the amount of temporal gaps. This will open potential research questions involving larger scale questions of specific areas.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

NG conducted the analysis and led the project. All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00109/full#supplementary-material

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Satellite Telemetry Reveals Spatial Overlap Between Vessel High-Traffic Areas and Humpback Whales (*Megaptera novaeangliae*) Near the Mouth of the Chesapeake Bay

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During winter months, humpback whales (*Megaptera novaeangliae*) frequent the coastal waters of Virginia near the mouth of the Chesapeake Bay. Located within the Bay is Naval Station Norfolk, the world's largest naval military installation, and the Port of Virginia, the sixth busiest container port in the United States. These large seaports, combined with the presence of recreational boaters, commercial fishing vessels, and sport-fishing boats, result in a constant heavy flow of vessel traffic through the mouth of the Chesapeake Bay and adjacent areas. From December 2015 to February 2017, 35 satellite tags were deployed on humpback whales to gain a better understanding on the occurrence, movements, site-fidelity, and overall behavior of this species within this high-traffic region. The tags transmitted data for an average of 13.7 days (range 2.7–43.8 days). Location data showed that at some point during tag deployment, nearly all whales occurred within, or in close proximity to, the shipping channels located in the study area. Approximately one quarter of all filtered and modeled locations occurred within the shipping channels. Hierarchical state-space modeling results suggest that humpback whales spend considerable time (82.0%) engaged in foraging behavior at or near the mouth of the Chesapeake Bay. Of the 106 humpback whales photo-identified during this research, nine individuals (8.5%) had evidence of propeller strikes. One whale that had previously been tagged and tracked within shipping channels, was found dead on a local beach; a fatality resulting from a vessel strike. The findings from this study demonstrate that a substantial number of humpback whales frequent high-traffic areas near the mouth of the Chesapeake Bay, increasing the likelihood of injurious vessel interactions that can result in mortalities.

Keywords: humpback whale, satellite telemetry, tagging, state-space modeling, ship strike, *Megaptera novaeangliae*, Chesapeake Bay

INTRODUCTION

The humpback whale (*Megaptera novaeangliae*) is a cosmopolitan species that undergoes long-distance seasonal migrations between high-latitude feeding grounds and low-latitude breeding and calving grounds. Many regional populations, having recovered from decades of commercial whaling (e.g., Stevick et al., 2003), were recently downgraded from "Endangered" to "Threatened" status under the United States Endangered Species Act (ESA) and some populations have been removed entirely from ESA listing (Federal Register, 2016). Globally, they are listed as 'least concern' under the IUCN (Cooke, 2018). Humpback whales in the North Atlantic, considered part of the West Indies distinct population segment (Bettridge et al., 2015) and removed from ESA listing (Federal Register, 2016), migrate from northern feeding grounds in the Gulf of Maine and off the coasts of Canada, Greenland, Iceland, and Norway to the waters of the West Indies during the winter months to mate and give birth (Katona and Beard, 1990; Christensen et al., 1992; Palsbøll et al., 1997). An unknown portion of the population does not migrate to Caribbean waters, but instead uses the coastal waters between New Jersey and North Carolina as a supplemental winter feeding ground (Swingle et al., 1993; Barco et al., 2002). Wiley et al. (1995) hypothesized that it could be an adaptive strategy for juvenile humpback whales to remain in the Mid-Atlantic region during winter months rather than migrating to breeding areas.

Ship strikes are a major cause of mortality for humpback whales worldwide (Bettridge et al., 2015; Hill et al., 2017). A database of global large whale ship strike records, compiled by Jensen and Silber (2004), found humpback whales to be the second most commonly struck species. In April 2017 the United States National Oceanic and Atmospheric Administration (NOAA) declared an unusual mortality event (UME) for humpback whales along the Atlantic east coast from Maine to Florida due to a larger-than-normal number of deaths (n = 93) between January 2016 through April 2019 (NOAA, 2019). The Mid-Atlantic states, including Virginia and North Carolina, account for roughly one third (n = 32) of that mortality. Approximately half of the humpback whales examined as part of the UME had evidence of human interaction, either from ship strikes or entanglement with fishing gear (NOAA, 2019). Historically, this region has documented numerous occurrences of ship strikes. The East Coast recorded the highest number of confirmed and possible ship strikes in North America, with the mid-Atlantic ranking second globally (Jensen and Silber, 2004). Wiley et al. (1995) determined that six of 20 (30%) humpback whales that stranded off the United States Mid-Atlantic and Southeast from 1985 to 1992 had serious injuries likely attributable to vessel strikes. These injuries ranged from propeller cuts to evidence of blunt force trauma, including a disarticulated skull, a fractured mandible, and areas of hemorrhage and extensive skeletal damage (Wiley et al., 1995). Another five of the 20 humpback whales from that study had injuries consistent with entanglement in fishing gear (Wiley et al., 1995).

Along the eastern seaboard of the United States, in the Mid-Atlantic region, is the entrance to the largest estuary in the country, the Chesapeake Bay (**Figure 1**). Located just inside the bay is the world's largest naval installation, Naval Station Norfolk, as well as the Port of Virginia, the sixth busiest container port in the United States (U.S. Army Corps of Engineers, 2017). These active seaports, combined with the presence of recreational boaters, as well as high numbers of commercial and recreational fishing vessels, result in a constant and often heavy flow of vessel traffic through the mouth of the Chesapeake Bay and adjacent waterways (**Figure 2**). From November through April there are ship-speed reduction rules in effect as part of a Seasonal Management Area (SMA) set up to protect ESA-listed North Atlantic right whales (*Eubalaena glacialis*) (NOAA, 2008). These speed restrictions are established along the entire eastern seaboard and require all vessels 65 feet (19.8 m) or longer to travel at 10 knots (18.5 km/h) or less when the whales are most likely to be present. The SMA in this study area begins at the mouth of the Chesapeake Bay and extends outwards to 37 km (**Figure 1**).

Understanding the occurrence and behavior of humpback whales within the Chesapeake Bay's high-traffic region is critical to mitigating potentially harmful impacts on the species. Funded through the United States Navy Marine Species Monitoring Program, in 2015 scientists at HDR Inc. began a long-term study of humpback whales that utilize the waters in and around the mouth of Chesapeake Bay to address questions of habitat use and identify potential conflicts associated with anthropogenic activities. Specifically, this project sought to document the behavior and movements of humpback whales, the level of overlap with high-traffic areas, evaluate site fidelity, and examine any discernable movement and habitat use patterns while taking into account age class and gender.

MATERIALS AND METHODS

Study Area and Field Methods

From January 2015 to February 2017, field effort occurred in each of the 3 years during the winter and early spring. Each field season is referred to herein as, e.g., the 2015/2016 season. Surveys were conducted using an 8.2-m fiberglass hybrid-foam-collar vessel that departed from Lynnhaven Inlet in Virginia Beach, Virginia. Field days were chosen based on optimal sea conditions (Beaufort Sea State of 3 or less and swell height less than 2 m) and time of year (November–March), when sightings of humpback whales in the area are most numerous. Field effort was conducted during daylight hours although start and end time varied based on suitable weather. The primary area of interest was in and around the mouth of the Chesapeake Bay (**Figure 1**). This area is relatively shallow, 30 meters (m) or less in depth in the shipping lanes and precautionary areas (although most range from 12 to 18 m), and 11 to 15 m outside of the shipping channels (provided by NOAA Office of Coast Survey[1], charts US5VA13M and US5VA19M). The mouth of the bay is approximately 120 km from the continental shelf break, and it is

[1] www.nauticalcharts.noaa.gov

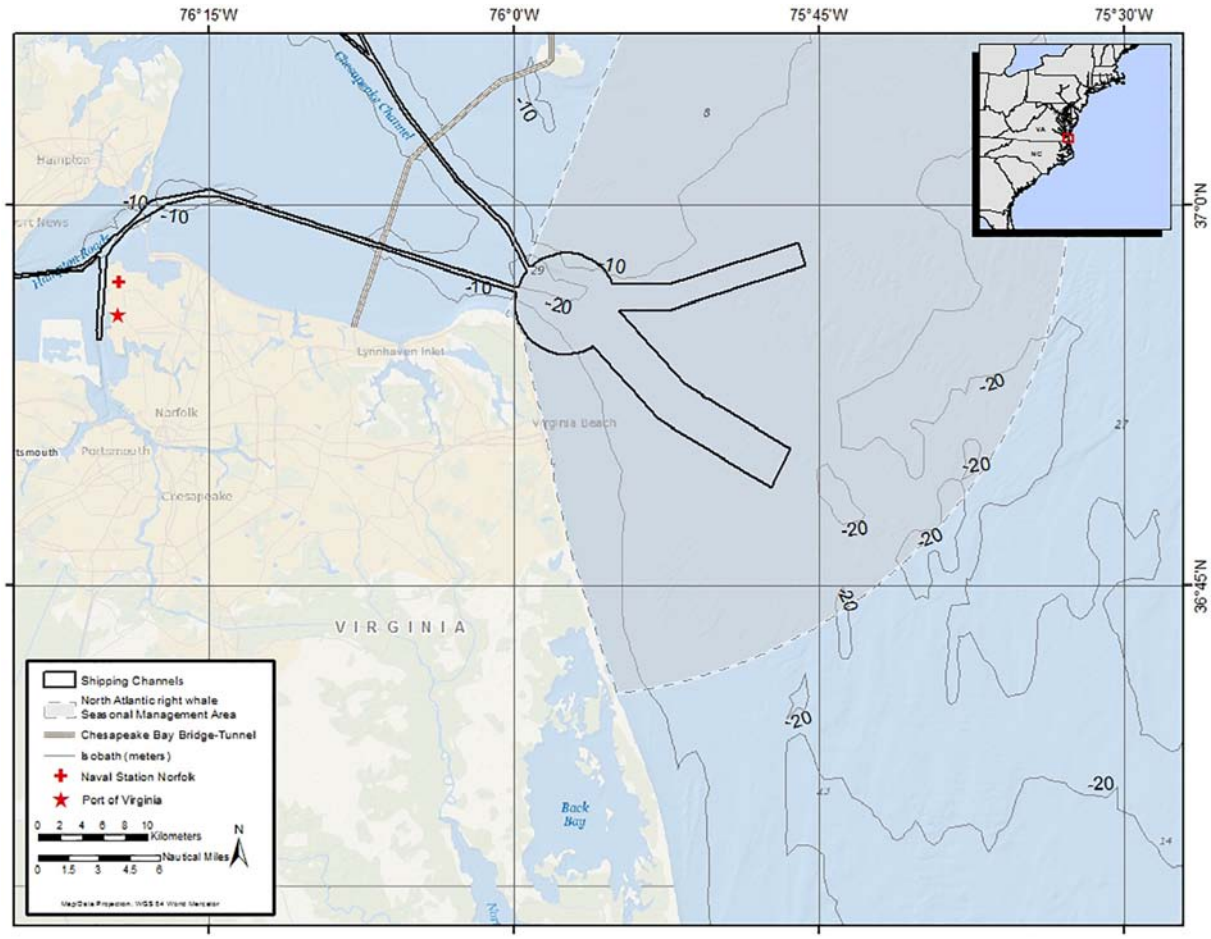

FIGURE 1 | Map of the primary study area, which includes waters in and around the mouth of the Chesapeake Bay off Virginia Beach, Virginia as well as the Port of Virginia and Naval Station Norfolk. Shipping channels are outlined in black and the North Atlantic right whale Seasonal Management Area is shaded. The 37-km Chesapeake Bay Bridge-Tunnel spanning the mouth of the Bay is also shown.

only east of the break that depth increases beyond 100 m. When no whales were observed within the primary study area, the field team would extend their search farther offshore (up to 70 km) or to the south near the North Carolina border. Surveys were non-systematic and no transect lines were followed. The vessel operated at a speed of approximately 25–40 km/h, with three to five observers scanning 360 degrees noting the presence of all marine mammal species.

All baleen whales observed were approached to confirm species and record group size, behavioral state, estimated age class, and GPS location. Age class for humpback whales was approximated by using the 8.2 m vessel as a reference. When approaching broad side to a whale, whales that were estimated to be a similar-size (± approximately 2 m) to the vessel were considered to be juveniles. Those estimated to be >2 m longer than the vessel were categorized as non-juveniles (i.e., either sub-adults or adults). Although subjective, these length estimates are in line with a study by Clapham and Mead (1999) who found males > 11.5 m and females > 11.9 m to be sexually mature. Whenever possible, identification photos – "photo-IDs" – of tail flukes and dorsal fins, using Canon DSLR cameras and 100–400-mm telephoto lenses, were obtained for all humpback whales encountered. Many humpback whales do not regularly lift their tail flukes above the surface in the study area, likely due to the shallow water depth. Photo-IDs were compared to HDR's catalog of unique individuals, which was kept on-board. Individual humpback whales were identified using unique markings on the dorsal fin (e.g., Wells and Scott, 1990; Würsig and Jefferson, 1990) and pigmentation and serration patterns on the ventral surface of the tail flukes (e.g., Katona et al., 1979). Based on a whale's identification (ID), previous encounter history, overall behavior, and health assessment, a determination was made if biopsy sampling and/or satellite tagging would be attempted. Individuals with known sighting histories were the preferred candidates for tagging, however, this was only possible about half of the time, and any animal deemed to be in good body condition was considered a potential candidate for tagging. Tissue samples were collected from tagged animals, whenever possible, as well as from individuals that were not tagged using either a 68-kg pull Barnett compound crossbow (Barnett Outdoors, LLC, Tarpon

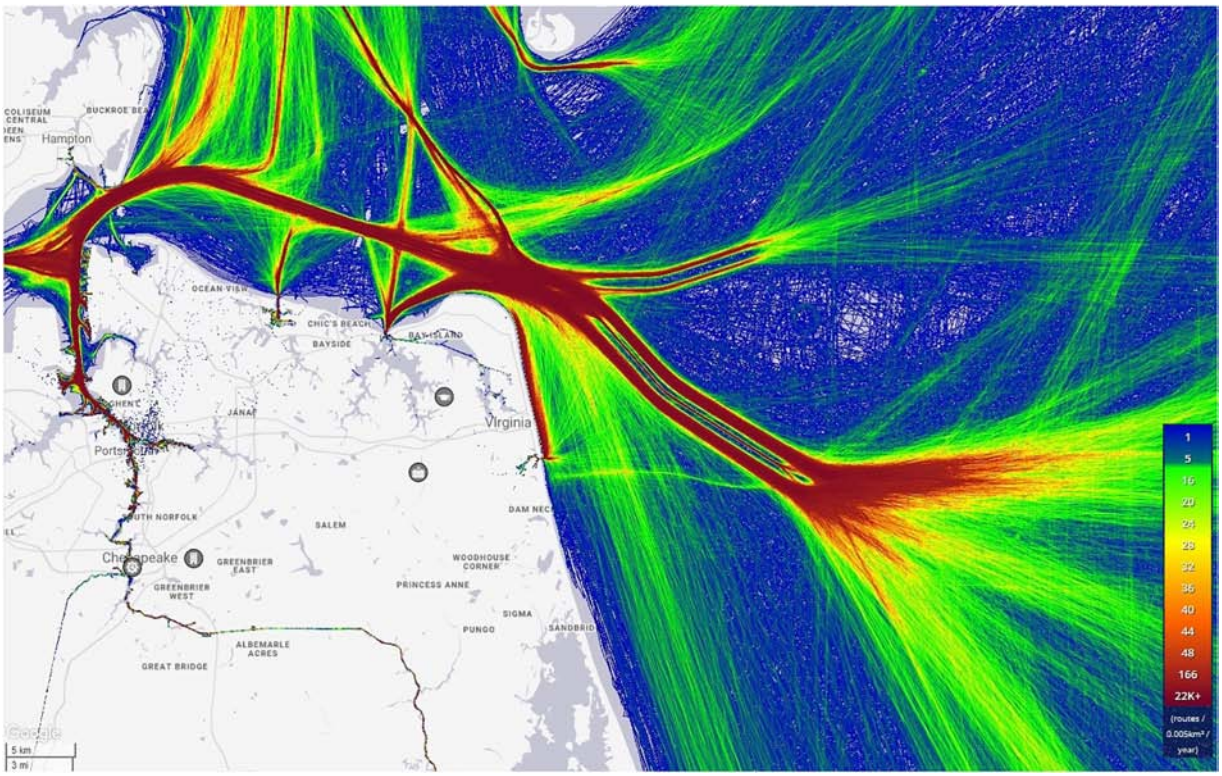

FIGURE 2 | Heat map showing vessel traffic movement in and out of the Chesapeake Bay for the year 2016. The red color indicates the highest density of vessel traffic. Yellow indicates a moderate volume of vessel traffic, and blue indicates the lowest density of vessel traffic. Image provided by www.MarineTraffic.com.

Springs, FL, United States) or a Paxarms biopsy rifle (Paxarms New Zealand Ltd., Cheviot, New Zealand). Skin samples were processed for gender determination at Duke University following the methods described in Waples (2017).

Satellite Tagging

Argos satellite-linked tags from Wildlife Computers (Redmond, Washington) in the Low Impact Minimally Percutaneous External-electronics Transmitter (LIMPET) configuration (Andrews et al., 2008) were used, with location-only Smart Position and Temperature (SPOT-240) tags comprising the majority (32) of tags deployed. A small number (3) of SPLASH10-F-333 tags, which, in addition to collecting location data also collected depth data, with a depth sensor resolution of 0.5 m, in pre-defined bins, were trialed in 2017. These anchored tags with the electronics package external to the skin (see Andrews et al., 2019) were remotely deployed using a modified air rifle DAN-INJECT JM25 pneumatic projector[2]. Two 6.8-cm surgical-grade titanium sub-dermal darts with six backward-facing petals were used to attach tags to the dorsal fin or just below the dorsal fin. Given existing information on attachment durations of these tags on humpback whales (e.g., Schorr et al., 2013), tags were expected to function over a period of a few days to weeks. Therefore, tags were programmed to maximize the number of transmissions and locations received during attachment rather than to extend battery life. Additionally, based on satellite availability in the area, tags were programmed to transmit continuously 20–22 h per day with the exception of one tag that was limited to 250 transmissions per day. Once a tag ceased transmitting, location and dive data were downloaded via the tag portal accessible on the Wildlife Computers website[3]. Locations of tagged individuals were approximated by the Argos system using the Kalman filtering location algorithm (CLS, 2016) and all Argos location classes were retained except for class Z. Additional filtering to remove locations corresponding to unrealistic swimming speeds was performed using the Douglas Argos Filter package provided within Movebank[4], where maximum swimming speed was set at 15 km/h (e.g., Noad and Cato, 2007). Unrealistic locations (i.e., those on land) were manually removed using tools provided within Movebank. For the tag data collected from each individual whale, the PTT ID (a unique six digit serial number) of the tag will be used for the purposes of identification in this study.

Using the Argos locations obtained post-filtering, a 'total distance' was calculated for each tagged whale by summing the cumulative distances between each Argos location. This distance was then divided by the number of days the tag transmitted to provide an 'average distance per day' that an individual whale

[2]www.dan-inject.com

[3]www.my.wildlifecomputers.com

[4]www.movebank.org

traveled. The 'max distance from initial location' was calculated as the furthest straight line distance between the first location and the farthest location, and the 'mean distance from initial location' was calculated as the mean distance of all Argos locations from the first tag location.

For the three SPLASH10-F-333 tags, the number of dives recorded were binned according to pre-determined depths. The dive-depth bins were defined as <5, 5–10, 10–15, 15–20, 20–25, 25–30, 30–35, and 35–50 m. Dive duration was also categorized in pre-determined 30 s time bins when greater than 1 min and less than 7 min. Dives shorter than 1 min and longer than 7 min were their own bin. Given these parameters, a dive to 22 m that lasted 4 min and 10 s would be logged in the 20–25 m depth bin and the 4–4.5 min duration bin. One histogram message was generated daily via Argos that contained the information for each of the dive-depth and dive duration bins.

Data Processing, Analysis, and State-Space Modeling

Photo sorting and matching were performed in ACDSee Pro v. 7 and 9[5]. Photos for each sighting were cropped and sorted by separating photos of different animals and matching all duplicates of the same individual in order to choose the best images for cataloging. Each unique whale was assigned an individual catalog ID. For each subsequent sighting, images were first compared to previously cataloged individuals to see if they matched before designating as a new individual and assigning a new ID. Any potential matches found were also verified by a second experienced reviewer. A spreadsheet was used to track additional details, such as sighting location (latitude and longitude); date and time of the sighting; whether dorsal fin or fluke photos or both were obtained; within-season or between-season re-sightings; age-class estimation; and whether the individual was tagged or biopsied. Within-season re-sights are defined as re-sightings of the same individual during the same winter season – e.g., an animal sighted in December 2016 and re-sighted in January 2017 would be considered the same season, despite occurring in different calendar years. Between-season re-sights are defined as re-sightings of the same individual during different winter seasons – e.g., an animal sighted in March 2017 and re-sighted in December 2017, although observed in the same calendar year, would be classified as a between-season re-sight.

A hierarchical state-space model (hSSM) was applied to the tag data from all tagged whales in order to gain inference on animal behavior and residency. As with other state-space approaches, the track is smoothed into equal time intervals, with the estimated locations taking Argos location error into account. The R package 'bsam' (Jonsen et al., 2005; Jonsen, 2016) was selected as it allows for hierarchical modeling of tag locations. This method estimates movement parameters for all animals jointly, as well as an individual effects parameter for each tag. This can be advantageous as it may allow shorter deployments that could not have been modeled individually to give realistic results, as was the case here. The model assumes that animal movement patterns are broadly similar. We suggest that this is reasonable as all tags were from the same species and region. Though it is possible that factors such as age, sex, and inter-annual environmental variability may affect movement patterns, our objective was to gain as much inference as possible on short deployments, which meant grouping tags together as much as possible.

To determine the appropriate time interval for the hSSM predictions, the average time between received locations amongst all tags was calculated. This average time was the smallest interval considered between predicted locations in candidate models. Tag deployments shorter than 4 days (n = 2) were not analyzed given the low number of reported locations and lack of discernable behavior.

Model diagnostics were examined to ensure that Monte Carlo Markov chains (MCMC) were mixing and that all movement and individual effect parameters were converging as expected. Tracks were examined *post hoc* and dropped from subsequent analysis if issues were identified with the output.

The model attempted to assign estimated locations into one of two behavioral states based on the two-dimensional movement of the animal, travel and area restricted search. Travel is characterized by faster movement and fewer direction changes whereas area restricted search (ARS) is characterized by slower movement and frequent turns. Area restricted search is often associated with foraging activity, and in this study behavioral observations of feeding whales, as well as their proximity to prey aggregations, support this assumption. Behavioral states were assigned following Jonsen et al. (2007) from the mean predicted behavioral state of all samples. Values less than 1.25 were classified as traveling. Values greater than 1.75 were classified as ARS.

Animal locations, both filtered Argos and modeled hSSM locations, were overlaid with major shipping lanes to determine the degree of overlap as a proxy for risk of ship strike. Initially, the "shipping lane study area" was defined by the Traffic Separation Scheme which defines inbound and outbound commercial traffic boundaries for the Chesapeake Bay. However, as tag locations showed movements out of the defined area but still within other shipping channels around the Bay, the area was extended using additional nautical charts and datasets, including the Traffic Separation Scheme, Coastal Maintained Channels in United States Waters (United States Army Corps of Engineers), and Shipping Fairways, Lanes, and Zones for United States Waters (National Oceanic and Atmospheric Administration) as guidelines. These revised boundaries are, hereafter, collectively referred to as shipping channels and were used to determine the percentage of animal locations that occurred within and outside of them.

RESULTS

Field Effort and Tagging Results

Seventy-two field days were conducted between January 02, 2015 and March 21, 2017 (**Table 1** and **Figure 3**). In total, there were 305 sightings of 442 humpback whales; 106 unique humpback whales were cataloged, 51 individuals were biopsied, and 35 satellite tags (32 SPOT-240 and 3 SPLASH10-F-333) were deployed. Humpback whale behavior was most often categorized

[5]www.acdsee.com

TABLE 1 | Summary of field effort and humpback whales sighted, photo-identified, satellite-tagged, and biopsied over three consecutive field seasons from 2015 to 2017.

	Season			
	2014/2015	**2015/2016**	**2016/2017**	**Totals**
No. field days	16	27	29	72
First field date	2-Jan-2015	1-Dec-2015	1-Nov-2016	—
Last field date	31-May-2015	9-May-2016	21-Mar-2017	—
Total effort (min)	6,847	9,877	11,830	28,554
Total trackline distance (kms)	1,485	2,456	2,857	6,797
Sightings (individuals)	41 (57)	96 (136)	168 (249)	305 (442)
Unique IDs	31	37	59	127
Unique IDs seen in previous seasons	N/A	8	20	28
Satellite tags deployed	0	9	26	35
Biopsy samples collected	12	11	28	51

as traveling (43.6%), followed by milling (24.2%), feeding (16.4%), socializing (2.6%), and resting (1.3%). Behavioral state was unknown in the remaining 11.8% of observations, primarily due to groups not being approached. Most tags were deployed at the mouth of the Chesapeake Bay with 16 of the deployments occurring in shipping channels, one deployment occurring inside the Bay, and one deployment occurring south of the primary study area near the North Carolina border (**Figure 4**). Satellite tags transmitted data for an average of 13.7 days (range = 2.7–43.8). In addition to humpback whales, fin whales (*Balaenoptera physalus*) were observed in the study area during the 2014/2015 and 2015/2016 season (8 sightings of 11 individuals across both seasons) and minke whales (*Balaenoptera acutorostrata*) were observed during the 2016/2017 season (3 sightings of 3 individuals). Sightings of minke whales occurred at the mouth of the Bay, just outside shipping channels and the SMA. Fin whale sightings occurred at the mouth of the Bay and slightly to the south, with half occurring within the shipping channels inside the SMA and half occurring just outside of shipping channels and the SMA (**Figure 3**).

Based on size estimates, all tagged humpback whales were judged to be juveniles or sub-adults and none were associated with a calf. Of the 51 biopsies obtained, 30 were collected from whales that were satellite tagged. Gender analysis was performed on a subset (n = 29) of the 51 samples and showed roughly equal gender ratios (14 females; 15 males) (Waples, 2017). Of the whales that were satellite tagged, eight individuals were females, 11 individuals were males (**Table 2**), and the remaining 15 samples are awaiting gender analysis. Tags deployed on males (n = 11) transmitted longer (mean = 12.0 days) than females (n = 8; mean = 7.3 days). For whales determined to be juveniles, the tags transmitted longer than tags deployed on whales classified as sub-adults. Tags deployed on juvenile males (n = 7) transmitted for the longest (mean = 13.3 days) and tags deployed on sub-adult females (n = 3) transmitted for the shortest duration (mean = 4.9 days). The SPLASH10-F tags deployed (n = 3) transmitted for shorter durations (mean = 7.7 days) than the SPOT-240C tags (n = 32; mean = 14.2 days).

The number of Argos locations obtained post-filtering ranged from 10 to 862 (mean = 280) per tag (**Table 2**). Whales, in general, remained close to their tagging location (mean = 33 km), but individual movements varied within and between years (**Table 2**). One whale (157917) traveled a maximum distance of 506 km from the initial tagging location over a 12.1-day period, whereas another whale (158683), tagged 3.6 km away 1 year later traveled a maximum distance of only 21 km from the initial tagging location during approximately the same amount of time (12.9 days). The average distance traveled per day ranged from 23.4 km–108.3 km (mean = 65.0 km). Juvenile whales traveled, on average, shorter distances (58.6 km/day) than sub-adults (86.7 km/day) and their maximum and mean distance traveled from initial tagging location was less (98.9 km; 28.3 km) than those of sub-adults (144.8 km; 47.1 km) (**Table 2**).

All 35 tagged whales had filtered Argos locations within the shipping channels at the mouth of Chesapeake Bay. Approximately one quarter of all locations were within the shipping channels (**Figure 4**). Four individuals (166678, 166679, 166681, and 168688) had more than half of their Argos locations occur within the shipping channels over periods of 18.4, 17.2, 11.6, and 21.9 days, respectively. On average, juveniles had more locations occur in shipping lanes (29.3%) than sub-adults (14.5%). Number of locations within the shipping channels by males and females were similar (**Table 2**).

Fifteen of the 26 (57.7%) tagged animals from the 2016/2017 season had Argos locations inside the Chesapeake Bay [west of the Chesapeake Bay Bridge-Tunnel (CBBT), a 37-km man-made structure that spans the mouth of the Chesapeake Bay with portions above and below water] (**Figure 1**). This was an increase when compared to the 2015/2016 field season where only two of nine (22.2%) tagged whales had locations west of the CBBT. Of the two individuals with locations west of the CBBT in 2015/2016, only one, 157923, spent considerable time in that area– approximately 2.4 days over the course of the 20.7 day February deployment. During the 2016/2017 season, five individuals spent > 2 days west of the CBBT during the months of January and February; 166671 (2.1 days), 166687 (2.4 days), 166675 (2.9 days), 166679 (3.8 days), and 166686 (5.2 days). The last location from 166686 was 37 km N of the CBBT (the farthest location recorded inside the bay during this study, 50 km N of the CBBT was from the same individual).

Movements out of the primary study area included offshore travel to the north (New York), south (North Carolina), and east (offshore to 178 km), where whales spent time in both the shallow waters over the continental shelf as well as deeper waters (>3,100 m) east of the continental shelf break (**Figure 5**).

A total of 9,781 dives were recorded from the three SPLASH tags. Nearly all (96.4%) dives were to depths of 20 m or less, with the majority (87.2%) to 15 m or less (**Figure 6**). Only one dive was recorded in the 30–35 m range. Dive durations were short and the majority (88.6%) were less than 3 min (**Figure 7**).

Re-sightings of humpback whales were noted both within- and between-seasons. Of the 106 cataloged individuals, 66 were seen on more than one occasion (excluding same-day re-sightings).

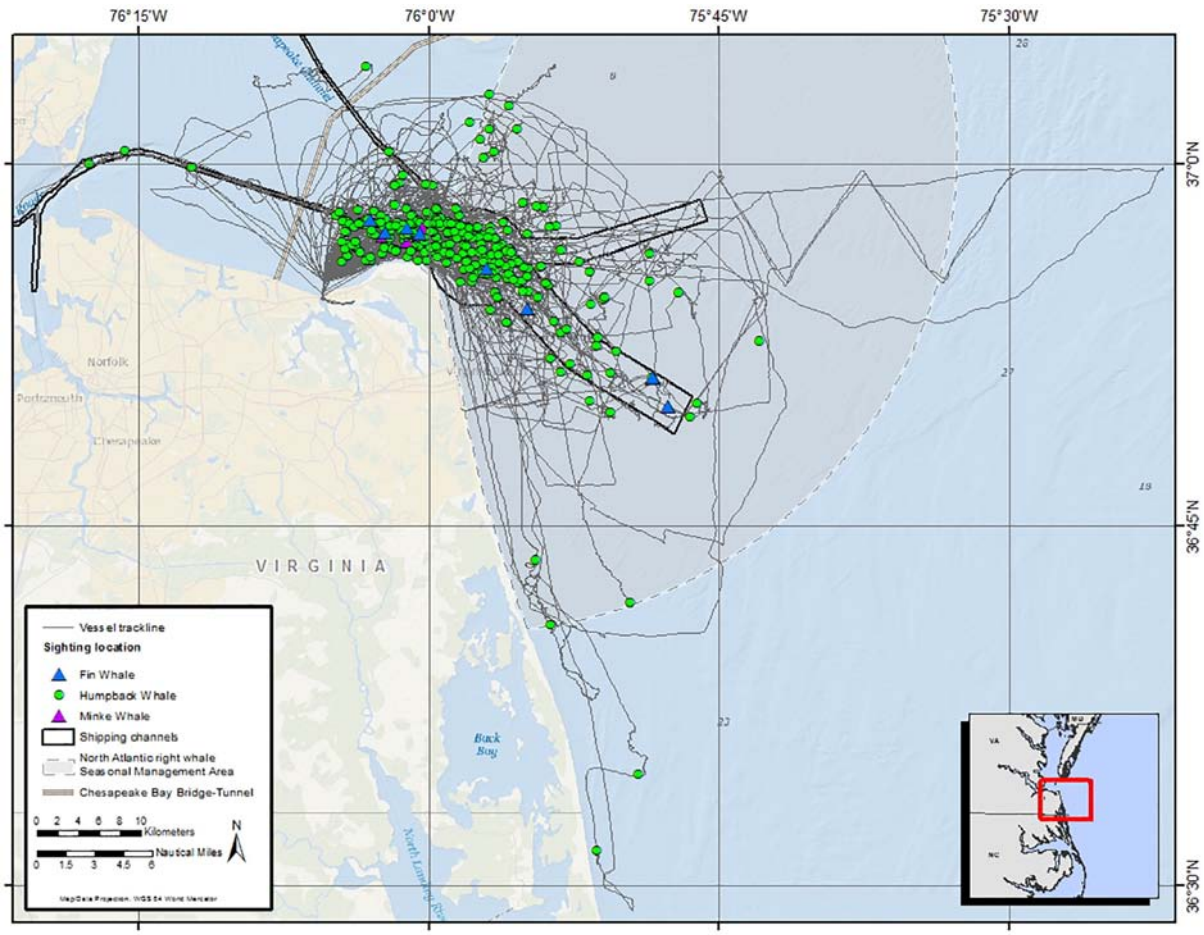

FIGURE 3 | Survey vessel trackline (gray line) and sightings of baleen whales for all field effort from December 2015 to February 2017, with humpback whales shown as green dots, minke whales shown as purple triangles, and fin whales shown as blue triangles.

Of those seen more than once, within-season re-sightings (from the 1st day observed to the last day observed) ranged from 1 to 94 days (mean = 29; median = 25). Eight individuals were re-sighted between the 2014/2015 and 2015/2016 seasons, and 20 individuals observed during the 2016/2017 season were seen in previous seasons. Using photographs obtained from the cataloging effort, obvious evidence of vessel interaction, such as propeller scarring, was apparent on at least nine of the 106 (8.5%) cataloged humpback whales.

State-Space Modeling Results

Two tags were omitted from the hSSM analyses completely (157922 and 158676) due to deployment durations of less than 4 days, a low number of reported locations, and no discernable behavior. On average 62 min passed between received locations, with the maximum gap being almost 1 day. As such, 1 h was the minimum time interval considered for an hSSM. However, the finest temporal scale model that converged successfully was a 3-h model. The selected model converged using 30,000 burn in samples and 15,000 samples. The 15,000 samples were thinned to retain 1,000 in total. A qualitative review of the tracks did not show excessive smoothing between Argos locations, with one exception. One tag, 166675, was also removed from the analysis after reviewing the results and the hierarchical model was rerun without it. This tag had a different duty cycle and few reported locations with long gaps between, which resulted in a modeled track that was artificially over-smoothed. Diagnostics for the updated model performed similarly to the one with the dropped tag. Overall the final model performed acceptably: all parameters converged, MCMCs were mixing, and autocorrelation between chains was low.

Visual inspection of hSMM results was also used to validate the outputs. Generally, the model predicted the behavior that would be expected from reviewing the Argos data qualitatively. Despite the study area being a complex estuarine system, location predictions did not cut across land significantly; as such, no locations were dropped from the model output. Indeterminate locations were most often found as animals were transitioning between traveling and ARS behaviors. Of 3,714 modeled locations, 458 (12.3%) were identified as traveling, 211 (5.7%) were indeterminate, and the remaining 3,045 (82.0%) were identified as ARS (**Figure 5**), which likely represented foraging

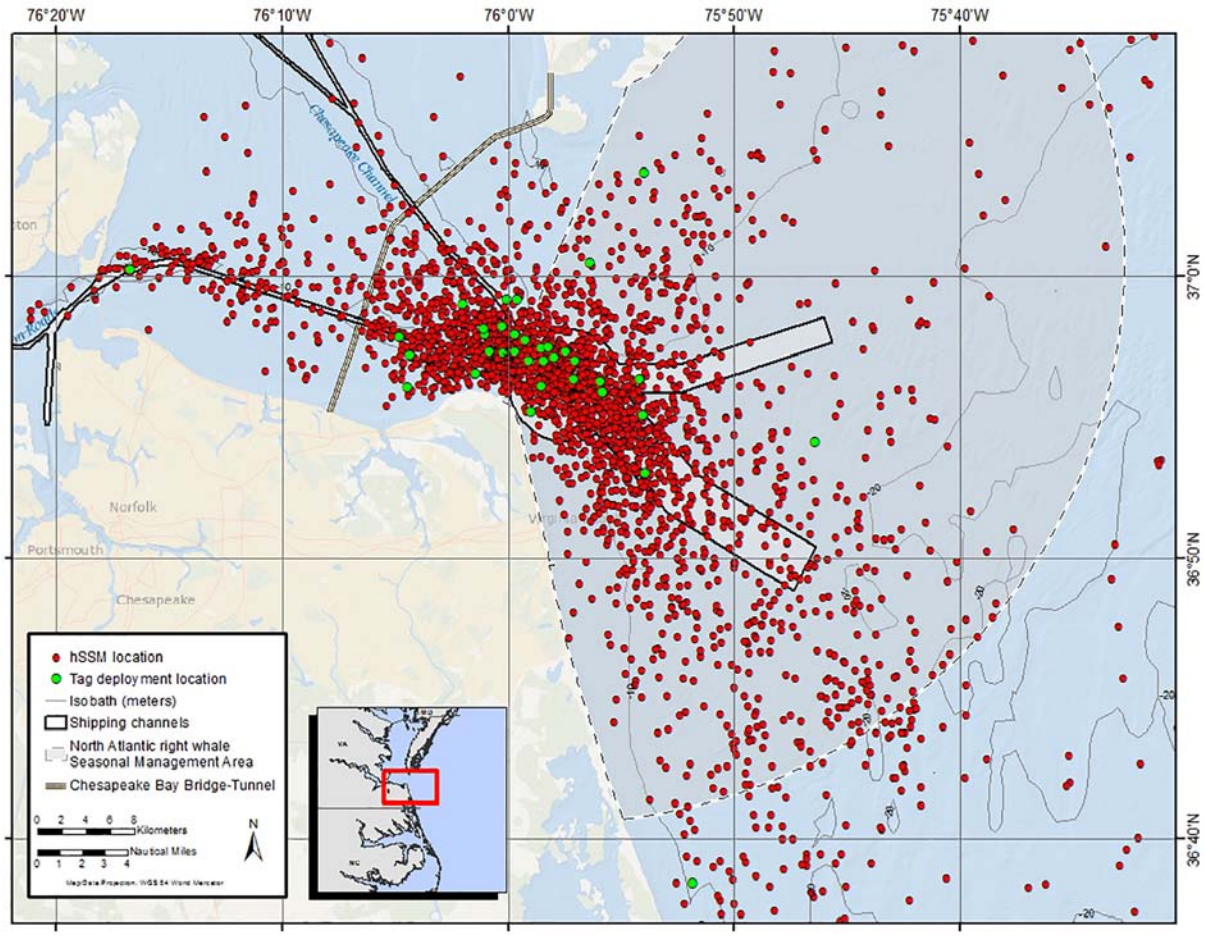

FIGURE 4 | All hierarchical state-space model (hSSM) locations (red dots) in the immediate vicinity of the shipping channels at the mouth of the Chesapeake Bay from 33 satellite tagged humpback whales included in the hSSM. Green dots show tag deployment locations from all 35 tagged whales.

based on numerous observations of feeding observed during field work. In addition to obvious feeding (i.e., lunges, which were observed during one third of all foraging observations), aggregations of prey, stunned fish at the surface, and diving Northern gannets (*Morus bassanus*) were other indications of likely foraging activity. The ARS locations were primarily centered around the mouth of the Chesapeake Bay, with 30.8% of ARS locations occurring within shipping channels. A smaller percentage (6.8%) occurred inside the Chesapeake Bay (west of the CBBT). Additional ARS locations also occurred outside of the primary study area, farther offshore and to the south. Modeled locations identified as travel were minimal in the primary study area, with only 1.2% occurring in shipping lanes, and less than 0.25% occurring inside the Chesapeake Bay.

Visual Observations of Presumed Vessel Interactions

On January 02, 2016 a humpback whale was observed and photographed within the shipping channels without any apparent injuries (**Figure 8A**). One week later, on January 09, 2016, the same individual was encountered, 6.3 km from its previous location, still within the shipping channels, but with a severe laceration across its back (**Figure 8B**). The deep wound, which appeared to have been caused by a large propeller, had sliced through the blubber layer and into the musculature of the whale. The injury was most likely life threatening, and this whale was not seen again.

A second humpback whale was first observed and tagged (157919) on December 20, 2015 (**Figure 8C**). During the 11.5-day deployment, this individual stayed within the primary study area and did not move farther than 13 km from the initial tag location (**Table 2**). On December 30, 2015, he was re-sighted and fluke photographs were obtained (**Figure 8C**). The individual was re-sighted four more times; on 15 January, 20 January, 6 February, and finally on March 03, 2016 when the tail flukes were photographed again, this time with severe left fluke lacerations and visible tissue that was clearly necrotic (**Figure 8D**). These injuries are consistent with a propeller strike. Elsewhere, humpback whales have been documented with portions or all of their tail flukes missing (e.g., Steiger et al., 2008), however, this individual was never re-sighted after the March 03,

TABLE 2 | Details of satellite tag deployments and results from GIS and hSSM analysis of 35 humpback whales from 2015 to 2017.

PTT ID	Age-class	Sex	Date deployed	# days transmitted	Number of locations post-filtering (Argos/hSSM)	Within shipping lane (%) (hSSM)	Total distance (Km)	Mean distance (Km)/day	Max distance from initial location (Km)	Mean distance from initial location (Km)
SPOT 240 Tags										
157916	Juvenile	Male	12/7/2015	14.2	212/114	16.7	714	50.3	42	12
157915	Juvenile	Unknown	12/9/2015	10.5	163/85	25.9	611	58.2	83	23
157917	Juvenile	Male	12/9/2015	12.1	149/96	11.5	786	65.0	506	104
157918	Juvenile	Female	12/10/2015	5.6	76/45	15.6	252	45.0	21	8
157919	Juvenile	Male	12/20/2015	11.5	163/93	29.0	497	43.2	13	5
157920	Sub-adult	Male	12/20/2015	17.6	210/142	2.8	943	53.6	242	81
157921	Juvenile	Male	2/6/2016	21.4	231/171	41.5	1360	63.6	344	40
157922	Juvenile	Male	2/6/2016	3.3	10/N/A	–	219	66.4	115	34
157923	Juvenile	Male	2/9/2016	20.7	305/166	39.8	1139	55.0	189	22
158676	Sub-adult	Female	11/1/2016	2.7	62/N/A	–	238	88.1	32	15
158677	Sub-adult	Male	11/1/2016	6.7	163/54	11.1	722	107.8	212	55
158678	Sub-adult	Male	11/1/2016	6.0	144/48	4.2	650	108.3	136	34
158675	Sub-adult	Female	11/3/2016	3.5	211/28	14.3	885	105.4	205	71
158679	Sub-adult	Male	11/3/2016	8.4	78.67	10.4	361	103.1	158	62
158680	Sub-adult	Female	11/18/2016	8.4	215/68	5.9	465	55.4	120	51
158681	Juvenile	Female	12/13/2016	9.3	253/75	44.0	536	57.6	20	8
158682	Juvenile	Female	12/21/2016	8.4	206/67	25.4	494	58.8	29	12
158683	Juvenile	Female	12/21/2016	12.9	292/103	35.9	727	56.4	21	9
166671	Juvenile	Unknown	12/28/2016	19.6	498/157	33.1	1210	61.7	49	13
166672	Juvenile	Female	12/28/2016	7.2	160/58	37.9	431	59.9	24	8
166673	Juvenile	Unknown	1/1/2017	38.7	724/310	16.5	1868	48.3	94	26
166674	Juvenile	Unknown	1/5/2017	19.2	319/152	10.5	1386	72.2	158	41
166675	Juvenile	Male	1/11/2017	10.0	84/N/A	–	234	23.4	53	20
166676	Juvenile	Unknown	1/16/2017	9.2	254/74	35.1	629	68.4	104	19
166677	Juvenile	Unknown	1/19/2017	11.5	265/93	6.5	759	66.0	111	41
166678	Juvenile	Unknown	1/19/2017	18.4	487/147	61.9	1072	58.3	40	13
166680	Juvenile	Unknown	1/21/2017	24.7	705/197	2.0	1694	68.6	179	97
166679	Juvenile	Unknown	1/25/2017	17.2	471/138	58.0	1066	62.0	26	8
166681	Sub-adult	Unknown	2/1/2017	11.6	303/93	52.7	836	72.1	53	8
166682	Juvenile	Unknown	2/2/2017	21.9	547/175	51.4	1541	70.4	41	12
166683	Juvenile	Unknown	2/2/2017	19.2	512/153	43.1	1106	57.6	39	9
166685	Juvenile	Unknown	2/14/2017	43.8	862/350	9.1	2754	62.9	238	128
SPLASH10-F-333 Tags										
168686	Juvenile	Unknown	2/17/2017	7.6	184/66	10.6	422	55.5	66	27
168687	Juvenile	Unknown	2/17/2017	10.5	200/88	21.6	656	62.5	40	11
168688	Juvenile	Unknown	2/24/2017	5.2	99/41	48.8	338	65.0	24	13
			Mean	**13.7**	**280/116**	**26.0**	**846**	**67.5**	**109**	**33**

2016 encounter, even when reviewing photo-ID effort beyond the timeframe of this project through 2019.

A third humpback whale was first observed and tagged (166675) on January 11, 2017 (**Figure 8E**) and during the 10-day deployment, spent time around the mouth of the Chesapeake Bay and up to 23 km west of the CBBT. This individual was re-sighted twice, on 21 and 25 of January 2017 east of the CBBT. On February 12, 2017 the whale washed ashore dead in Virginia Beach. A large incision across its back exposing internal organs suggested a propeller strike from a large ship (**Figure 8F**). Post-mortem examination supported this determination (National Marine Fisheries Service [NMFS], 2017).

DISCUSSION

Results from satellite tagging and photo-ID during 3 years of effort show both within-season and between-season site fidelity in the study area for individual whales and a high level of occurrence within the shipping channels. Because Argos satellite locations have error associated with them, ranging from <250 m to >1,500 m (CLS, 2016), the hSSM locations were also examined to reduce bias and determine risks associated with humpback whale presence/absence within the high traffic shipping channels. Results from both were nearly identical, further supporting the high use of this particular habitat by humpback whales.

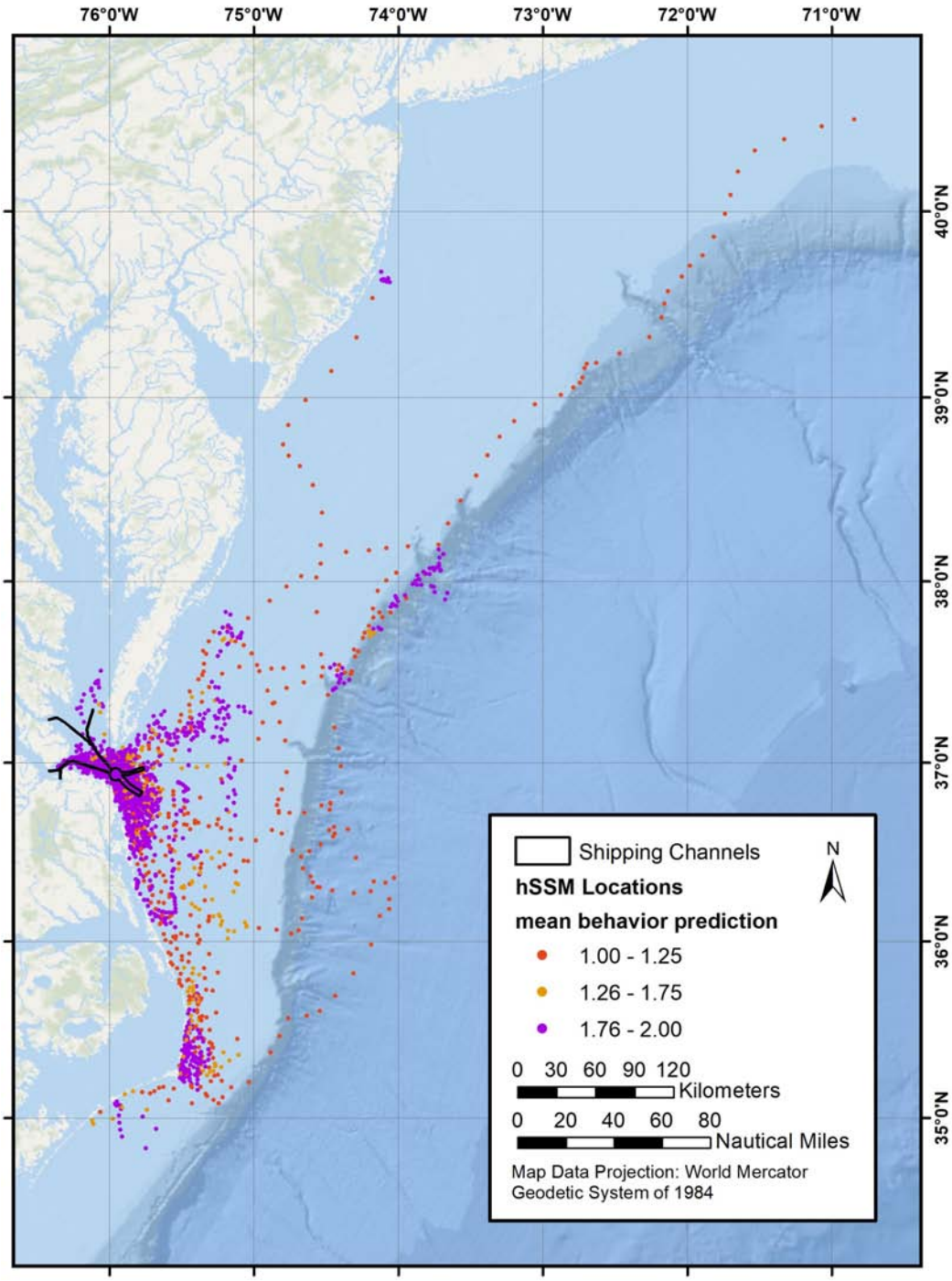

FIGURE 5 | Mean predicted behavioral state from the hSSM for 33 tagged whales along the eastern seaboard from Cape Hatteras, North Carolina to Long Island, New York, showing travel (red dots), area-restricted search (purple dots), and indeterminate behavior (orange dots).

The hSSM analysis provided valuable insight regarding the behavior of all but the shortest (or sparsely reporting) tagged humpback whales in this study. Humpback whales showed variable movement patterns, though the most common was ARS centered around the mouth of Chesapeake Bay, highlighting that this is an important foraging area for this population. This is where most of the tags were deployed and it may also be that tags were shed before significant movement was undertaken. Other movement strategies observed when examining all tracks included looping down near the Outer Banks of North Carolina to presumably feed and then returning north, foraging further inside the bay, and long-distance directed

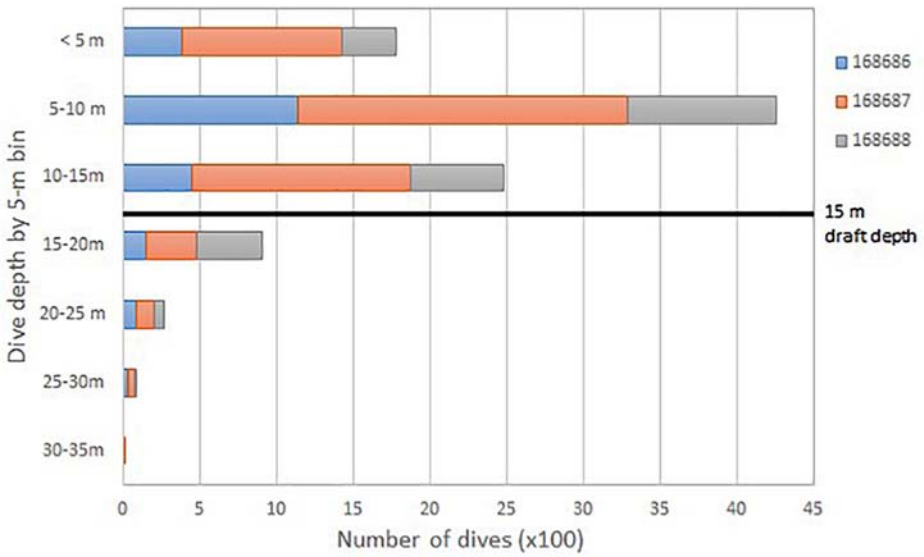

FIGURE 6 | Total number of dives (100×) for three satellite-tagged humpback whales (identified by PTT ID) grouped by 5-m depth bins. Maximum large-ship draft is noted by the black line.

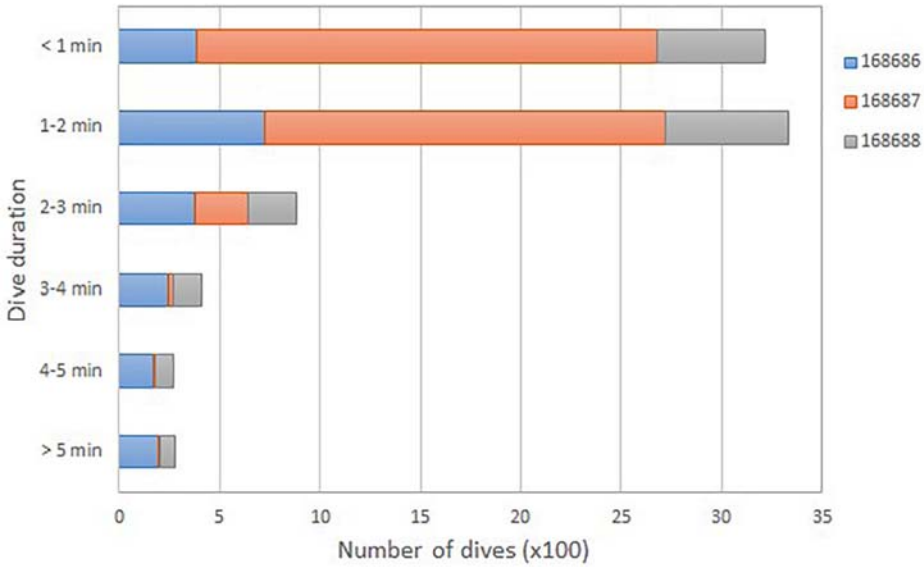

FIGURE 7 | Total number of dives (100×) for three satellite-tagged humpback whales (identified by PTT ID) grouped into 1-min dive duration bins.

movements northwards along the coast and the shelf break before engaging in ARS in other locations.

Because tag deployments were on the order of days to weeks, it is important to take into account the potential for tagging bias with these results. Whales may be more likely to occur in close proximity to where they were tagged, at least initially (e.g., Kennedy et al., 2013). In this study, whales with the shortest tag durations were omitted from the hSSM analyses to help reduce this bias. One shortfall of LIMPET tags is that they tend to have shorter retention times on large whales than tags designed to anchor in the muscle below the facia layer. For comparison, Kennedy et al. (2013) deployed 28 transdermal 'consolidated' tags (see Andrews et al., 2019) on humpback whales in the North Atlantic. The mean tag longevity was 26 days for these consolidated tags, almost twice the mean tag retention of 13.7 days in this study. However, prior to commencing this effort, the authors felt that the greater depth penetration by consolidated tags was not preferred for a variety of reasons, including the fact that the vast majority of whales in the area were known to be juveniles or sub-adults (Swingle et al., 1993; Barco et al., 2002). Because the goal of this study was to

FIGURE 8 | (A) Humpback whale observed January 02, 2016 with no apparent injuries, and **(B)** again on January 09, 2016 with severe laceration across its back; **(C)** humpback whale observed fluking on Decmber 30, 2015 without injury, and **(D)** again on March 03, 2016 with nearly severed tail fluke; **(E)** humpback whale photographed and tagged on January 11, 2017, and **(F)** dead on a Virginia Beach with large incisions and exposed organs on February 12, 2017.

assess where humpback whales are spending their time while in the study area, rather than where they go once they leave the area, the shorter retention time of the LIMPET tags was not considered prohibitive in addressing the primary study objectives and bias due to shorter retention times is considered nominal.

Many humpback whale sightings, and subsequently tag locations, occurred within the deeper shipping channels suggesting these may be areas of preferred prey aggregations. A fishery for Atlantic menhaden (*Brevoortia tyrannus*) exists in and around the Chesapeake Bay (Smith and O'Bier, 2011). During this study approximately one third of humpback whale feeding observations were accompanied by lunging and the presence of small schooling fish species, including Atlantic menhaden. At times, these observations were in close proximity to the commercial fishing fleet, although this was not systematically recorded during the initial survey years. Whale defecations were regularly observed, further supporting that foraging is actively occurring in the region. An analysis of stable isotope signatures from biopsied skin samples collected from humpback whales near the mouth of the Chesapeake bay during this project by Waples (2017) found that the mean $\delta^{15}N$ value for humpback whales were comparable to those collected from humpback whales in the Gulf of St. Lawrence, a well-known foraging habitat, during summer months (Gavrilchuk et al., 2014). In the Gulf of St. Lawrence humpback whales were believed to primarily be feeding on American sand lance (*Ammodytes americanus*), northern krill (*Meganyctiphanes norvegica*), capelin (*Mallotus villosus*) and Atlantic herring (*Clupea harengus*) (Gavrilchuk et al., 2014). Although not conclusive, the similarity in stable isotope values implies that whales in both locations are feeding at similar trophic levels and lends support that the humpback whales biopsied during this study are feeding during winter months.

Dive data from the three SPLASH10-F-333 tagged whales revealed that the majority of dives were to depths of 15 m or less. The current maximum draft for commercial and military vessels extends to 15 m. The spatial overlap of humpback whales in this study area with transiting ships, results in an increased likelihood for interactions (**Figure 6**). McKenna et al. (2015) noted blue whale behavior in commercial shipping lanes off southern California and found that whales showed no horizontal movements away from oncoming ships, rather they exhibited a shallow dive response in 55% of the recorded observations in close proximity to transiting commercial vessels. Even if humpback whales in the mouth of the Chesapeake Bay dive to avoid ships, there is minimal water depth between the vessel and the seafloor where a collision can be avoided.

During the winter months, when humpback whales are most likely present, large ships moving into and out of the Chesapeake Bay are required to reduce their speed to 10 knots in order to be compliant with the North Atlantic right

whale SMA guidelines (NOAA, 2008). A review of historical records by Laist et al. (2001) concluded that lethal collisions of whales with ships sharply increased when ships were moving at speeds of 10–14 knots (18.5–25.9 km/h) and were rare at speeds below 10 knots. However, the speed restrictions within the SMA do not apply outside of those boundaries (Code of Federal Regulations 33 [Cfr] § 165.501, 2018), which, as this study has shown, are areas humpback whales are still actively foraging within (**Figure 4**). This may put whales at an increased risk for ship strike by faster-moving vessels transiting into or out of the Bay, outside the SMA. Laist et al. (2001) also found that whales are typically not seen prior to collision, or are seen too late to be avoided. In the coastal waters at the mouth of the Chesapeake Bay, with already poor visibility in turbid water, it is unlikely transiting ship crews would be able to see or avoid humpback whales. Silber et al. (2010) also found whales submerged at one to two times the depth of a ship's draft were at an increased probability of coming into contact with the hull or propeller of a ship. In other regions where ship-strike risks are high, such as southern California (e.g., Berman-Kowalewski et al., 2010) and Sri Lanka (Priyadarshana et al., 2016), studies showed or suggest that re-routing ship traffic has the potential to reduce ship strikes. However, re-routing vessel traffic into the mouth of the Chesapeake Bay is not practical; thus leaving speed reductions of transiting vessels as the primary mechanism for reducing humpback whale strike in this region. Speed restrictions from the SMAs have proven to reduce deaths of both North Atlantic right whales and humpback whales (Laist et al., 2014). Based on the results of this study, if the mid-Atlantic SMA was extended further into the Chesapeake Bay, it may reduce ship strikes in this region.

Approximately half of the humpback whales examined to date as part of the UME had evidence of human interaction, either due to ship strike or entanglement (NOAA, 2019). Prior to the UME, the Gulf of Maine humpback whale injury rate was calculated to be 9/year (Henry et al., 2015). The actual number of vessel-related injuries on large whales is most likely under-reported due to a proportion of dead individuals that do not wash ashore, animals that are too decomposed or otherwise inaccessible for assessment, and interactions that go unreported (Laist et al., 2001; Henry et al., 2015). One complication with stranding data is that it is often impossible to determine the location where the interaction occurred, especially for animals that undertake long-distance movements or migrations. Within-season re-sightings of humpback whales occurred, on average, over the course of 29 days during this study, often allowing for multiple opportunities to re-sight, and "monitor" the same individual throughout the season. We documented three instances of injuries and a fatality observed from whales that had been previously seen unharmed. This level of monitoring has the potential to significantly augment data gaps that can occur when strandings are the only source of information on mortalities and injuries.

In total, nine of the 106 (8.5%) humpback whales in our humpback whale catalog have scars or injuries indicative of propeller or vessel strikes. While it is impossible to conclude if these injuries occurred outside of the study area, the evidence from this study highlights different instances where humpback whales were observed in the study area without injuries and re-sighted within the same season with vessel-related injuries. Such examples support the notion that those injuries likely occurred in the primary study area near the mouth of the Chesapeake Bay and suggests that animals are at an increased risk of deleterious interactions with localized shipping traffic.

More than three-quarters of the humpback whales identified and satellite tagged during the first 3 years of this study were estimated to be juveniles. The large percentage of juveniles observed matches both historic stranding data (e.g., Wiley et al., 1995) and observational data (e.g., Swingle et al., 1993; Barco et al., 2002) for this area. In this study, juvenile humpback whales spent more time (i.e., had more tag locations) in shipping channels and stayed closer to their initial tag location when compared to sub-adults. Laist et al. (2001) noted that eight of ten humpback whales struck by ships were juveniles, estimated to be 3 years of age or less, suggesting this is a particularly vulnerable age class for this species. It is possible that these younger animals, with less experience, have not yet learned to avoid ships, whereas older, presumably more experienced animals, have better acquired that ability. Based on the gender analysis to-date, humpback whales were approximately equal ratios of male and female (Waples, 2017) suggesting both sexes are equally vulnerable to potential vessel interactions.

Interactions with vessels, both large and small, are a significant cause for concern for humpback as well as other baleen whale species encountered in the study area. Although the satellite tagging effort focused on humpback whales, other baleen whale species, including minke whales and ESA-listed fin whales, were also documented in the study area. ESA-listed North Atlantic right whales are also known to occur near the mouth of the Chesapeake Bay (Mallette et al., 2017; Hayes et al., 2018) and although the SMA's are in place for this region, the results of this study underscores the need to consider additional protections for other baleen whale species utilizing these waters each winter.

While much of the tagging data corroborates sighting location 'hot spots' in and around the shipping channels, the amount of time some tagged individuals spent west of the CBBT was somewhat unexpected. This is an area where live observations of humpback whales have not previously been reported in the literature, and only occasional sightings have been anecdotally reported by local fisherman or tour operators. The extensive network of bridge pilings appear to create a physical barrier with regards to passage by whales to waters west of the CBBT. Observations of whales passing through the unobstructed non-pile shipping channel openings directly over the CBBT tunnels are not unexpected given their preference to remain in the deeper channels to forage. Although less field effort was conducted in waters west of the CBBT, it should be considered an area of interest in future years given the high traffic rate of large vessels,

reduced speed restrictions, and extent of marine-based military training exercises occurring in this part of the Bay. Increased presence of humpback whales west of the CBBT may be attributed to a combination of possible factors, including, but not limited to: a short-term distributional shift related to overall oceanographic conditions causing prey to become more concentrated farther into the Bay than in previous years, better documentation of whale presence through increased field effort or an increased number of deployed satellite tags, or simply an overall increase in the number of humpback whales in the study area.

CONCLUSION

The number of sightings of humpback whales and other baleen whales (including ESA-listed fin whales), as well as the level of interaction between whales and vessel traffic to-date, support the need for further documenting habitat use and movement patterns in this region. Satellite-tag data have signified that the mouth of the Chesapeake Bay is an important habitat for humpback whales during winter months. The hSSM results suggest that many of the modeled locations centered at the mouth of the Bay represent foraging behavior for these whales, which is further supported from visual observations and stable isotope analyses. This segment of the population clearly engages in diverse feeding and movement strategies, which also needs to be taken into account when mitigating anthropogenic impacts and determining effective management actions. At the time of deployments, the SPLASH10-F-333 tags used in this study were programed to collect only binned depth data. Research is ongoing, and future tagging effort will incorporate behavioral dive profiles to give a more detailed picture of how humpback whales spend time beneath the shallow waters of the Chesapeake Bay. A small unmanned aerial system has also been added to the study with the goal of obtaining more precise length estimates and therefore improving and validating age class estimations. Future hSSM analyses will focus on temporal patterns of use, increasing sample size with more tag deployments, simulating longer tracks, and exploring individual space use further. Additional United States Navy-funded collaborative efforts will also involve deploying digital acoustic recording tags to collect information on received levels of ship noise, as well as determining behavioral states and assess possible avoidance responses. All of this information will provide a better understanding of the occurrence and behavior of humpback whales within these heavily transited waters.

The waters around the mouth of the Chesapeake Bay are a busy area for transiting commercial and military ships, as well as recreational boats. Seasonal speed restrictions established as part of the North Atlantic right whale SMA limit the speed of large vessels only at the mouth of the Bay, but speed restrictions are not in place in other areas that humpback whales actively utilize nor do they pertain to vessels <19.8 m. Extending the SMA farther into the Bay and farther offshore has the potential to improve protection for humpback whales, as well as other baleen whale species utilizing this habitat.

DATA AVAILABILITY STATEMENT

Data generated by this study are represented or included as summarized data in the article/supplementary material. Sighting data is available through OBIS SEAMAP and tag location data are available for viewing through both Movebank and the Animal Telemetry Network.

ETHICS STATEMENT

All tagging and survey methods were conducted under a scientific research permit #16239 issued to Dan Engelhaupt by the National Marine Fisheries Service under the Marine Mammal Protection Act. Prior to tagging, procedures were reviewed and approved by an Institutional Animal Care and Use Committee as part of the Animal Welfare Act.

AUTHOR CONTRIBUTIONS

JA, DE, and AE: field work, analysis, and writing. AD: analysis and writing. TP, MR, and JB: field work and writing.

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frontiers in Marine Science

REVIEW
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A Global Review of Vessel Collisions With Marine Animals

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Concern about the effects of maritime vessel collisions with marine animals is increasing worldwide. To date, most scientific publications on this topic have focused on the collisions between large vessels and large whales. However, our review found that at least 75 marine species are affected, including smaller whales, dolphins, porpoises, dugongs, manatees, whale sharks, sharks, seals, sea otters, sea turtles, penguins, and fish. Collision incidents with smaller species are scarce, likely as a result of reporting biases. Some of these biases can be addressed through the establishment of species-specific necropsy protocols to ensure reliable identification of collision-related injury, particularly blunt force trauma. In addition, creating a ship strike database for smaller species can assist in identifying the species most frequently involved in collisions, identifying high-risk areas, and determining species-specific relationships between vessel speed and lethal injury. The International Whaling Commission database on collisions with large whales provides a good example of this type of database and its potential uses. Prioritizing the establishment of a species-specific necropsy protocol and a database for smaller species as well as the identification of high-risk areas for species other than large whales, would be a valuable step toward the mitigation of collisions with smaller species.

Keywords: collisions, ship strikes, marine animals, injury, mortality, high-risk areas, mitigation measures, information gaps

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INTRODUCTION

A vessel collision or strike is defined as any impact between any part of a watercraft (most commonly bow or propeller) and a live marine animal (Peel et al., 2018). Collisions often result in physical trauma to- or death of the animal (e.g., Lightsey et al., 2006; Byard et al., 2012; Neilson et al., 2012; Towner et al., 2012; Moore et al., 2013) and may cause serious damage to the vessel, while people on board are at risk of injury and mortality (Neilson et al., 2012; Ritter, 2012).

Concerns about the effects of collisions on marine animals and their populations primarily originate from the extensive and growing utilization of the world's oceans by commercial and recreational vessels. Between 1890 and 2018, the number of globally registered large commercial vessels (>100 gross tonnage) increased from 11 108 to just over 94 000 (United Nations Conference on Trade Development, 2018; Lloyds Register of Shipping 1992 in Laist et al., 2001). The largest increase in commercial vessels took place between 1950 and 1980, which coincided with an increase in the amount of ship strikes fatal to large whales, mainly baleen whales (Mysticeti: hereafter referred to as whales) (Laist et al., 2001). In 2005, vessel strikes were identified as a priority by the International Whaling Commission Conservation Committee (IWC-CC) who established the Ship Strike Working Group (SSWG: International Whaling Commission, 2005). The main aim of

the SSWG is to understand and reduce the threat of vessel strikes to cetaceans, especially whales. One essential contribution has been the establishment of an international centralized database (ship strike database) that contains validated information on cetacean (i.e., whales, dolphins, and porpoises) ship strikes worldwide. Although reporting of incidents to the ship strike database can still be improved, the collation of global data provides valuable insight into the scale of the problem, the factors involved in collisions, population specific vessel strike mortality, and the identification of areas where collisions are commonly observed (Jensen and Silber, 2003; Cates et al., 2017; Panigada and Ritter, 2018).

To date, most scientific publications have focused on collisions between vessels and North Atlantic right whales (*Eubalaena glacialis*: e.g., Kraus et al., 2005; Parks et al., 2012; van der Hoop et al., 2012, 2015; Davies and Brillant, 2019), fin whales (*Balaenoptera physalus*: e.g., Williams and O'Hara, 2010; David et al., 2011; Redfern et al., 2013, 2019; Sierra et al., 2014; Panigada et al., 2017), blue whales (*Balaenoptera musculus*: e.g., Berman-Kowalewski et al., 2010; Ilangakoon, 2012; Redfern et al., 2013, 2019; Priyadarshana et al., 2015), humpback whales (*Megaptera novaeangliae*: e.g., Wiley et al., 1995; Alzueta et al., 2001; Neilson et al., 2012; Redfern et al., 2013, 2019; Hill et al., 2017), sperm whales (*Physeter macrocephalus*: e.g., Carrillo and Ritter, 2010; Fais et al., 2016; Di-Méglio et al., 2018; Frantzis et al., 2019), and Florida manatees (*Trichechus manatus latirostris*: e.g., Laist and Shaw, 2006; Lightsey et al., 2006; Rommel et al., 2007; Edwards et al., 2016). However, there is increasing evidence that more marine species are at risk of collision, especially within coastal areas frequented by smaller vessels. Our review aims to provide an overview of all marine animal species involved in collisions and evaluates whether our knowledge of vessel strikes with whales can assist in understanding and mitigating vessel strikes with smaller species. We conclude with recommendations for priority actions to address essential information gaps. It should be noted that we acknowledge all work conducted on ship strikes by various intergovernmental organizations [e.g., IWC, Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic (ACCOBAMS), Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS)] from which annual-, workshop-, and technical-reports have been produced. However, wherever possible, we referred to peer-reviewed publications. Consequently, we do not reference reports that discuss published work.

SPECIES OF MARINE ANIMALS COLLIDING WITH VESSELS

One of the first collision reports dates back to 1877, when a steamship collided with a small unidentified whale (Allen 1916 in Laist et al., 2001). The first identified species was a sperm whale in 1908 (Laist et al., 2001), after which a gradually increasing number of baleen whale species were identified as struck by vessels. Collisions with smaller marine animals were only recognized around 1980, when Hartman (1979) proposed collisions as the most serious threat to Florida manatees in the U.S. Around the same time, the New York State Marine Mammal and Sea Turtle Stranding Program started a sea turtle stranding database, revealing that 10.6% of all turtles exhibited evidence of propeller wounds (Gerle and DiGiovanni, 1998). To date, necropsy data, eye-witness collision reports, and anecdotal data suggest that at least 75 marine species have been struck by vessels (**Table 1**), including baleen whales, smaller toothed whales (Odontoceti: e.g., Parsons and Jefferson, 2000; Stone and Yoshinaga, 2000; Kemper et al., 2005; Byard et al., 2012; Lair et al., 2014), manatees and dugongs (Sirenia: e.g., Ackerman et al., 1995; Meager and Limpus, 2012a), carpet sharks (Orectolobiformes: e.g., Graham and Roberts, 2007; Rowat et al., 2007; Speed et al., 2008), mackerel sharks (Lamniformes: e.g., Speedie et al., 2009; Towner et al., 2012), seals and sea otters (Carnivora: e.g., Kreuder et al., 2003; Byard et al., 2012; Wilson et al., 2017), turtles (Testudines: e.g., Gerle and DiGiovanni, 1998; Chaloupka et al., 2008; Meager and Limpus, 2012b), penguins (Sphenisciformes: Cannell et al., 2016), and even fish (Perciformes: e.g., Brown and Murphy, 2010; Clarey, 2014).

Collision reports for smaller marine species are generally scarce likely due, at least in part, to a reporting bias rather than collisions with smaller species being less frequent. We know that collisions between large vessels and whales may not be reported because vessel crew are not aware of the collision (Dolman et al., 2006). Lack of awareness of a collision is even more likely for smaller species. In addition, fatal collisions with most cetaceans, whale sharks (*Rhincodon typus*), and sea turtles likely go unnoticed because carcasses of these species sink quickly (van Waerebeek et al., 2007; Speed et al., 2008; Williams et al., 2011; Nero et al., 2013). Even if carcasses float, they may be consumed by scavengers or too decomposed to reach shore. Whether strandings of small and large species are reported with the same probability is also unknown. It is possible that the general public may be less concerned about reporting smaller species, such as penguins and sea turtles, than about reporting large whales and dolphins. Finally, there is no global encouragement nor a global database, like the ship strike database for cetaceans established by the IWC, to report collisions with smaller marine species. These factors make it even more challenging to assess the frequency and consequences of collisions with smaller species than it is for large whales.

POSSIBLE CONSEQUENCES OF COLLISIONS

Collision incidents have led to concerns about animal welfare, animal conservation, safety of people on board the colliding vessel, and economic consequences as a result of vessel damage. In general, three types of consequences are distinguished: direct (i.e., consequences that are the immediate result of collision), long-term (i.e., decrease in animal fitness over time), and population consequences. Direct consequences can further be categorized as injuries to the animal, injuries to vessel crew, and damage to the vessel.

TABLE 1 | Table of species identified as struck by vessels.

Order/common name	Latin name	Status (International Union for the Conservation of Nature, 2019)	Frequency	Reliability	References
Perciformes					
Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Near Threatened	Frequent locally	External examination	Brown and Murphy, 2010; Balazik et al., 2012
Sunfish	*Mola mola*	Vulnerable	Rare	Anecdotal (sailors)	Porcasi and Andrews, 2001; Lulham, 2006; Clarey, 2014
Sphenisciformes					
Magellanic penguin	*Spheniscus magellanicus*	Near threatened	Rare	Anecdotal (unknown)	Newshub, 2010
Little penguin	*Eudyptula minor*	Least concern	Noticeable locally	Necropsy	Cannell et al., 2016
Testudines					
Loggerhead sea turtle	*Caretta caretta*	Vulnerable	Frequent scattered	Necropsy/external examination	Gerle and DiGiovanni, 1998; Orós et al., 2005; Tomás et al., 2008; Casale et al., 2010; Cardona et al., 2012; Meager and Limpus, 2012b; Foley et al., 2019
Green sea turtle	*Chelonia mydas*	Endangered	Frequent scattered	Necropsy/external examination	Gerle and DiGiovanni, 1998; Davenport and Davenport, 2006; Chaloupka et al., 2008; Meager and Limpus, 2012b; Denkinger et al., 2013; Work et al., 2015; Segniagbeto et al., 2017; Monzón-Argüello et al., 2018; Foley et al., 2019
Kemp's Ridley sea turtle	*Lepidochelys kempii*	Critically endangered	Frequent scattered	Necropsy	Cannon, 1998; Gerle and DiGiovanni, 1998; Witzell, 2007; Foley et al., 2019
Olive Ridley sea turtle	*Lepidochelys olivacea*	Vulnerable	Rare	Anecdotal (researcher)	Segniagbeto et al., 2017
Hawksbill sea turtle	*Eretmochelys imbricata*	Critically endangered	Noticeable locally	Necropsy	Meager and Limpus, 2012b; Foley et al., 2019
Flatback sea turtle	*Natator depressus*	Data deficient	Rare	Necropsy	Meager and Limpus, 2012b
Leatherback sea turtle	*Dermochelys coriacea*	Vulnerable	Frequent locally	Necropsy	Gerle and DiGiovanni, 1998; Deem et al., 2006; Foley et al., 2019
Carnivora					
Northern sea otter	*Enhydra lutris kenyoni*	Endangered	Rare	Anecdotal (researcher)	Neilson et al., 2012
Southern sea otter	*Enhydra lutris nereis*	Endangered	Noticeable locally	Necropsy	Kreuder et al., 2003
Harp seal	*Pagophilus groenlandicus*	Least concern	Rare	Necropsy/clinical examination	Swails, 2005
Harbor seal	*Phoca vitulina*	Least concern	Noticeable locally	Necropsy/clinical examination	Goldstein et al., 1999; Swails, 2005; Carretta et al., 2012; Barcenas-De la Cruz et al., 2017
Caspian seal	*Pusa caspica*	Endangered	Noticeable scattered	Eyewitness (researcher)	Wilson et al., 2017
Gray seal	*Halichoerus grypus*	Least concern	Noticeable locally	Necropsy/clinical examination	Swails, 2005
South African fur seal	*Arctocephalus pusillus*	Least concern	Rare	Eyewitness	Wickens and Sims, 1994
Southern fur seal	*Arctocephalus forsteri*	Least concern	Rare	Necropsy	Byard et al., 2012
Steller sea lion	*Eumetopias jubatus*	Near threatened	Rare	Anecdotal (unknown)	Allen and Angliss, 2013
California sea lion	*Zalophus californianus*	Least concern	Noticeable locally	Necropsy	Goldstein et al., 1999; Carretta et al., 2012; Barcenas-De la Cruz et al., 2017
Northern elephant seal	*Mirounga angustirostris*	Least concern	Noticeable locally	Clinical examination	Carretta et al., 2012; Barcenas-De la Cruz et al., 2017

(Continued)

TABLE 1 | Continued

Order/common name	Latin name	Status (International Union for the Conservation of Nature, 2019)	Frequency	Reliability	References
Orectolobiformes					
Indian Ocean whale shark	*Rhincodon typus*	Endangered	Frequent scattered	Anecdotal/pictures (researcher)	Gudger, 1938a,b; Graham and Roberts, 2007; Rowat et al., 2007; Speed et al., 2008; Ramírez-Macías et al., 2012; Department of Parks Wildlife, 2013
Lamniformes					
Basking shark	*Cetorhinus maximus*	Vulnerable	Noticeable locally	Anecdotal/pictures (researcher)	Kelly et al., 2004; Speedie and Johnson, 2008; Speedie et al., 2009
White shark	*Carcharodon carcharias*	Vulnerable	Rare	Eyewitness	Towner et al., 2012
Sirenia					
Dugong	*Dugong dugon*	Vulnerable	Noticeable locally	Necropsy	Borsa, 2006; Meager and Limpus, 2012a; Owen et al., 2012; Meager, 2016
American manatee	*Trichechus manatus*	Vulnerable	Frequent locally	Necropsy	Ackerman et al., 1995; Ávila-Canto et al., 2017; Runge et al., 2017
Cetacea					
Harbor porpoise	*Phocoena phocoena*	Least concern	Noticeable locally	Necropsy	Jepson, 2005; Camphuysen and Siemensma, 2011; Fenton et al., 2017; Deaville et al., 2018
Burmeister's porpoise	*Phocoena spinipinnis*	Near threatened	Rare	Anecdotal (researcher)	van Waerebeek et al., 2007
Finless porpoise	*Neophocaena phocaenoides*	Vulnerable	Rare	Necropsy	Parsons and Jefferson, 2000; Jefferson et al., 2002; Morimura and Mori, 2019
Indo-Pacific humpback dolphin	*Sousa chinensis*	Vulnerable	Rare	Necropsy	Jefferson, 2000; Parsons and Jefferson, 2000
Atlantic bottlenose dolphin	*Tursiops truncatus*	Least concern	Noticeable locally	Anecdotal/pictures (researcher)	Lockyer and Morris, 1990; Morgan and Patton, 1990; Bloom and Jager, 1994; Fertl, 1994; Wells and Scott, 1997; McFee and Hopkins-Murphy, 2002; van Waerebeek et al., 2007; Currey, 2008; Wells et al., 2008; Bechdel et al., 2009; Dwyer et al., 2014; Félix et al., 2018
Indo-Pacific bottlenose dolphin	*Tursiops aduncus*	Data deficient	Rare	Necropsy	Kemper et al., 2005; van Waerebeek and Leaper, 2008; Byard et al., 2012
Short beaked common dolphin	*Delphinus delphis*	Least concern	Noticeable locally	Necropsy	Jepson, 2005; Kemper et al., 2005; van Waerebeek et al., 2007; Ritter, 2012; Martinez and Stockin, 2013; Common Wealth of Australia, 2017; Deaville et al., 2018
Commerson's dolphin	*Cephalorhynchus commersonii*	Least concern	Rare	Anecdotal/picture (unknown)	van Waerebeek et al., 2007
Heaviside's dolphin	*Cephalorhynchus heavisidii*	Near threatened	Rare	Anecdotal/picture (researcher)	Elwen and Leeney, 2010
Hector's dolphin	*Cephalorhynchus hectori*	Endangered	Noticeable locally	Necropsy	Stone and Yoshinaga, 2000; van Waerebeek et al., 2007; International Whaling Commission, 2019b

(Continued)

TABLE 1 | Continued

Order/common name	Latin name	Status (International Union for the Conservation of Nature, 2019)	Frequency	Reliability	References
Striped dolphin	*Stenella coeruleoalba*	Least concern	Noticeable locally	Anecdotal (unknown)	van Waerebeek et al., 2007; Common Wealth of Australia, 2017
Spinner dolphin	*Stenella longirostris*	Least concern	Rare	Anecdotal/picture (researcher)	Camargo and Bellini, 2007
Atlantic spotted dolphin	*Stenella frontalis*	Least concern	Rare	Anecdotal (researcher)	van Waerebeek et al., 2007; Luksenburg, 2014
Pantropical spotted dolphin	*Stenella attenuata*	Least concern	Rare	Necropsy	Obusan et al., 2016
Guiana dolphin	*Sotalia guianensis*	Data deficient	Noticeable locally	Eyewitness	van Waerebeek et al., 2007; Santos et al., 2010; da Costa Toledo et al., 2017
White-beaked dolphin	*Lagenorhynchus albirostris*	Near threatened	Rare	Anecdotal (necropsy unconfirmed)	Jepson, 2005
Peale's dolphin	*Lagenorhynchus australis*	Least concern	Rare	Anecdotal/picture (researcher)	van Waerebeek et al., 2007
Pacific white-sided dolphin	*Lagenorhynchus obliquidense*	Least concern	Rare	Necropsy	International Whaling Commission, 2019b
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	Least concern	Rare	Anecdotal	Evans et al., 2011
Risso's dolphin	*Grampus griseus*	Data deficient	Rare	Necropsy	Deaville et al., 2018
False killer whale	*Pseudorca crassidens*	Near threatened	Rare	Anecdotal (researcher)	Luksenburg, 2014
Killer whale	*Orcinus orca*	Data deficient	Noticeable scattered	Eyewitness	Visser, 1999; Ford et al., 2000; Visser and Fertl, 2000; Jensen and Silber, 2003; van Waerebeek et al., 2007; Williams and O'Hara, 2010; Carretta et al., 2012; Weir and Pierce, 2012; Allen and Angliss, 2013
Short-finned pilot whale	*Globicephala macrorhynchus*	Least concern	Noticeable locally	Necropsy	Carrillo and Ritter, 2010; Ritter, 2012; Díaz-Delgado et al., 2018
Long-finned pilot whale	*Globicephala melas*	Least concern	Rare	Anecdotal (unknown)	van Waerebeek et al., 2007
Beluga whale	*Delphinapterus leucas*	Least concern	Noticeable locally	Necropsy	Neilson et al., 2012; Lair et al., 2014
Pygmy sperm whale	*Kogia breviceps*	Data deficient	Noticeable locally	Necropsy	Borsa, 2006; Carrillo and Ritter, 2010; Arbelo et al., 2013; Sierra et al., 2014; Díaz-Delgado et al., 2018
Dwarf sperm whale	*Kogia sima*	Data deficient	Rare	Anecdotal/picture (unknown)	Borsa, 2006; Carrillo and Taverna, 2011
Sperm whale	*Physeter macrocephalus*	Vulnerable	Frequent locally	Necropsy	Félix and van Waerebeek, 2005; Borsa, 2006; Abdulla and Linden, 2008; Carrillo and Ritter, 2010; Carretta et al., 2012; Weir and Pierce, 2012; Arbelo et al., 2013; Sierra et al., 2014; Nanayakkara and Herath, 2017; Deaville et al., 2018; Díaz-Delgado et al., 2018; Di-Méglio et al., 2018; Peel et al., 2018
Cuvier's beaked whale	*Ziphius cavirostris*	Least concern	Noticeable locally	Necrospy	van Waerebeek et al., 2007; Carrillo and Ritter, 2010; Neilson et al., 2012; Arbelo et al., 2013; Sierra et al., 2014; Nanayakkara and Herath, 2017; Deaville et al., 2018; Díaz-Delgado et al., 2018

(Continued)

TABLE 1 | Continued

Order/common name	Latin name	Status (International Union for the Conservation of Nature, 2019)	Frequency	Reliability	References
Stejneger's beaked whale	*Mesoplodon stejnegeri*	Data deficient	Rare	Genetic	Honma et al., 1999; Neilson et al., 2012
Gray's beaked whale	*Mesoplodon grayi*	Data deficient	Rare	Anecdotal/picture (researcher)	Dalebout et al., 2004
Hector's beaked whale	*Mesoplodon hectori*	Data Deficient	Rare	Anecdotal (unknown)	van Waerebeek et al., 2007
Andrew's beaked whale	*Mesoplodon bowdoini*	Data deficient	Rare	Anecdotal (unknown)	van Waerebeek et al., 2007
Sowerby's beaked whale	*Mesoplodon bidens*	Data deficient	Rare	Necropsy	Deaville et al., 2018
True's beaked whale	*Mesoplodon mirus*	Data deficient	Rare	Necropsy	Carrillo and Ritter, 2010
Gervais beaked whale	*Mesoplodon europaeus*	Data deficient	Rare	Necropsy	Moore and Barco, 2013; Díaz-Delgado et al., 2018
Arnoux's beaked whale	*Berardius arnuxii*	Data deficient	Rare	Anecdotal/picture (researcher)	van Waerebeek et al., 2007
Blue whale	*Balaenoptera musculus*	Endangered	Frequent locally	Necropsy	van Waerebeek et al., 2007; Berman-Kowalewski et al., 2010; Carretta et al., 2012; Ilangakoon, 2012; de Vos et al., 2013; Nanayakkara and Herath, 2017; Peel et al., 2018
Fin whale	*Balaenoptera physalus*	Vulnerable	Frequent scattered	Necropsy	Laist et al., 2001; Jepson, 2005; Panigada et al., 2006; Neilson et al., 2012; Allen and Angliss, 2013; Sierra et al., 2014; Deaville et al., 2018; Díaz-Delgado et al., 2018; Peel et al., 2018
Gray whale	*Eschrichtius robustus*	Least concern	Frequent locally	Necropsy	Stroud and Roffe, 1979; Rugh et al., 1999; Laist et al., 2001; Bradford et al., 2009; Ritter, 2012; Scordino et al., 2017
Bryde's whale	*Balaenoptera edeni brydei*	Least Concern	Locally frequent; noticeable scattered	Necropsy	Laist et al., 2001; Félix and van Waerebeek, 2005; van Waerebeek et al., 2007; Carrillo and Ritter, 2010; de Vos et al., 2013; Constantine et al., 2015; Nanayakkara and Herath, 2017; Jeri et al., 2019
Sei whale	*Balaenoptera borealis*	Endangered	Noticeable scattered	Necropsy	Laist et al., 2001; Félix and van Waerebeek, 2005; van Waerebeek et al., 2007; Douglas et al., 2008; Carrillo and Ritter, 2010; Moore and Barco, 2013; Fais et al., 2016; Pirotta et al., 2018
Common Minke whale	*Balaenoptera acutorostrata*	Least concern	Noticeable scattered	Necropsy	Laist et al., 2001; Bogomolni et al., 2010; Hazevoet et al., 2010; Allen and Angliss, 2013; Deaville et al., 2018; Díaz-Delgado et al., 2018
Antarctic Minke whale	*Balaenoptera bonaerensis*	Near threatened	Noticeable locally	Anecdotal	Borsa, 2006; Peel et al., 2018
Bowhead whale	*Balaena mysticetus*	Least concern	Rare	External examination	George et al., 1994

(Continued)

TABLE 1 | Continued

Order/common name	Latin name	Status (International Union for the Conservation of Nature, 2019)	Frequency	Reliability	References
Southern right whale	*Eubalaena australis*	Least concern	Frequent scattered	Necropsy	Best et al., 2001; Greig et al., 2001; Rowntree et al., 2001; van Waerebeek et al., 2007; McAloose et al., 2016; Cates et al., 2017; Figueiredo et al., 2017; Peel et al., 2018
Northern right whale	*Eubalaena glacialis*	Endangered	Frequent scattered	Necropsy	Knowlton and Kraus, 2001; Moore et al., 2004; Bogomolni et al., 2010; van der Hoop et al., 2015; Sharp et al., 2019
Humpback whale	*Megaptera novaeangliae*	Least Concern	Frequent scattered	Necropsy	Wiley et al., 1995; Neilson et al., 2012; Díaz-Delgado et al., 2018; Groch et al., 2018; Peel et al., 2018; Jeri et al., 2019

Frequency indicates whether collision reports were rare (i.e., reported for a limited number of locations with <3 reports in total), noticeable scattered (i.e., >3 reports but scattered over several locations, with generally <3 reports per location), noticeable locally (i.e., >3 reports at one location, but not known as the most common cause of mortality), frequent scattered (i.e., reported throughout distribution range), or frequent locally (i.e., reported as a common cause of mortality within specific areas of overall distribution). Reliability reflects the most reliable data source for all reports: necropsy data, genetic evidence, eye-witnessed collision reports, anecdotal accounts with photographs, or pure anecdotal accounts. It should be noted that the list of references is not exhaustive, but rather reflects the amount of literature available for each species.

Direct Consequences

Injury to the Animal

Animals incur sharp and blunt force injuries when colliding with a vessel (**Figure 1**), which can be lethal immediately upon impact as well as several hours, days, or weeks after the incident (Campbell-Malone et al., 2008; Martinez and Stockin, 2013; Dwyer et al., 2014). Both sharp and blunt force injuries have been extensively described for whales and manatees. Sharp force injuries include external gashes and severed tail stocks or fins, mainly originating from contact with a rotating propeller (e.g., Beck et al., 1982; Moore et al., 2004; Lightsey et al., 2006; Rommel et al., 2007; Campbell-Malone et al., 2008; Douglas et al., 2008; Hill et al., 2017). Blunt force injuries predominantly originate from contact with the bow, hull, skeg, or rudder, and are classified as abrasions (i.e., removal of the epithelial layer of the skin), contusions (i.e., hemorrhages), lacerations (i.e., tearing of the skin), and bone or skull fractures (DiMaio and DiMaio, 2001; Lightsey et al., 2006; Moore et al., 2013). Criteria to identify vessel strikes as the cause of death have predominantly been developed from comprehensive necropsies on whales (Moore et al., 2013) and reliable ways to separate ante-mortem from post-mortem injuries are becoming increasingly more established (Sierra et al., 2014; Arregui et al., 2019). Furthermore, necropsies and observations of whales and manatees surviving a vessel strike have provided information about the relationship between the severity of injury and depth of laceration (i.e., into the skin, blubber, or muscle), anatomical site of injury, and vessel speed (Rommel et al., 2007; Vanderlaan and Taggart, 2007; Conn and Silber, 2013; Wiley et al., 2016; Combs, 2018). Injured animals experience a reduced welfare because of pain, stress, and possible associated negative psychological states, with the extent of welfare reduction being directly related to the type, severity, and duration of an injury (de Vere et al., 2018).

Our knowledge of vessel strike injuries in whales and manatees has contributed to the identification of vessel-related sharp force injuries in smaller species. The finding that almost every smaller marine species listed within **Table 1** has been observed with sharp force injury is likely a result of the relatively easy identification of these injuries. However, there is a lack of knowledge of the species-dependent relationship between the severity of injury and depth as well as location of external gashes. Fatal sharp force injuries on Florida manatees were generally deeper than 17 cm (Rommel et al., 2007), but a similar wound would be less likely to cause fatal injury to a whale with a thick layer of blubber. In contrast, a similar injury could easily decapitate a fish or penguin. There is also a lack of knowledge of the species-dependent relationship between lethality of injury and vessel speed (see section Vessel-related factors below). Finally, interspecific differences in bone strength may result in different risks of incurring blunt force trauma (Clifton et al., 2008).

Injury to Vessel Crew

Crew on vessels that collide with whales may get thrown around or thrown into the water, incur injuries, or even die (Neilson et al., 2012; Ritter, 2012; Peel et al., 2018). Reports of these instances are dominated by, but are certainly not exclusive to, small vessels. For example, in March 2019, a Japanese fast ferry was said to have collided with a whale, resulting in serious injuries to at least 13 passengers (Rahim, 2019). In contrast, vessel crew being thrown off their feet because of collision with a smaller marine animal has only been reported anecdotally for fast racing yachts colliding with sunfish (*Mola mola*) and potentially sharks (Clarey, 2014).

Vessel Damage

Vessel damage (e.g., cracked hulls, damaged hydrofoils, rudder damage) is commonly reported for collisions between whales and large vessels as well as small vessels (Laist et al., 2001; Neilson et al., 2012; Ritter, 2012). Vessel damage from collisions with smaller marine animals is less frequently reported. However, South African fur seals (*Arctocephalus pusillus*) occasionally

FIGURE 1 | Examples of vessel collision injuries observed on live cetaceans. (**Top left**) Southern right whale (*Eubalaena australis*) with a healed propeller injury (Photo credit: Renée Schoeman). (**Top right**) humpback whale (*Megaptera novaeangliae*) with suspected blunt force trauma (Photo credit: Titus Shaanika). (**Bottom**) bottlenose dolphin (*Tursiops truncatus*) with suspected propeller injury (Photo credit: Michelle Caputo).

come too close to the propellers of fishing trawlers while foraging. These close interactions can result in fur seals incurring propeller cuts and bending or breaking the propeller (Wickens and Sims, 1994). Sunfish are well-known for their damaging effects on sailing yachts and even a large cement carrier incurred paint damage to the hull from colliding with this species (Porcasi and Andrews, 2001; Lulham, 2006). It is surprising that no vessel damage, especially propeller damage, has been reported for collisions between small vessels and sea turtles considering their hard carapace. As mentioned under the section "Species of marine animals colliding with vessels", the limited number of vessel damage reports could be due to reporting biases.

Long-Term Consequences for Individual Animals

Long-term consequences of collisions for individual animals are not well-understood, but several species indicate long-term locomotive impairments and possible reduced fitness. Locomotive impairments may result from injuries to flukes, flippers, fins, turtle carapace, and even feather disruptions (Jacobson, 1998; Moore et al., 2013; Cannell et al., 2016). These impairments can potentially prevent effective foraging and could ultimately result in death from starvation. An additional concern is that open wounds and bone fractures increase an animal's energy expenditure (Robbins, 1993 in Visser, 1999; Towner et al., 2012). Consequently, more energy is transferred toward body maintenance, while less energy is available for growth and reproduction, which will eventually cause a decrease in individual fitness (Duffus and Dearden, 1993). It is difficult to assess long-term consequences in more detail, but animal tagging as well as photo-ID records of injured individuals could provide more insight into long-term effects of collision injuries on behavior and survival.

Population Consequences

The impact of collision-related mortality on species and (sub)populations is currently not well-understood (Thomas et al., 2016). A decrease in population growth rate can be caused by a high mortality rate or a decline in fertile animals. The latter is of particular concern for long-lived marine species, which generally have low recruitment rates and an older age of sexual maturity (Heppel et al., 1999). Over time, it is possible that vessel-related mortality might exceed the recruitment rate, either through contributing to a cumulative mortality rate (i.e., mortality from both natural and human-related causes) or on its own (e.g., Kraus et al., 2005; Guo, 2006; van der Hoop et al., 2013; Fais et al., 2016). For species such as North Atlantic right whales (Moore et al., 2004), Hauraki Gulf Bryde's whales (*Balaenoptera edeni bryde*, New Zealand: Constantine et al., 2015), North Pacific blue whales (Redfern et al., 2013; Rockwood et al., 2017), North Pacific humpback whales (Rockwood et al., 2017), North Pacific fin whales (Rockwood et al., 2017; Keen et al., 2019), and Canary Island sperm whales (Spain: Fais et al., 2016), tools like comprehensive ship strike reporting systems, stranding databases, and modeled risk analyses have helped to identify populations for which ship strike rates may exceed population recruitment rates.

The impact of mortality rates is starting to be understood for some populations of smaller species, including the Little penguin (*Eudyptula minor*) population in Perth (Australia: Cannell et al., 2016), the sea turtle populations around Florida (U.S.: Foley et al., 2019), and the Florida manatee (Lightsey et al., 2006). However, for most smaller species, this impact still needs to be assessed.

THE RISK OF COLLISION

The risk of collision is defined as the probability that a collision occurs, combined with the probability that such a collision will lead to a serious outcome (i.e., major injury, mortality, or damage to the vessel: International Whaling Commission, 2011). Assessing the risk of collision between animals and vessels is an essential step toward the implementation of appropriate mitigation measures in relevant geographical areas (Cates et al., 2017; Crum et al., 2019). Assessments of collision risk require information on animal and vessel distribution patterns and, ideally, on specific vessel- (e.g., size and speed) and animal-related factors (e.g., time spent at or near the surface and behavioral response to vessels) (Martin et al., 2016; Crum et al., 2019). However, this information is typically not known and targeted research may be needed prior to assessing a species' collision risk.

Identification of High-Risk Areas

The probability of collision between a vessel and marine animal increases with a higher vessel and/or animal density (e.g., Lagueux et al., 2011; Redfern et al., 2013, 2019; Bezamat et al., 2014; Priyadarshana et al., 2015; Nichol et al., 2017; Rockwood et al., 2017; Di-Méglio et al., 2018). A first important step in collision risk analyses is therefore the identification of high-risk areas: areas where a high number of vessels (shipping lanes, shipping routes, and port approaches) and a relatively high number of animals (areas where a large proportion of the population aggregates or return in high numbers on a regular basis) converge (Cates et al., 2017). At present, 14 high-risk areas are listed by the IWC, based on overlaps in the distribution of large vessels (>100 gross tonnage and typically >30 m) and whales/whale strandings (Cates et al., 2017). Collision reports between large vessels and smaller marine animals are rare (Brown and Murphy, 2010; Balazik et al., 2012; Wilson et al., 2017). Hence, overlap between large vessel and smaller species distribution patterns requires further research.

Overall, there has been a focus on large vessels, because reports have shown that these pose a higher risk to whales (Laist et al., 2001; Jensen and Silber, 2003). While large vessels may indeed increase the risk of lethal injury, there is sufficient evidence that all vessels collide with whales (Best et al., 2001; Neilson et al., 2012; Wiley et al., 2016; Peel et al., 2018) and that even vessels <15 m can cause fatal injury when traveling at high speed (Ritter, 2012). In addition, various marine species in **Table 1** frequent estuarine and coastal waters, either permanently (e.g., estuarine fishes, penguin species with colonies near or on a mainland, seal species with haul-out sites near or on a mainland, manatees, dugongs) or temporarily (e.g., marine turtles during their nesting season, whale sharks, basking sharks, some dolphin species, and mother-calf pairs of some whale species). Species that occur in coastal waters are specifically at risk of collision with small- and medium-sized vessels that occur in high densities near urbanized coastal regions. Efforts should therefore also be put into the identification of high-risk areas based on small vessel traffic, especially in areas where these dominate collision reports (Neilson et al., 2012).

Information on small vessel distribution patterns is difficult to obtain, compared to data on large vessels that emit regular GPS position data and concentrate in shipping routes, shipping lanes, and port entrances. However, simultaneous studies on animal and vessel distribution patterns as well as local vessel registry data have previously assisted in the identification of areas of concern (Preen, 2000; Maitland et al., 2006; Foley et al., 2019).

Factors Affecting the Risk of Collision

After high-risk collision areas are identified, a risk-analysis can be performed that ideally constitutes two steps: (1) modeling the probability of a collision based on encounter rate theory and (2) modeling the probability that the collision is fatal (Martin et al., 2016; Crum et al., 2019). Risk analyses based on encounter rate theory, model the probability that an animal and vessel will be close enough in time and space for an encounter (Martin et al., 2016). Whether an encounter (a) results in a collision and (b) is lethal, depends on both vessel-related (e.g., speed, draft, size) and animal-related factors (e.g., dive pattern, vessel avoidance: Martin et al., 2016; Crum et al., 2019). However, most studies have assessed the overlap between vessel and whale distributions to calculate the probability of an encounter (Fonnesbeck et al., 2008; Redfern et al., 2013; Di-Méglio et al., 2018; Pirotta et al., 2018; Frantzis et al., 2019), or have combined overlap in distributions with vessel speed to model probabilities of a lethal collision (Vanderlaan et al., 2008; van der Hoop et al., 2012; Currie et al., 2017; Nichol et al., 2017; Redfern et al., 2019). Below we list the most important factors, discussing potential differences between species.

Vessel-Related Factors

A broad range of vessel types are involved in collisions (**Table 2**). A vessel poses a higher risk when traveling at a higher speed, because higher speeds result in a stronger impact (i.e., higher force) and increase the risk of serious blunt force trauma (Wang et al., 2007). However, the relationship between vessel speed and severity of injury is a species-dependent relationship that varies between vessel types. For example, Vanderlaan and Taggart (2007) found that the probability of a lethal injury for whales decreased to <50% when large vessels slowed down to 10 knots. Small vessels traveling at a speed of 10 knots are likely to have an even lower probability of lethal injury for whales. In contrast, small vessels (3–6 m in length) had to slow down to at least 7.5 knots to decrease the probability of lethal injury to loggerhead sea turtles (*Caretta caretta*: Work et al., 2010), highlighting a clear difference between species. The species-dependent relationship between vessel speed/type and lethal injury is not well-understood and needs further investigation to support the development of mitigation measures that are appropriate for each species. In addition to a higher probability of lethal injury, high

TABLE 2 | Summary of vessel classes involved in collisions with marine animals, including their typical length in meters (modified after Laist et al., 2001; Lammers et al., 2003; Neilson et al., 2012) and characteristics contributing to collision risk.

Vessel class	Length	Examples	Characteristics contributing to collision risk
Un-motorized small	<15 m	Canoes/kayaks/paddle skies	Silent
Un-motorized medium	15–30 m	Medium sailing	Silent, potentially high speed (racing yachts)
Motorized small	<15 m	Commercial fishing, police, recreational high-speed, research, and wildlife watching vessels	Coastal waters, some intentionally come close to animals, potentially high speed, potentially erratic movements
Motorized medium	15–30 m	Coast guard, commercial fishing, navy, research, and whale watching vessels, passenger ferries	Coastal waters, some intentionally come close to animals, potentially high speed
Motorized large	30–80 m	Coast guard, commercial fishing, navy, and research vessels, passenger ferries, icebreakers, whaling boats	Mass, potentially high speed, manoeuverability
Motorized very large	>80 m	Aircraft carriers, coast guard vessels, container vessels, cruise ships, dredgers, navy vessels, passenger/car ferries (normal, jetfoils, catamaran, trimaran), research vessels, steam ships, tankers	Mass, potentially high speed, manoeuverability

vessel speeds result in a decreased probability of detection of marine animals by vessel operators and vice versa, resulting in a higher probability of collision (Hazel et al., 2007; Gende et al., 2011). Even if vessel operators are aware of an animals' location, the ability to avoid that animal will depend on the detection distance, vessel speed, and vessel maneuverability (i.e., vessel type). Small vessels may be able to move out of the way, even when an animal is close and the vessel is going at high speed, due to better maneuverability. In contrast, large vessels have less maneuverability (i.e., greater response time to initiate and adjust an avoidance maneuver and a greater turning angle) and would need large distances to avoid an animal (Agreement on the Conservation of Small Cetaceans of the Baltic North Seas, 2011).

Large vessels also have deeper drafts and thus, a larger strike zone (i.e., position of animal beneath the surface at which encounters with a vessel result in a collision). The species-specific extent of the strike zone depth in relation to a vessels' draft is currently not known. Silber et al. (2010), however, conducted limited tests of hydrodynamic effects in collisions using scale models of a container ship and a right whale, with the whale at depths up to two times the draft of the vessel. Over 50% of trials resulted in propeller strikes if the whale was considered to act as a rigid body. The effect of vessel draft on the risk of collision needs further research as increased strike zone depths increase estimated species mortality rates (Rockwood et al., 2017). Other vessel-related factors that could play a role are the vessels' acoustic signature (i.e., acoustic signal produced by a vessel mainly from onboard machinery and propeller cavitation), which affects the probability that an animal will hear the upcoming vessel (Leal et al., 2015). Hydrodynamic forces may also be important (Silber et al., 2010; Allen et al., 2012). These forces are likely of significance for small and slow moving species, such as sea turtles, near large vessels and fast-moving small vessels (Work et al., 2010).

Animal-Related Factors

Which animal-related factors (e.g., time spent at surface, type of behavior at surface, behavioral response to vessels, hearing capabilities) affect the risk of collision is not well-understood. One important factor is the amount of time a species spends at or near the surface. Surface time within species may follow a diurnal pattern related to behavior (e.g., Izadi et al., 2018; Keen et al., 2019), but can also vary between individuals of different age classes (e.g., Wiley et al., 1995; Knowlton and Kraus, 2001; Kreuder et al., 2003; Hazel and Gyuris, 2006; Panigada et al., 2006; Carrillo and Ritter, 2010; Neilson et al., 2012; Foley et al., 2019) and sexes (e.g., Kreuder et al., 2003; Panigada et al., 2006). For example, Bryde's whales in the Hauraki Gulf (New Zealand) spent 75–100% of night-time hours in an inactive state (i.e., resting) with dive depths of <9 m (Izadi et al., 2018). Keen et al. (2019) modeled fin whale ship strike risk in the California Current System considering diel patterns of surface use and found that night-time collision risk was twice as high as the daytime risk. The dominant behavior of North Atlantic right whale mother-calf pairs during a calf's first 9 months is comprised of surface resting and near-surface feeding behavior (45–80% of time: Cusano et al., 2019). Similarly, lactating female humpback whales in Exmouth Gulf (Australia) spent 53% of their time within 3 m of the surface (Bejder et al., 2019). Animals at or near the surface are at risk of collision because they are within reach of a vessels' hull and propeller.

One question that remains difficult to answer is why animals do not move out of the way of approaching vessels. Behaviors such as resting, foraging, nursing, and socializing likely distract animals from risk detection (Dukas, 2002). Furthermore, animals potentially do not hear approaching vessels when near the surface. Sound from a vessel reaches an animal via direct and surface-reflected paths leading to constructive and destructive interference (i.e., Lloyd's mirror effect), with moments when vessel noise may be inaudible to the animal (Gerstein et al., 2005; Thorpe, 2010; Erbe et al., 2016). In addition, acoustic shadows in which radiated ship noise levels approach or fall below ambient noise levels, may form ahead of a vessel (Gerstein et al., 2005), leaving that vessel undetectable to animals in its direct path, especially when at the surface.

As mentioned under section "Vessel-related factors," vessels have variable acoustic signatures and animals have variable hearing capabilities (Ketten, 2002). Thus, the distance at which certain vessel types can be detected acoustically likely differs between species. However, even if animals can hear/are aware of a vessel, they may not avoid the approaching vessel or they may take avoidance measures that have limited or adverse effects (Stone and Yoshinaga, 2000; Nowacek et al., 2004). A study of tagged blue whales near shipping lanes off the coast of southern California found that whales do not avoid areas of heavy ship traffic (McKenna et al., 2015). McKenna et al. (2015) also found that blue whales at the surface were limited in their ability to avoid collisions with fast ships because individuals responded to approaching ships with a slow descent and no lateral movement away from the ship. There is no single factor that can tell us why some individuals or species are more prone to vessel collisions than others and more species-specific research is needed to understand interspecific differences.

CURRENT MITIGATION MEASURES, THEIR EFFECTIVENESS, AND SUITABILITY FOR SMALLER SPECIES

A wide variety of mitigation measures that aim to reduce the risk of collisions between vessels and marine animals exist today (e.g., Silber et al., 2012b; Couvat and Gambaiani, 2013; McWhinnie et al., 2018), most of which were developed with a focus on whales. The most suitable mitigation measure(s) depends on the geographic area, environmental conditions, vessels involved, species targeted, time-pressure to implement a mitigation measure, and cost of mitigation (e.g., Weinrich et al., 2010; Silber et al., 2012a; Constantine et al., 2015; McWhinnie et al., 2018). Below we list the mitigation measures that have been developed today and discuss whether they have been effective in the protection of whales as well as whether they could be applied to smaller marine species.

Geographical Measures

Re-routing Measures

Once areas of greatest collision risk have been identified, vessel traffic can be re-routed provided that alternative routes do not compromise safe navigation (e.g., Vanderlaan et al., 2008; Redfern et al., 2013, 2019; Frantzis et al., 2019). Proposals from coastal states to establish or amend routing measures outside, or partially outside, territorial waters need to be submitted to and endorsed by the International Maritime Organization (International Maritime Organization, 1986). In 2009, the IMO published a guidance document to inform member governments about principles to consider when developing actions to reduce collision risk and which guidance documents should be consulted when preparing routing proposals (International Maritime Organization, 2009). Where routing measures fall within territorial waters, decisions can be made directly by coastal states, although these measures may also be submitted to the IMO for revision and approval. Routing measures can be permanent or seasonal, mandatory or recommended, and may apply to all vessels or a sub-set of certain vessel type(s).

Permanent mandatory rerouting measures to prevent ship strikes with whales include Traffic Separation Schemes (TSSs). In June 2003, the TSS in the Bay of Fundy (Canada) was rerouted around the Grand Manan Basin to reduce the risk of lethal encounters between vessels $\geq$300 gross registered tonnage and North Atlantic right whales (International Maritime Organization, 2003; Vanderlaan et al., 2008). Since then, a TTS has been established or amended near Boston (MA, U.S.: International Maritime Organization, 2006, 2007a), within the Santa Barbara Channel (CA, U.S.: International Maritime Organization, 2012), off San Francisco (CA, U.S.: International Maritime Organization, 2012), and in the approach to Panama City (Panama: Guzman et al., 2012; International Maritime Organization, 2014) to protect North Atlantic right, blue, and humpback whales. Year-round recommended routes have been implemented to and from the Port of Auckland to reduce collision risks with Bryde's whales (Ports of Auckland, 2015; Maritime New Zealand, 2019). Seasonal re-routing measures have also been used to protect whales. Specifically, seasonal, voluntary two-way routes were established in Cape Cod Bay and in coastal waters of the southeast U.S. (SEUS) to protect the North Atlantic right whales (Fonnesbeck et al., 2008; Lagueux et al., 2011).

Vessel traffic exclusion zones aim to reduce the number of vessels in an area. Examples are the permanent, voluntary Area To Be Avoided (ATBA) that was adopted by the IMO in 2017 to protect humpback whales near Costa Rica (International Maritime Organization, 2017) and the seasonal ATBAs in the Great South Channel (off Cape Cod Bay, MA, U.S.) and Roseway Basin (south of Nova Scotia, Canada) to protect North Atlantic right whales (International Maritime Organization, 2007b, 2008; Vanderlaan et al., 2008; Vanderlaan and Taggart, 2009). A less common rerouting measure is the establishment of Dynamic Management Areas (DMAs) in the U.S. DMAs are temporary (i.e., 15 days) management areas, established by the U.S. National Marine Fisheries Service (NMFS) for the protection of North Atlantic right whales from collisions with large vessels (Federal Register, 2008). When a (group of) right whale(s) is sighted, a circle providing an area of 44.5 km^2 per whale (i.e., radius of circle is adjusted for the number of right whales in the group), is drawn around the group. Any circle or group of contiguous circles with more than three right whales qualifies to be demarcated as a DMA with a minimum radius of 27.8 km (Federal Register, 2008). All vessels are asked to voluntarily avoid a DMA (or to reduce their speed to $\leq$10 knots while transiting the area, see section Speed restrictions) (Federal Register, 2008; Laist et al., 2014).

Rerouting vessel traffic around areas with known concentrations of whales is an effective mitigation measure (International Whaling Commission, 2014; International Maritime Organization, 2016). The risk of collision can be reduced by 60-95% when compliance with a routing measure is high (e.g., Vanderlaan et al., 2008; Vanderlaan and Taggart, 2009; Guzman et al., 2012; van der Hoop et al., 2012); which generally seems to be the case for IMO adopted routing measures (Silber et al., 2012b). However, compliance with voluntary routing

measures implemented by coastal states varies. Lagueux et al. (2011) found that compliance of tankers and cargo vessels with recommended routes in the SEUS, increased from 51.7 to 96.2% over the first 3 years of implementation. In contrast, compliance with DMAs off the U.S. coast as well as a voluntary 'No Go Area' (NGA) in the St. Lawrence Estuary (Quebec, Canada) was low (Silber et al., 2012a; Chion et al., 2018). It should be emphasized that rerouting is not always feasible (i.e., safety of navigation) and that some rerouting measures only apply to large commercial vessels (most TSSs and ATBAs). Therefore, they do not decrease the risk of collision with small vessels. Even if rerouting measures apply to small vessels, it is more difficult to assess compliance because their location and speed are challenging to monitor. Increased compliance may require enforcement, which is difficult to achieve when the geographic area is relatively large (i.e., large area or multiple smaller exclusion zones spread across a large geographic area), or when a country or state does not have the capacity to apply enforcement actions. In addition, several studies have indicated that rerouting measures assisting one species could increase the risk of collision for other species, highlighting the need for a multi-species research approach when assessing the efficacy of rerouting measures (e.g., Redfern et al., 2013; Priyadarshana et al., 2015; Ritter et al., 2019).

Reducing the overlap between vessel traffic and aggregations of animals can also be a successful mitigation method for smaller species. Rerouting TSSs is not possible for coastal species, where most overlap will be found around port entrances. However, vessel traffic exclusion zones can provide opportunities for risk reduction. A small number of no-go-zones was established to protect the Florida manatee (Florida Fish Wildlife Conservation Committee, 2018). This type of measure can potentially be implemented for a much wider variety of animals that aggregate year-round or seasonally in particular areas. Vessel traffic could, for example, be excluded year-round from important dugong habitat in Queensland (Australia), or seasonally (March-July) around loggerhead and green turtle (*Chelonia mydas*) nesting beaches along the U.S. Florida coast (Maitland et al., 2006; Foley et al., 2019).

Source-Based Mitigation Measures

Speed Restrictions

Implementations of vessel speed restrictions have been suggested to provide animals and vessel crew with more time to detect and avoid each other as well as to reduce the severity of injury (Hazel et al., 2007; Vanderlaan and Taggart, 2007; Gende et al., 2011; Conn and Silber, 2013). The implementation of reduced vessel speeds was first proposed by Laist et al. (2001). Vanderlaan and Taggart (2007) modeled the relationship between vessel speed and probability of lethal injury from collision reports with large whales. They found that the probability of lethal injury decreased to <50% when vessels traveled at speeds ≤10 knots. Conn and Silber (2013) used a slightly larger database and found similar results. They also found that the ship strike rate went down as vessel speed decreased. Implementation of vessel speed restrictions to protect whales from collisions with large vessels have been numerous, with vessel speed restrictions ranging from ≤13 to ≤10 knots (e.g., Federal Register, 2008; McKenna et al., 2012; Ports of Auckland, 2015; Currie et al., 2017; Ritter et al., 2019). Similar to rerouting measures, proposals from coastal states to implement vessel speed restrictions outside territorial waters need to be submitted to and endorsed by the IMO (Silber et al., 2012b). Vessel speed reductions can also be voluntary or mandatory as well as permanent or seasonal.

A reduction in vessel speed has been successful in reducing collision risk and is the preferred measure to implement when vessels cannot be re-routed (International Whaling Commission, 2014; International Maritime Organization, 2016). Humpback whale surveys conducted with a small vessel traveling at speeds between 5-20 knots revealed that whales were three times more likely to be sighted beyond the close encounter distance of 300 m when vessels traveled at speeds ≤12.5 knots (Currie et al., 2017). In addition, the mean detection distance of close encounters (i.e., ≤300 m) increased from 190 m to 211 m (Currie et al., 2017). A reduction in vessel speed is the only mitigation measure that has been recommended for a variety of smaller marine species, such as manatees (Calleson and Frohlich, 2007), dugongs (Hodgson, 2004), sea turtles (Hazel et al., 2007; Work et al., 2010), and fish (Brown and Murphy, 2010). However, compliance with vessel speed restrictions can be low (e.g., Gorzelany, 2004; Jett and Thapa, 2010; Lagueux et al., 2011; McKenna et al., 2012; Freedman et al., 2017). Vessels traversing DMAs, for example, generally did not reduce their speed to the recommended 10 knots (Silber et al., 2012a). Initial compliance with a mandatory 10 knots speed restriction in Seasonal Management Areas (SMAs) was also low, but improved with targeted enforcement programs (Silber et al., 2014). Similar boater compliance issues were found in speed restriction zones to protect manatees; compliance varied between Sarasota and Lee County sites as well as between vessel types, with a lower compliance by smaller vessels (Gorzelany, 2004; Jett and Thapa, 2010). However, compliance increased in the presence of law enforcement, highlighting that enforcement efforts are important to assure effectiveness of speed reduction measures (Gorzelany, 2004; Jett and Thapa, 2010).

As noted under section "Vessel-related factors," there is no known relationship between vessel speed and collision risk for smaller marine species, mainly because the data needed to infer such relationships have not been collected. Research on sea turtles has indicated that individual turtles are more likely to flee from an approaching vessel when speeds are reduced to 2 knots, while the probability of lethal injury decreased by 60% for vessels idling at 4 knots (Hazel et al., 2007; Work et al., 2010). Large differences in the relationship between vessel speed and collision risk can therefore be expected between species and more species-specific research is needed to identify these relationships.

Animal Detection Onboard the Vessel

Collisions with animals can be avoided if animals are detected and appropriate avoidance measures are adopted by the vessel operator. Vessel crew are generally not trained to detect and identify marine animals and are likely focussed on other aspects of the voyage. Placing a trained, dedicated observer onboard a vessel has been suggested to help increase the detection rate of whales along a vessel's route during day-light hours.

The effectiveness of placing trained, dedicated observers on a ship's deck or bridge to detect whales has been tested for high-speed ferries and commercial cargo vessels (Mayol et al., 2008; Weinrich et al., 2010; Flynn and Calambokidis, 2019). Observers were found to detect more whales than standard vessel crew and often at larger distances from the vessel. This early detection provides vessel crew more time to take avoidance measures. However, as highlighted under section Vessel-related factors, large vessels have less maneuverability and may not be able to effectively avoid whales despite observers effectively locating animals. In contrast, small vessels have greater maneuverability, but observers are closer to the sea-surface reducing the effective sighting distance. Onboard observers are therefore only suitable for vessels that are large enough to provide observers with an elevated platform that enables detecting animals over a sufficient range, but small enough to effectively maneuver. Even in the presence of trained observers, collisions with whales occur when they are not seen or seen too late to take avoidance measures (Wiley et al., 2016). This risk is higher for species that spend more time near the surface instead of at the surface.

During night hours, observers could make use of infrared cameras that create images from infrared radiation emitted by a whale, which is a function of both body and/or blow temperature and spectral emissivity (Cuyler et al., 1992; Horton et al., 2017). The effectiveness of infrared imagery has been addressed in various studies (e.g., Barber et al., 1991; Burn et al., 2009; Graber, 2011; Yonehara et al., 2012), but more data is needed to assess its use for effective mitigation against collisions with vessels (Horton et al., 2017). Woods Hole Oceanographic Institution is currently experimenting with new automatic infrared detection techniques (Lubofsky, 2019). Another around-the-clock detection method for marine animals is active sonar: a method to detect objects of various sizes by releasing acoustic energy into the marine environment and subsequently receiving the echoes that bounce off the object (Kozak, 2012). However, the release of acoustic energy is of concern as increased levels of noise are known to negatively affect all species (Popper and Hawkins, 2012, 2016).

Animal detection measures are unlikely to result in a significant decreased risk of collision for smaller species because they are less easily sighted by observers at the distances needed to implement an avoidance measure. In addition, infrared cameras will not work on small-bodied animals, especially without the extra cue of a large blow. The detection range of active sonars decreases with decreasing water depth. Thus, active sonar is unsuitable for shallow coastal areas frequented by species, such as manatees, dugongs, and turtles (Gerstein, 2002).

Deterrent Devices

Deterrent devices can be installed directly on vessels to alert a marine animal to- and deter them from an approaching vessel without vessel crew needing to detect the animal.

Nowacek et al. (2004) tested the effect of an acoustic alerting stimulus on North Atlantic right whale behavior and found that individuals moved to the surface. This behavioral response would increase, rather than decrease, their collision risk. Lagerquist et al. (2013) did not observe any avoidance of an acoustic deterrent device by Gray whales (*Eschrichtius robustus*) migrating along the Oregon coast. Hence, there is currently no evidence that acoustic deterrent devices work for whales. Gerstein and Blue (2004) developed a Manatee Alert Device (MAD) that sends out a low intensity, highly directional sound. Ninety-five percent of manatees elicited an avoidance response during test trials (Gerstein and Gerstein, 2017). In addition, manatees avoided the active MAD at a greater mean distance (20 m) in comparison to non-active controls (6 m: Gerstein and Gerstein, 2017). Lenhardt (2002) developed an alerting device for sea turtles that emits acoustic signals from 0.2 to 15 kHz as well as a visual deterrent cue to a) initiate a fleeing response and b) direct animals away from the vessel because turtles flee in the direction that they are facing. However, there are no data available on the effectiveness of this turtle alerting device. There are also concerns that sounds emitted by acoustic deterrent methods potentially cause acoustic trauma (i.e., hearing loss), displace animals from important habitats, or affect non-targeted acoustically sensitive marine species (Johnston and Woodley, 1998; Morton and Symonds, 2002; Olesiuk et al., 2002; Barlow and Gisiner, 2006). In addition, marine animals may get habituated to the deterrent signal, which would render the device ineffective. We therefore conclude that deterrent devices are not an effective means of mitigating collisions with any marine animal.

Propeller Guards

Propeller guards, such as cages and ducts, can be installed around a propeller as a physical boundary between the propeller blades and an animal. The use of propeller guards has not been tested for large whales. Work et al. (2010) tested the ability of propeller guards to protect loggerhead sea turtles from being injured by small vessels. Propeller guards helped to reduce the risk of lethal injury from 40 to 10% for vessels at idle speed (i.e., 4 knots), but no reduction in risk was seen at planing speed (i.e., 22 knots), because of an increased risk of blunt force trauma. In addition, propeller guards of a different design were not as effective, even at idle speed. These results highlight the need for further research into the best designs for propeller guards. However, in combination with a reduction in speed, propeller guards could effectively reduce sharp force injuries.

Technological Data and Information Systems

Technological data and information systems have primarily been developed to aid the mitigation of collisions with large whales, although some may also protect smaller species. In general, these systems are used to alert mariners that they are entering an area with a high density of animals prone to collisions (Ward-Geiger et al., 2005), to alert mariners of recent animal sightings (National Marine Fisheries Service, 2005; Ward-Geiger et al., 2005; van Parijs et al., 2009; Souffleurs d'Ecume, 2012; Conserve.iO, 2019), to gather data on vessel abundance and distribution (van der Hoop et al., 2012), and to gather data on vessel compliance with mitigation measures (Lagueux et al., 2011; McKenna et al., 2012; Silber et al., 2014).

Mandatory Ship Reporting (MSR)

There are two MSR areas along the eastern U.S. coastline that surround critical North Atlantic right whale habitat: 1 year-round area off the state of Massachusetts and one seasonal area within the SEUS (Ward-Geiger et al., 2005). These measures were adopted by the IMO in 1998 and represent the first involvement of the IMO in implementing measures to protect whales from collisions. The MSR system requires all ships ≥300 gross tonnage to report to a shore-based station when entering the areas. A land-based station stores all incoming ship reports and returns an automated message on steps to avoid collisions with whales (i.e., keep a look-out and reduce speed) as well as recent whale sightings.

MSR systems themselves are not an effective mitigation measure to protect right whales from collisions, but have been regarded as a successful method to educate mariners on ship strike issues and measures to decrease the risk of collision (International Whaling Commission, 2011). MSR systems also provide the opportunity to gather ship transit data (i.e., ship route, ship speed, and primary destinations), which can assist with the development of mitigation measures and assessment of compliance with mitigation measures (Ward-Geiger et al., 2005; Silber et al., 2015). However, ship transit data can now be derived from AIS data for large vessels. We expect the effects of MSR systems to be similar for smaller species. Although MSR systems can help to educate mariners, there are other, less costly ways available to achieve education goals.

Early Warning System (EWS)

The EWS is an aerial survey network operated within the SEUS (from Georgia, south along the coast of Florida), the Great South Channel, and Cape Cod Bay (Boston, MA) (National Marine Fisheries Service, 2005). The EWS was established to reduce ship strikes with North Atlantic right whales by providing whale sighting information to the U.S. Navy (USN), U.S. Coast Guard (USCG), U.S. Army Corps of Engineers (USACE), harbor pilots, port authorities, and other maritime organizations. Sighting information is subsequently distributed to commercial and recreational vessel crew. If the EWS sights whales near shipping lanes, vessels are requested to reduce their speed and where possible, to undertake avoidance measures to prevent collision with- or serious injury to whales.

There is no evidence that this reporting system has reduced the number of collisions with right whales along the east coast of the U.S. (Lagueux et al., 2011). In addition, aerial surveys are costly as well as restricted to good weather conditions. Furthermore, communication of animal sightings to smaller vessels is challenging because they are often less well-equipped for radio communication. Hence, despite many smaller marine animal species also being visible from aerial surveys (e.g., dolphins, dugongs, manatees, sharks, and even sea turtles: Irvine and Campbell, 1978; Preen, 2000; Kessel et al., 2013; Martins et al., 2013), we do not recommend that these systems are specifically implemented to mitigate vessel collisions. An EWS may work when aerial surveys are already flown for other research purposes. However, solving the issue regarding communication with smaller vessels will require development of alerting systems, such as mobile phone apps (see section Recent mobile phone alerting systems).

Passive Acoustic Buoy Systems

Passive acoustic buoy systems can be used to improve the detection of marine animals. The Cornell Laboratory of Ornithology and Woods Hole Oceanographic Institution in the U.S. developed a real-time passive acoustic buoy system that specifically recognizes North Atlantic right whale calls (van Parijs et al., 2009). One of these systems is moored to the seafloor along the TSS approaching Boston harbor, where large tankers cross a primary right whale feeding habitat. The buoys listen to whale calls and communicate their data to a shore-based laboratory for whale call verification. All information is forwarded to the right whale Sighting Advisory System (SAS); a multi-institutional effort to monitor right whale populations within northeast U.S. waters. The SAS will alert mariners to the presence of right whales via verbal updates to commercial vessels, 24-h radio broadcasts, and postings on several websites (van Parijs et al., 2009).

Similar to the EWS, there is currently no evidence that passive acoustic buoy systems help to reduce collision risk. In addition, vocalizations for some species have strong temporal patterns or depend on an individuals' behavior, resulting in inconsistent acoustic detection probabilities (e g., Baumgartner and Fratantoni, 2008; Feng and Bass, 2016; Webster et al., 2019). Furthermore, the detection range of vocalizations is reduced for vocalizations at a higher frequency, thereby reducing the potential effective mitigation range for Odontocete species. For other species, we lack knowledge of their acoustic repertoire (Ferrara et al., 2014). Buoy systems also come at a cost and require regular maintenance to prevent deterioration. Considering these constraints, we recommend that further research be conducted to determine the utility of passive acoustic buoy systems in reducing collision risk.

Real Time Plotting of Cetaceans (REPCET)

REPCET is a software system developed to reduce ship strikes with whales in the Pelagos Sanctuary (Mediterranean Sea: Mayol et al., 2008; Mayol, 2012) and can be installed on commercial as well as recreational vessels. Once an animal is sighted, the observer inserts the GPS position in the REPCET system, which subsequently transmits the sighting data to a shore-based station. From the shore-based station, the information is sent to other ships equipped with a REPCET system within the sighting area. The onboard receiver automatically processes the data and displays the sightings on a digital map, including an associated risk zone. Each vessel with REPCET will automatically receive a warning signal upon entry of a risk area.

To date, the effectiveness of REPCET in the prevention of collisions has not been verified (Couvat and Mayol, 2014). However, although designed to protect large whales, the system appears to function well in distributing sighting information of both large and small cetaceans to mariners (Couvat and Mayol, 2014). We think it will be valuable to evaluate the effectiveness of REPCET in reducing collision risk with whales and smaller cetaceans, as animal positions are relayed in a seemingly faster manner then via MSR, the EWS, or passive acoustic methods. The

illustrative display of sightings and automated warning signal may also be an easier and more effective way to encourage mariners to slow down, be cautious, and undertake an avoidance maneuver. However, we are concerned with the use of such systems on small recreational and commercial vessels. These smaller vessels are often attracted to or even specifically aim to view charismatic species (e.g., whale- and dolphin-watching vessels, ecotour vessels, sunset cruises), and may use REPCET to find and potentially harass these animals.

Recent Mobile Phone Alerting Systems

Mobile technology continuously progresses and has been used to spread information on mitigation measures and animal sightings since 2012. The app "Whale Alert" was the first mobile technology to provide the U.S. shipping industry with information on North Atlantic right whale management areas, required reporting areas, recommended routes, and ATBAs. It also provides near real-time warnings of right whale detections from the passive acoustic buoy system near the Boston TSS (Conserve.iO, 2019). At present, the app is being diversified to include warnings for multiple whale species and to cover a larger geographical area. The sister app "Manatee Alert" alerts boaters to manatee management areas and provides a means to easily report an injured or distressed manatee. In recent years, several other apps (e.g., SpotterPro, Seafari, WhaleReport) have become available that allow people to log marine animal sightings directly from their mobile phone. These applications could potentially be converted into similar alerting systems as "Whale Alert."

At present, it is unknown whether "Whale Alert" or "Manatee Alert" have helped to reduce collision risks with whales and manatees. However, reporting apps do have the potential to aid in voyage planning and to provide information about animal distribution patterns via public reporting (i.e., citizen science). Citizen science data is often characterized by challenges, such as misidentification of species and the absence of effort data. These challenges can be overcome by applying sighting selection criteria (e.g., photo of species, detailed description of species) to decrease species identification biases and by applying background sampling techniques (e.g., including a proxy of human densities) to account for effort biases (Derville et al., 2018). Thus, reporting apps may be useful to identify species hotspots and therefore, assist in the identification of potential high-risk areas. In addition, apps can provide an easy means to report marine animal strandings or sightings of injured animals.

Education and Awareness

As mentioned under section Mandatory Ship Reporting (MSR), one of the earliest efforts to educate and create awareness with mariners about collision risks was the broadcasting of messages via the MSR system. Since then, education and creating awareness initiatives have been started globally. Global efforts have been undertaken by the IMO, who published a collision guidance document that member governments were encouraged to circulate further to stakeholders and interested parties (International Maritime Organization, 2009). The IWC is putting continuous efforts into global public outreach initiatives, which have been a topic of attention since the start of the SSWG (International Whaling Commission, 2007). While global efforts will ensure that mariners receive consistent information about collision risks, it can take a considerable amount of time to compose and distribute internationally relevant data. Hence, local efforts to educate mariners on the risk of collision with a specific species or within a specific area are a faster way to create awareness and help mitigate collisions in local hotspots. An example of a more localized effort is the development of an education module for maritime academies, as well as certification and licensing courses, by the New England Aquarium under a contract issued by the NMFS in 2003. This module aims to educate vessel officers and crew about the potential for vessel strikes with North Atlantic right whales and the regulatory measures in place to protect these whales (Knowlton et al., 2007). The merchant marine trainer module has been introduced to various marine academies in the U.S. as well as to international maritime schools that are likely to train mariners who transit the east coast of the U.S. and Canada (Knowlton et al., 2007). Identifying and evaluating these types of programs for use on a wider scale or for other marine user groups is included in the latest IWC strategic plan (Cates et al., 2017). In New Zealand, a special Bryde's whale ship strike working group has been established to investigate and share information on the cause of ship strikes with Bryde's whales as well as to develop and discuss feasible mitigation measures (Constantine et al., 2015). This working group includes individuals from industry, government, academic institutions, non-government organizations, and local Māori tribes. During a joint ACCOBAMS/Pelagos workshop, shipping company representatives highlighted the importance of educating captains and vessel crew on the risk of collision (Weinrich et al., 2005).

Education is the fundamental basis for the implementation of mitigation measures and for compliance with regulations, because people need to understand the risk to animals, vessels, and vessel crew as well as the locations where vessel crew are likely to encounter marine animals, and what they can do to avoid a collision (Ritter, 2012; Flamm and Braunsberger, 2014). It is difficult to assess quantitatively how education and awareness reduce collision risk, but it is generally known that education leads to active engagement. We therefore suggest that more effort is dedicated to creating awareness about collision risks with marine animals, regardless of species. Whether education efforts are developed globally or locally should depend on factors, such as the species distribution, number of locations in which a species is at risk of collision, and types of vessels involved in collisions.

ISSUES AND KNOWLEDGE GAPS

Assessing the Extent of Collision Incidences

A total of 75 marine species have been identified to collide with marine vessels, which illustrates that collisions with marine life may comprise a much larger problem than initially thought. However, for most smaller species, we know little about the extent of collision incidences. This knowledge gap should be addressed. Most collision reports involving smaller species were

based on signs of sharp force trauma. Although differences in the most prevalent trauma may exist between species, it is highly unlikely that smaller marine species are not subject to blunt force injury. The absence of blunt force trauma suggests that the information needed to identify this type of trauma in smaller species may be absent, which would result in an underreporting of collision incidents. Detailed criteria have been developed for the identification of both sharp and blunt force trauma in whales and manatees as well as for sharp force injuries in dolphins, seals, and sea turtles (Moore et al., 2013; Foley et al., 2019). However, there is a need to develop species-specific necropsy protocols that will allow for the identification of collision-related blunt force trauma in smaller species.

The next step is to establish an international collision database for smaller marine species, as has been done by the IWC for whales. The IWC database has been shown to be a valuable tool for identifying the species most affected, vessels involved in collisions, and correlations between vessel speed and collision risk (Jensen and Silber, 2003). Hence, establishing a database for other marine species could provide similar valuable information. Unlike large whales, many populations of smaller species are at risk of collision within smaller geographic regions. We therefore think that the establishment of a database for small species will work best via mandatory and standardized reporting protocols that are implemented and managed by government authorities. Local databases should then annually be submitted to an international database.

Long-Term Consequences of Collisions

Injuries relocate energy from growth and reproduction to body maintenance (van der Meer, 2006), but there is a lack of information on how non-fatal injuries affect individual fitness over prolonged timeframes. In addition, for many populations it is unknown how collision-related mortality contributes to the overall mortality rate. There is an urgent need to move beyond the quantification of the type of injury and to assess population level consequences. Once we can begin to assess population consequences, we will then be able to consider how these consequences affect ecosystem structure, function, and stability (Wong and Candolin, 2015).

The Risk of Collision

The identification of high-risk areas is an important step toward the implementation of mitigation measures, but has so far focussed on whales. In addition, the identification of high-risk areas is likely biased because of global information gaps on vessel as well as animal abundance and distribution. The distribution and abundance of smaller vessels is poorly understood because they do not have to use designated shipping lanes and are not required to carry an AIS transponder that transmits their position (Lagueux et al., 2011). A lack of data on small vessel distribution patterns prohibits the identification of high-risk areas for coastal species. A long-term option to trace small vessels could comprise mandatory installation of simple, cost-effective, GPS-tracking systems on small vessels to monitor general movement patterns. However, implementation of such a system will take time. A quicker solution could be to start simultaneous surveys on animal and small vessel distribution patterns in areas where collisions are frequently reported or in coastal areas where species aggregations are known to overlap with vessel traffic. Identified Important Marine Mammal Area's (IMMAs: discrete portions of habitat, important to marine mammal species, that have the potential to be delineated and managed for conservation) may be a good starting point to identify high-risk areas for smaller marine mammals (International Whaling Commission, 2019a).

As mentioned under section "Assessing the extent of collision incidences", assessment of the extent of collision events and factors affecting the risk of collision can be facilitated by an international database (Jensen and Silber, 2003). Currently there is not enough information about smaller marine species to model collision risk as a function of vessel speed and to assess which types of vessels collide with smaller species. However, it should be highlighted that collisions with large vessels are unlikely to be reported, regardless of the establishment of a comprehensive database, because crew on these vessels will be unaware of collisions with smaller species. Reporting biases should be considered when making inference from a collision database (Peel et al., 2018).

Mitigation Measures

Two mitigation measures have been identified to successfully mitigate collisions with whales: re-routing of vessel traffic around areas of greatest relative risk and a reduction in vessel speed. Similar mitigation measures will be effective for the protection of other marine species. Several studies have highlighted the non-compliance of smaller vessels with mitigation measures unless there is enforcement (Gorzelany, 2004; Jett and Thapa, 2010). Successful mitigation of collisions with smaller species therefore requires careful consideration of methods to ensure that compliance is high. Education and enforcement are key to compliance. Education can start with handing out information brochures when issuing skippers tickets or permits to operate in specific areas.

It should be highlighted that animals may change their distribution, timing of migration, expand their range etc. Thus, a constant re-evaluation of implemented mitigation strategies is important (Record et al., 2019). More information is needed about unintended consequences as well as potential benefits associated with the implementation of specific mitigation measures. A reduction in vessel speed, reduces the risk of lethal injury, greenhouse gas emissions, and noise at low frequencies (10-100 Hz) (Joy et al., 2019; Leaper, 2019). However, lower speeds also result in increased transit times and may result in a higher probability of a collision for species that do not avoid vessels (Gerstein et al., 2005; Martin et al., 2016). In addition, reduced vessel speeds reduce noise at higher frequencies (10–100 kHz) less effectively and may therefore result in prolonged exposure with consequent negative effects on species sensitive to high frequency noise (Joy et al., 2019). There is also a paucity of information on how mitigation measures implemented to protect one species affect other species within the same area. Re-routing vessel traffic around one species habitat, for example, may increase risk to a different species (Redfern et al., 2013; Ritter et al., 2019).

Which mitigation measures should be applied depends on the species involved, other species within the area, vessel traffic (i.e., predictability and manageability), the geographic and environmental features of the area, and the economic impacts of the mitigation measure (Laist and Shaw, 2006; Couvat and Gambaiani, 2013; Constantine et al., 2015). The effectiveness of mitigation measures depends on their design and the level of compliance. Selection of effective mitigation measures requires a multi-species approach and active interactions between relevant stakeholders so that individual priorities can be identified and addressed (Constantine et al., 2015; Redfern et al., 2019).

CONCLUSIONS

To date, most scientific publications on collisions have focused on the interactions between large vessels and large whales. Consequently, over the years we have gained valuable insights on the risk of collision to large whales as well as how to effectively mitigate collisions with large whales. Our review found that at least 75 marine species, including smaller whales, dolphins, porpoises, dugongs, manatees, whale sharks, sharks, seals, sea otters, turtles, penguins, and fish have collided with vessels. To date, data on collisions with smaller marine species is scarce, which is likely more a result of reporting biases than a reflection of the true extent of the collision problem. Reliable reporting requires the establishment of species-specific necropsy protocols to accurately identify collision-related injury and mortality, especially for blunt-force trauma. The establishment of an international standardized database, like the IWC database for large whales, could aid in filling information gaps on frequency of collisions, vessels involved, and potential high-risk areas for smaller species. In addition, technology and research are needed to estimate risk in areas used by smaller vessels, smaller species, and less well-studied species. Prioritizing these three areas (i.e., species-specific necropsy protocols, a database for smaller species, and identification of high-risk areas for smaller species) would be a valuable step toward the mitigation of collisions with species other than large whales.

AUTHOR CONTRIBUTIONS

RS conceived and designed the study, collected and analyzed the literature, produced the tables and figures, authored and reviewed the drafts of the manuscript, and approved the final manuscript. CP-A produced the tables and figures, reviewed the drafts of the manuscript, and approved the final manuscript. SP conceived and designed the study, reviewed the drafts of the manuscript, and approved the final manuscript.

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