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Widespread exposure of marine parks, whales, and whale sharks to shipping

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ABSTRACT

Context. Shipping impacts are a major environmental concern that can affect the behaviour and health of marine mammals and fishes. The potential impacts of shipping within marine parks is rarely considered during the planning process. **Aims.** We assessed the areal disturbance footprint of shipping around Australia, its overlap with marine parks, and known locations of megafauna, so as to identify areas of concern that warrant further investigation. **Methods.** Automatic Identification System (AIS) shipping data from 2018 to 2021 were interpreted through a kernel-density distribution and compared with satellite data from ~200 individuals of megafauna amalgamated from 2003 to 2018, and the locations of marine parks. **Key results.** Over 18% of marine parks had shipping exposure in excess of 365 vessels per year. Around all of Australia, 39% of satellite-tag reports from whale shark and 36.7% of pygmy blue and humpback whale satellite-tag reports were in moderate shipping-exposure areas (>90 ships per year). Shipping exposure significantly increased from 2018 despite the pandemic, including within marine parks. **Conclusions.** These results highlight the wide-scale footprint of commercial shipping on marine ecosystems that may be increasing in intensity over time. **Implications.** Consideration should be made for assessing and potentially limiting shipping impacts along migration routes and within marine parks.

Keywords: acoustic pollution, AIS, marine parks, satellite tag, shark, ship strikes, shipping, whale.

Introduction

Migrating marine wildlife are exposed to global shipping activity throughout the world's oceans. Those particularly vulnerable include large whales, whale sharks and basking sharks, also known as marine megafauna or marine giants (Pirotta *et al.* 2019). These marine animals are capable of widespread ocean movements and share similar traits such as large body size and time spent at the surface feeding, breathing or basking that makes them susceptible to shipping activity (Pirotta *et al.* 2019). Direct interactions with shipping can result in ship strike with marine giants, which can cause serious injury or fatalities. For example, in the waters around Canada and the United States, shipping interactions with North Atlantic right whales (*Eubalaena glacialis*) are directly limiting the recovery of the population (Meyer-Gutbrod and Greene 2018). Shipping also has a number of indirect consequences that can affect marine megafauna, including acoustic pollution from ship engines (Wilcock *et al.* 2014), which can interfere with whale communication (Tennessen and Parks 2016; Tsujii *et al.* 2018) and lead to changes in behaviour or movement (Guzman *et al.* 2020; Martin *et al.* 2022), and chemical pollution, such as, for example, oil spills (Liubartseva *et al.* 2015). The impacts of direct interactions with marine giants can be observed because they leave visible marks on vessels and animals (Peel *et al.* 2018), but many animals sink when killed and thus deaths from ship strikes are heavily under-reported (Speed *et al.* 2008). The more diffuse and perhaps pervasive impacts of indirect shipping effects are less well understood.

Australian waters are home to many of the world's marine megafauna and UNESCO-listed world heritage marine parks such as the Great Barrier Reef and Ningaloo Coast world heritage areas. The shipping density around Australia is low compared with that in other parts of the world, with fewer vessels operating and fewer large ports (Wang and Wang 2011), and, therefore, any risk in this geographic region would be low in relation to global impacts. However, the potential for shipping interactions with marine megafauna is high. For example, eastern coast humpback whale populations (*Megaptera novaeangliae*) conduct annual migrations from their feeding grounds in Antarctica to northern Australian waters (Chittleborough 1965; Dawbin 1966). Whale sharks (*Rhincodon typus*) migrate along the western coast of Australia past Perth to Ningaloo Reef during the austral autumn and winter (Norman *et al.* 2016; Reynolds *et al.* 2017). Other species of baleen whales that migrate through Australian waters include the dwarf minke whale (*Balaenoptera acutorostrata*, eastern coast migration only), the southern right whale (*Eubalaena australis*) and the pygmy blue whale (*Balaenoptera musculus brevicauda*, western coast; Double *et al.* 2014). During their migrations, many of these species travel close to the shoreline, following narrow migratory corridors (Pirotta *et al.* 2016) and passing some of Australia's largest cities and shipping ports. Other species of marine megafauna persist within Australian waters; however, many of these species remain data deficient and, therefore, interactions with shipping are simply unknown (Harcourt *et al.* 2014).

Marine protected areas (MPAs), also known as marine parks or marine reserves, are designed to create pockets of ocean free from, or with reduced levels of, disturbance. These parks can have limited or no fishing, controlled recreational use, and low industrial use. When planned and enforced effectively, marine protected areas can increase the diversity and abundance of fauna and flora (Malcolm *et al.* 2018) and have flow-on effects outside of their boundaries (Russ *et al.* 2003; da Silva *et al.* 2015; Di Lorenzo *et al.* 2016). However, there is growing evidence that many marine parks are not properly enforced (Harasti *et al.* 2019) or are poorly planned, with zoning not covering appropriate areas or providing adequate management protections (Gill *et al.* 2017). The fishing, mining and tourism impacts within future and current marine protected areas are generally considered, yet there is often little consideration of potential impacts from commercial shipping (Erbe *et al.* 2012) or smaller recreational vessels. This may be due to a lack of awareness of the scale of the shipping impact footprint in marine environments.

As a result of the COVID pandemic, shipping traffic temporarily decreased (Notteboom *et al.* 2021). Previously, as a direct result of the 2008 financial crisis where many vessels were ~20% smaller than they are today (Notteboom *et al.* 2021), commercial shipping vessels around the world adopted 'super-slow steaming' (travelling at ~12 knots or ~22.2 km h^{-1} rather than ship-designed cruising speeds of ~20 knots or ~37 km h^{-1}) as a means of lowering fuel costs and emissions (Corbett *et al.* 2009; Maloni *et al.* 2013). At these speeds, shipping noise levels (and their associated impacts) are lower (~10 dB less than normal) than at normal cruising levels (~190 dB re 1 μPa^2 at 1 m) (McKenna *et al.* 2013; Gassmann *et al.* 2017). Slower shipping speeds also reduce likelihood of ship strikes on marine megafauna (Conn and Silber 2013; Laist *et al.* 2014; Crum *et al.* 2019), although they may not reduce the lethality of collisions (Kelley *et al.* 2021). Global shipping speeds and volume will likely increase beyond pre-pandemic levels in the years following the onset of the pandemic. This study determined the extent and intensity of the shipping footprint in Australia in relation to the placement of marine parks and known movement patterns of marine giants, and assessed whether exposure to shipping impacts has increased or decreased since the start of the COVID pandemic.

Materials and methods

The aim was to create an areal map of exposure to shipping footprint to quantify its distribution and determine how often marine areas and marine giants were exposed to shipping.

Data sourcing

Automatic Identification System (AIS) shipping data from each month of 2018, 2019, 2020, 2021 were obtained from the Australian Maritime Safety Authority (AMSA) website. As of 2004, all ships weighing more than 300 gross tonnes travelling in international waters are always obliged to have an AIS activated and transmitting. Cargo vessels greater than 500 gross tonnes and any commercial passenger ships inside national waters are also obliged to have operational AIS. In this context, our analyses exclude all smaller vessels that are not required to carry these systems.

AIS data were in the form of individual point records, with vessel identification, size, bearing, speed, and time of logged data at ~1-h intervals (this varied between 60 and 90 min). Higher-resolution data with higher-frequency recording were available, but only for select locations. These 60-min recording intervals were too coarse to capture and classify vessel activity, and, although other commercial providers do have higher-frequency data available, these are typically costly to obtain, whereas data from AMSA are freely available. In addition, with ships generally moving at ~10 knots (~18.5 km h^{-1}) (Maloni *et al.* 2013), the 60–90-min record intervals meant that a ship travelling at that rate would not be recorded repeatedly within 10 nautical miles (~18.5 km), which is relevant for the methodology explained below. Individual shapefiles for each month were collated into a single year. Ships with a movement speed of less than 1 knot (~1.9 km h^{-1}), suggesting they were at anchor, were removed because they were unlikely to be producing significant amounts of noise or be a strike risk to megafauna. This resulted in data sets with over 6.2 million data points in 2018, 6.6 million in 2019, 6.6 million in 2020, and 7.3 million in 2021. These data were within the Australian Search and Rescue zone and ranged through the Australian Exclusive Economic Zone (EEZ) 200 nautical miles (~370.4 km) offshore, including north to Papua New Guinea and Indonesia, west to the middle of the Indian Ocean, south to Antarctica, and east past Lord Howe Island and nearly to New Zealand (Fig. 1). Isolated MPAs that were outside the extent of the available AIS data (e.g. Norfolk Marine Park in the east) were excluded.

In QGIS (ver. 3.10.0, see https://github.com/qgis/QGIS), the resulting shapefile was analysed using kernel-density distribution with a radius of 10 km. This distance is analogous with a conservative range estimate from which shipping can affect marine mammals (Pine *et al.* 2018; Putland *et al.* 2018; Pirotta *et al.* 2019). Here by 'affect' we imply any change from natural conditions that can range from short-term behavioural reactions to direct ship strikes, and we expect a relationship between increasing exposure to shipping and more deleterious impacts. The kernel density of ships within the 10-km radius, over the year of collated data, thus represents a measure of rate of exposure to shipping within a 10-km radius, also the distance from which marine megafauna such as baleen whales may be affected by shipping noise. This is a presumed generalised distance from which whales may be affected, and is highly dependent on the species, sound source levels, and environment.

The resulting kernel-density distribution was classified into the following three different overlapping levels of exposure to shipping: low exposure (more than one vessel per year), moderate exposure (more than 90 vessels per year or one vessel every 4 days on average), and high exposure (more than 365 per year or more than one vessel per day on average). These disturbance levels were chosen to correspond with short-term stress events that would induce minor behavioural responses or a low likelihood of ship strike. Moderate rates of exposure were likely to induce seasonal higher stress levels or avoidance behaviours. High exposure to shipping levels would be analogous to conditions in high-throughput shipping lanes, with a higher probability of ship strike and consistent changes to stress hormones, communication abilities, and induce permanent range shifts (Rolland *et al.* 2012; Gomez *et al.* 2016). We assumed that

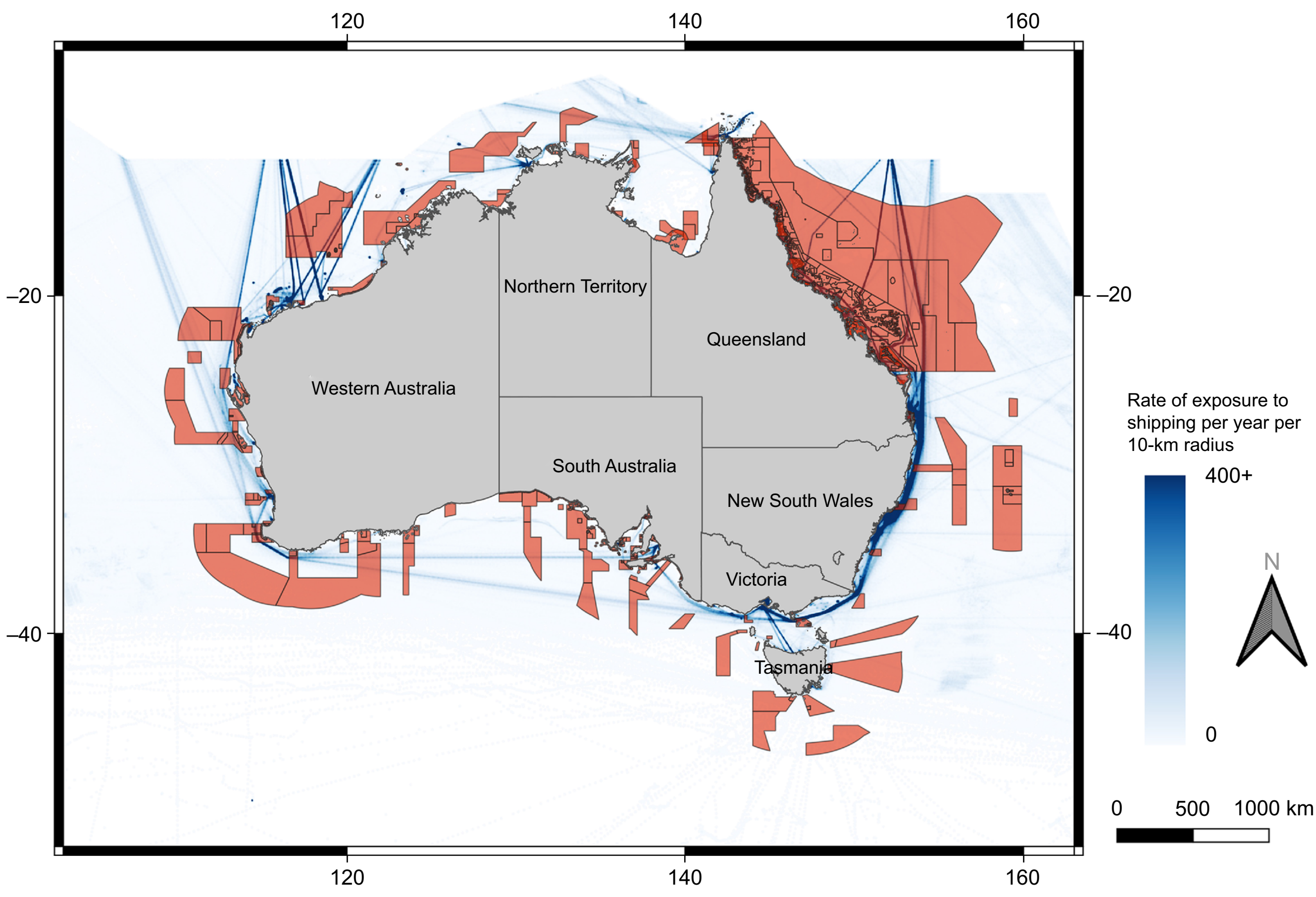

Fig. 1. Heatmap of Australia with exposure to shipping (vessels per year per 10 km) likely to have impacts on wildlife. Created using all available AIS data for ships moving faster then 1 knot (~1.9 km h^{-1}) in 2018. Marine parks of Australia are overlayed in red.

the vessel records were evenly distributed throughout the year, although in some cases daily vessel numbers may be higher during busy shipping periods.

Marine megafauna tag data

We targeted species affected by shipping, either directly through vessel strikes or indirectly through anthropogenic noise, and species listed as threatened locally or internationally, including whale sharks (*R. typus*), pygmy blue whales (*B. m. brevicauda*), and humpback whales (*M. novaeangliae*). Although satellite-tagging of these animals is becoming more common, the number tagged relative to total populations is relatively small (Sequeira *et al.* 2019). Determining ecosystem-scale patterns from satellite-tag programs with small sample sizes can be difficult, especially because marine systems are affected by inter-annual changes, which may mean that the results from a single year are not representative of other years (Yurkowski *et al.* 2016). To overcome these limitations, data from multiple species over multiple years were amalgamated in a fashion similar to that of Queiroz *et al.* (2019), who grouped AIS data from over 6 years and satellite-tag data from over 12 years to determine spatial use patterns.

Satellite tags commonly used for tracking marine giants only ping location data when they are directly exposed to the air, and thus when the animal carrying it is at the surface. Movement between detected satellite pings can be inferred; however, there is a large degree of uncertainty on the individual's location between these points, especially for animals such as whale sharks that do not need to surface periodically to breathe. The frequency of detections is highly variable ranging from multiple times within a day to weekly. As a result, only direct tag-detection points were included for this component and possible trajectories between detection points were not considered. This approach is thus conservatively estimating the impacts of exposure to shipping on marine megafauna, because the true area covered by the tagged animals is larger than that indicated by satellite data.

Satellite tracking data were collected by the Argos CLS satellite network from 52 whale sharks (*R. typus*) tagged at Ningaloo Reef and Shark Bay from 2010 to 2018. All deployed tags were satellite-linked SPOT5 tags (Wildlife Computers Inc., Redmond, WA, USA), except for one tag deployed at Ningaloo Reef in 2010, which was a SPLASH tag (Wildlife Computers Inc.). These two tags deployed in 2010 were made positively buoyant and tethered to the flank of each whale shark by using a wire connected to a dart inserted subcutaneously in the flank of the shark, below the dorsal fin. All other tags were attached to a negatively buoyant clamp designed to be mounted on the dorsal fin of the shark and then to detach within ~6–12 months after deployment (for more information, see Norman *et al.* 2016; Reynolds *et al.* 2017). The detections of the tagged sharks were mapped in ZoaTrack (www.zoatrack.org; Dwyer *et al.* 2015). Detections that occurred on land or those that were too distant from earlier or later, more accurate, detections to be biologically possible were excluded from further analyses.

Pygmy blue whale (*B. m. brevicauda*, $n = 18$) and humpback whale (*M. novaeangliae*, $n = 137$) tag data were collected from 2003 to 2016 (Smith *et al.* 2012; Double *et al.* 2014; Weinstein *et al.* 2017). Satellite-tag models were Spot 5 produced by Wildlife Computers Inc. and were attached by an Air Rocket Transmitter System (Double *et al.* 2014). For consistency with whale shark data, only direct detections were used in subsequent analyses. ARGOS tag data were cleaned for unreliable readings by using the Argosfilter package (ver. 0.62, C. Freitas, see https://cran.r-project.org/package=argosfilter) in R (ver. 3.4.4, R Foundation for Statistical Computing, Vienna, Austria, see https://www.R-project.org/).

Ethics statement

Whale shark tagging was undertaken according to Murdoch University Animal Ethics (permit numbers W2058/07, W2402/11 and R2926/17) and The University of Queensland Animal Ethics (permit number SBS/085/18/WA/INTERNATIONAL). Permission to tag whale sharks at Ningaloo Reef and Shark Bay, WA, was granted by the Western Australian Department of Environment and Conservation (permit number SF007471/007949/008572/009184/009897) and The Western Australian Department of Parks and Wildlife (permit numbers SF010414/010781, 08-000533-2, 01-000193-1 and 08-002082-2).

Humpback and pygmy blue whale tagging was undertaken in strict accordance with the approvals and conditions set by the Antarctic Animal Ethics Committee of the Australian Antarctic Division for this project, Australian Antarctic Science project 2941. Additionally, this study was also conducted in strict accordance with the approvals and conditions set by the Western Australian Department of Environment and Conservation Animal Ethics Committee for this project, 30/2008. Fieldwork was undertaken in Commonwealth Waters with the permission of the Australian Government under EPBC permits 2007-006 and 2007-007 and in Western Australia state waters under DBCA Permit SF010439 and SF009946.

Data analyses

These data were overlapped with a shapefile layer containing all Australian state and national marine parks of any kind, irrespective of protection level. In total, there were 3340 identified marine parks or marine zones, and occasionally these overlapped each other. The relative and absolute area of marine parks exposed to shipping was determined by overlaying the shipping exposure (high, moderate, low) layers on the marine park layer. Using the clip tool in QGIS, the surface area of marine parks exposed to each level of shipping impact was extracted in square kilometres. Marine giant track data were overlayed onto the three exposure layers created previously and the 'count in polygon' tool was used to determine the number of points that occurred within the ship exposure-level polygon. To assess whether there were differences year-on-year, and during the COVID-19 pandemic in particular, subsequent rasters of shipping footprint (expressed in number of exposures to ships per year) for the years 2019, 2020 and 2021 were subtracted from the 2018 raster by using the 'raster calculation' tool. To determine whether exposure had changed in marine parks across these years, the mean difference within each park was calculated with the raster zonal statistics tool. Results were then log-transformed and annual within-park means compared year-on-year by using a linear model in R (ver. 4.2.1, R Foundation for Statistical Computing).

Results

Calculated level of exposure to shipping ranged from 0 to >2000 vessels per year per 10-km radius. Assuming that all vessels are associated with similar levels of risk to wildlife, irrespective of ship size or speed, high levels of exposure to shipping (>365 vessels per year per 10-km radius) were more common along the coastline than in open ocean. Most of the east, and parts of the western coast of Australia, had high rates of exposure to shipping. In comparison, the Great Australian Bight and the Northern Territory had relatively lower exposure to shipping (Fig. 1). The mean proportion of marine park area that was exposed to shipping impacts was 86 ± 32% s.d. (at least one ship per year), and 15.7% of total marine park area was affected by moderate levels of exposure to shipping (more than 90 ships per year) with less than 5% of total marine park area being affected by high levels of exposure to shipping (>365 ships per year; Table 1).

Of the 34 831 ARGOS detections of tagged blue whales and humpback whales that remained after data cleaning, 12 872 or 36.7% of all detections were within areas that were affected by at least moderate exposure to shipping (Fig. 2, 3). Whale detections on the eastern coast of Australia were close to shore in areas with very high levels (>400 vessels per year) of exposure to shipping. This pattern was not evident on the western coast of Australia where shipping was lower. In marine parks, blue whales and humpback whales were exposed to moderate levels of shipping across 2922 or 8.3% of all blue whale and humpback whale detections. Of the 4559 surface detections of 52 tagged whale sharks, 1818 detections or 39% of all detections were within areas that were affected by at least moderate exposure to shipping (Fig. 3). In marine parks, whale sharks were moderately exposed to shipping across 968 or 21.2% of all whale shark detections. Thus, although whale shark, pygmy blue whale and humpback whale detections occurred at similar proportions within areas with moderate exposure to shipping, whale sharks were nearly three times more likely to be detected in areas with moderate exposure to shipping within marine parks.

Exposure to shipping per year per 10-km radius was consistent between 2019 and 2018 (1.41 ± 31.71, mean difference ± s.d.) but differed in 2020 and 2021. In 2020 and 2021, there was a progressive increase in shipping exposure in the north-west of Australia except for one shipping route that appeared to be less used (mean difference ± s.d.: 1.56 ± 44.52 in 2020, 4.28 ± 64.95 in 2021). Exposure to shipping also increased north of the Northern Territory, likely being a result of fishing activities. Exposure to shipping increased on the eastern coast of Australia, with seas just north of Sydney and Brisbane seeing increases greater than 400 vessels per year but decreasing slightly close to the coastline (Fig. 4). Within marine parks, exposure to shipping significantly changed year-on-year relative to 2018, with a mean relative difference of 2.79 ± 0.04 in 2019 (d.f. = 2_{6593}, $t = 65.19$, $P < 0.001$), which decreased significantly in 2020 (2.52 ± 0.07, d.f. = 2_{6593}, $t = -4.12$, $P < 0.001$) and increased significantly in 2021 (3.35 ± 0.06, d.f. = 2_{6593}, $t = 9.43$, $P < 0.001$).

Discussion

Marine parks around Australia are almost all exposed to shipping to some degree, often at high rate of exposure over much of their area. Marine giants within Australian waters may spend a significant proportion of their time with moderate–high levels of exposure to shipping, and the increased risks of ship strike and pollution this entails. The proportion of marine megafauna exposed to shipping appeared to be lower within marine parks; however, shipping exposure increased rather than decreased within marine parks despite the pandemic. Because shipping levels around Australia are lower than in other oceans such as the

Table 1. Summary of results for threshold layers of exposure to shipping calculated from AIS shipping data and corresponding overlaps with Australian marine parks.

Exposure rate to shipping vessels	Percentage total marine park area covered (%)	Percentage individual marine park area covered (mean ± s.d., %)	Proportion of marine parks with 100% exposure (of 3340; %)
>1 per year (low)	79.9	86 ± 33	77.3
>90 per year (moderate)	15.7	60 ± 46	49.9
>365 per year (high)	4.7	26 ± 41	18.7

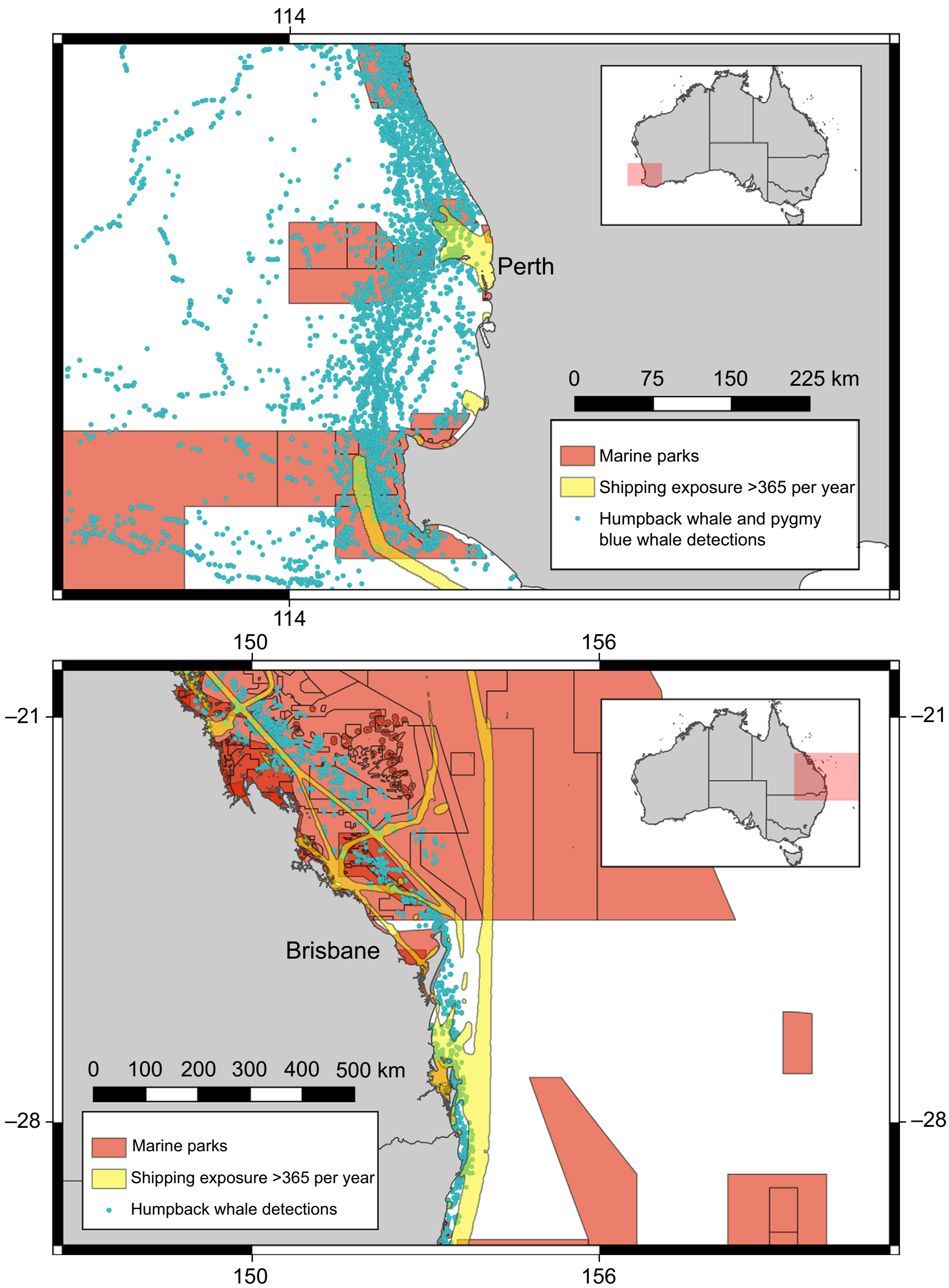

Fig. 2. Areas exposed to >365 vessels per year per 10-km radius (yellow) in relation to baleen whale movements. Waters surrounding Perth in south-western Australia (top). Southern Great Barrier Reef Marine Park (bottom). Satellite-tag record locations of pygmy blue and humpback whales (n = 155) indicated with teal points. State and national marine parks of Australia are overlayed in red.

Atlantic or the South China Sea (Corbett *et al.* 2007), and the analyses presented here exclude both smaller vessels without AIS and other ship-borne pollution that can have a greater reach (seismic testing, acoustic positioning), it is likely that the shipping footprint identified here is an underestimate of the total risk that occurs on a global scale and at higher exposure levels elsewhere. In this context, this study suggests that exposure to shipping should be considered during marine park zoning and for the conservation and management of marine giants on a scale similar to that of climate change and other global anthropogenic threats to natural environments.

Iconic and large marine parks such as the Great Barrier Reef and Ningaloo are likely to be exposed to shipping

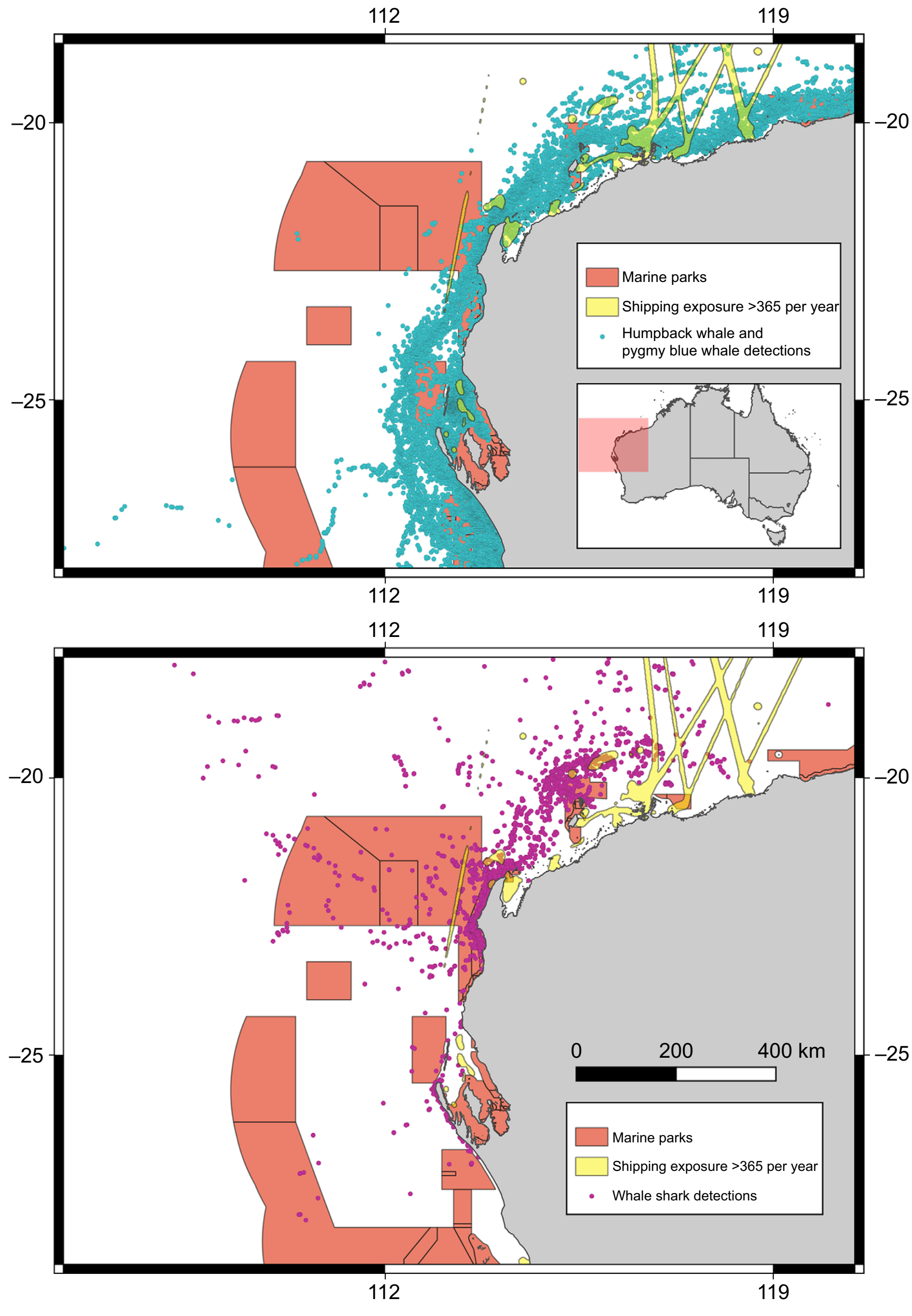

Fig. 3. Areas exposed to >365 vessels per year per 10-km radius (yellow) in relation to baleen whale movements (teal) and whale shark movements (purple), with a focus on the Ningaloo Marine Park (in the centre) and other marine parks (red zones). Satellite-tag record locations of whale sharks ($n = 52$ sharks, purple, bottom) and pygmy blue and humpback whales ($n = 155$, teal circles, top). State and national marine parks of Australia are overlayed in red.

across large proportions of their area, despite shipping having been previously identified as a threat to these environments (Grech *et al.* 2013). Often, the primary impact of shipping is considered to be port construction and management, and, to some degree, vessel strikes and pollution (Negri and Marshall 2009; Brodie and Waterhouse 2012). Our study did not assess the effects of the level of protection offered by these marine parks (e.g. limited fishing *v.* exclusion zones); however, the areal extent of these subclassifications is often smaller than the exposure footprint of vessels that may transit adjacent to these zones (10 km in this study, with whales elsewhere showing avoidance behaviours

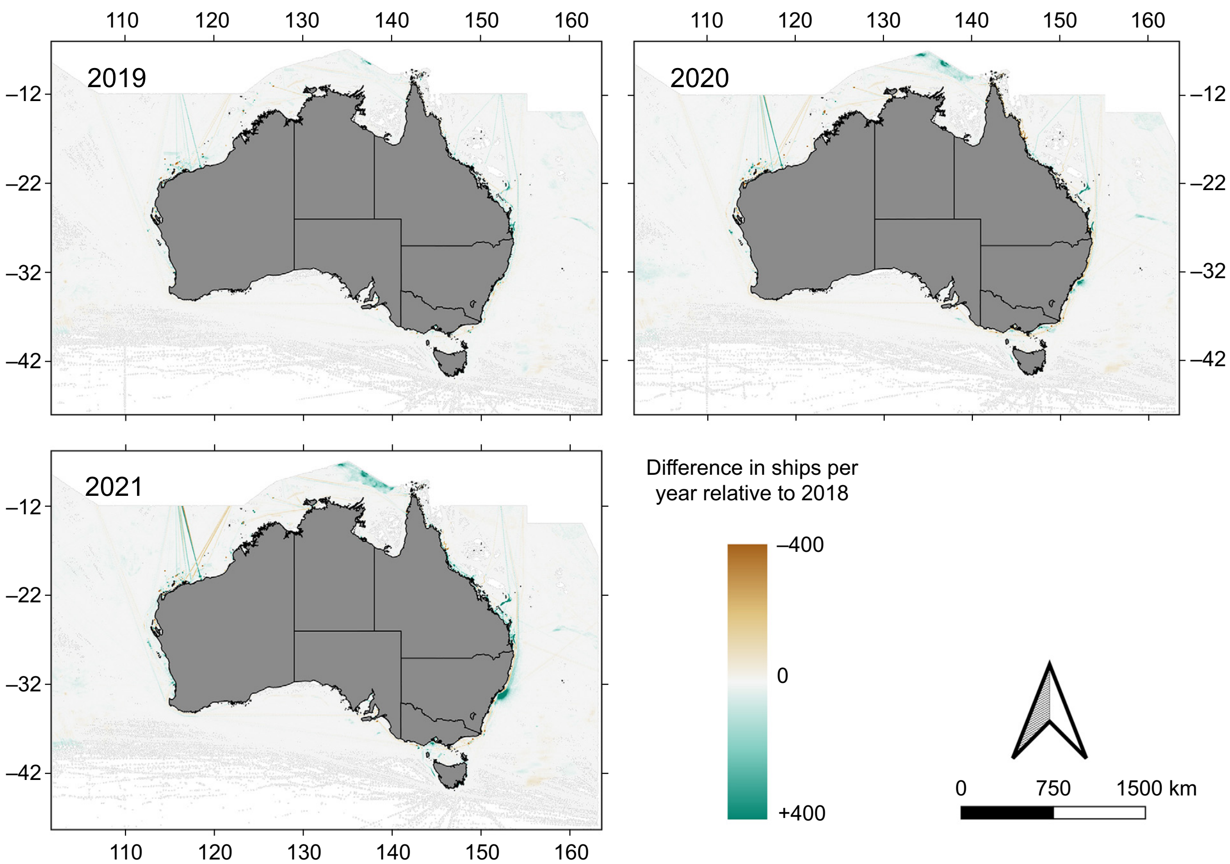

Fig. 4. Difference in ship exposure per year per 10-km radius in 2019, 2020 and 2021, relative to 2018.

>30 km from ships; Martin *et al.* 2022), with few of these zones (preservation zones in the Great Barrier Reef) explicitly excluding vessel transit. Marine park zoning should, therefore, explicitly include restrictions on large-vessel operation in addition to current restrictions, or design buffer zones to limit the effects of shipping on otherwise highly protected zones of interest. Our results suggest that existing commercial shipping routes should be considered during marine park zoning or that shipping should be managed in a way that reduces its marine footprint.

Exposure rates of marine megafauna to shipping appeared to be reduced within marine parks relative to outside, sometimes by as much as 75%, but marine parks did not eliminate exposure to shipping for marine megafauna. Different species of marine megafauna were not evenly protected by marine parks, with whale sharks found more frequently within moderate exposure areas in marine parks. This occurrence of whale sharks in areas with high shipping densities was also observed in Womersley *et al.* (2022), and could be due to a lack of avoidance mechanisms of these sharks, as in some whales (McKenna *et al.* 2015), although avoidance behaviours have been observed in odontocetes (Martin *et al.* 2022). Although being strongly dependent on vessel design, reduced shipping speeds have been successful at lowering vessel strikes of marine giants (van der Hoop *et al.* 2015), and would presumably lower shipping noise (McKenna *et al.* 2013) but would also increase temporal exposure. Alternatively, vessels could be encouraged to travel over a reduced number of corridors, or narrower corridors, to lower the areal extent of the shipping footprint. The International Maritime Organisation already has emissions targets that should result in reductions in speed or numbers of ships (Leaper 2019), and should combine these objectives with reductions of the footprint of fisheries where possible.

The movement patterns of marine giants suggest that they are exposed to shipping at over a third of the locations where tags recorded positions. Whale sharks (*R. typus*) and baleen whales (*M. novaeangliae*, *B. m. brevicauda*) were exposed at comparable proportions (37 and 39% respectively). Although it is not possible with this data set to determine whether these animals show any avoidance to high exposure to ships or shipping noise because satellite detections were assessed across 1 year, *M. novaeangliae* on the eastern coast travel closely along the shoreline where shipping-exposure rates are high. On the western coast of Australia, which had relatively lower exposure to shipping, this pattern was not evident. These data suggest that humpback whales continue to migrate along the western and eastern coast of Australia despite the shipping exposure but may be squeezed closer to shore as a response to high shipping exposure on the eastern coast. This may be a result of biological urges to travel north and reproduce, or whales may have acclimatised to a variety of noises in a modified acoustic environment (Smith *et al.* 2012; Pirotta *et al.* 2016). The recovery of the western and eastern coast humpback whales also means that more whales are likely to be exposed to shipping, because both populations continue to grow annually (Bejder *et al.* 2016). The effects of shipping and anthropogenic noise on elasmobranchs are less well known; however, shipping and its various direct and indirect impacts are considered one of the major threats to whale sharks globally (Speed *et al.* 2008). Whereas many of the effects of shipping on marine megafauna are unknown, the results from this study have highlighted that they are likely to be wide-ranging and could produce ecosystem-scale effects, especially because teleosts, other marine mammals and elasmobranchs are likely to be affected across a similar areal extent. Future studies should examine tracking data from a broad range of species and examine seasonal patterns of movement and their relation to shipping exposure.

The possible dampening effects of the COVID pandemic on shipping exposure appear to have been small, with exposure levels in marine parks only decreasing in Australia in 2020, before increasing beyond pre-pandemic levels in 2021. This aligns somewhat with global patterns in shipping, which initially declined as a result of the pandemic, but had mostly recovered by the end of 2020 (Notteboom *et al.* 2021). The distribution of exposure to shipping also changed in that time, with some shipping lanes being apparently abandoned entirely (e.g. in Western Australia), whereas some areas saw large increases in exposure (e.g. north of Sydney). This shows that, although shipping exposure globally may be increasing slowly, the way it is distributed is able to change rapidly, meaning ecosystems that have historically had low exposure rates may rapidly be exposed to high shipping traffic. These rapid increases in shipping exposure may limit the ability of marine communities to adapt to increased threats, and shipping exposure within Australian waters is increasing. Marine parks should be designated as spaces where protections are in place against rapid increases in shipping exposure to protect marine communities from these rapid increases in threatening processes.

The approach used in this study assumed that the area of effect of a ship was equivalent for all ships, and considered only ships with AIS data. It also assumed the impact footprint area of 10-km radius was appropriate; however, for some species of whales and sharks this footprint may be larger or smaller depending on their sensitivity to noise and other impacts of shipping (Pirotta *et al.* 2019) and the mean effect size across taxa is difficult to estimate. Vessel speed also affects risk of collision and noise levels (Pine *et al.* 2018; Putland *et al.* 2018) and was excluded from analyses here; however, future studies should incorporate those effects to assess areas of greater vulnerability. Ignoring ships without AIS is likely to result in large underestimates of shipping footprints in areas near the coasts where smaller vessels without AIS are more likely to operate, although identifying these vessels is becoming feasible in combination with satellite technology (Park *et al.* 2020). Although the areal extent of shipping impacts from smaller vessels is likely to be reduced relative to that from large cargo vessels (≥500 gross tonnes), smaller vessels are known to affect animal behaviour (McCormick *et al.* 2018; Fakan and McCormick 2019), can also cause lethal boat strikes (Schoeman *et al.* 2020; Fuentes *et al.* 2021), and small vessels often occur in higher densities more frequently. Because animal data were surface detections, they are also more representative of areas where collisions would be likely to occur, even if movements extend beyond this range. Thus, even though there are uncertainties around the shipping-exposure footprint identified here, in many environments the levels identified here are conservative.

The results from our study have highlighted the probable scale of exposure to shipping and underlined that it should be considered during marine park planning and for marine megafauna conservation and management. Although the direct impact of shipping is considered to be lower than other current marine conservation issues (Boonstra *et al.* 2015), this may be a result of prioritisation of research on other stressors and therefore a greater uncertainty on impacts. Shipping at current levels has been occurring on a global scale for decades, and, although its impacts on large marine megafauna such as those examined here have received attention, shipping effects on other marine fauna such as teleosts and invertebrates are poorly understood. Shipping impacts are also diverse and difficult to additively assess, including direct (ship strike) and chemical or acoustic pollution. Over time, exposure to shipping may have deleterious effects on coastal ecosystems, increasing the susceptibility of these ecosystems to other environmental stressors. New approaches such as the one presented here highlight the extent and distribution of shipping impacts on regional and, by extrapolation, on global scales, and will hopefully draw more attention to this global environmental issue.

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