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REVIEW PAPER

The direct physical, chemical and biotic impacts on Australian coastal waters due to recreational boating

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Abstract In economically developed countries it is projected that by around 2015 over 50% of a person's lifetime will become available for leisure. Demand for leisure needs, already strong, will continue to increase. One segment of the market, outdoor nature-based recreation (including tourism), is growing strongly worldwide. A substantial proportion of these activities are water-based. The associated demand for recreational vessels has increased rapidly in recent years and is projected to continue to trend upwards. Australian trends mirror those internationally. Using Australia as a case study, we review the direct physical, chemical and biotic impacts associated with recreational boating in coastal water environments. Major physical impacts include disturbance due to movement of craft in shallow waters (e.g., turbulence) and the effects of anchoring/drag, noise/interference/collision that impacts on wildlife. The most critical chemical impacts result from pollution due to fuels and oils, defouling treatments (even those not legislated in-country), and human waste (e.g., sewage effluent). Important biotic impacts are the potential continued introduction and secondary spread of non-native species. We conclude that while greater research effort will provide more environmentally benign products, with the increasing popularity of recreation vessels, it will be beyond the resources of Australian governments to police legislation effectively. However, based on Australian's demonstrated engagement with government in terrestrial environmental management, with their deliberate engagement with the boating fraternity, the impacts of recreational boating would be lessened.

Keywords Sewage discharge · Anti-fouling treatments · Species translocation · Fuel and oil pollution · Animal kill · Boating impacts · Tourist impacts · Hydrocarbon pollution · Recreation impacts on wildlife · Community engagement

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Introduction

Throughout human history, continuous advances in technology and management have meant that automation and organised labour have produced and transported more goods and services, faster and further than in each preceding generation. This has resulted in greater individual leisure time in developed countries, from over 23% of an individual's lifetime in the 1770s, to approximately 41% in the 1990s. Based on current trends it is projected that by around 2015 technological and organisational advances, together with longer life expectancy, earlier retirement and fewer children will mean that over 50% of a person's lifetime will become available for leisure in economically developed countries (Molitor 2000). The associated continuing escalation of the leisure industry poses substantial risks for environmental impacts since leisure is already the single largest driver of anthropogenic global carbon dioxide emissions, measured by end user need (The Carbon Trust 2006).

One segment of the overall leisure market, outdoor nature-based recreation (including tourism), is growing strongly worldwide (Buckley 2003; Cole 1996; IUCN 1996; Pickering and Hill 2007). A substantial proportion of such activities are water-based. One such water-related recreational activity with substantial potential for ecological impact is boating. Recreational vessels range from personalised watercraft for one or two individuals, through sailing dinghies, speedboats, cabin cruisers and houseboats designed to carry four to five individuals on lakes, rivers or within coastal waters, through to larger powerboats and sailing yachts capable of international travel.

Despite growth in recreational boating (ABS 2009) and the economic importance of tourism (Buckley 2003), Australian research into recreational impacts on the ecology of aquatic ecosystems tends to lag behind that of other countries, especially the United States of America (US) and the United Kingdom (Hadwen et al. 2006; Hardiman and Burgin 2010; Sun and Walsh 1998). With Australian's penchant for outdoor recreation (particularly water-based; Hadwen et al. 2006), effective management of aquatic ecosystems is critical. In this paper we review the physical, chemical and biotic impacts of recreational vessels as they pertain to the Australian coastal and estuarine waters to (a) enhance awareness and understanding of such impacts; and (b) identify and prioritise future research needs.

Recreational boating trends

The majority of Australians participate in some form of water-based recreation. It is generally accepted that there is a disproportionate focus on aquatic resources compared to alternative outdoor recreational opportunities. Between 26 and 46% participate in recreational boating (Hadwen et al. 2006) and this activity continues to increase in popularity. Between 1999 and 2009, boat ownership grew nationally by 36.4% to over 800,000 (Table 1). While Australia-wide the greatest number of boats is generally correlated with areas of most dense population, boat ownership in Queensland grew at twice the speed of New South Wales, the most populous State. In the last 10 years growth has been twice the rate of New South Wales despite the population of Queensland being only 62% of the population of New South Wales (ABS 2009).

Australia-wide most recreational vessels are small, typically <7 m long, for example they constitute 85% of vessels in Tasmania (Marine and Safety Tasmania 2010) and on the Great Barrier Reef Catchment Area (GBRMPA 2009). In Tasmanian the growth in boat

Table 1 Number of recreational vessels registered annually for the decade 1999–2009 in each Australian State and Territory where data were obtainable

Year	NSW	Qld	Vic	Tas	SA	WA	Total
1999–2000	179,346	150,034	134,641	15,928	50,181	64,616	589,346
2000–2001	179,835	157,615	139,001	18,841	48,314	66,067	609,673
2001–2002	184,225	167,035	143,959	19,931	49,095	69,075	633,320
2002–2003	190,720	175,659	145,978	21,045	50,574	71,109	655,085
2003–2004	196,234	184,279	141,607	22,179	51,511	74,097	669,907
2004–2005	203,260	193,964	135,783	23,407	52,661	76,327	685,402
2005–2006	209,382	202,958	158,508	24,628	54,207	81,446	731,129
2006–2007	213,387	212,545	167,406	25,365	55,250	86,811	760,764
2007–2008	217,074	222,381	174,686	26,072	55,280	88,410	783,903
2008–2009	222,322	228,869	178,053	27,342	54,179	93,023	803,788
% increase[a]	24.0	52.5	32.2	71.7	8.0	44.0	36.4

Information supplied by respective State/Territory Marine Safely Department as personal communications. Data unavailable for the Northern Territory, estimate based on The National Recreational Fishing Survey for 2001 was 11,000. Australian Capital Territory does not require registration

NSW New South Wales, *Qld* Queensland, *Vic* Victoria, *Tas* Tasmania, *SA* South Australia, *WA* Western Australia

[a] Percent increase for the decade

ownership is broadly similar across all boats types and sizes; however, there has been especially high growth in the ownership of personal watercraft such as jet-skis. In this state the level of ownership of these watercraft (<7.5 m) increased by 533.3% in the decade to 2008–2009 (Table 2; Marine Safety Tasmania 2010). Since the trend in recreational vessel ownership has been on an upward trajectory in all states over the last decade (Table 1), the trends observed in Tasmania (Table 2) are very likely to be mirrored across Australia, although the rate of growth does vary (Table 1).

Impacts from recreational boating activities

Recreational impacts on aquatic ecosystems include physical, chemical, and biotic factors. They may be direct or indirect and typically result in decreased water quality, the introduction of alien species, physical disturbance and/or destruction of flora and fauna (Gergel et al. 2002). Detecting and measuring these impacts is inherently problematical. This is because, unlike terrestrial ecosystems, aquatic environments may not have clear boundaries. For example, estuaries are linear without defined upstream or downstream boundaries, 'edge effects' are extensive, and many aspects of aquatic health depend upon the management of the adjoining catchment (NRC 1995). Attributing impacts to a specific activity may also be difficult if the resource is used for multiple recreational activities (e.g., boating, swimming and/or fishing; Liddle 1997). Impacts (e.g., altered floral/faunal assemblages) are typically not as obvious in the aquatic ecosystem as in the terrestrial environment, and may also indirectly affect aquatic-dependent biota such as nesting, feeding, and/or breeding water birds. Impacts may also be localised (e.g., physical damage from a boat anchor (Dinsdale and Harriott 2004; Davenport and Davenport 2006) or dispersed (e.g., pollution from anti-fouling paints, Evans et al. 2000) while sewage from

Table 2 Number and type of recreational vessels that were registered annually in the Australian state of Tasmania between 1997–1998 and 2008–2009, together with percent increase over the period and type

Length	1997–1998	1998–1999	1999–2000	2000–2001	2001–2002	2002–2003	2003–2004	2004–2005	2005–2006	2006–2007	2007–2008	2008–2009	% decade increase
Dinghies and runabouts													
<7.5	11,173	12,813	14,075	16,211	17,133	18,124	19,018	19,672	20,680	21,646	22,161	23,132	18.2
>7.5 < 10	22	26	21	195	69	51	52	63	68	86	104	100	0.4
>10 < 20	0	0	0	0	0	0	0	0	0	0	0	0	
>20	0	0	0	0	0	0	0	0	0	0	0	0	
Total	11,195	12,839	14,096	16,406	17,202	18,175	19,070	19,735	20,748	21,732	22,265	23,232	85.0
% Craft	84.9	85.1	88.5	87.1	86.3	86.4	86.0	84.3	84.2	85.7	85.4	85.0	
Yachts and motor sailing craft													
<7.5	774	863	585	737	818	859	878	1265	1286	914	903	910	3.3
>7.5 < 10	587	634	530	776	903	918	946	917	917	1010	940	988	3.6
>10 < 20	514	588	526	662	662	667	757	899	1010	949	1091	1232	4.5
>20	4	11	11	5	5	7	6	11	13	8	12	11	0.02
Total	1879	2096	1652	2180	2388	2451	2587	3092	3226	2881	2946	3141	11.5
% Craft	14.3	13.9	10.4	11.6	12.0	11.6	11.7	13.2	13.1	11.4	11.3	11.5	
Personal watercraft													
<7.5	105	153	180	255	341	419	522	580	654	752	861	969	3.5%
% Craft	0.8	1.0	1.1	1.4	1.7	2.0	2.4	2.5	2.7	3.0	3.3	3.5	3.
Total craft registered or all sizes and percentage change over the decade													
Total	13,179	15,088	15,928	18,841	19,931	21,045	22,179	23,407	24,628	25,365	26,072	27,342	
%		14.5	5.6	18.3	5.8	5.6	5.4	5.5	5.2	3.0	2.8	4.9	

Source: Marine and Safety Tasmania 2010

coastal on-shore effluent disposal or from recreational vessels are typically indistinguishable (Warnken and Byrnes 2004). Threshold stress levels are also difficult to measure (Davis and Tisdell 1996). Understanding such impacts in aquatic ecosystems and their biological significance over the long term is especially lacking, and short-term behavioural responses of flora and/or fauna cannot necessarily be extrapolated to the longer term (Bejder et al. 2006; Hadwen et al. 2006, 2008).

Physical impacts of recreational vessels on the environment

Direct physical impacts from boating activities may damage plants and/or animal life (Antos et al. 2007). The intensity of such recreational impacts is dependent on factors such as craft speed, wave height, soil type, riparian vegetation density, and root depth. For example, in a discussion on the impacts of power boating on lakes and reservoirs, Mosisch and Arthington (1998, 2004) documented that detrimental effects on fauna can be caused by the impact of waves and wash that may directly destroy floating bird nests. These impacts may also result in bank soil erosion, physical damage to emergent and floating water plants, erosion of littoral plant roots, and changes in distribution of aquatic vegetation. Localised erosion also occurs at boat launching sites, typically due to boat propellers causing re-suspension of sediments. The severity of turbidity also depends on boat hull design, motor power output, water depth (more severe in shallow water) and sediment composition (e.g., clay suspenoids stay suspended for substantially longer than other sediment of larger particle size). Elevated sedimentation may also occur due to bank slumping and damage to habitats as a result of the destruction and/or disruption of bank integrity, plant communities and potentially changes to water quality. The resulting turbidity may have direct affects on fauna (e.g., fish, macroinvertebrates). For example, there may be macrophyte loss due to reduced light penetration. Alternatively, there may be loss of microhabitat. We assume that power boats have equivalent impacts in coastal environments, at least in estuaries and shallow bays.

Mooring also creates physical impacts on coral assemblages and/or other bottom fauna, for example when mooring directly over reefs. Impacts also typically extend beyond the point of impact and may cause wider damage as the anchor chain drags along the sea bed and/or the boat shifts with the wind, tide and current (Davenport and Davenport 2006).

The recently introduced propellerless jet-skis are able to travel (noisily) in very shallow water at high speed, and even over low banks through salt marsh vegetation. This has further extended the impacts of craft into habitats, even in marine protected areas, that were previously refuges from boat traffic for fish and birds (Burger 1998; Preen 2001). Such areas, for example salt marsh or mangrove ecosystems, are important nurseries for fish and other organisms, and their disturbance may consequently affect fish catch and overall biological diversity. Disruption to these ecosystems, including the substrate may displace sediment, affect hydrology, and render the shoreline more prone to storm surge erosion (Hall 2001). Although data are scant on the physical impacts of vessels, there is evidence that trampling in the intertidal zone (e.g., rocky shores—Povey and Keough 1991; coral flats—Kay and Liddle 1989; mangrove forests—Ross 2006) has a substantial impact. Vessels hulls and anchors are likely to have an even greater impact on coral reefs, seagrass beds and in mangrove forests.

Direct impacts of recreational boating on aquatic organisms are not restricted to bottom dwelling taxa. Many cetaceans, seals, turtles, and dugongs are killed due to collision with recreational vessels, especially powered boats. Turtles and dugongs are slow-swimming and are especially vulnerable, even to sail boats. Green turtles *Chelonia mydas* and dugong *Dugong dugon* (endangered with small populations) often browse in near-shore seagrass meadows where recreational boat traffic is concentrated and thus are frequently injured, even in marine protected areas (Davenport and Davenport 2006). Along with direct animal kill, the main negative impact of recreational boating is the destruction and/or pollution of the animals' seagrass beds used for grazing, although there is evidence that they also may be displaced from their habitat, including being driven from nurseries, owing to their sensitivity to and avoidance of boat-generated noise (Preen 2001).

Although data for the impact of recreational boating are unavailable, during the period 1999–2002 at least 65 turtles, mainly green and loggerhead *Caretta caretta*, were killed annually by vessel strike on the Queensland east coast. This figure is broadly comparable to the deaths that occurred in commercial trawl nets before the introduction of mandatory turtle-exclusion net devices (Hazel and Gyuris 2006). There are also on-going validated deaths of dugongs due to boat strike. For example, between 1996 and 2007, there were 34 (0–7 annually) such dugong deaths in Queensland waters (Greenland and Limpus 2007). These figures are likely to under-represent the true mortality of even recreational vessel strike since they were inferred from recorded stranded, dead or moribund animals. Others causalities would not have been encountered.

Marine mammals may also die from the indirect impacts of recreation. Fishing, including from boats, is a popular activity in Australia, estimated to attract one-third of the population. Associated activities generate substantial amounts of (mostly non-biodegradeable) aquatic rubbish (KESAB, undated). Nylon line, lead/steel sinkers, hooks and plastic bags dropped by anglers cause pollution and/or animal death. For example, although not necessarily attributable to recreational boating, Greenland and Limpus (2007) found that in addition to the 34 dugongs killed by boat strike (see above), at least two dugong deaths resulted from the ingestion of fishing line/hooks and a further five from entanglement in float lines and ropes.

Noise and visual disturbance created by motorised craft may also directly impact on local fauna and cause elevated stress, behavioural change and displacement, together with reduced reproductive success (Preen 2001), for example, among waterbirds (Mosisch and Arthington 1998; Suski and Cooke 2007). The noise of power boat motors also impacts on hearing capability in fish and even model power boats may disturb water birds or cause them to move away temporarily (Suski and Cooke 2007). For example, Bamford et al. (1990) found that birds moved temporarily, even in response to the noise of model power boats.

The effect of human recreation on shorebirds has been the specific topic of many northern hemisphere studies in recent years (for examples see Chan and Dening 2007) although published Australian studies specifically associated with direct impacts of recreational boating appear scarce. In a related study, undertaken in Moreton Bay to research the diversity and relative abundance of non-breeding (overwintering) tern species on estuarine sandbanks, researchers noted that the terns overwintering coincided with the period of highest human recreational use of the waterway. Recreational activities included boating, kite surfing, and the use of jet-skis and similar motorised personal watercraft, each of which may impact on the birds' behaviour (Chan and Dening 2007).

Recreational boating may also cause impacts on the behaviour of aquatic mammals and fish. Cetaceans (e.g., whales, dolphins and porpoises), seals, sealions, and the world's largest fish, the whale shark *Rhincodon typus* may be affected (Liddle 1997). The more obvious disturbance to such animals is due to recreational 'watching' animals in their natural habitats from boats (Liddle 1997). In Australia, there are at least 175 boat-based whale-watching commercial tour permits in New South Wales, Queensland and Western Australia (Valentine et al. 2004), and the number of watchers had increased by 15% annually to reach more than 1.5 million in 2003, consequently this industry contributes approximately AUD$300 million to the Australian economy (IFAW 2004). The 'swim with whale shark' experience at Ningaloo (Western Australia) began in 1989, and this industry had 15 licensed commercial boat operators with an annual income to the region of AUD$15 million in 2005 (CALM 2005).

The potential for such interaction, especially over extended periods, to adversely affect animal behaviour is acknowledged but not well understood. Management of tour operators is currently via specific 'codes of behaviour' (Catlin and Jones 2010; Valentine et al. 2004). Watching and interaction codes of behaviour for commercial tour boat operators are targeted differently for small whales (e.g., dolphins, porpoises), large whales (e.g., minkes, humpbacks) and the whale shark (Catlin and Jones 2010; Davenport and Davenport 2006; Valentine et al. 2004). Such codes are intended to reduce animal stress and allow the animals, as far as possible, to determine and control the level of interaction. Enforcing such codes is difficult in practice, however, and non-compliance is often recorded. For example, among commercial dolphin swimming and watching tour operators in Port Phillip Bay (Victoria), operators were found to be non-compliant with all four permit conditions (Scarpaci et al. 2003).

Such problems, together with increasing understanding of marine mammal displacement behaviour, have resulted in calls for revisions to existing codes and/or permits (Hawkins and Gartside 2008). Plans of management for many Australian protected areas acknowledge the potential impacts on native animals and place restrictions on boat speed, noise, and mooring, together with limits on visitor numbers, frequency of visits, behaviour and restrictions on approach distance to native fauna (e.g., GBRMPA undated). However there are increasing numbers of visitors seeking close contact with aquatic animals as part of an 'interactive experience', either watching them from boats (Liddle 1997) or swimming with them (CALM 2005) and there is therefore on-going pressure to allow greater opportunities for such marine mammal 'watching'.

Chemical pollution due to recreational vessels

Engine chemicals

Major pollution sources from recreational boating are fuel, oil and other chemicals discharged from powered boats. Such pollution occurs in water in the form of reduced water quality and/or is accumulated in the sediments, and in the air via exhaust fumes. Pollution is worst from 2-stroke engines (which use a mix of petrol and oil). These have been traditionally used on small vessels with outboard engines. The more widespread use of small 4-stroke engines (now 70–80% of outboard motor sales, Marine Engine Digest 2010), which are cleaner and more efficient, together with increased use of unleaded fuels has helped ameliorate the worst of the pollution from engine chemicals (EPA 2008a).

The main chemical contaminants present in the fuel are methyl tertiary butyl ether (MTBE), an additive used to replace lead in fuel and to reduce exhaust emission, and various polycyclic aromatic hydrocarbons (PAH) that are present in fuels and lubricants (EPA 2008a). MTBE evaporates relatively quickly in surface waters (EPA 2008b); however, their impacts on marine or estuarine aquatic organisms are apparently unstudied (Davenport and Davenport 2006). Some compounds of PAH are known to be potentially carcinogenic (Berko 1999; Davenport and Davenport 2006). Such compounds (e.g., fluroanthene, phenanthrene, benzo(a)pyrene and pyrene) generally have low solubility in water. They may not, therefore, persist in the water column but instead bind with organic/inorganic particulate matter and deposit in bottom sediments (Mastran et al. 1994; Mosisch and Arthington 2001). Contamination may therefore be immediate or may accumulate in the sediments over time. Accumulation occurs particularly under anoxic conditions, for example, in shallow confined areas (e.g., seagrass marshes, mangrove swamps), habits that are important to invertebrate communities and juvenile fish (Burger 1998; Preen 2001).

These are areas that are now increasingly penetrated by the new generation of propellerless personal watercraft (Burger 1998; Preen 2001). Such disturbance continues to provide a pollution source even after the use of the fuel has ceased (Davenport and Davenport 2006; Mastran et al. 1994; Mosisch and Arthington 2001). Determining the contribution of recreational boating to such chemical pollution is problematic as PAH can occur naturally and anthropogenically (Berko 1999); however, chemical impacts are best monitored in the sediments except where the pollution is recent and acute (Mosisch and Arthington 2001; Matthai et al. 2009).

Anti-fouling treatments

Chemical pollution is also derived from anti-fouling treatments used on the hulls of all boats used in the marine environment to prevent encrusting organisms such as barnacles and limpets attaching to the hull (Mosisch and Arthington 1998, 2001, 2004; Saphier and Hoffmann 2005). Fouling causes drag, reduces speed and increases fuel consumption and, therefore, greater exposure to pollution. To combat fouling, hulls need to be scrubbed regularly (typically annually) and treated, most commonly with copper-based paint or, since its introduction in the 1960s (Terlizzi et al. 2001) one containing the chemical Tributyltin (TBT). As with previous antifouling paint formulations (e.g., DDT, lead, organo-mercury compounds), TBT has proved effective and has been widely used. It is, however, highly toxic to non-target marine organisms (Evans et al. 2000). Even in minute quantities it may bioaccumulate in the food chain (see Mosisch and Arthington 1998, 2001, 2004 for detailed reviews of the impacts in freshwater ecosystems).

Concentration, release and longevity of TBT are influenced by various factors, for example, its water solubility. It is strongly affected by pH; toxicity from TBT is lowest at neutral pH while it is more soluble and toxic at lower and higher pH values (0.75–31 μg/g over the pH range of 2.6–8.1; Uhler et al. 2000). When present the compound biodegrades relatively slowly. This implies continuous delivery to the water column. Sewage and humic substances also slow its photo-degradation and create a particulate source for filter feeding organisms (Noller 2003). The presence of TBT in an actively polluting form is not; however, necessarily linked to its level in silt and mud sediments as it may be released from the boat hull in flakes which are deposited in high energy sandy areas, such as shipping channels. It may therefore potentially be present in measurable quantities in sediments in areas not otherwise exhibiting elevated concentrations of inorganic analytes or organic compounds (Matthai et al. 2009).

Pollution by TBT has caused mortality in the Pacific oyster *Crassostrea gigas*, a commercial species farmed widely in eastern Australian states. While adult oysters may tolerate low concentrations in the surrounding waters, with typical uptake in meat at 10–20 μg Sn/g, spat have shown toxic effects at as low as 5 μg/l and mortality at greater than 200 μg/l (Noller 2003). This compound has also been implicated in causing ‘imposex’ in gastropods, an endocrine disruption syndrome which results in the development of male organs in females. The condition has been recorded in around 120 gastropod species worldwide and can, in extreme cases, result in sterility leading to population decline or local extinction (Garaventa et al. 2006; Makita and Omura 2006). Induction of imposex appears to be dose-dependent and may occur at very low ambient concentrations, and at as little as a few nanograms per litre (Garaventa et al. 2006). The condition can also occur after relatively short-term exposure and persist (Gibson and Wilson 2003).

Toxic effects of TBT, for example on oyster mortality and/or gastropod imposex have been observed to be more frequent, severe and/or more persistent in enclosed 'hotspots' such as harbours or marinas, compared to open sites (Gibson and Wilson 2003; Noller 2003). For example, following translocation to Sydney Harbour from open coastal locations, imposex was induced in the common predatory intertidal whelk *Thais orbita* within 9 weeks and was still found to be widespread at other nearby sites 10 years after a partial ban on TBT-based paints (Gibson and Wilson 2003).

Internationally, TBT has been implicated in causing mortality, monstrosities and/or hormonal imbalance in a wide range of organisms including dolphins, crabs, lobsters, invertebrate larvae, seagrasses and algae (Evans 1999; Evans et al. 2000). However, it is still debatable whether TBT is the primary causal agent of such impacts, or whether additional factors (e.g., heavy metals, changed environmental conditions, parasitic infestations and/or other androgenic compounds) may contribute (Garaventa et al. 2006).

Since TBT was banned globally by the *International Maritime Organisation* (IMO) in 2008 (Champ 2003; Kotrikla 2009), the impacts of this chemical will ultimately decline; however, because of the past extensive international use of the chemical on the hulls of boats, amelioration will be long-term.

Paints containing a variety of heavy metal oxides are also used to protect the hulls of boats. Many such compounds, especially copper (Cu), lead (Pb) and zinc (Zn) are environmentally toxic and are persistent as pollution sources due to their affinity for, and immobilisation within, anaerobic sediments (Irvine and Birch 1998). Although studies of heavy metals in sediments are relatively rare in Australia (Matthai et al. 2009), significantly higher levels of some metals (e.g., Cu, Pb, Zn) were found at boating 'hot spots' such as wharfs and boat slipways, compared to open water sites, in some cases exceeding sediment quality guidelines in the lower Hawkesbury—Nepan River (Matthai et al. 2009).

One factor that may influence the persistence of heavy metals in the sediments is the presence of mangroves. Common in Australian estuarine ecosystems, particularly in northern Australia, they are robust to heavy metals and able to tolerate relatively high accumulation before exhibiting visible signs of impact such as reduced growth and/or reduced photosynthesis. Metal accumulation and partitioning for Cu, Pb and Zn have been found to be broadly similar across genera and families (MacFarlane 2002; MacFarlane et al. 2007). These plants may therefore play a role as phytostabilisers and aid in toxic metal retention and/or reduced transport to adjacent ecosystems (MacFarlane et al. 2007). Sufficiently high heavy metal uptake may; however, cause cellular damage and/or wider phytotoxic effects in individual plants and wider ecosystem effects due to the production and distribution of carbon-based products along the estuarine food chain via detrital export (MacFarlane 2002).

Organic biocides

While acknowledging the environmental impacts of TBT and heavy metals, concerns have also been voiced regarding the organic biocides introduced as their replacement (e.g., Evans et al. 2000; Matthai et al. 2009; Thomas 2001). Specific concerns include limited knowledge of possible toxicity, sub-lethal impacts, persistence, and synergistic effects. Concentrations of biocides have already been detected in areas of high boating activity internationally, for example in recreational marinas and harbours (Konstantinou and Albanis 2004). The main source of biocide release from vessel hulls is probably from normal wear but the loss can be exacerbated, for example with the use of high-pressure cleaning hoses, potentially releasing paint particles that may become incorporated into the sediments (Thomas et al. 2003).

Biodegradability can vary considerably among the various biocides, ranging from readily biodegradable (e.g., Sea-Nine 211) to non-biodegradable (e.g., diuron and Irgarol 1051; Matthai et al. 2009). Apart from leaching rates, factors influencing risk and degree of potential pollution include the number of vessels treated, water movement, rate of degradation, and adsorptive behaviour. Although biocides are generally found at lower concentrations in the water column than in sediments (Voulvoulis et al. 2002), a lack of reliable data on their solid-phase—dissolved-phase partitioning makes it difficult to determine concentration sorption affect partitioning (Matthai et al. 2009). Monitoring biocide fate in water and sediment and their effects on organisms (e.g., sea urchins, marine oysters), is therefore currently limited.

Current status antifoulants

Despite limited data, the European Commission has banned some non-biodegradable antifouling biocides (e.g., diuron) because of demonstrated toxicity and has restricted the use of several others (e.g., dichlofluanid, zinc pyrithione, zineb). Although such organic biocides are not routinely assessed in the Australian marine environment and data are therefore scarce, future use of diuron is under review while the use of other biocides (e.g., dichlofluanid, chlorothalonil) has been restricted (APVMA undated).

The risk of pollution of Australian marine ecosystems from antifouling chemicals discussed here is substantial and is recognised in state/territory and federal legislation governing commercial and recreational vessels. Visiting international vessels also pose risks. For example, although unavailable in Australia, the biocide Irgarol 1051 has been detected in the Hawkesbury-Nepean estuary. Environmental impacts can also persist. Banned in New South Wales since 1989, elevated concentrations of TBT, together with heavy metals (e.g., Cu, Pb, Zu) and organic booster biocides (e.g., diuron, chlorothalonil, Irgarol, dichlofluanid) used to augument Cu-based antifoulants since TBT was banned have been also recorded at recreational boating sites in the Hawkesbury-Nepean estuary (Matthai et al. 2009).

Organic pollution

Chemical impacts may also result from sewage discharge (Davenport and Davenport 2006) which may alter algal, coral and other communities, causing blooms and other pollution related problems (An et al. 2002). In addition to each of the more than 800,000 registered recreational vessels in Australia providing a source of machine waste, there is also a constant source of human sewage and other organic waste. An indication of the amount generated is indicated by the research of Leon and Warnken (2008) undertaken over one year at 20 popular anchor sites in the eastern part of the Moreton Bay Marine Protected Area (Queensland). Approximately 10,000 recreational vessels (>6 m length) produced an estimated annual load of 141 + 46 kg of copper (from Cu-based antifouling paints), 22.2 + 13.3 tonnes human faecal matter, 102.1 + 13.3 tonnes urine and 1.17 + 0.38

tonnes of nitrogen (from sewage). This load is approximately equivalent to a modern sewage treatment plant serving a 4.5 km^2 mostly residential development and associated infrastructure (Leon and Warnken 2008). Such vessel activity (and hence pollution load) is also likely to be highly variable, with spikes that coincide with periods of highest activity, typically weekends and school/public holidays, most notably during Christmas/New Year and Easter periods (Widmer and Underwood 2004).

Discharge of raw sewage into waterways is generally prohibited in Australian waters. Sewage must be captured in onboard holding tanks and disposed of via approved pump-out facilities at harbours/marinas. Management typically includes boat owner educational campaigns and fines (e.g., New South Wales—$750 individual, $1500 corporate; NSW Maritime, undated a, b). Despite these requirements, the proportion of boats fitted with sewage holding equipment is unknown. However, since approximately 90% (excluding personal watercraft) of Australian recreational vessels are ‘small’ (i.e. < 7.5 m long), it is assumed that the proportion with pump-out facilities on-board is small. This assumption is supported by anecdotal evidence from marina operators who have suggested that the use of shore-based sewage pump-out facilities is low (Leon and Warnken 2008). Sewage is known to often be discharged untreated into the local waters, for example at anchor sites. Since these areas are usually located in calm, protected waters with low tidal flushing, the pollutants tend to sink to the seabed (Pratt et al. 2007). As indicated above, the load dumped would depend on recreational vessel activity which is highly variable (Widmer and Underwood 2004).

Biotic impacts associated with recreational vessels

The introduction and spread of non-indigenous species is considered to be a key conservation management issue worldwide (Lintermans 2004). Transport of such species on fouled hulls has been identified as a major route for their spread by recreational vessels. The extent of such fouling depends on a hull’s susceptibility to recruitment by aquatic organisms and the local availability of competent planktonic propagules (Floerl and Inglis 2003).

More than 250 marine non-indigenous species now inhabit Australian waters. These range from plankton and algae to fish, and are spread throughout all habitats (Thresher 1999). While the scale, relative importance of transport mechanisms, and underlying biological processes is less well known for aquatic compared to terrestrial ecosystems (Hewitt et al. 2004), alien species may be introduced on any fouled hull entering Australian waters. Once established, the associated biota may be translocated to other destinations, for example adjacent bays/harbours, often with harmful effects. The potential for further invasions continues to grow with the increasing popularity of recreational boating (Thresher 1999). Once established in Australian coastal or estuarine waters, such taxa may rapidly out-compete native biota. The outcome may dramatically alter diversity and abundance, potentially over substantial geographic areas, and some may cause widespread economic impacts (Cohen et al. 2000; Hewitt et al. 2004). For example, the Northern Pacific Seastar *Asturias amurensis* and the Giant Fan Worm *Sabella spallanzanii* became established in the Derwent River estuary (Tasmania), probably in ships’ ballast water in the early 1980s (Morrice 1995). Predator of native oysters and other sedentary species, *A. amurensis* reduces the abundance and commercial harvests of these native species. Once *S. spallanzanii* is established it excludes native biota by smothering seafloor habitat (Environment Australia 2001). To date, attempts to eradicate these species have failed and by 1998 *A. amurensis* had extended its range to Port Phillip Bay (Victoria), a large temperate embayment that Hewitt et al. (2004) suggested had more alien species than any other marine ecosystem in Australia.

Long-term studies of Port Phillip Bay have shown that the aquatic community has changed substantially over time. Some 160 alien species have been identified. These are concentrated around the ports of Geelong and Melbourne (Hewitt et al. 2004). Non-indigenous species comprised over 13% of the total recorded species; four groups, bryozoans, cnidarians, chordates, and crustaceans, represent over 75% of total non-indigenous species (Hewitt et al. 2004). Such species were also widespread and abundant in the epibenthos and seven of 23 taxa (35% of individuals) were alien (Cohen et al. 2000). In comparison, the Derwent River estuary had 70 alien species in 2008 (Whitehead 2008) and Sydney's Botany Bay had around 45 such species in 1998 (Pollard and Pethebridge 2002). Although quantification of the rate of invasion of non-indigenous species is problematical for example, due to difficulties of taxonomic identification and/or sampling frequency, at least three new benthic alien species probably become established in Port Phillip Bay annually (Hewitt et al. 2004; Thresher 1999).

Recognition of the potential devastation of such introduced species has resulted in the classification of alien species as a key threatening process (DEWHA 2009).

Transport vectors

Discharge of foreign-sourced ballast water has long been recognised as a major source of alien species introduction (Bax et al. 2001; EPA 2010). Although such tanks are often absent in the small boats typically used for coastal recreation, they are used in large yachts. Another major source of such introductions is on fouled hulls which have historically and consistently been shown to be a dominant transport vector (Coutts et al. 2003; Hewitt et al. 2004). However, along with fouled hulls and ballast tanks, infection on recreational vessels can include internal water systems, anchor/chain cabinets, sea chests or ropes. For example, the bivalve *Corbula gibba*, although considered a 'ballast water' non-indigenous species, has been discovered in sea chests (Coutts et al. 2003).

Once introduced, biota may be translocated (Hayes et al. 2004, 2007). For example, following the introduction of a cold-water tolerant strain of *Caulerpia taxifolia* into Port Hacking, near Sydney (Millar 2000), the species has spread to nine other locations on the New South Wales coast, all estuaries or sheltered embayments. Since the infestations are especially concentrated around boat launching ramps and jetties, it is assumed that translocation was due to movement of recreational craft (Creese et al. 2004).

The black-striped mussel *Mytilopsis sallei*, a close relative of the zebra mussel *Dreissena polymorpha,* accidentally introduced into marinas near Darwin (Northern Territory), is thought to have been introduced on fouled hulls or in the seawater piping of visiting international yachts (Thresher 1999). Although detected in the early stages of infestation, its eradication necessitated the efforts of 270 people and the decontamination of hundreds of recreational and fishing boats at a cost of more than AUD$2 million. These activities resulted in the destruction of all taxa within three enclosed marinas (Bax et al. 2001; Goggin 2004). A similar eradication operation was required in 2001 following the discovery of the invasive Asian green mussel *Perna viridis* on a hull in Cairns Harbour (Goggin 2004; Neil 2002).

Some species have proved impossible to eradicate. For example, a serious threat is posed by the northern Pacific seastar, *Asterias amurensis* (NIMPIS 2002a). Introduced from Japan into Tasmania in the 1980s, this pest had spread to Port Phillip Bay by 1998.

Eradication attempts have failed, and due to that port's importance as a major hub for domestic commercial and recreational traffic, its secondary infestation of other Australian ports is inevitable (Thresher 1999). Although not such a major pest because it is not highly invasive, the Caribbean tubeworm *Hydoides sanctaecrucis* has been introduced into the Cairns Harbour and other Australian ports and marinas, where it has become a hull-fouling nuisance for boat owners (Goggin 2004).

Recognition of the potential for recreational vessels to act as transport vectors for alien species is growing (Glasby and Lobb 2008; Hewitt et al. 2007; Pollard and Pethebridge 2002). Although the size of their contribution to the problem appears to have had limited research (Johnson et al. 2001, Minchin et al. 2006), their infection risk is known to differ by (i) vessel type (moored vessels higher risk than trailered) and; (ii) species (Glasby and Lobb 2008). Risk also depends on owners' awareness and preventative care; type and age of antifouling paint used (Floerl et al. 2005); the vessel hull's susceptibility to recruitment by aquatic organisms, and local availability of competent planktonic propagules (Floerl and Inglis 2003). However, even well maintained vessels may act as vectors in the translocation of introduced species (Hayes et al. 2004, 2007). Studies of marinas in Cairns and Townsville have suggested that the degree of fouling may be influenced by the design of the harbour in which the boat is moored. Recruitment of sessile invertebrates to available surfaces has been shown to be several orders of magnitude more abundant in enclosed marinas (e.g., breakwaters) than in unenclosed marinas. The exception was barnacles, which were found to recruit more densely in open locations (Floerl and Inglis 2003). This may be because although entrapment of water in enclosed marinas may limit the dispersal of planktonic propagules, it also increases propagule pressure to available surfaces, including resident boat hulls. Harbour design may therefore have an important influence on the size and frequency of inoculation events of non-indigenous species.

Assessment and monitoring programs

Problems in monitoring non-indigenous species in Australia include (i) natural inter-annual or long term ecosystem change; (ii) anthropogenic impacts such as dredging (Currie and Parry 1999); (iii) large uninhabited coastal areas and; (iv) wide range of climate zones. Tropical zones are particularly at risk of invasion by alien species in 'wet' seasons, owing to changes in water salinity, temperature and stratification of local waters, which increase micro-environmental niches (Marshall et al. 2003). Longitudinal studies are essential, and Australia's National *Introduced Species Port Baseline Surveys Program* provides such data by monitoring non-indigenous invasions in the country's 62 most important 'first ports of call' for foreign vessels (Hewitt et al. 2004). Australia's management of alien species will also be enhanced with the adoption of a National System for the *Prevention and Management of Marine Pest Incursions*. At the time of writing, implementation of this System had been reached by the Federal Government, Northern Territory and all state governments except New South Wales, the latter pending final agreement (NIMPIS 2009b).

Australia is also a signatory to the *2005 International Ballast Water Management Convention* (NIMPIS 2009b). All vessels operating in Australian waters are governed by *Australian Ballast Water Management Requirements*, managed by the Australian Quarantine and Inspection Service (AQIS). Separate requirements apply to vessels that are less than 25 m or greater than 25 m in length, regardless of whether they are used for recreational or commercial purposes. All boats are subject to similar regulations governing discharge of ballast water tanks and on-shore disposal of sewage. However, those over 25 m in length are required to use approved port facilities, while for owners of smaller vessels it is voluntary. Smaller boats are also expected to voluntarily manage hull-antibiofouling, although it is expected that this will become mandatory (AQIS 2010a, b).

Discussion

The issues associated with recreational boating in Australia are not unique to Australia or to recreational vessels. All of the physical, chemical and biotic impacts on aquatic ecosystems due to recreational boating that we have discussed also potentially occur elsewhere. This has been recognised. For example the *International Maritime Organisation*, the United Nations body that administers the international regulatory regime for shipping, has adopted a treaty to ban the global use of TBT (Champ 2003; Kotrikla 2009). Australia supported this ban and has investigated alternative treatments to its use on naval vessels. In the process over 120 potential alternatives were assessed (Lewis 2008). The outcomes of such research ultimately become available commercially.

The potential for ballast water to act as a vehicle for the translocation of organisms was first proposed in the early 1900s, although it was not until 1975 that Medcof published the results of the first sampling of ballast water. In 2002, Gollasch et al. published a review of past European shipping studies and reported that over 1000 species had been recorded to be transported in ballast water. The first international guidelines were introduced by the *International Marine Organisation* in the early 1970s, and more recently (2004) the *International Convention for the Control and Management of Ships' Ballast Water and Sediments* was adopted, and member states of the *International Maritime Organisation* have been invited to become signatories (Gollasch et al. 2007). Australia became a signatory in 2005 and is currently preparing the legislative and administrative changes required to give effect to this Convention (DIF 2010).

The enhanced awareness of these issues will therefore ultimately permeate down to the community. However recognition of the role that recreational boating plays in environmental impacts on coastal waters tends to be ignored, even among world authorities. For example, the most recent report commissioned by the *European Confederation of Nautical Industries* (Moreau et al. 2007) generally restricted its discussion of the impacts of recreational vessels to oil and fuel emissions and waste water management (the concerns they identified as potential issues), and reported that these chemicals were negligible compared to other sources of such pollution. The focus of the report was the direct impact on humans, with only limited consideration of the impacts on the aquatic environment.

With the recorded and projected increasing demand for recreational boats, associated with increased leisure time and the associated demand for water-based recreation, there is an urgent need to raise awareness of the potential impacts of associated activities in coastal environments. This requires greater research effort to enhance our understanding of the problems, and provide the tools to better manage recreational boating. For example, marinas designed to minimise the invasion of non-invasive species should be investigated. Further research and development of antifouling treatments that are not toxic to non-target species including for example, research associated with hull design and treatments that inhibit settlement of organisms on boat hulls, together with the more sophisticated design and/or handling of water inlets (e.g., bilge, sea chests) is needed. Mechanisms to underpin legal instruments (e.g., policies, laws, treaties) based on the appropriate research should be undertaken (where lacking), and consideration of reducing the time taken to develop such instruments at the regional, national and international levels is required.

More environmentally benign products and suitable legislative tools should reduce the chemical and biotic environmental impacts of recreational boating in coastal waters. However some impacts, including on-shore waste disposal, together with the physical impacts on animals require more than just ‘codes of conduct’ or voluntary compliance underpinned by legislation. It requires changed attitudes of those involved in recreation associated with boating.

While the Federal and state governments provide the legislative incentives, with the continued increase in numbers of recreational boaters predicted, the resources required for enforcement will continue to be lacking. Real change will only occur with the enhanced awareness of the issues by the recreational boating fraternity and their involvement in compliance. This requires raising awareness in the community of the cumulative impact that is occurring due to the actions of individuals who undertake recreational boating activities.

There are models within Australia of governments and communities working together to support voluntary environmental management, most notably through Landcare. The focus of Landcare is to more sustainably manage natural resources and the network encompasses a range of ‘care’ groups that reflect the interests of the participants (e.g., Rivercare, Dundcare, Bushcare; Burgin et al. 2005). In December 2010, in New South Wales alone, there were 2130 groups (57,454 individuals) registered under Landcare (LandcareNSW 2011), and communities in each state and territory participate (Burgin et al. 2005). In 2010, Clear Up Australia Day attracted 588,000 volunteers (1 in 50 Australians) to remove rubbish from 7,000 registered sties (Landcare Australia Ltd, undated). These figures indicate that volunteer Australians have the potential and are willing to participate in environmental management. We therefore predict that with the support of governments to guide, engage and educate the boating fraternity changes in attitudes could be readily achieved. This, in turn, would lessen the impacts of recreational boating.

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