Impacts of recreational motorboats on fishes: A review

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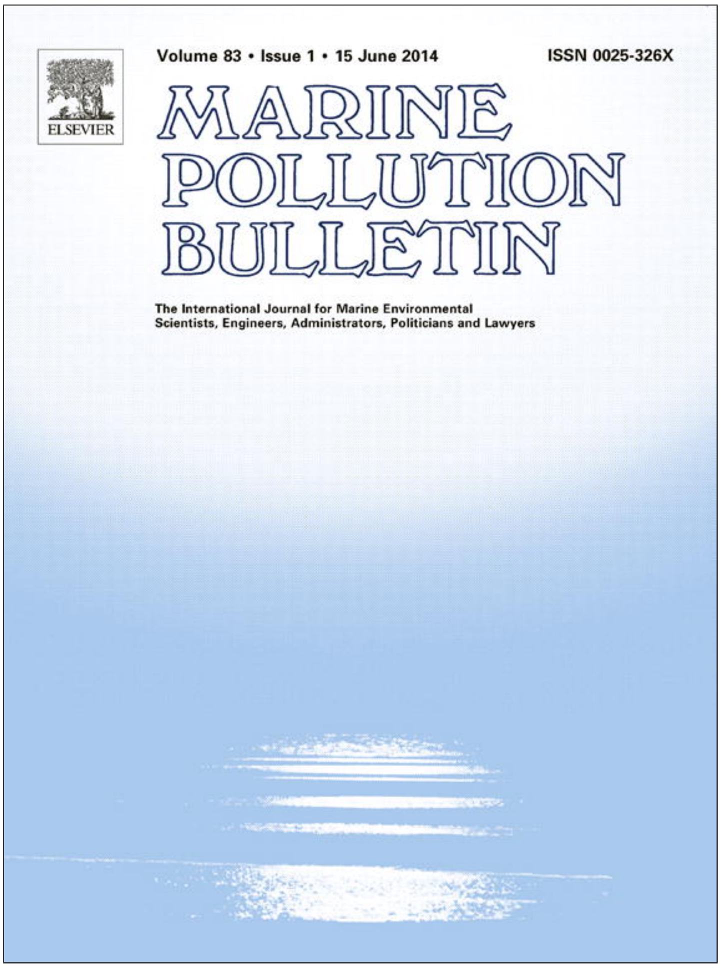
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Review

Impacts of recreational motorboats on fishes: A review

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A B S T R A C T

A considerable amount of research has been conducted on the impacts of recreational boating activities on fishes but little or no synthesis of the information has been undertaken. This review shows that motor boats impact on the biology and ecology of fishes but the effects vary according to the species and even particular size classes. Direct hits on fishes by propellers are an obvious impact but this aspect has been poorly documented. Alterations in the wave climate and water turbidity may also influence fishes and their habitats, especially submerged and emergent plant beds. Sound generated by boat motors can also influence the communication and behaviour of certain species. Pollution arising from fuel spillages, exhaust emissions and antifouling paints all have detrimental effects on fishes. Finally, the use of recreational boats as vectors of aquatic invasive organisms is very real and has created major problems to the ecology of aquatic systems.

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1. Introduction

There has been considerable conjecture among scientists, environmental managers and members of the public regarding the possible influence of recreational boat traffic on fishes and their aquatic environment (Lloret et al., 2008). Since much of this speculation is not based on facts or direct research evidence, the need to collate available information is both overdue and, in some instances, urgent. This review is also timely because coastal regions are experiencing unprecedented human population growth, with densities within 100 km from the ocean now three times greater than the global average (Small and Nicholls, 2003). In addition, there has been a significant rise in coastal recreation and tourism (Davenport and Davenport, 2006), including boating, with recreational motor boats accounting for a large percentage of boating traffic (Sidman and Fik, 2005; Lloret et al., 2008; Gray et al., 2011; Balaguer et al., 2011). For example, in Sydney Harbour, it was found recreational boats accounted for 70% of all boating activity (Widmer and Underwood, 2004). In the USA alone there

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are more than 12 million registered powerboats (NMMA, 2004), with a further 2.5 million in Canada (NMMA Canada, 2006). In freshwater ecosystems, recreational activities such as waterskiing are also increasing in popularity, with demonstrated environmental impacts (Mosisch and Arthington, 1998). In the opinion of these authors it is obvious, based on their review, that the biological impacts of power boating and water skiing on inland waters have been underestimated and that there is a need for more focused research within this field.

Some evidence is available (e.g. Sarà et al., 2007; Zamani-Ahmadmahmoodi et al., 2013) which suggests that motorboat traffic and the associated disturbance and pollution caused by such activities are having a negative impact on fishes associated with a range of aquatic systems, including freshwater, estuarine and marine waters. Unfortunately, relatively little work appears to have been conducted within this research field when compared to the speculation about the potential harm that such activities may bring to fish and the associated water bodies. For example, it has been proposed that motor boating may have been an important factor in the decline of fishes in the Danube River (Kiwek, 1995) but little scientific information is available to link cause and effect. Similarly, recreational fishers in Barnegat Bay (New Jersey) were of the opinion that personal watercraft (jet skis) were a severe environmental problem that affected fish within this system but could not offer any research evidence that backed up this perception (Burger et al., 1999).

We have chosen to focus our review on motor boat effects on fish, excluding other aquatic animals such as marine mammals and reptiles. The effect of boat noise on marine mammals has received significant previous research attention (e.g. Jensen et al., 2009; Buckstaff, 2004) and will not be covered in this paper. Similarly, we have chosen to limit this review to the effects of all sizes and types of recreational motor boats on fishes and will not assess the potential impacts of larger vessels such as cruise liners, commercial shipping and dredgers. Boating activities have both direct and indirect effects on fishes (Fig. 1). What we have attempted here is to cover the more immediate and direct effects in the first three subheadings after the Introduction and then move on to the more indirect effects in the final five subheadings. It should be noted, however, that both direct and indirect effects are present in all sections of the review and across all the time scalers (Fig. 1).

2. Motorboat traffic and direct hits

Despite this being the most obvious direct impact on fishes, very little work has been done at quantifying fish strikes at different boat speeds and by boats of different sizes. Direct strikes by motorboat propellors on fish have been noted in certain scientific studies (Balazik et al., 2012) and may occasionally reach high levels (Killgore et al., 2011). In addition, some fish species are so disturbed by motorboat traffic that they begin jumping in the same manner as if escaping from an underwater predator. The flathead mullet *Mugil cephalus* is well known to undertake such behaviour (Hoese, 1985) and numerous specimens have been recorded jumping into moving boats in estuaries, especially at night (personal observation).

There is also little information on the direct impact of rapidly rotating boat propellers on delicate fish larvae, especially at night when the ichthyoplankton is concentrated in surface waters. Are these fish too small to be impacted by the propeller blades, or do the water vortices created by the rapid rotation of propellers cause instantaneous mortalities amongst the larvae? A study by Jude et al. (1998) noted that the early life stages of at least three species of fishes in the Great Lakes of North America may be affected by large boats which cause resuspension of sediments, dislodge eggs and can lead to the premature emergence of larvae.

3. Motorboat traffic and fish behaviour

Some fish species do not appear to respond behaviourally to the presence of powered outboard engines, e.g. lake trout *Salvelinus namaycush* in a small Canadian lake did not respond boat traffic, even during detailed manual tracking of individual fish

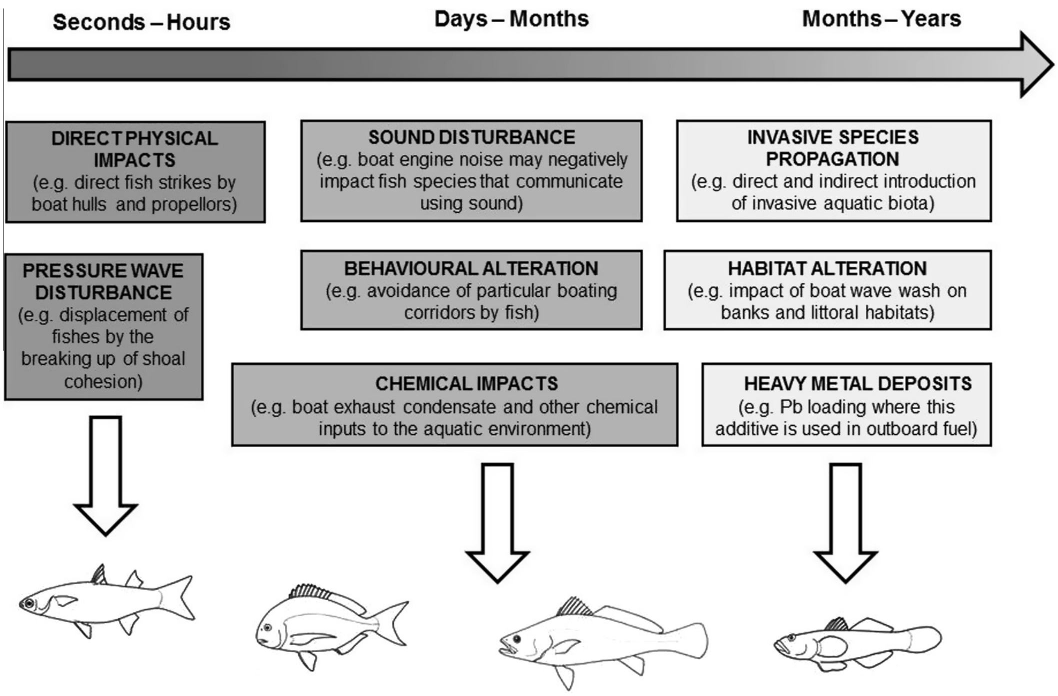

Fig. 1. Likely influences and impacts of power boating activities on fishes and their habitats and the likely time frame over which the impacts may act (for details and references see text).

Table 1
Summary of the major findings relating to recreational motorboat activities and fishes.

Key references	Major findings
Motorboat traffic & direct hits	
Balazik et al. (2012) Killgore et al. (2011)	Evidence of direct hits by boats. Very few studies have quantified fish strikes by boats at various speeds or the fish sizes that are affected. This is an area needing further research
Motorboat traffic & fish behaviour	
Becker et al. (2013a,b) Erbe (2013) Holles et al. (2013) Sebastianutto et al. (2011) Picciulin et al. (2010) Slabbekoorn et al. (2010) Popper and Hastings (2009) Codarin et al. (2009) Vasconcelos et al. (2007) Smith et al. (2004)	The effect of motorboat traffic on the behaviour of fish is probably the most studied aspect of boat impacts on fish. Noise emitted from engines may increase stress levels in fishes, and underwater noise has also been linked to disruption in the reproductive behaviour of certain fishes. Noise has been found to influence all fish life history stages, including the larvae. Most studies have been conducted in laboratories but recent examples from field based studies have provided real data for the testing of hypotheses. Further research is required on fish size related responses to boat movements, as well as which species are most negatively affected by boat traffic
Heavy metals	
Superville et al. (2014) Eisler (2010) Leon and Warnken (2008) Shazili et al. (2006)	Sources of heavy metals in aquatic ecosystems arising from boats include antifouling paints and exhaust emissions, as well as the resuspension of contaminated sediments by boat propellor action and wave wakes. More research is needed to link levels of boating activity to Pb and other metal concentrations in the aquatic environment
Motorboat byproducts	
Mattos et al. (2010) Khan (2003) Kempinger et al. (1998) Tjarnlund et al. (1995, 1996) English et al. (1963)	Engine exhaust is the most prominent byproduct of motorboats. Diesel can influence gene expression in fish, while multiple studies have found that other petroleum based products can adversely affect the health of fish. Carbon monoxide poisoning has been linked to fish kills and this may be a particular threat in systems with high boat traffic and low flushing rates
Invasive species propagation	
Belz et al. (2012) Kerfoot et al. (2011) Hardiman and Burgin (2010) Boltovskoy et al. (2006)	Transport of invasive fish species overland from one water body to another is a major issue, with this often being done deliberately. However the inadvertent transport of fish diseases and parasites on/in boats and associated equipment is a topic which has not received research attention and is in need of urgent investigation
Boat Infrastructure	
Demers et al. (2013) Gladstone and Courtenay (2014) Becker et al. (2013a,b) Fowler and Booth (2013) Montefalcone et al. (2008) Clynick et al. (2007) Creed and Amado Filho (1999)	Infrastructure which facilitates boating activities such a piers, moorings, ramps and marinas can impact fish assemblages. Removal of natural habitat to construct infrastructure has the greatest impact, with fish and invertebrate assemblages on man-made structures rarely the same as those found in natural habitats. Research has also been conducted on the negative effects of mooring sites and anchoring chains on seagrass beds. The use of swing mooring has been shown to greatly reduce these impacts
Impacts on aquatic habitats	
Gabel et al. (2011) Bishop and Chapman (2004), Bishop (2007) and Bishop (2008) Bell et al. (2002)	Moving boats can impact aquatic habitats by increasing turbidity, eroding banks with wave wash, and scouring aquatic macrophyte habitats with boat propellors Invertebrates in seagrass exposed to boating activity have been found to have lower diversity than control sites, which can have important implications for fish productivity Wave wash from boats can be mediated by restricting the speed of boat traffic in sensitive areas

(Blanchfield et al., 2005). However, the passage of boats has been shown to break up fish schools and result in increased energy expenditure as they attempt to move away from the disturbance. Manipulative work examining nest guarding behaviour of longear sunfish found passing boats caused fish to leave their nests for longer periods than during control times (Mueller, 1980). A recent study on the effects of passing motorboats on the abundance of different sized fish within the main channel of a South African estuary revealed that the 100–300 mm and >500 size classes had no change in their abundance following the passage of boats (Becker et al., 2013a). However, the mid-sized fishes (301–500 mm) decreased in abundance, a displacement which was attributed to a number of factors, including noise, bubbles and the approaching boat itself. Other studies have indicated that, under high boat traffic conditions, long-term changes in fish abundance can occur (Gutreuter et al., 2006) and will result in altered trophic functioning within such a system.

Disturbance effects of motorboats on fishes can be linked to several factors (Fig. 1), including noise levels (Scholik and Yan, 2002; Codarin et al., 2009; Slabbekoorn et al., 2010). Noise is of particular concern as fish have sensitive auditory organs and anthropogenic noise has the potential to cause physiological and behavioural responses (Popper and Hastings 2009; Purser and Radford, 2011). *In situ* recording of powerboat noise spectra indicate that outboard sounds can be detected by species such as cyprinids at a distance of hundreds of meters (Amoser et al., 2004). Noise from boats may also increase fish stress levels (Graham and Cooke, 2008; Smith et al., 2004) or even have a direct impact on the breeding behaviour of certain fish species (Bruinjes and Radford, 2013).

In addition, boat noise has been shown to adversely affect the territorial behaviour of fish species such as *Gobius cruentatus*, which uses sound production as an effective tool for territorial defense (Sebastianutto et al., 2011) and nest caring by *Chromis chromis* (Picciulin et al., 2010). Indeed, both Codarin et al. (2009) and Vasconcelos et al. (2007) concluded that boat engine noise is most pronounced in the frequency range where communication by vocal fishes takes place and therefore has a potentially negative affect on these particular species.

Larval fish may also susceptible to boat noise. In choice chamber experiments, settlement stage *Apogon doryssa* were subjected to five noise types, with boat noise disrupting the ability of these larvae to actively settle on their native reef habitat (Holles et al., 2013). In contrast, Jung and Swearer (2011) used *in situ* manipulative experiments to show that boat noise neither repelled certain larval fish, nor inhibited their ability to navigate towards biological sounds.

To date there is very limited work conducted on the environmental effects of powered water craft (PWC), also known as 'jet skis', and this is certainly an area for future research. While the air noise pollution of PWC is often complained about by the public (Blomburg et al., 2003), noise emitted by these craft in the water may also be of concern (Erbe, 2013). While the noise from the rotating blades may be somewhat muffled by the plume of bubbles created, there are concerns that the bubbles themselves may create enough noise to have environmental impacts (Erbe, 2013). Also, pilots of PWC tend to change speed and direction far more frequently than those piloting typical recreational boats, thus giving rise to unpredictable changes in sound pitch and volume as well as craft direction. Additionally the hull of PWC tends to strike the water surface harder and with greater frequency than typical motorboats, all of which is likely to cause more confusion in nearby fish schools. Given that there is also evidence that bubbles emitted by motorboats may impact fish behaviour (Becker et al., 2013a,b), we recommend future research be directed at this topic.

Investigations on the possible impact of motor boat wakes on fishes have also been conducted. Boat wakes in the Rogue River, Oregon, did not appear to change the thermal properties of thermal refuges of juvenile Chinook salmon *Oncorhynchus tshawytscha* nor did these fish exhibit startle responses, provided the boat was more than 3 m from the individual salmon (Reid, 2007). Information on trolling activities from mobile marine boats (Akimichi, 1975) suggests that pelagic game fish capture normally occurs when the bait or lures are within the wake of a trolling boat.

4. Heavy metal levels and fishes

Trace metals are a natural component in the bodies of fishes and have been extensively studied by a number of authors (see Eisler, 2010 for a review). However, heavy metal inputs to aquatic environments became a major issue following the industrial revolution and in the modern era these pollutants can come from a variety of sources, including boats. In terms of recreational boating, in the recent past, the major sources of heavy metals were antifouling paints and boat exhaust emissions. In addition, resuspension of sediment bound metals by boat wake, and direct sediment disturbance by boat engine operations in shallow water, have accentuated the problem. Fortunately, considerable progress has been made in reducing toxic metals from paints and lead from petroleum products, thus reducing pollution from these sources.

Increasing lead (Pb) levels in lakes, rivers and estuaries are perhaps one of the most obvious potential indicators of environmental pollution by outboard motors, particularly where lead is used as an additive to the fuel (Rashed, 2001). Although it has been hypothesized that heavy boating traffic might contribute significantly to Pb concentrations in fishes and the aquatic environment, little published information is available on this topic (Zamani-Ahmadmahmoodi et al., 2013). Nevertheless, *Tilapia nilotica* from the High Dam Lake in Egypt were found to have elevated levels of cadmium and lead, with the latter pollutant attributed to gasoline leaks from fishery boats (Rashed, 2001).

Lead in the aquatic environment from exhaust waste is most likely to occur in a relatively insoluble form (Daines et al., 1970), with Pb accumulating in the sediments and being potentially assimilated by certain benthic biota. This view is supported by the work of Sin et al. (1991) who showed higher amounts of metals in invertebrate species occupying the Singapore River bed, while fish at higher trophic levels accumulated lower amounts of these metals. If biomagnification were to occur, then fish tissues would be expected to contain increasing concentrations of metals but this does not appear to be the case, e.g. heavy metal measurements (including lead) in fish from Malaysian waters indicated that no contamination had occurred (Shazili et al., 2006). Similarly, an early scientific paper on lead loading in selected fish species from several Nebraskan lakes, which received differential boating pressures, showed that Pb levels in the fish muscle ranged from less than 0.05 to 1.35 ppm and did not differ significantly between fish in the controlled and non-controlled boating lakes (Oates, 1976).

Boat traffic can also result in the resuspension of heavy metals from polluted sediments. For example, boat traffic in the Deûle River in northern France has been directly linked to the resuspension of sedimentary particles that significantly increased electrolabile Pb and Zn into the overlying water (Superville et al., 2014). Elevated Cu levels in Lake Texoma water were attributed to antifouling based paint used on boats and high Cu levels at specific locations in marinas around the lake appeared to be associated with recreational boat repair activities (An and Kampbell, 2003). Similarly, Leon and Warnken (2008) estimated Cu leaching from antifouling paints on recreational boats in Moreton Bay, Australia, reached 141 kg per annum at the 20 most popular anchor sites in this bay.

Leaching of tributyltin-containing antifouling paints used on boats into coastal waters is a major problem for certain invertebrate species (Bhosle et al., 2004) and the enzyme system activities of certain fish (Axiak et al., 2000). Molluscs appear to be most affected by tributyltin and its degradation products, with fish having low levels of contamination (Morcillo et al., 1997). However, because fish invertebrate prey is negatively affected by tributyltin, it is likely that fish stocks will also be impacted. Fortunately, legislation in many countries has seen these toxic paints being replaced by more environmentally acceptable alternatives (Rees et al., 2001; Birchenough et al., 2002; Dafforn et al., 2011).

5. Fish exposure to other motorboat products and byproducts

Motorboats are usually powered by either diesel or a petroleum and oil based mixture, both of which are sometimes accidentally spilt into waterways when filling up tanks or servicing engines close to the water body. In addition, both types of fuel emit exhaust fumes into or onto the water when the motorboat is underway which can affect fish eggs, larvae and juveniles, especially in surface waters

Diesel is an important fuel used by both small and large boats in coastal areas and has the potential to influence gene expression in fishes (Mattos et al., 2010). Similarly, petroleum contamination of the surrounding water by small boats was found to negatively influence the health of winter flounder (*Pleuronectes americanus*) in Placentia Bay, Newfoundland (Khan, 2003). Accidental spillage of motorboat fuel directly into aquatic ecosystems is a reality and will always remain a water pollution risk that can negatively influence periphytic algal biomass (Nayar et al., 2004) and other fish food sources.

Laboratory tests conducted in 1960 by English et al. (1963) showed that bluegill sunfish were killed when outboard fuel consumption reached 530 L per million litres of water. However, fish flesh could be tainted by outboard motor exhaust wastes at much lower levels. These tests showed that 90% of persons in a taste panel noted objectionable flavour in fish exposed to cumulative fuel consumption levels of 2.8 L per million litres of water. Follow-up field studies by Surber (1971) indicated that tainting of fish occurred at a daily fuel-use rate of approximately 0.2 L per million litres of water. Similar work by Lüdermann (1968) found that detrimental changes in the flavour of the flesh of freshwater fishes exposed to the exhaust emissions of outboard motors disappeared after a few days of the fish living in clean freshwater.

Studies by Tjarnlund et al. (1995, 1996) on the exposure of rainbow trout to *n*-hexane extract of outboard engine exhaust condensate indicated disruption of biological functions at different levels, e.g. cellular and subcellular processes (DNA-adduct levels and enzyme activity) and physiological functions (carbohydrate metabolism). These authors concluded that the biological effects of exhaust emissions represent a serious threat to the aquatic environment. Although the exhaust emissions from boats have been identified as a possible source of heavy metal pollution in the Singapore River, particularly of the sediments, contamination of fishes in this system was at a much lower level than invertebrate species occupying the river bed (Sin et al., 1991).

Carbon monoxide was attributed to fish kills near an outboard testing facility on the Fox River and the suggestion made that such events could to be exacerbated by low river flows and high temperatures (Kempinger et al., 1998). This indicates that there may be potential for carbon monoxide poisoning in areas with very high boat traffic and low flushing rates.

Disposal of sewage from recreational vessels can also cause nutrient pollution to waterways that may affect fish species. In Moreton Bay, Australia, it was estimated that over one year approximately 10,000 recreational craft over 6 m in length generated at least 1 tonne of N into the system via sewage waste at 20 overnight anchor sites (Leon and Warnken, 2008). The impact of this additional N input on estuarine food webs and fishes is unknown.

6. Motorboats and invasive species propagation

Recreational motorboats are a very effective transmission vehicle for aquatic invasive biota, mainly through the transport of alien fish species, plants and invertebrates (Hardiman and Burgin, 2010; Belz et al., 2012). This transport is usually facilitated by overland travel from one water body to another and may be either deliberate in the case of fishes, or inadvertent in the case of plants and invertebrates (Boltovskoy et al., 2006; Kerfoot et al., 2011). Deliberate translocation usually occurs by anglers wishing to stock adjacent water bodies with favoured sport fish (e.g. largemouth bass *Micropterus salmoides*) or the use of live fish bait captured from another system. Studies have shown that trailered recreational boats are an important vector in spreading aquatic invasive species and that most boaters have not yet adopted effective cleaning methods for their equipment (Rothlisberger et al., 2010). There is also a strong possibility that fish diseases and parasites are also translocated by boats from one water body to another and could have significant effects on fish populations, but this topic has been poorly studied.

The increased connectivity between waterways due to the construction of shipping canals and small boat access routes also increases the risk of invasive species introductions, including fishes. One of the management options to reduce this risk as far as fishes are concerned is the use of electrical, acoustic, chemical, visual and hydrological deterrence techniques to prevent fish invasions (Noatch and Suski, 2012). Such behavioural deterrent systems can then be evaluated mathematically to determine their effectiveness, e.g. laboratory experiments using a bubble curtain to inhibit *Cyprinus carpio* movement provided data for a model that assessed the effectiveness of this system as a barrier to common carp movements (Zielinski et al., 2014).

7. Spatial and volumetric implications

The severity of impacts of motor boats on fish will vary greatly depending on the dimensions and hydrodynamics of the location where boat traffic is occurring. For example, in small estuaries with low flushing rates and restricted dimensions, the impact of limited boat traffic is likely to be higher than along an open coast under the influence of much higher traffic volumes. In estuaries with restricted mouths, pollutants have a longer retention time (Abrahim et al., 2007), while the smaller dimensions of some systems would offer fish fewer options in terms of alternate habitats for refuge from boat noise or other disruptive effects (Becker et al., 2013a,b).

Along an open coast, the density of even much higher numbers of boats is likely to be low, and the options for fish to escape motor boat disturbance would be much higher. Pollutants and re-suspended nutrients also have far greater of volumes of water in which to disperse, and this would be further facilitated by coastal currents. Therefore a potential disparity in the impacts of motorboats exists in terms of the location of the fish and the size and nature of the surrounding ecosystem. It could be similarly argued that fish within narrow rivers might also be subjected to potentially higher disruptive effects when compared to large lagoons or lakes (Miranda, 2011).

8. Boat infrastructure requirements

With increasing numbers of recreational boats, comes a need for increasing infrastructure to facilitate motorboat activities. These facilities include boat ramps, jetties and piers, breakwaters, marinas and moorings. The construction of such infrastructure and resultant habitat loss can have a variety of impacts on fish. Permanent boat moorings have been demonstrated to have serious deleterious effects on seagrass meadows worldwide (Walker et al., 1989; Montefalcone et al., 2008). It is well established that seagrass beds support a significant number of fish species at various life history stages and the loss of seagrass beds can have serious implication for fish which rely on these habitats (Gillanders, 2006).

The impact of permanent motorboat moorings on seagrass may be greatly reduced by the use of 'screw' moorings which allow a boat to pivot around a vertical pole, rather than have an anchor chain drag through seagrass beds (Hastings et al., 1995; Demers et al., 2013). Significant damage may also be caused through short term anchoring (e.g. Francour et al., 1999; Lloret et al., 2008), not only within harbours and marinas but also in the more remote areas of aquatic ecosystems where boating activities occur (Creed and Amado Filho, 1999). On balance, the use of permanent mooring buoys is preferred to the use of anchor chains because the latter method tends to create more damage to benthic habitats. Legislation and management plans banning anchoring in areas consisting of fragile habitat may also be an option in some instances to reduce the impact of boats.

Piers and jetties often replace natural habitat which is destroyed during the construction phase and act as an artificial structure in a natural ecosystem (Clynick et al., 2007). While the destruction of habitat can clearly have negative impacts on fish assemblages, it is not always clear what ecological role the artificial structures perform (Clynick et al., 2007; Rilov and Benayahu, 2000). While piers and jetties do provide unique structural complexity, they normally differ from the natural habitat in their vertical nature, shading and homogenous surfaces (Able et al., 1998; Clynick et al., 2008). However, some breakwaters have been found to harbour similar or even a richer diversity of fish than nearby natural reef habitat, although the ichthyofaunal assemblages themselves may differ (Burt et al., 2009, 2013; Fowler and Booth, 2013). This indicates that although the replacement of natural reefs with structures, such as breakwaters, may not result in significant shifts in fish assemblages, their failure to mimic all aspects of local communities may lead to shifts in ecosystem functioning. Furthermore, the construction and altered hydrodynamics may influence nearby soft-sediment fish assemblages and vegetation cover (Pérez-Ruzafa et al., 2006).

Dredging for recreational boat harbours and approaches to boat mooring areas are also of concern (Gladstone and Courtenay, 2014). Such activities may impact directly on fishes through changes in water turbidity and dissolved oxygen, but may also affect the food resources (Jones, 1986) and vegetated habitats (Erftemeijer and Lewis, 2006) of fishes in a negative manner.

Lighting from infrastructure such as jetties and marinas may also influence fish assemblages around these types of artificial structure, potentially leading to changes in trophic interactions among fish (Becker et al., 2013b). The impact of boat lights on fishes, as well as their invertebrate prey, is less well studied and requires further investigation.

9. Boat impacts on aquatic habitats and other biota

Moving boats can have direct and indirect effects on fish assemblages through the impacts on the immediate environment occupied by the fish and their prey. Such impacts include the physical effects of boat wake, including temporary changes in water turbidity and the local wave climate, as well as direct scour and hydrodynamic changes in surface waters caused by spinning propellers. For example, studies by Nedohin and Elefsiniotis (1997) and Yousef et al. (1980) showed that motor boat activity creates enough disturbance of bottom sediments in shallow lakes to release stored phosporous into the overlying water. Such activities have the potential to accelerate the rate of eutrophication in such systems. However in deeper waters these impacts appear to be less of a threat (Yousef et al., 1980).

Boat wake can have a number of negative indirect effects on fish through the physical impact on aquatic systems. Wake wash and bed sediment resuspension often changes water turbidity (Garrad and Hey, 1987; Garel et al., 2008) and this can have direct and indirect effects on the biota (Bishop and Chapman, 2004; Bishop, 2007; Eriksson et al., 2004). The abundance and diversity of invertebrates (e.g. amphipoda and polychaeta) within seagrass beds exposed to boat wake was found to be significantly lower than at control sites within an estuary, with direct implications for fish production (Bishop, 2008).

Pulsed waves produced by boats differ from wind generated waves, with the latter being more continuous and generally of a smaller size. Experimental studies have shown that changing the wave frequency and size can alter predator–prey interaction among fish and invertebrates, with the potential to thereby influence and change fish assemblages (Gabel et al., 2011). Particularly in restricted inland and estuarine waters, wash from boats can lead to significant bank erosion (Liddle and Scorgie, 1980; Nanson et al., 1994; Murphy et al., 1995). This may uproot riparian vegetation which forms important habitat for many freshwater fish (e.g. Hawkins et al., 1983). In areas where riparian vegetation has already been removed, bank erosion from boat wake is likely to be more severe and will significantly contribute to turbidity levels. The impact of boat wake on bank erosion can, in many cases, be reduced by restricting the speeds of passing boats (Mosisch and Arthington, 1998).

Another boat impact on the environment is the creation of underwater 'paths' through aquatic plant beds by outboard engines, particularly when these craft attempt to plough through submerged beds in shallow water (Zieman, 1976). Indeed, propeller damage to seagrass beds can lead to habitat fragmentation (Bell et al., 2001) which has the potential to influence the associated biota, including fishes. Although Bell et al. (2002) could not detect any negative impacts of propellor scarring on seagrass (*Thallassia testudinum*) beds on fish and shrimp abundance in Charlotte Harbour and Tampa Bay, they did suggest that higher levels of scarring may lead to the degeneration of seagrass beds which would then affect the associated nekton.

10. Conclusions and future research needs

The available evidence suggests that motor boats do have an impact on many aspects of the biology and ecology of fishes (Fig. 1, Table 1). Multiple lines of evidence point to direct disturbance effects varying according to the species and sometimes only certain size classes of fishes seem to respond to outboard powered craft. Alterations in the local wave climate and water turbidity may also influence fishes and their habitats, especially submerged plant beds. The sound generated by boat motors can influence the communication and behaviour of certain species but others appear to be largely unaffected. Boat pollution arising from fuel (diesel or petrol) and antifouling paint used on boat hulls also has direct and indirect detrimental effects on fishes, although the elimination of Pb additives from petrol and the banning of TBT paints in many countries have reduced the negative impacts of these particular management problems. Finally, the use of boats as intentional and unintentional vectors of aquatic invasive organisms (including fishes) is very real and has created major problems to the natural ecology of freshwater systems in particular.

Future research needs to address the urgent requirements around the management of aquatic ecosystems in a sustainable and ecologically acceptable manner. In this regard, motorboat engines and their fuel are becoming ever more efficient and less of a pollution threat to fishes than was the case in the past. Similarly, the banning of toxic antifouling paints in many countries has gone a long way towards improving aquatic ecology where large numbers of recreational boats are present. No research appears to have been devoted to the effect of propeller swash and exhaust condensates on pelagic fish eggs and larvae, despite the fact that such impacts have the potential to be extensive and debilitating on future fish cohort strengths in aquatic systems. Perhaps the area most in need of research attention is the use of boats as vectors for the transport of alien aquatic organisms into new water bodies and the methods that need to be applied to reduce this risk. In some cases, alien invasions are devastating natural aquatic ecosystems and, if boats are instrumental in the spread of these species, then new legislation and management responses are required.

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