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A Comparison of the Costs and Effectiveness of Prevention, Eradication, Containment and Asset Protection of Invasive Marine Species Incursions

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The Australian Government is currently reviewing the national biosecurity arrangements that are designed to safeguard Australia's maritime industries and the marine environment from non-indigenous marine species (NIMS). As part of this review, ABARES was commissioned to assess the costs and effectiveness of actions to prevent NIMS incursions, compared with the costs and effectiveness of eradication and the costs of living with the NIMS should they become established.

This study is restricted to ballast water management as a preventative measure. While biofouling is also recognised as a major pathway for the introduction of NIMS, the costs and effectiveness of biofouling prevention are not included in this study. This is because it is understood that currently the primary reason for investment in biofouling prevention is to manage fuel efficiency and vessel safety. Australia does not currently have a formal national system for the management of biofouling for the purposes of biosecurity.

In this analysis, the costs and effectiveness of different management approaches to NIMS are compared using three different methods. While conceptually simple, this type of study is challenging because of significant uncertainty about the magnitude of many of the critical factors. To account for this uncertainty a range of values was considered, based on values derived from a comprehensive review of available literature and from estimates derived from primary data sources. Values included:

- The costs of prevention through ballast water management are estimated at $36.2 million a year for exchange and $0.8 million a year for compliance monitoring.
- Incursion rates in the presence and absence of prevention by ballast water exchange (0.04 – 1.5 a year in the absence of ballast water exchange, with an 80 – 90 per cent reduction in the presence of ballast water exchange).
- The costs and effectiveness of eradication attempts ($5 million – $20 million, with a 5 – 20 per cent chance of successful eradication).
- The costs of living with NIMS, which include any loss of production from marine industries (for example aquaculture); non-market impacts (for example impacts on the environment); plus any management costs directed at controlling the NIMS ($4 million - $1 billion per incursion).

The first method compares the total cost of living with all high impact NIMS with the total cost of a prevention or eradication approach. Because prevention and eradication approaches are unlikely to be 100 per cent effective at avoiding the establishment of NIMS, the total cost of these approaches includes the cost of living with those high impact NIMS that still become established despite the approach being used. The results are highly dependent on the parameter values considered, particularly for prevention. These range from highly favourable, i.e. cheaper, for prevention relative to the other two approaches when the incursion rate of ballast water pests is high and/or the assumed average cost of living with a high impact incursion is high, to highly unfavourable for either prevention or eradication relative to living with all high impact NIMS when incursion rates are low and/or the assumed average cost of living with a high impact incursion is low. High average costs of living with NIMS could arise if the public places a high value on non-market impacts. When eradication is favoured over living with all high impact NIMS it is only favoured slightly, because most eradication attempts are likely to fail.

The second method compares the cost-effectiveness of prevention vs. eradication, i.e. money spent per establishment avoided. For this method it is not necessary to estimate the average cost of living with NIMS, but it is critical to define the management objective. If the objective is to avoid the establishment of all NIMS regardless of their impact, then eradication cannot achieve the same outcomes as prevention. This is because eradication would only ever be attempted for potentially high impact NIMS, while prevention covers all NIMS. If the concern is only with high impact incursions, then a break-even point can be estimated where an amount of money spent on eradication attempts achieves the same reduction in establishments of NIMS as the prevention system. In this case the break-even point requires a 90 per cent chance of eradicating all high impact marine incursions, but it is unlikely that this would be achievable regardless of the amount of money spent. Hence, in terms of cost-effectiveness, prevention is clearly favoured over eradication.

The third method compares the benefit-cost ratios (BCRs) of prevention or eradication approaches. BCRs indicate the losses avoided relative to the amount of money spent, but they do not provide a good basis for comparing between eradication and prevention, because they ignore the impacts of failures to either prevent or eradicate. For prevention, results were similar to the first method, but for eradication the results showed that when considering any individual incursion it may be worth considering attempting eradication.

The evidence from the scientific literature suggests that, to date, there has been a low rate of incursion by high impact NIMS in Australia. Assuming this is correct, and that the average cost of living with high impact NIMS is low, then living with all NIMS may be the better option. If, however, the average cost of living with high impact NIMS is found to be high, or if living with NIMS is considered undesirable, then prevention is generally preferred over eradication. Even with a prevention system in place however, eradication could still be considered as a backup in the event of the prevention system failing as incursions arise.

A major consideration for the addition of an eradication approach is how much or whether to invest in an 'early warning' system to improve the likelihood of eradication being successful. That was beyond the scope of this study, but would benefit from future work.

1 Introduction

1.1 Background

The Australian Government is currently reviewing the national biosecurity arrangements that are designed to safeguard Australia's maritime and marine industries and the marine environment from non-indigenous marine species (NIMS). As part of this review, ABARES was commissioned to assess the costs and effectiveness of actions to prevent NIMS incursions, compared with the costs and effectiveness of eradication and the costs of living with the NIMS should it become established. This report presents results from the ABARES analysis.

1.2 Invasive marine species

Non-indigenous marine species that have large impacts are referred to as invasive marine species (IMS). IMS transported around the world on hulls of vessels and in ballast water are considered a major threat to biodiversity by displacing and preying on native species, altering marine ecosystems and affecting water quality (Carlton and Geller, 1993, Bax et al., 2003). They are also considered a threat to human systems by preying on aquaculture species, fouling ships hulls and urban and industrial infrastructure (Gollasch, 2011, Rilov and Crooks, 2009, Molnar et al., 2008, Bax et al., 2003) and some species pose risks to human health and public amenity values (Nunes and van den Bergh, 2004, Hallegraeff, 1992).

The identification of a recently established population of an invasive species is often difficult. It is estimated that it took at least six years for the introduced population of northern Pacific seastar (*Asterias amurensis*) in the Derwent estuary in south-eastern Tasmania to be identified, as it was mistaken during that time for the native rough seastar (*Uniophora granifera*) (Turner, 1998). For eradication of an incursion of an invasive marine species to have any chance of success it must be detected early while eradication is still technically and logistically feasible. For example, the successful eradication of black-striped mussel (*Mytilopsis sallei*) in Darwin in 1999 was largely because of the fortunate restriction of the incursion to marinas with lock gates, but not before it had reproduced at least once (Bax et al., 2002). This is one of only a small number of successful marine eradications globally. In most cases the discovery of an incursion occurs too late for eradication to be feasible (Turner, 1998) or an eradication attempt fails (e.g. Coutts and Forrest, 2007).

Because invasive marine species can have large impacts, and eradication is considered very challenging, minimising the likelihood of the arrival and establishment of IMS has often been asserted as being a cheaper and more effective option than either attempting eradication or control once established (Finnoff et al., 2007, Hewitt et al., 2007, Vander Zanden et al., 2010). Leung et al. (2002) found that the benefits from a prevention program to achieve a modest reduction in the risks of zebra mussel becoming established in a lake exceeded the costs, but there is little other documented evidence to confirm the assertion that prevention is better.

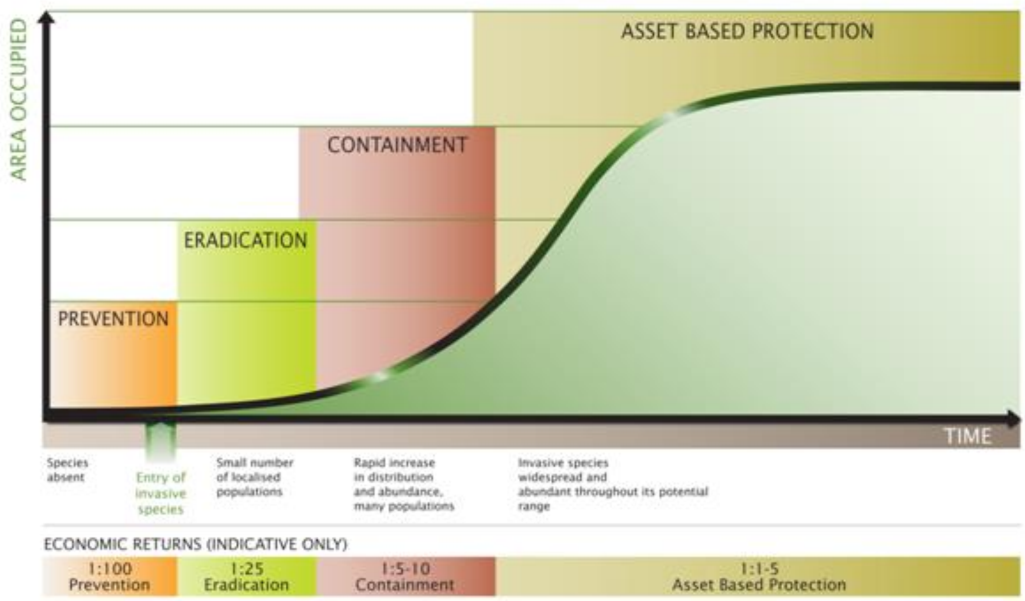
1.3 Outline of approach taken

The purpose of this study is to bring together information on the costs and effectiveness of prevention when considered against the potential costs and effectiveness of eradication, containment and protection of assets – the elements of the generalised invasion curve (Figure 1). The costs of prevention considered in this study focus on the total costs of ballast water exchange or treatment by vessels arriving from international ports; and the costs to the government of monitoring compliance with this system. These costs were estimated based on vessel transit data combined with costs of ballast water exchange for different types of vessel.

Biofouling, like ballast water, has been recognised as a major pathway for invasive marine species (Hewitt et al., 2011, Coutts, 1999). It has also been recognised by the IMO, which has developed the Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species to provide a globally consistent approach to the management of biofouling (IMO, 2011). The costs of biofouling prevention are not, however, included in this study. This is because it is understood that the primary reason for investment in biofouling prevention is to manage fuel efficiency and vessel safety. The history of biofouling prevention on ships hulls to improve sailing efficiency goes back to at least the 17th Century (Chambers et al., 2006). While it may be the case that not all vessels are as rigorous about biofouling prevention as is desirable for biosecurity control, it was decided that since a) the majority of vessels carry out regular measures to prevent biofouling build-up and b) there is no precedent for the proportional allocation of biofouling prevention costs to fuel efficiency and biosecurity, and c) Australia does not currently have a formal national system for the management of biofouling for the purposes of biosecurity, it should be considered that all biofouling prevention costs should be allocated to fuel efficiency. If, at a later date, it is determined that some proportion of the costs of biofouling prevention should be allocated to biosecurity and Australia adopts a formal system for managing the biosecurity risk associated with biofouling, further research will be required.

While estimating the costs of ballast water management is relatively simple, estimating the likely effectiveness of these treatments is more problematic. We draw on past published studies to estimate incursion rates in the presence and absence of preventative measures, but given the significant uncertainty we present results for a range of possible values. Effectiveness of prevention is presented in section 3.

Figure 1 Stages of invasion and generalised invasion curve.



Source: Department of Environment and Primary Industries, Victoria.

Estimating both the costs and effectiveness of eradication is also difficult, and again we draw on past published studies, but given the significant uncertainty we present results for a range of possible values. The costs and effectiveness of eradication are presented in section 4.

If eradication fails, or if it is decided not to proceed with eradication, the remaining management responses to marine pest incursions in Australia are: attempting to contain it to the area of incursion; or deciding to let the pest spread and manage any impacts that arise. Both could be attempted, but here we consider them as separate management decisions. We term the latter approach ‘living with the pest’. When ‘living with it’ is the option, Australia will be subjected to the impacts of the pest. These impacts include any loss of production from marine based industries (for example aquaculture); non-market impacts (for example impacts on the environment, social amenity); plus any management costs directed at controlling the pest. For each local entity where the pest will have an impact, managers will decide how much to invest in management vs. the level of residual impact on production/non-market impact they will accept. The sum of management costs plus the residual impacts determine the cost of living with the pest. The costs of living with the pest could be considered the same as the costs of asset protection.

While we also attempt to value these impacts, properly assigning monetary values to them is beyond the scope of this study. This would be an expensive and complex process, requiring the identification of appropriate non-market valuation methods that would capture all aspects of non-monetary values that could be affected and put them in monetary terms. Hence, section 2 focuses on available existing information about the likely impacts of marine pests, including non-market impacts, as well as presenting results from a case study of impacts of already established marine pests in Port Phillip Bay.

Based on all these values, we compared the costs and effectiveness of the different management approaches to NIMS with three different methods, presented in section 5. In the first method the total cost of living with all high impact NIMS is compared with the total cost of an eradication or prevention approach. Because eradication and prevention approaches are unlikely to be 100 per cent effective at avoiding establishments of NIMS, the total cost of these approaches included the cost of living with those high impact NIMS that still become established despite the approach being used. In the second method, the cost-effectiveness of prevention vs. eradication is compared by estimating the cost per establishment avoided. In the third method benefit-cost ratios (BCRs) of prevention and eradication approaches are estimated. BCRs indicate the losses avoided relative to the amount of money spent on each approach.

2 Impacts of invasive marine species

In this section we review the literature on impacts of invasive marine species and present results from a limited case study of Port Phillip Bay.

2.1 Evidence of impacts

Economic and environmental impacts of invasive species in terrestrial systems are well known, but impacts of invasive marine species have received less attention (Pimentel, 2011). Many non-indigenous marine species are now found throughout the world but reported impacts exist for a relatively small proportion of these. For those that do exist, reported economic and environmental impacts can be very large, as described below.

Large economic impacts include, for example, the US$200m a year (in 1992 dollars) impact of naval shipworm (*Teredo navalis*) on ships and docks in the U.S.A. in the early part of the 20th century (Cohen and Carlton, 1995). The introduction of the ctenophore (*Mnemiopsis leidyi*) to the Black Sea in the 1980s resulted in a 90 per cent decline in catch of Black Sea anchovy (*Engraulis encrasicolus*) with estimated losses of "in present values terms, … [of] hundreds of millions of US dollars over several decades" (Knowler, 2005). The annual costs of managing the impacts of the fresh water zebra mussel (*Dreissena polymorpha)* and quagga mussel *(Dreissena rostriformis bugensis*), which were accidently introduced into the Great Lakes in North America in the late 1980s in ballast water (Hebert et al., 1989), was reported at more than US$500 million. Impacts included fouling of power plants, water systems, industrial complexes, and on boats and docks in the Great Lakes. The cost to electricity generation and drinking water plants alone was a total of more than US$267 million for the period from 1989 to 2004 (Connelly et al., 2007). Significant environmental impacts have also been reported. For example, the introduction of the ctenophore to the Black Sea also caused environmental impacts (Zaitsev, 1992). The invasive algae *Caulerpa taxifolia* has had major impacts on ecosystems in the Mediterranean (Meinesz, 2002). The European green crab (*Carcinus maenas*) greatly reduced the abundance of susceptible native prey species (5 to 10 fold declines) in California (Grosholz et al., 2000), although this impact may be restricted to areas where large native crabs do not occur (Jensen et al., 2007). Other environmental impacts are less certain, with reported 'impacts' ranging from 'positive' (increasing diversity including of native species) to highly negative (Thomsen et al., 2009, Maggi et al., 2015, Katsanevakis et al., 2014). Historic environmental impacts may never be known because many marine pest invasions occurred more than 50 years ago (Hewitt et al., 2004), while some impacts may not have been evident because of the low statistical power of studies conducted to assess impacts (Davidson and Hewitt, 2014), creating some uncertainty about their true extent.

Impacts of invasive marine species in Australia are also uncertain and examples from overseas may not represent likely scenarios in Australia. It is, however, impossible to predict where or when the next major incursion will occur. In an attempt to address this Hayes et al. (2005) estimated the potential impact of species in terms of their perceived human health, economic and environmental impacts, using interval analysis and a web-based questionnaire sent to international and domestic experts. Assessors were asked to score likely impacts on a scale of 0 to 1 (divided into 10 intervals), and Hayes et al. (2005) used interval analysis to aggregate scores across standardised impact categories, while maintaining the assessors' uncertainty. Between one and six assessors provided scores for each species depending on the number of experts that could be found for a given species. Assessors were asked to score impacts on human health (1 category), economic impacts (5 categories), and environmental impacts (9 categories).

Their results were presented in terms of averages for each broad classification, that is human health, economic, and environmental impact, and for a combined score that adds the three together. Averaging across the broad scores could obscure the magnitude of some impacts; for example, a pest might have major impacts on aquaculture through predation, but cause no damage to marine structures, which would result in average economic impacts falling somewhere between. The mid-point for the overall score for the 10 highest ranked marine pests currently in Australia ranged from 0.61 – 1.61 (a maximum possible score is 3). Many pests had low scores for human health impacts, so these overall scores are relatively high considering the human health component was close to zero.

Despite these high scores, evidence of the impacts of some of these species in Australia is lacking or doesn't conform to the scores. Potential impacts of some species currently considered the most significant in Australia are most comprehensively summarised in National Control Plans[1]. These summaries also emphasise the uncertainty around the impacts of these species in Australia. To summarise these plans: environmental and economic impacts for the northern Pacific seastar (*Asterias amurensis*) could be significant in certain areas as the seastar is a major predator of both commercial and non-commercial shellfish species; environmental and economic impacts of the European shore crab (*Carcinus maenas*) may be high locally, under specific circumstances, but evidence for widespread impacts is lacking, particularly for mainland Australia; economic impacts of the Asian date mussel (*Musculista senhousia*) appear likely to be minor. Environmental impacts may occur, but given the species does best in eutrophic environments (lagoons and estuaries) impacts may only occur in systems already affected by other processes; environmental and economic impacts of the European fan worm (*Sabella spallanzanii*) were generally considered minor; major economic impacts of Japanese kelp (*Undaria pinnatifida*) appear unlikely. Significant impacts on the environment and public amenity may occur where Japanese kelp forms dense populations but these areas are likely to arise in response to some initial disturbance; environmental and economic impacts of the European clam (*Varicorbula gibba*) appear unlikely. These invasive species can form very dense populations but these tend to occur in environments that have already been disturbed by human activity (e.g. Clark and Johnston, 2009).

2.2 Valuing non-market impacts

Both terrestrial and marine pest incursions can result in significant non-market (environmental and social) costs. These costs are those that are not explicitly priced in the market and include, for example, environmental damage caused by both incursion and eradication, and the social costs that such damage impose on communities (McLeod, 2004, Gong et al., 2009, Lightfoot, 2010). As most of the environmental and social impacts from pest incursions are not readily expressed in dollar terms, non-market valuation techniques are usually used to evaluate these costs.

A number of recent choice modelling studies on the non-market impact of invasive species in Australia have shown that the Australian community places a significant value on reducing or preventing the impact of invasive species on the Australian environment. For example, Akter et al. (2011) found that the willingness to pay for dealing with invasive species in the natural environment in Queensland, Victoria and New South Wales was valued at $25 per household per year to protect one native species from the threat. The same study also estimated the willingness

[1] National Control Plans have been developed for the following species: *Asterias amurensis*, *Carcinus maenas*, *Musculista senhousia*, *Sabella spallanzanii*, *Undaria pinnatifida* and *Varicorbula gibba*.

to pay of $4 per household per year to eliminate weeds from one per cent of landscape and water bodies. These willingness to pay values were estimated per household per year and when multiplied by the number of households, number of years of protection as well as a number of species and area of cover protected show that the Australian public places a high value (in millions of dollars) on the protection of the natural environment. Although these values refer mostly to terrestrial biodiversity, they are an indication that the Australian community places significant values on keeping invasive species out of Australia.

There have also been a number of non-market valuation studies that estimate the value of marine related recreational activities in Australia (Hailu et al., 2011, Ezzy and Scarborough, 2011, Barrett et al., 2010, Rolfe and Windle, 2009, Gazzani and Marinova, 2007, Ernst and Young, 2006, Rolfe and Prayaga, 2006). The general consensus from these studies is that Australians place a significant value on marine recreational activities. Ernst and Young (2006) estimated a total net economic benefit of $13.4 million for New South Wales recreational anglers fishing for striped marlin in 2002-03. Bennett and Gillespie (2011) estimated an aggregate willingness to pay of $400 million for the marine protected areas in the South-west marine region, while Rolfe and Windle (2009) showed that the non-market value to the Queensland public to improve the condition of the Great Barrier Reef is valued at $11 million a year for each 1000km^2 of improvement.

While a number of studies assess the types of non-market impacts of IMS (Miehls et al., 2009, Verween et al., 2010, Schaffelke and Hewitt, 2007, Barbiero and Tuchman, 2004, Ross et al., 2002, Ross et al., 2003), there are only a limited number of studies that estimate the non-market value of the impact of IMS. For example, the study by Nunes and Markandya (2008) used non-market valuation methods to estimate the potential loss of recreational use values and marine ecosystem amenity from a marine bio-incursion in Rotterdam harbour in the Netherlands. The non-market impact on the North Holland coast from this incursion (mostly resulting from harmful algal blooms) in Rotterdam harbour and further coastal spread was estimated at €326 million for the year 2000, which equated to 0.08 per cent of the Netherlands' GDP.

Valuing non-market impacts of invasive marine species in Australia is likely to be important for understanding the true value of management activities undertaken in Australia. A key question will be: does the Australian public value keeping out as many non-indigenous marine species (NIMS) as possible, or are they concerned more (or only) with those that have (or are likely to have), significant impacts on the environment and public amenity? We discuss this more in section 5.

2.3 Port Phillip Bay case study

Port Phillip Bay has been described as "one of the most invaded marine ecosystems in the Southern Hemisphere" (Hewitt et al., 2004). Hewitt et al. (2004) report on surveys carried out in Port Phillip Bay in 1995/1996 that identified 99 introduced and 61 cryptogenic (undetermined origin) species. Introduced species in Port Phillip Bay include seven species on the former Consultative Committee on Introduced Marine Pest Emergencies (CCIMPE) trigger list and six of these have had national control plans developed, including the northern Pacific seastar (*Asterias amurensis*) (http://www.marinepests.gov.au/national-system/how-it-works/Pages/Ongoing-management-and-control.aspx.) It may be expected, therefore, that if invasive marine species are having an impact on infrastructure or operations anywhere in Australia they would have been experienced in Port Phillip Bay.

A number of stakeholders that would be aware of current impacts of IMS on infrastructure or operations in Port Phillip Bay were contacted and asked to provide information on impacts. This was not intended to be a comprehensive survey, which was outside the scope of this project, but a scoping survey designed to be escalated if impacts were identified. The case study did not consider environmental impacts. Stakeholders were asked:

"Are there any impacts from any of the invasive marine species resident in Port Phillip Bay on your infrastructure or your operations that require a response that incurs a cost?"

Impacts on infrastructure or operations

City West Water, which operates the sewage and waste water treatment plant at Altona in north-western Port Phillip Bay advised that the outfall was "teeming with life, including native marine species, but no fouling was noted. The duckbill valves are specifically designed to eliminate biofouling" (Beatrixe Fisher, City West Water, pers. comm.).

Port of Melbourne Corporation identified no significant impacts of invasive marine pests on its operations (Ulrich Storch, Port of Melbourne Corporation, pers. comm.).

Impacts on fisheries - sand flathead and shellfish aquaculture

Sand flathead (*Platycephalus bassensis*) was once both an important commercial fishery and the largest recreational fishery in Port Phillip Bay (Hirst et al., 2014). Stocks in Port Phillip Bay declined by 80–90 per cent over the period from 2000 to 2010 and the popular perception was that this was related to the massive increase in the population of the northern Pacific seastar (*A. amurensis*) during the late 1990s. However, Hirst et al. (2014) found no evidence that the decline of sand flathead stocks was related to the presence of the seastar, but attributed it to a prolonged drought.

Shellfish aquaculture has been practiced in Port Phillip Bay since about 1980 with blue mussel (*Mytilus edulis*) the main species cultivated. Aquaculture is practiced at a number of sites around Port Phillip Bay, including around Bellarine Peninsula (Clifton Springs and Grassy Point) and in eastern Port Phillip Bay (Pinnace Channel, Dromana, Mount Martha and Beaumaris) in dedicated aquaculture fisheries reserves. An aquaculture business was contacted to ask about the impacts of invasive marine species on its business. It was unable to allocate a cost for losses resulting from predation by the northern Pacific seastar, for example, and noted that the native seastar (*Coscinasterias muricata*) also predates on its stock, although not to the same extent as the northern Pacific seastar and in different circumstances if the mussel ropes touch the bottom, northern Pacific seastar larvae settle on the mussel ropes but as the mussels grow quickly juvenile seastars are unable to penetrate older mussels' shells. The business also commented on an incursion by an alien hydroid weed that "devastated the industry" in 2000, but that problem seems to have dissipated.

Conclusions of the case study

A limited number of businesses were contacted and the low response rate from those contacted and the minor impacts reported indicate that economic impacts from invasive marine species are relatively minor. Impacts on the ecosystems and environment were not part of this study, so current or past impacts on these cannot be ruled out. In overall terms, the costs to businesses to manage impacts of invasive marine species in Port Phillip Bay appear to be minor.

3 Prevention of marine pest incursions

3.1 Approaches and their costs

The costs of prevention include:

- The costs of measures taken by ship owners, recreational vessel owners and others with the aim of reducing the risks of transporting invasive marine species to Australia, both in ballast water and as biofouling, and
- The costs of measures taken by governments, such as inspections, to ensure compliance with legislation or regulations.

Here we consider one of the two key pathways implicated in the transport of invasive marine species into Australia: ballast water. As noted above (Section 1.3) biofouling prevention is only briefly considered in this section; a brief overview of the issues to be considered in calculating antifouling application and treatment costs is included in Appendix A. Other vectors, including illegal entry vessels, ghost nets[2] and marine debris, will not be considered in this report as they are impossible to systematise. There are also minor risks with the import of live organisms, for example the ornamental marine fish trade, which will also not be considered.

Preventative measures are generally not targeted at individual species but aim to be effective against all potentially invasive species. Nevertheless, some species or groups of species are more likely to be transported in one transport mechanism than the other, as discussed below.

3.1.1 Ballast water

Most ships carry ballast water for stability and trim purposes but only three types of vessel carry large volumes of ballast water when they are not carrying cargo (Snell, 2015):

- Bulk carriers, including wood chip carriers;
- Tankers; and
- Liquefied Natural Gas (LNG) and Liquefied Petroleum Gas (LPG) carriers.

Other types of ships carry ballast water that is largely retained on board and used for maintaining trim and balancing loads. Container ships are the most numerous of the latter type of ship. Container ships almost invariably carry cargo as they load and discharge at every port they visit and ballast water is largely retained on board for trim purposes, is infrequently discharged, and then only in small volumes (Verling et al., 2005). There are likely to be exceptions to these generalisations, however, as nearly 4000 vessels made more than 26,000 voyages to Australia in 2007 and some vessels, which normally retain their ballast water, may have decided to pump it all out at some point.

The role of ballast water in the introduction of invasive marine species has been well documented (Dunstan and Bax, 2008, David and Perkovic, 2004, Carlton and Geller, 1993, Carlton, 1985). The International Convention for the Control and Management of Ships' Ballast Water and Sediments (the “Ballast Water Convention”), which was adopted by the International

[2] Ghost nets are nets that have been lost overboard but continue to entangle marine species as they drift.

Maritime Organization (IMO) in 2004, but has not yet entered into force[3], requires ships to manage their ballast water and prohibits the release of unmanaged ballast water in port (IMO, 2004). There are two management options:

- Regulation D-1 – Ballast water exchange standard
- Regulation D-2 – Ballast water performance standard

The D1 system is due to be phased out and replaced with a performance standard (Regulation D2), following the coming into force of the Ballast Water Convention (IMO, 2004). Regulation D-3 sets out the approval requirements for ballast water management systems that are effectively treatment systems. A number of commentators suggest that given the fact that the Ballast Water Convention has not yet entered into force, the costs of purchasing, installing and running ballast water treatment systems and other factors such as exemptions sought for older vessels, ballast water exchange is likely to be continued to be used for some time yet (e.g. Albert et al., 2013).

Ballast water exchange

Ballast water exchange is where ballast water taken up in port is exchanged for surface waters in mid-ocean. As a preventative measure there are concerns about its effectiveness (Lodge et al., 2006, Dickman and Zhang, 1999) but in the absence of ballast water treatment it achieves risk reduction (Gray et al., 2007, Taylor et al., 2007) and is better than no management, as well as being mandatory. Ballast water exchange is also considerably cheaper than the current generation of treatment systems, see below.

There are three methods of exchanging ballast water that have been approved by the IMO (2004):

- Sequential ("empty and refill") method – a process by which a ballast tank intended for the carriage of ballast water is first emptied and then refilled with replacement ballast water to achieve at least a 95 per cent volumetric exchange.
- Flow-through method– a process by which replacement ballast water is pumped into a ballast tank intended for the carriage of ballast water, allowing water to flow through overflow or other arrangements. This requires pumping three times the ballast water volume through the ballast water tanks, although a lesser volume may be acceptable as long as 95 percent volumetric exchange can be demonstrated.
- Dilution method (also known as the Brazilian dilution method) – a process by which replacement ballast water is filled through the top of the ballast tank intended for the carriage of ballast water with simultaneous discharge from the bottom at the same flow rate and maintaining a constant level in the tank throughout the ballast exchange operation(Hay and Tanis, 1998). This system requires extra pumps and pipes to be fitted so is rarely used (Snell et al., 2015).

The sequential ("empty and refill") method is the cheapest and probably the most effective method of exchanging ballast water in that the tank is almost completely emptied before re-filling. It is not the most widely used, however, as some ballast tanks when empty introduce stresses in the ship's hull and potentially cause instability and it is therefore a safety hazard (Waite et al., 2003). The flow-through method is therefore the method that is used by most ships (Snell, 2015). This requires pumping 300 percent of the ballast water volume through the ballast water tanks.

[3] The current status of the IMO Ballast Water Management (BWM) convention can be found here:
http://www.imo.org/About/Conventions/StatusOfConventions/Documents/Summary%20of%20Status%20of%20Conventions.xls

The Australian Maritime Safety Authority (AMSA) transit dataset for 2013 was used to provide data on ship movements and estimates of ballast water capacity. Although Lloyds Maritime Intelligence Unit (LMIU) data are generally considered the most reliable, the most recent data holding is from 2007, which was considered to be out of date. According to the AMSA dataset, in 2013 a total of 11,034 voyages by ballast water carrying vessels visited Australia. Table 1 lists the numbers of ballast water carrying vessel types.

Table 1 Numbers of ballast water carrying vessel types entering Australian ports in 2013

Vessel type	No. of vessels
Bulk carrier	9134
Tanker[1]	1313
Gas carrier (LNG and LPG)[2]	434
Wood-chip carrier	146
MODU or FPSO[3]	5
Combination carrier[4]	2
TOTAL	11 034

Notes:
1. Includes oil, chemical and noxious liquid tankers.
2. Liquefied Natural Gas and Liquefied Petroleum Gas.
3. MODU = Mobile Offshore Drilling Unit. FPSO = Floating Production, Storage and Offloading unit (see glossary).
4. A vessel capable of carrying both liquid and solid bulk cargoes.

The costs of ballast water exchange were provided by Captain A.G. Snell and Associates (Snell et al., 2015). There are many factors involved in calculating the costs of ballast water exchange, including the type and size of the vessel, the ballast tank configuration and the method of exchange but the major cost is that of the fuel required to run the pumps.

The following assumptions have been made:

- The following vessels carry ballast water that must be discharged when being loaded. The ballast water on these ships must be managed, i.e. exchanged or treated.
    - Bulk carriers of all types. 30% of deadweight carried.
    - Wood chip carriers. 50% of deadweight carried.
    - Gas carriers. 30% of deadweight carried.
- The following vessels carry ballast water but are not likely to discharge it in Australian waters:
    - Container ships
    - Most other trading vessels, i.e. cargo ships
    - Other ships
- Most non-bulk carriers, e.g. container ships, do not arrive in Australia in full ballast, i.e. empty of cargo. Unless there is high value cargo to be loaded, it is not economic to send an

empty ship. Bulk carriers are different because there are few bulk materials that Australia needs and cannot supply itself.

Snell (2015) provided a ballast water exchange cost calculator with which to calculate the costs of exchange for six types of vessel:

- Handymax, Panamax, Newcastlemax and Wozmax size classes of bulk carrier (see glossary)
- Woodchip carriers
- LNG carriers

The exchange cost calculator uses 12 parameters to calculate the costs, which include the deadweight tonnage of a typical vessel of that class, the volume of ballast water carried and the volume to be exchanged, pump capacity, hours of pumping, pump efficiency and the fuel cost. Ships generally burn the cheapest type of fuel on ocean voyages and reserve a relatively small quantity of higher grade (and more expensive) fuel oil for use when manoeuvring in port. Fuel prices were accessed from http://shipandbunker.com/prices on 14 January 2015 and, selecting Singapore prices, the heaviest fuel oil (IFO380) was priced at US $274/tonne and marine gasoline oil (MGO), the lightest, was priced at US $S486/tonne. Using the ballast water exchange cost calculator, the average cost of exchanging ballast water was found to range from US $0.017 (AU $0.021 at $US0.81)/tonne for a 60 000 tonne woodchip carrier to US $0.029 (AU $0.036 at $US0.81) for a 70,000 tonne LNG carrier.

It is also assumed that vessels arriving from international ports of origin did not have to incur any costs from having to deviate or delay their voyages in order to exchange ballast water prior to arriving in Australian waters.

The total volume of ballast water exchanged by the 9134 bulk carriers coming to Australia from international ports of origin in 2013 was calculated to be 450 million m^3. The total cost of ballast water exchange by bulk carriers was calculated as $31.5 million. The average volume of ballast water exchanged per vessel in 2013 was therefore 49 195 m^3 with an average cost per vessel of $3444.

A further 1900 vessels that carry bulk ballast water, including all types of tanker, gas carriers, wood chip carriers, floating production storage and offloading units (FPSOs) and mobile offshore drilling units (MODUs), also arrived in Australia from international ports of origin. Assuming that all these vessels also exchanged their ballast water without deviation or delay, the total costs of ballast water exchange for these vessel types was $4.75 million with an average volume exchanged of 35 634 m^3. The average cost of ballast water exchange for these types of vessels was $2494. The total cost of ballast water exchange by all vessels is therefore estimated as $36.2 million.

There is considerable variation in the numbers of vessels carrying ballast water entering Australian ports. Table 2 lists a selection of ten ports, including the four ports with the greatest tonnages of visiting vessels and therefore the highest rates and therefore costs of ballast water exchange.

Table 2. A selection of ten ports listing the number of visiting bulk carriers, tankers and other vessels carrying large volumes of ballast water.

Order	Port[1]	Nos. of vessels	Total BW exchanged (million m^3)	Cost BW Exchange[2] (million $)	Cost BW Treatment[3] (million $)
1	Port Hedland	1731	288.58	$ 8.84	$13.75
2	Dampier	1211	194.62	$ 5.96	$9.27
3	Newcastle	1561	161.41	$ 4.94	$7.69
4	Hay Point	861	102.13	$ 3.13	$ 4.87
11	Port Kembla	215	17.93	$0.55	$0.85
14 18	Melbourne & Geelong	452	26.20	$0.80	$1.25
15	Darwin	130	14.04	$0.43	$0.67
26	Port Botany	82	3.06	$0.09	$0.15
47	Hobart	18	0.59	$0.02	$0.03

Note.
1. Ports are ordered by the total volume of ballast water exchanged, which is derived from vessel deadweight tonnage.
2. Costs of ballast water exchange were derived using the method described in Snell (2015).
3. Costs of ballast water treatment were derived from King et al. (2012). It is assumed that ship owners would opt for the lowest price solution so a cost of $0.14/m^3 of ballast water treated was used.

Source: AMSA transit data for 2013.

Ballast water treatment

The Ballast Water Convention accepts ballast water treatment systems as substitutes for ballast water exchange and provides performance standards for these systems. As of May 2014, 42 ballast water treatment systems (BWTS) have received approval certification from their respective administrations (IMO, 2014). The capital costs of purchasing a treatment system with a similar pump capacity to that of ballast water exchange, i.e. approximately 2000 m^3/hour, range from US $650 000 to nearly US $3 million (Zagdan, 2014).

Lloyd's Register (2010) reviewed current BWTS and sixteen manufacturers or suppliers provided information on capital expenditure and operating costs[4]. From this information, Lloyd's Register (2010) found that for systems that could treat 200 m^3/hour, mean capital costs were $382 667 and for systems that could treat 2000 m^3/hour, mean capital costs were $1 038 667 000. Operating costs for treating 1000 m^3 of ballast water ranged from $0, where waste heat is used, to $173. The mean operating cost was $0.04/m^3.

[4] Prices in $AU converted from $US at 0.75, FY 2008-09 prices.

King et al.(2012) also reviewed BWTS from a US perspective. They found that the average purchase price[5] of four types[6] of BWTS varied between $621 359 for a Filtration & Chemical system to $906 149 for a Filtration & UV light system. They noted that discounts of about 5–10 per cent were applied to bulk purchases. Installation costs were listed ranging from $17 476 to $48 544 for a new construction general cargo ship in a non-US yard to $93 204 to $191 262 for a retrofit in service of a non-US Very Large Crude Carrier (VLCC). Annual operating costs were the same across all vessel types for each BWTS except filtration/chemical. Annual costs were $10 680 for filtration/UV, $8 738 for deoxygenation/cavitation and $16 505 for electrolysis/electrochlorination. Annual costs for filtration/chemical systems ranged from $30 097 for general cargo ships to $287 379 for VLCCs. Finally, the authors calculated life cycle costs per metric tonne based on a 25 year life cycle. For a Capesize[7] bulk carrier, the size class that accounts for more than 50% of bulk carriers visiting Australia, costs ranged from $0.14 to $0.37 across the various systems. Although these costs are approximately 60 per cent higher than for ballast water exchange, it is anticipated that they will fall as the technology develops, becomes more efficient and the manufacturing costs decline.

The current Australian Ballast Water Management Requirements (Version 5) (http://www.agriculture.gov.au/biosecurity/avm/vessels/quarantine-concerns/ballast/australian-ballast-water-management-requirements) do not mention whether ballast water that has been treated by an IMO approved ballast water treatment system is acceptable for discharge into an Australian port. Approval of ballast water that has been treated by an approved ballast water treatment system is, however, included in the Biosecurity Act 2015 that will commence on 16 June 2016 (http://www.agriculture.gov.au/biosecurity/legislation/new-biosecurity-legislation). It is anticipated, therefore, that ballast water that has been treated by an approved ballast water treatment system will be accepted in Australia in due course.

3.1.2. Biofouling

The costs of biofouling treatment have been excluded from this study on the basis that the application of antifouling coatings and the treatment of internal water systems are measures that have been adopted by the shipping industry for reasons of fuel efficiency and vessel safety rather than preventing the spread of marine pests. This conclusion notwithstanding, the IMO and shipping industry generally recognise that biofouling on shipping is a potential vector for the introduction of IMS. As a consequence the IMO has released “Guidelines for the control and management of ships' biofouling to minimise the transfer of invasive aquatic species” to provide a globally consistent approach to the management of biofouling (IMO, 2011). A brief discussion of some of the issues involved in the prevention of biofouling and the costs involved is included in Appendix A.

3.1.3 Compliance

The costs of compliance, i.e. verification by the regulator that ships have exchanged their ballast water relate to the costs of ship inspections and clearance, these costs being recovered from the vessel operators. There are currently no inspections relating to biofouling occurring at the Commonwealth level, as there is no legislative requirement for biofouling management. Ballast water compliance inspections are carried out by officers of the Department of Agriculture.

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[5] Prices in $AU converted from $US at 1.03, FY 2011-12 prices.

[6] Filtration & UV light, Filtration & Chemical, Deoxygenation & Cavitation and Electrolysis & Electrochlorination. A fifth type: Filtration, Deoxygenation & Cavitation was not costed.

[7] Capesize bulk carriers range in size from about 80,000 to 300,000 deadweight tonnes.

Inspections

Australian Government

Vessels coming into Australia are required to complete a Quarantine Pre-Arrival Report[8] that includes two questions about ballast water:

- Do you intend to discharge ballast in Australian waters? Yes or No
- Have you maintained accurate records of ballast exchanges? Yes or No

The pre-arrival report is subject to documentary clearance, which is carried out at the Maritime National Co-ordination Centre in Adelaide. The cost of documentary clearance, which is currently $850, is recovered from vessels. Vessels making maiden voyages to Australia are subject to a two hour inspection regime with 20 minutes devoted to questions of ballast water management. The cost of the inspection is recovered from the vessel being inspected. The current cost of an inspection for a vessel greater than 25m length is $1210 and for vessels less than 25m is $380. For a commercial vessel, i.e. greater than 25m length, the cost of the 20 minutes devoted to ballast water is therefore 20/120 x $1210 = $201.67.

The Department of Agriculture maintains a compliance regime whereby those vessels visiting Australia regularly are subject to documentary clearance only but with random inspections at a rate of about one in four visits. There are a number of exemptions to this, however, including cruise ships and livestock carriers, which require a physical inspection every visit. The cost of documentary clearance is $850 for which one hour is allocated. It is estimated that 10 minutes is allocated to checking responses to the questions on ballast water and the ballast water summary sheets and that therefore the cost of compliance against this measure is $141.67 per vessel. Assuming that one out of every four vessels is inspected and the other three are subject to documentary clearance only, the average cost of compliance, i.e. inspection or documentary clearance is $201.67 \times 0.25 + 141.67 \times 0.75 = \$156.67$.

For example, in 2007, 829 vessels visited Dampier from overseas ports. The total cost of compliance allocated to that port is $129 879. The estimated cost of ballast water exchange by vessels entering Dampier is $6m so the costs of compliance are approximately 2.2 per cent of that cost. We use this proportional value together with the estimate of total cost of ballast exchange from section 3.1.1 to derive an Australia-wide estimate for compliance monitoring of approximately $36.2\text{m} \times 0.022 \cong \$800\,000$ a year.

The Department of Agriculture is developing the Maritime Arrival Reporting System (MARS), which will streamline documentary clearance and ballast water reporting. It is not yet possible to determine what the costs of preventative measures relating to ballast water through this system will be.

3.2 Incursion rates of marine pests with and without prevention

For IMS to successfully invade a new area they must first be transported to the new location and then become established. Establishment involves being released into the new environment where environmental, biological and physical conditions are appropriate and then for a self sustaining population to develop (e.g. Wonham et al., 2013). The entry and establishment of IMS has been common in the past in some areas; for example, Hewitt et al. (2004) identified 99

[8] Full information about the biosecurity requirements for vessels is on the Department of Agriculture website: http://www.agriculture.gov.au/biosecurity/avm/vessels/

introduced and 61 cryptogenic species in Port Phillip Bay representing more than 13 per cent of the recorded species of the bay, while Ruiz et al. (2000) identified 298 non-indigenous species of invertebrates and algae that have established in marine and estuarine waters in North America. Worldwide, Hewitt and Campbell (2010) identified 1781 marine and estuarine species that have been introduced into bioregions outside their native range. There is also a suggestion that the rate of IMS introductions is increasing (Ruiz et al., 2000, Cohen and Carlton, 1998), although ballast water management has become more widespread since these studies were done.

Understanding establishment rates of IMS, particularly in a quantitative sense, is challenging, and our understanding of current patterns is affected because sampling effort has not been evenly spread over time and the marine environment is not as easily observable as terrestrial environments (Ruiz et al., 2000). Nonetheless introductions through ballast water and biofouling are believed to be one of the major drivers (Hewitt and Campbell, 2010, Ruiz et al., 1997, Mineur et al., 2008, Lewis et al., 2003). However, while invasion success is clearly linked to the rate of arrivals, the actual relationship between arrival rate and invasion success has not been quantified (Clark and Johnston, 2009, Wonham et al., 2013). Biological processes (known as stochastic extinction and Allee effects) operating on small populations may mean threshold levels of introduction must be exceeded before pests can successfully establish. In addition, local environments are also likely to influence establishment rates. For example, there is increasing evidence that human disturbance is a factor in successful establishment of invasive marine species (e.g. Clark and Johnston, 2009).

These complications make it challenging to estimate establishment rates of NIMS, let alone understand how preventative measures change these rates. Despite these challenges, some attempts have been made to estimate establishment rates. Cohen and Carlton (1998) estimate that between 1961 – 1995 one new non-native species established in the San Francisco Bay and Delta system every 14 weeks (almost 4 per year). Hewitt (2011) estimated an establishment rate for NIMS in Australia since 1960 of about 4.9 a year; a rate similar to the estimate of Cohen and Carlton. (While one estimate is for a large bay and the other for an entire country, we do not know the relative vector traffic to be able to comment in more detail on the comparability of these rates). Hewitt further broke this down, estimating that between 3.4 and 4.1 establishments of NIMS a year could be the result of biofouling, with the balance (0.8 – 1.5 a year) largely the result of ballast water. Not all NIMS are likely to have major impacts, so Hewitt estimated the proportion of NIMS establishing that are likely to have impacts as those on the Australian Species of Concern (SOC) list, divided by an estimate of the total global pool of NIMS established elsewhere but not currently in Australia. He quoted a biofouling species of concern (SOC) list of 58, and an estimated total global pool of 1070 NIMS not present in Australia , and estimated the probability of at least one of the arrivals in a year being a SOC (which is 1 – probability that none are a SOC) as (Hewitt, 2011):

$$1-\left(1-\frac{SOC}{Tot.Pool}\right)^{\left(\frac{NIMS}{yr}\right)} = 1-\left(1-\frac{58}{1070}\right)^{4.05} = 0.2$$

Alternatively, the arrival rate of biofouling SOCs could be expressed as:

$4.1 \times 58/1070 \cong 0.2$ a year or about one every five years. Assuming similar proportions of SOC for ballast water, then a high rate for ballast water pests could be expressed as $1.5 \times 58/1070 \cong 0.08$ a year. Clearly from this formula the rate is directly proportional to the size of the SOC list, which is currently closer to 40 – this would bring the respective rates down to 0.15 a year (biofouling) and 0.06 a year (ballast water).

However, Lewis (2011) questions this rate because he believed the methodology used by Hewitt (2011) was inappropriate. Lewis suggests that the time of detection (which Hewitt equated with establishment time) was likely to be strongly influenced by changes in surveillance effort and technology employed when looking for NIMS and hence not reflective of establishment time. This is a problem confronting all current estimates of incursion rate, and if establishment occurred well before first detection the actual establishment rate since 1960 would be lower.

Given the uncertainty about incursion rates, in our subsequent analyses we present a range of rates from 0.04 a year to 1.5 a year, with the range for a particular analysis also varying depending on whether the analysis considers all NIMS or only high impact NIMS.

Effectiveness of ballast exchange

It is clear that ballast exchange can be highly effective at reducing the concentration of planktonic organisms from a source location, with a comprehensive study of the literature plus organisational research (Ruiz and Reid, 2007) indicating an 80-95 per cent reduction in the concentration of organisms, using either the sequential or flow-through methods. However, the overall impact on establishment rates is not clear because of: (i) uncertainty about the relationship between propagule pressure and establishment rate; and (ii) the presence of organisms in bottom sediments where the dilution effect of ballast exchange is less, for example these organisms could become resuspended during a subsequent ballast water release. We assume either an 80 per cent or 90 percent reduction in rate of establishment to represent the effectiveness of ballast exchange in subsequent analyses.

Effectiveness of Biofouling management

The estimates for establishment rates of non-native species from both Cohen and Carlton (1998) and Hewitt (2011) are for periods where there was extensive use of anti-fouling treatments on vessels for the primary reasons of protecting the underlying metal and reducing fuel consumption (Section 2). Rates prior to the adoption of this technology are unknown. Nor is it known why incursion rates were still relatively high during this period despite the use of this technology. Ballast water introduction is one reason, but Hewitt (2011) suggests that a much higher proportion of species introduced during this period were more likely to have been introduced through biofouling. Possibilities include biofouling in niche areas on vessels, poor biofouling treatments on some vessels, despite the benefits to owners, and biofouling of vessels that may not benefit as much from antifouling treatments, for example slow moving vessels such as barges. Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species were introduced by the IMO in 2011(IMO, 2011). The Department of Agriculture is currently considering policy options to implement the IMO guidelines for vessels entering Australian waters, and future work will be required to determine the costs of these approaches and how effective they are at reducing the biosecurity risk posed by biofouling from its current level.

4 Eradication of marine pest incursions

Eradicating marine pest incursions is extremely difficult. This was summed up by Wittenberg and Cock (2001) who stated: “It should be stressed that eradication in marine waters is only possible in extremely unusual circumstances that allow treatment of an effectively isolated population in a relatively contained area... In the great majority of cases, eradication has been and will remain impossible”.

Some of the successful and unsuccessful attempts at eradication are summarised in Williams and Grosholz (2008). Some of the notable successes include the successful eradication of black-striped mussel (*Mytilopsis sallei*) from enclosed marinas in Darwin at a cost of A$2.2 million plus unaccounted personnel costs, mainly through chemical treatment of the enclosed waters (Bax et al., 2002); the successful eradication of a sabellid polychaete (*Terebrasabella heterouncinata*) from a small area in California by removing hosts as well as infected material (Culver and Kuris, 2000, Moore et al., 2013); the successful eradication of the ‘invasive algae’ *Caulerpa taxifolia* from two areas in California (one with non-contiguous coverage totalling approximately 1 ha and one with non-contiguous coverage totalling approximately 1000 m^2), using covering and chemical treatment, for a reported US$6 million (Anderson, 2005, Williams and Grosholz, 2008); the successful eradication of *Undaria pinnatifida* from a sunken trawler in the Chatham Islands, direct treatment of the seaweed reportedly cost NZ$423,500, plus NZ$2.5 million spent unsuccessfully trying to salvage the vessel; and the successful eradication of the brown mussel *Perna perna* from a small area established as a result of de-fouling of a barge, by dredging to densities below which the mussel could not recover from Allee effects (Hopkins et al., 2011). The cost of the latter operation was not published.

In many cases these successes were for incursions that covered isolated populations in relatively contained areas, as suggested by Wittenberg and Cock (2001), but another feature was often the ability to successfully deal with the governance issues required to mount an eradication (Anderson, 2005, Wotton et al., 2004, Coutts and Dodgshun, 2007). However, there are also examples of failed eradication attempts (Williams and Grosholz, 2008) and possibly many cases where it was decided eradication would not be attempted or failures have not been reported. This sort of bias is known to occur for terrestrial eradication datasets (Tobin et al., 2014).

Recently an analysis has been conducted on the probability of eradication of aquatic non-indigenous species given characteristics of the incursion, based on a dataset including 143 case studies where 52 per cent resulted in successful eradication (Drolet et al., 2014). To give an insight into how the case studies were constructed (data are available online at http://datadryad.org/resource/doi:10.5061/dryad.1rh77/1), the black-striped mussel eradication in Darwin was considered as 3 separate successful eradications (3 case studies), based on eradication from the 3 different areas it was found in Darwin (Ferguson, 2000), rather than a single eradication. Drolet et al. (2014) comment that this dataset is likely to be biased towards successes, as discussed above. This type of bias would overestimate the probabilities of success for a given level of a particular explanatory variable, but may still provide reasonable insights into relative probabilities.

Unfortunately Drolet et al. (2014) provide no detail on the resultant statistical model from their analysis, other than to say it provided a good fit to the data. Some information can be gleaned from a subsequent paper (Drolet et al., 2015); when considering characteristics of incursions in isolation, that which most explained the likelihood of successful eradication was the spatial extent of the incursion, with smaller areas more likely to be successfully eradicated than larger areas. The next ranked characteristic was 'method employed' for eradication, but this received minimal support relative to the spatial extent of incursion.

Because of uncertainty about eradicating invasive marine species, Crombie at al. (2008) developed a tool to predict the costs of eradication based on 22 expert assessments of seven hypothetical species, each of which was given three incursion scenarios, that is 21 incursion scenarios in total. The scenarios were designed to cover a range of attributes thought to influence the cost (and likelihood of success) of any eradication program. Experts were asked to estimate the likelihood of success of eradication given eradication budgets ranging from $25,000 – $5,000,000. The responses were then analysed to develop a model to predict expenditure required for a 95 per cent chance of eradication based on the underlying attributes of an incursion. From the final model, which best explained cost required for eradication, the prediction of median cost (with approximate upper and lower 95% confidence intervals) is based on four factors, with differing numbers of levels:

- Area of incursion: Very small (<100 m^2), Small (100 – 1000 m^2), Medium (1000 m^2 – 10 000 m^2 (1 ha)) and Large (> 1 ha)
- Level of exchange of water: Minimal, Low and High
- Depth of available habitat: <15m and >15m
- Pest habit: Grouped and Solitary

From the model there are 48 combinations of these four variables (4 x 3 x 2 x 2), and while estimates of cost are provided for each, it should be noted that in only 14 of the 48 combinations is the median cost estimate less than the maximum budget presented to the experts for their assessment ($5 million). When the area of infestation on discovery is greater than 1 ha ('Large'), all estimates are much greater than $5 million and only 3 combinations with an estimated area of infestation of 1000 m^2 – 1 ha ('Medium') predict a budget <$5 million.

Summerson et al. (2013) applied a 'bottom-up' approach to estimating the costs of an eradication response to hypothetical incursions of black-striped mussel (*Mytilopsis sallei*) in four Australian ports: Cairns, Darwin, Dampier and Fremantle. The costs were based on the time and quantities of materials estimated to be required to respond to a range of scenarios in each port, taking the experiences from a range of eradication attempts of a number of species in Australia and New Zealand. While the aim was to estimate costs required for a high probability of eradication, Summerson et al. (2013) acknowledged there was significant uncertainty about this probability, and hence presented subsequent results comparing eradication with containment based on the full range of probabilities for success of eradication. In general estimated costs exceeded $5 million, with the exception of the scenario most similar to the historical incursion of black-striped mussel in Darwin (Table 3).

Table 3. Estimated costs of eradication of a hypothetical incursion of black-striped mussel (*Mytilopsis sallei*) in three Australian ports (Summerson et al., 2013).

Location	Facility type	Estimated eradication cost ($ million)
Darwin	Closed marina	0.2 [1]
Darwin	Commercial wharf	42.5
Darwin	Commercial wharf (cruise ships)	9.8
Cairns	Large open water marina	118.3
Cairns	Small open water marina	7.4
Dampier	Commercial wharf	55

[1]This cost was < 10% of the cost of the actual incursion in Darwin (Bax et al., 2002), because the cost of the original response was inflated by the development of eradication techniques.

Taken together, these studies are consistent with the idea that eradication may be feasible for some marine pest incursions if they cover very small areas when discovered (something less than 1ha) and can be contained for treatment. However, if incursions cover large areas then successful eradication is unlikely, and at best, probably prohibitively expensive. To account for this we assume eradication attempts cost on average either $5 million or $20 million, and have a probability of being successful of either 0.05 (a 5 per cent chance) or 0.2 (a 20 per cent chance).

Monitoring

Monitoring is a separate component of the National System (http://www.marinepests.gov.au/national-system/how-it-works/Pages/Monitoring.aspx) and substantial costs are required to implement it. A National Monitoring Network of 18 locations has been established with guidelines and a manual prepared to aid monitoring design, sampling procedures, etc. One aim of monitoring is to allow early detection of incursions, which theoretically should increase the likelihood of eradication success. The design of the current system is unlikely to achieve this early warning, and as Arthur et al. note (2015), there is a question of cost-effectiveness, as currently there is no evidence that an early-warning system increases the chance of finding a marine pest at an earlier stage of invasion that makes a difference, relative to having no system. The current monitoring system is currently under review and as a consequence the costs of the system have not been explicitly included in this study.

5 Cost comparisons

In this section we use the information from the preceding sections to provide economic assessments that compare the costs of three different approaches to invasive marine species: having a prevention system; having an eradication system; or 'living with them'. As discussed previously, prevention systems are aimed at preventing pests from arriving in Australia at rates that would allow them to establish if no other management actions were taken. Eradication systems are aimed at detecting pests that have become established and then subjecting them to management actions that will remove the established population. 'Living with IMS' means accepting the presence of an IMS in the knowledge that Australia will be subjected to the impacts of the pest. These impacts include any loss of production from marine based industries (for example aquaculture); non-market impacts (for example impacts on the environment, social amenity); plus any management costs directed at controlling the pest. For each local entity where the pest will have an impact, managers will have to decide how much to invest in management vs. the level of residual impact on production/non-market impact they are prepared to accept. The sum of management costs plus the residual impacts determine the cost of living with the pest.

The cost of living with each IMS will develop over time as the pest becomes more established and expands its range, so to compare this cost with the cost of prevention or eradication we express the cost in present value terms. We represent the average present value cost of each IMS as $Cost_{LWI(pv)}$. As indicated throughout the report, the comparison of costs is complicated by significant uncertainties about the costs and effectiveness of each approach to IMS. To deal with these uncertainties a range of values was tested in this study. These values, shown in Table 4, were chosen based on the preceding sections and to span values where the preferred management approach changes.

Table 4 Parameters and their values used in cost comparisons.

Parameter	Value
Ballast water incursion rate in absence of prevention inc_o	0.04, 0.08 and 0.25
Cost of ballast water prevention per year ($Cost_p$)	\$36.2 million and \$0.8 million
Cost of eradication per incursion ($Cost_e$)	\$5 million and \$20 million
Probability of eradication failure ($\Pr(erad_f)$)	0.95 and 0.80
Effect of prevention on incursion rate (Eff_p)	0.80 and 0.90 (proportional reduction in rate)
Average cost of living with each high impact IMS ($Cost_{LWI(pv)}$)	A range from \$5 million to \$1 billion

The impacts of IMS on the Australian environment are generally unknown (Section 2), making it difficult to estimate the value of living with the impacts in monetary terms. Many previous non-market valuation studies show that the Australian public generally holds positive willingness to pay to protect the natural environment (Section 2.2). These values are generally high and vary from tens to hundreds of millions of dollars for specific case studies, which are often conducted at regional scales. Hence, scaling up to larger spatial scales assuming the same values per household could produce much larger values because of the higher number of households affected. To account for the fact that the non-market value could be high, but is also uncertain, we test values of $Cost_{LWI(pv)}$ from $5 million to $1 billion.

The incursion rate of 0.08 was used as a possible rate of incursion of ballast water pests (section 3.2), but we also consider a lower incursion rate of 0.04, and a higher incursion rate of 0.25, which is close to Hewitt's (2011) estimated rate for high impact bio-fouling IMS (section 3.2).

Two different costs of prevention were used in the calculations. The higher cost of prevention is based on ballast water exchange and includes both costs to business and the administrative cost to the government of the existing ballast water management of international vessels coming to Australia (section 3.1). The lower cost only includes the administrative cost to the Australian government, based on the vessel-specific costs applying in the future because vessels are required to comply with the International Convention for the Control and Management of Ships' Ballast Water and Sediments (IMO, 2004) rather than an Australian system. Relatively high proportional reductions in incursion rate because of ballast water management have been assumed based on direct evidence of the effect of ballast water exchange on concentrations of potential invasive marine species in ballast tanks (Taylor et al., 2007)(section 3.2), but it is acknowledged that there may not be a direct translation between the concentration and incursion rates. There is currently no system in place to manage biofouling of vessels arriving in Australia other than the IMO guidelines (IMO, 2011), but systems are being considered. If the costs and effectiveness of the proposed systems fall within the range of values considered in this study, then the analyses presented here could be used to provide an economic assessment of that approach.

Costs of eradication will vary depending on each particular incursion, but on average eradication campaigns are likely to be expensive and have a low probability of success. Except in very constrained circumstances $5 million was not considered enough to provide a high probability of eradicating a marine pest incursion (section 4). Hence, we show results for 2 budgets, $5 million and $20 million, each with 2 probabilities of eradication being successful (0.05 and 0.2).

The three approaches considered are not the only approaches for dealing with IMS. For example, another approach would be to attempt to contain the pest to the point of incursion. Containment is a specific form of control where an attempt is made to restrict an invasive species to a limited area (Wittenberg and Cock, 2001). There are two forms of containment in the marine environment: the first is to attempt to contain the species to its point of incursion; the second is to control its spread by managing vector movements. The former may be achieved by reducing the population size to induce an Allee effect (Drake and Lodge, 2006); the second by vessel inspections, hull cleaning and ballast water management. There are many factors that would influence the success or otherwise of these approaches, including the life history of the species, its population size when discovered and how amenable it is to these methods. The costs of these approaches are another critical factor. In general terms marine species would be difficult to contain because of the broadcast spawning method of reproduction, which is likely to disperse juveniles over a large area (Shanks, 2009). Containment, if successful, would slow the spread to other ports, but not eliminate it, because there is also a high probability that containment will fail. The total cost of a containment approach could also be considered, but costing containment is extremely complicated, particularly for a system that will only apply containment measures to infested ports (the currently proposed approach for Australia's domestic ballast water system).

The complication arises because as each new port becomes infested it enters the containment system, unless it already has other pests and hence is already part of the containment system.

An alternative would be to consider a system where, for example, domestic ballast water treatment is mandatory. This would essentially have a set cost to apply rather than a cost that changed as more ports became infested. However, the difficulty with estimating the overall cost of this approach is that it would affect the rate of spread of pests and hence the rate at which their impacts and the associated cost of those impacts would develop. There is too much uncertainty to provide a reasonable assessment of the costs of containment, so it is not considered further in this study.

5.1 Overall costs of different systems (method 1)

If no prevention or eradication systems are in place, then the costs of living with all pests that would establish in Australia are incurred. Considering a particular year, if the incursion rate in the absence of preventative measures is inc_0, then the total present value cost of incursions from that year would be:

$inc_0 \times Cost_{LWI(pv)}$.

So, with a ballast water IMS incursion rate of 0.08 per year (section 3.2) and an assumed average present value cost of living with an IMS of \$1000 million (i.e. \$1 billion), the total present value cost for a year would be:

$0.08 \times \$1\ billion = \$80\ million$.

Prevention

Preventative measures are aimed at preventing the successful introduction of marine pests including those that will have large impacts, but they would not completely remove the possibility of introduction. The expected incursion rate in the presence of preventative measures (inc_p) will be based on the effectiveness of the prevention measures, i.e. $inc_p = inc_o \times (1 - Eff_p)$. The cost of preventative measures is $Cost_p$ per year. So, if a prevention system was in place, the yearly 'cost' of that approach would be the cost spent on prevention plus the present value cost of incursions not prevented in that year,

$Cost_p + inc_p \times Cost_{LWI(pv)}$.

For a cost of prevention of \$37 million and an effectiveness of 0.9, the total cost of the system would be:

$\$37\ million + 0.008 \times \$1\ billion = \$45\ million$.

Dividing the cost of the prevention approach by the cost of living with all IMS incursions in the absence of a prevention system gives the relative cost of the prevention approach to living with all IMS. For these parameter values, the prevention approach 'costs' about 56 per cent ($45/80 \times 100$) or 0.56 of the cost of living with all IMS and is clearly favoured.

The cost analysis assumes the system is in a steady state, with incursion rates neither increasing nor declining over time. Incursion rates might decline over time as more pests become established and the pool of potential invaders declines. If this were the case more sophisticated analyses that account for the system switching from prevention to eradication would be appropriate. However, reductions in the rate are likely to be trivial over the time scales relevant to this analysis. Indeed, an increase in incursion rate might be more likely as vessel traffic increases.

Technical note: The calculations above focus on 1 year of prevention and are relevant if the prevention system is to continue; if the cost of the prevention system is incurred every year, then the present value cost is:

$$\sum_{t=0}^{T}(Cost_p + inc_p \times Cost_{LWI(pv)})/(1+\delta)^t =$$

$$(Cost_p + inc_p \times Cost_{LWI(pv)}) \times \sum_{t=0}^{T} 1/(1+\delta)^t$$

where δ is the discount rate. Similarly, the cost of living with all incursions would be incurred every year, so the present value cost is:

$$\sum_{t=0}^{T}(inc_0 \times Cost_{LWI(pv)})\,/(1+\delta)^t =$$

$$(inc_0 \times Cost_{LWI(pv)}) \times \sum_{t=0}^{T} 1/(1+\delta)^t$$

When one is divided by the other the term $\sum_{t=0}^{T} 1/(1+\delta)^t$ cancels out and we are left with the ratio calculated above.

Eradication

If a marine pest arrives, a decision can be made to eradicate it; but there is a probability that the eradication will fail, $Pr(erad_f)$. The cost of eradication is $Cost_e$ per incursion. Clearly these values will vary depending on the pest and how early it is found post entry, but here we use average values.

If no prevention system was in place, but eradication was attempted for every incursion that was likely to have a high impact, then the total cost of the eradication approach would be the cost of the eradication attempts plus the present value cost of incursions for every eradication attempt that failed,

$inc_0 \times Cost_e + inc_0 \times \Pr(erad_f) \times Cost_{LWI(pv)}$.If eradication attempts on average cost \$20m and have a 20 per cent chance of success (an 80 per cent chance of failure), then the total cost of the approach would be:

$0.08 \times \$20\ million + \ 0.08 \times 0.8 \times \$1\ billion = \$65.6\ million.$

For these parameter values, the eradication approach 'costs' about 82 per cent ($65.6/80 \times 100$) or 0.82 of the cost of living with all IMS and is more costly than a prevention approach.

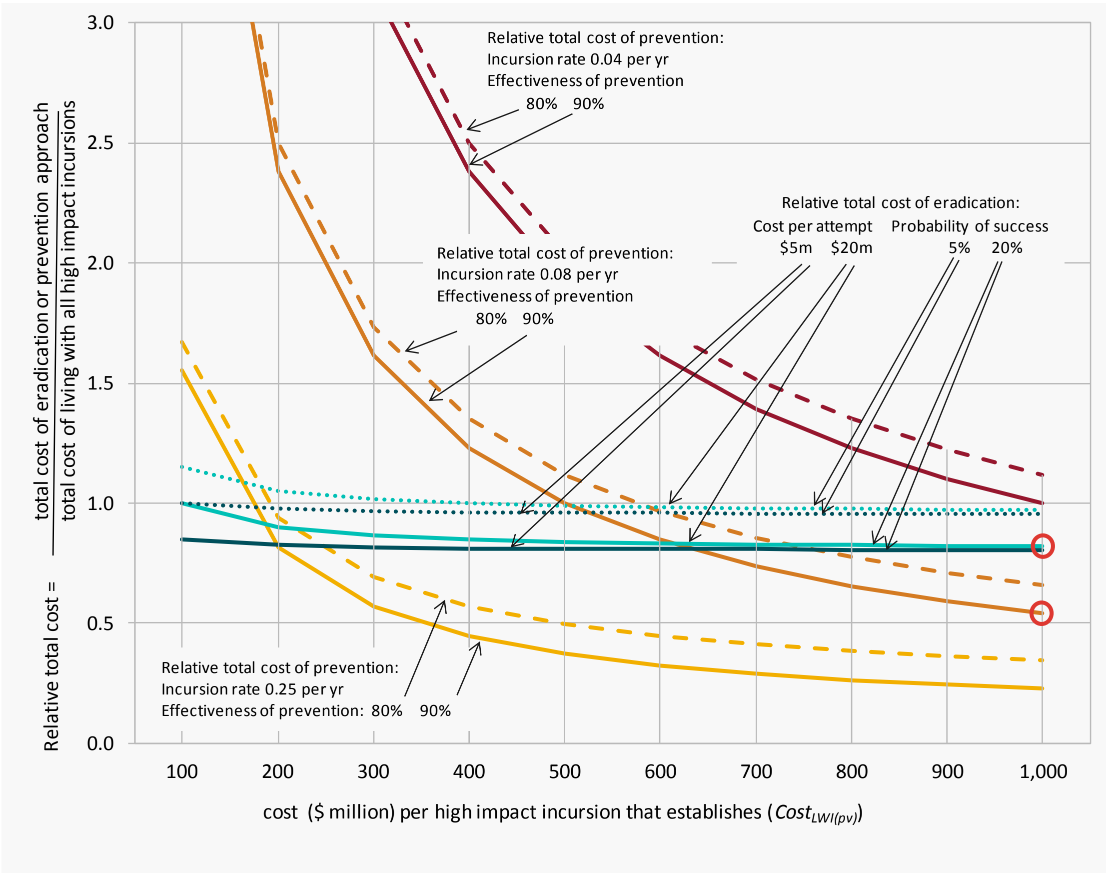
Of course, another option is to have both a prevention and eradication approach. If a prevention system was in place and eradication was attempted for every incursion, then the total cost of the approach would be the cost of prevention plus the cost of eradication attempts on incursions not prevented plus the present value cost of incursions for every eradication attempt that failed,

$$Cost_p + inc_p \times Cost_e + \; inc_p \times \Pr(erad_f) \times \; Cost_{LWI(pv)}$$

For the example parameter values considered above, the total cost is:$37 $million$ + 0.008 × $20 $million + \; 0.008 \times 0.8 \times \; 1 \; billion = \$43.6 \; million$.

This is only marginally less costly than a prevention only system and the joint approach is not considered further for total cost comparisons.

Figure 2 Total cost of a prevention or eradication approach relative to the total cost of living with all high impact incursions for different values of $Cost_{LWI(pv)}$ given a yearly cost of prevention ($Cost_p$) of $37 million.



Notes. The total cost of the eradication or prevention approach includes the cost of living with the high impact IMS that either eradication or prevention fail to prevent establishing. The cost of prevention ($Cost_p$) used to generate this figure was $37million per year. A value of '1' corresponds to when the total costs of the eradication or prevention approach equals the total cost of living with all high impact incursions. Different lines represent different values for key parameters. The red circles indicate the worked examples from the text. Incursion rate refers to the incursion rate in the absence of prevention. An effectiveness of prevention of 80 per cent means the prevention approach reduces the incursion rate by 80 per cent.

Costs of a prevention system of $37 million per year

Figure 2 shows results covering the range of parameter values shown in Table 4, but with the cost of a prevention system ($Cost_p$) fixed at $37 million per year. The two values that correspond to the relative costs calculated above are circled. The figure shows that a prevention approach that results in a 90 per cent or more reduction in incursion rate is a better option than living with the impact of all IMS if the present value of expected impacts is $1 billion per incursion and the incursion rate is 0.04 per year or higher. However, with an incursion rate of 0.25 per year, a prevention approach that results in an 80 per cent or more reduction in incursion rate is a better option than living with the impact of all IMS if the present value of expected impacts is higher than about $200 million per incursion. For the incursion rate of 0.08 per year, a prevention approach that results in a 90% or more reduction in incursion rate costs about the same as living with the impact of all IMS if the present value of the latter is about $500 million per incursion.

For the range of values considered, the relative total cost with an eradication approach was most influenced by the assumed effectiveness of the eradication attempt (Figure 2). With a 5% chance of success an eradication approach cost about the same as living with the impact of all IMS. Larger improvements relative to living with the impact of all IMS were seen with a 20% chance of success. If the probability of eradication is at least 20% and the cost no more than $20 million, then eradication is a better option than living with the impact of all IMS if the present value of expected impacts is higher than $100 million per incursion. Whether an eradication approach was favoured over a prevention approach or vice versa depended on the incursion rate and the average expected impact of an incursion, with higher incursion rates and higher expected impacts favouring prevention.

Costs of a prevention system of $0.8 million per year

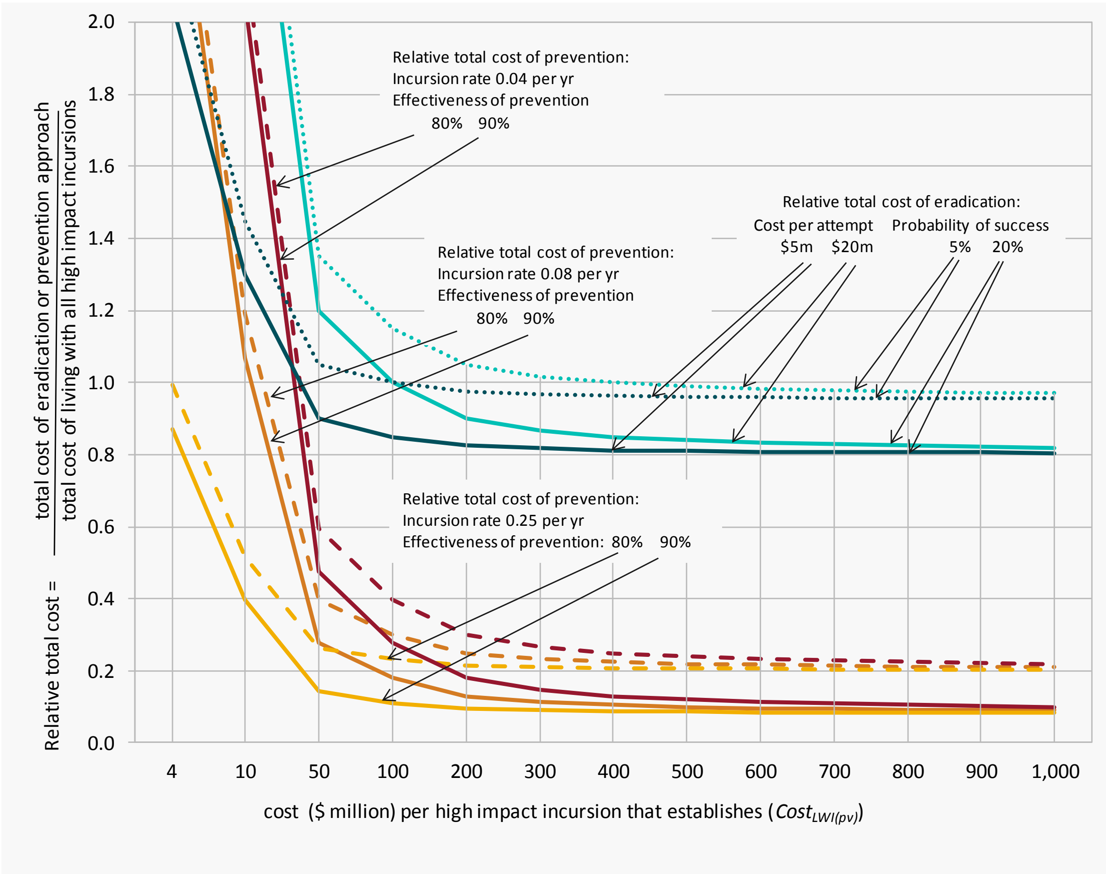
Figure 3 shows results covering the range of parameter values shown in Table 4, but with the cost of a prevention system ($Cost_p$) fixed at $0.8 million per year. Considering only the administrative/regulatory costs of prevention has a large effect on the results. In this case a prevention approach that results in an 80% or more reduction in incursion rate is a better option than living with the impact of all IMS if the present value of expected impacts is higher than $50 million per incursion and the incursion rate is 0.04 per year or higher. With an incursion rate higher than 0.25 per year a prevention approach that results in an 80% or more reduction in incursion rate is a better option than living with the impact of all IMS if the present value of expected impacts is higher than $4 million per incursion.

For all three incursion rates included in our scenarios prevention led to a substantial reduction in overall cost relative to living with the impact of all IMS for expected impacts of $50 million per incursion or higher. For expected impacts of $200 million per incursion or higher, overall costs of prevention were about 10 – 30 per cent of the overall cost of living with the impact of all IMS. Changing the cost of prevention does not impact on eradication, so under these scenarios prevention tends to have a much greater effect on overall costs than eradication does.

Summary

Prevention is more clearly favoured over living with the impact of all pests when only the cost of administering a prevention system is considered. When the cost on industry is also taken into account, the results are more clearly dependent on the parameter values considered, ranging from highly favourable for prevention when the incursion rate of ballast water pests is high and/or the assumed average cost of a high impact incursion is high, to highly unfavourable for either prevention or eradication when incursion rates are low and/or the assumed average cost of a high impact incursion is low. When eradication is favoured over living with all high impact NIMS it is only favoured slightly, because most eradication attempts are likely to fail.

Figure 3 Total cost of a prevention or eradication approach relative to the total cost of living with all high impact incursions for different values of $Cost_{LWI(pv)}$ given a yearly cost of prevention ($Cost_p$) of $0.8 million.



Notes. The total cost of the eradication or prevention approach includes the cost of living with the high impact IMS that either eradication or prevention fail to prevent establishing. The cost of prevention ($Cost_p$) used to generate this figure was $0.8m per year. A value of '1' corresponds to when the total costs of the eradication or prevention approach equals the total cost of living with all high impact incursions. Different lines represent different values for key parameters. Incursion rate refers to the incursion rate in the absence of prevention. An effectiveness of prevention of 80 per cent means the prevention approach reduces the incursion rate by 80 per cent. Note the x axis is not to scale, but shows critical values of $\boldsymbol{Cost_{LWI(pv)}}$ for which the relative cost was calculated.

5.2 Cost-effectiveness (method 2)

Cost-effectiveness analysis, like cost analysis, is another way of putting different management approaches on the same scale so they can be compared. In this case the questions are: how much needs to be spent on prevention vs. eradication to achieve the same reduction in establishments of either NIMS, or of IMS that are likely to have large impacts? One advantage of cost effectiveness analysis is that it is not necessary to estimate the present value of impacts per incursion.

If the concern is with the incursion rate of all non-indigenous ballast water species (that is all species regardless of their level of impact) to Australia, then the estimate for the incursion rate in the absence of prevention is about 1.5 per year (section 3.2). If we assume prevention reduces this to about 0.15 per year, then over a 10 year period we spend about $368 million (at an annual rate of $36.8 million) to reduce incursions from about 15 to about 2. If we assume 4% of these 15 incursions are 'species of concern' (that is, about 1), then eradication would not even be attempted for most of the 15 incursions, so this type of reduction (15 down to 2) cannot be achieved through eradication. Prevention is the only way to achieve this sort of reduction.

If the concern is with 'high impact' incursions, the rate of 'high impact' incursions is about 1 every 10 years and the existing ballast water rules might reduce this rate to about 1 in 100 years. Hence, a prevention approach costs $37 million (or $0.8 million administration cost) every year for 100 years to avoid 9 'high impact' incursions. The alternative is to ask the question – how much would we need to spend on average to have a 90 per cent chance of eradicating high impact incursions? This defines the break-even point, where spending $370 million ($8 million) per incursion on the 10 incursions over 100 years results in 9 of them being successfully eradicated. In other words a prevention system would cost $37 million ($0.8 million) per year for 100 years = $3 700 million ($80 million) and an eradication approach would cost $370 million ($8 million) per incursion for 10 incursions = $3 700 million ($80 million) and both would reduce the number of establishments from 10 down to 1. Money for an eradication system would be spent on an early warning surveillance system so that incursions could be detected before they become too large, plus the act of attempting to eradicate each of the 10 incursions. However, it is not clear that a 90 per cent chance of eradicating all high impact marine incursions could be achieved regardless of the amount of money spent - certainly $8 million would not achieve this (section 4).

5.3 Benefit-cost ratios (method 3)

Benefit-cost ratios can also be calculated for the prevention or eradication approaches. Benefit-cost ratios tell us the losses avoided relative to the amount of money spent. Because of this they indicate for each approach whether the avoided losses exceed the amount of money spent on the approach, but they don't provide a good basis for comparing between approaches, because they ignore the impacts of failures to either prevent or eradicate. As seen in section 5.1, these impacts can be substantial and are an integral part of the direct comparison between the approaches.

The benefit-cost ratio (BCR) of prevention is the avoided losses from having a prevention system, divided by the costs of having that system. We avoid the impact of $inc_0 - inc_p$ incursions at a cost of $Cost_p$. Hence:

$$BCR = \frac{(inc_0 - inc_p) \times Cost_{LWI(pv)}}{Cost_p}$$

If $Cost_{LWI(pv)} = \$1\ billion,\ inc_0 = 0.08,\ inc_p = 0.008\ (90\%\ reduction\ in\ rate)$, and $Cost_p = \$37\ million$, then the benefit-cost ratio of prevention is:

$$BCR = \frac{(0.08 - 0.008) \times \$1 billion}{\$37\ million} = 1.95$$

So, for every dollar spent on prevention we avoid twice as many losses.

The benefit-cost ratio of eradication is the avoided losses from those incursions we successfully eradicate, divided by the costs of attempting to eradicate them, which reduces to the benefit-cost ratio given an incursion has occurred:

$$BCR = \frac{inc_0 \times \left(1 - \Pr(erad_f)\right) \times Cost_{LWI(pv)}}{inc_0 \times Cost_e} = \frac{((1 - \Pr(erad_f)) \times Cost_{LWI(pv)}}{Cost_e}$$

If $Cost_{LWI(pv)} = \$1\ billion$, $Cost_e = \$20\ million$, and $\Pr(erad_f) = 0.80$, then the benefit-cost ratio of eradication is:

$$BCR = \frac{((1 - 0.8) \times \$1\ billion}{\$20\ million} = 10$$

So, for every dollar spent attempting eradication we avoid 10 times as many losses. When considered in isolation, eradication is worth considering, but is certainly not more preferable than a prevention approach even though the BCR is greater, because for the set of parameters chosen prevention is far more effective at avoiding establishments of IMS (section 5.1).

If $Cost_{LWI(pv)} = \$500\ million,\ inc_0 = 0.08,\ inc_p = 0.008\ (90\%\ reduction\ in\ rate)$, and $Cost_p = \$37\ million$, then the benefit-cost ratio of prevention is:

$$BCR = \frac{(0.08 - 0.008) \times \$500\ million}{\$37\ million} = 0.97$$

While the benefit-cost ration for eradication ($Cost_e = \$20\ million$, $\Pr(erad_f) = 0.80$) is 5. However, if only the cost of administering the prevention system is considered the benefit-cost ratio is:

$$BCR = \frac{(0.08 - 0.008) \times \$500\ million}{\$0.8\ million} = 45$$

If $Cost_{LWI(pv)} = \$500\ million$, $Cost_e = \$20\ million$, and $\Pr(erad_f) = 0.95$, then the benefit-cost ratio of eradication is:

$$BCR = \frac{((1 - 0.95) \times \$500\ million}{\$20\ million} = 1.25$$

As for the cost analysis (section 5.1), the benefit-cost ratios are highly dependent on the parameter values considered.

6 Conclusions

There is considerable uncertainty about many of the parameter values required for an economic evaluation of approaches to managing NIMS. These include uncertainty about incursion rates of NIMS both with and without prevention measures, the magnitude of impacts from NIMS and how rapidly they will develop, and the likely costs and effectiveness of eradication attempts. Nonetheless, taken together, the cost comparisons conducted in this study indicate that prevention is generally the preferred approach to eradication because prevention focuses on a broader range of NIMS than eradication and has higher rate of success than eradication. Even with a prevention system in place, eradication could still be considered as incursions arise.

Eradication of IMS is extremely challenging because of the difficulties of detection and treatment in the marine environment. Successful eradication would almost certainly require very early identification of an incursion before the IMS covers an area that is too large to contemplate eradication (perhaps as little as 1 ha). Hence a major consideration for the addition of an eradication approach is how much or whether to invest in an 'early warning' system to improve the likelihood of eradication being successful. Some of the attributes of an early warning system were considered in Arthur et al. (2015), which also suggested that the cost-effectiveness of an early warning system would need to be determined if one were to be adopted.

For the broader question of the economic value of a prevention system relative to having no prevention system, the answer is complicated by the apparently low incursion rates to Australia of high impact species to date - at least those with obviously high impacts. Any economic evaluation is likely to come down to how much the public values avoiding establishment of NIMS, including those that may have large environmental or social amenity impacts and the many more that will not.

Appendix A: Biofouling

Biofouling, like ballast water, has been recognised as a major pathway for invasive marine species by the IMO, which has developed the Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species to provide a globally consistent approach to the management of biofouling (IMO, 2011). As noted in the main body of the report the costs of biofouling treatment have been excluded from this study on the basis that the application of antifouling coatings and the treatment of internal water systems are measures that have been adopted by the shipping industry for reasons of fuel efficiency and vessel safety. In the event that these costs are required the following is a brief summary of some of the issues involved in the prevention of biofouling and the costs involved.

Objects placed in seawater are typically colonised by marine organisms, a process termed biofouling (Marszalek et al., 1979). The process of biofouling develops over time with a sequence of fouling organisms that typically begins with a slime layer or biofilm onto which macrofouling organisms may become attached (Chambers et al., 2006). Marszalek et al. observed that physically and biologically inert materials such as glass and stainless steel were fouled by bacteria within four hours of immersion in seawater. The development of coatings to inhibit growth and to prevent corrosion of the surface has been the subject of research and development since at least the 17th Century (Chambers et al., 2006).

The economic losses caused by biofouling have resulted in considerable investment in antifouling coatings. These coatings have a two-fold purpose: to prevent corrosion and the build-up of biofouling which has a severe negative effect on fuel consumption. The problem of biofouling was thought to have been more or less solved with the introduction in the 1960s of self-polishing copolymer based antifouling coatings with tributyltin (TBT) as the biocide (Townsin, 2003). By the 1980s, however, reports of declines in oyster production, problems with larval development and shell abnormalities (e.g. Alzieu, 1991) led to restrictions on the use of TBT, especially on recreational vessels (Dafforn et al., 2011). Its use in antifouling coatings was finally banned on all vessels by the International Convention on the Control of Harmful Anti-fouling Systems on Ships (IMO, 2001). The ban on TBT-based antifouling coatings led to a resurgence in research into alternative coatings, which can be divided into three different approaches:

- Inclusion of a biocide
- Fouling-release coatings that are too slippery to allow fouling to adhere
- Ultra-hard coatings without biocide that rely on regular in-water cleaning

For biocide-containing coatings, copper has again become the predominant antifouling biocide (Dafforn et al., 2011), but concerns about its toxicity have led to recommendations being made to limit its usage (Srinivasan and Swain, 2007). In response to the potentially conflicting objectives of protecting environments from the toxicity of antifouling coatings and preventing spread of marine pests, the International Maritime Organization introduced the Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species (IMO, 2011).

Chambers et al. (2006) list the requirements for an optimal antifouling coating. It must be:

- Anticorrosive
- Antifouling

- Environmentally acceptable
- Economically viable
- Have a long life
- Compatible with underlying system
- Resistant to abrasion/biodegradation/erosion
- Capable of protecting regardless of operational profile and
- Smooth

It must not be:

- Toxic to the environment
- Persistent in the environment
- Expensive
- Chemically unstable
- A target for non-specific species

It was estimated that TBT-based coatings once provided antifouling protection for 70% of the world's shipping fleet (DNV, 1999) but since it was banned a number of approaches to the development of antifoulant coatings have been followed. There are a number of ways of classifying the new generation of coatings, for example, Dafforn et al. (2011) categorise antifoulant coatings as follows:

- Conventional coatings
- Self-polishing copolymer coatings
- Booster biocides
- Foul-release coatings
- Biomimetic coatings

The result of this research and development activity has been wider choice but not necessarily more effective antifouling coatings.

Antifouling coatings have a limited service life for the following and other reasons (Thompson Clarke Shipping Pty. Ltd. et al., 2007):

- Type and thickness of the coating
- Leaching of the biocide from the coating (normal and predictable)
- Wear and tear from rubbing against wharves, minor collisions, etc
- Poor surface preparation, base layer application and antifouling coating application
- Corrosion
- Level of activity

The service life depends on a number of factors, not the least of which is the specification (and cost) of the coating. The typical service life of an antifouling coating for commercial shipping use could be 36 or 60 months and new generation coatings offer protection for up to 90 months (http://www.hempel.com/en/marine/underwater-hull). Antifouling coatings can only be applied when the ship is in dry dock. There are few dry docks available world-wide for large ships, e.g. Capsesize bulk carriers, and scheduling access requires bookings years in advance. In-water cleaning is often used as an interim measure between dry dockings. Dry docking is required at periodic intervals, typically three years, by ship classification societies for survey and many ship owners use this as an opportunity to clean and re-coat their ships' hulls.

Typical procedures associated with antifouling coating application (CET-Hamburg GmbH, 2014) are listed below. Each of these procedures incurs a cost:

- Voyage costs to dry dock location
- Tug and pilot to enter dry dock
- Hull cleaning (high pressure water spray)
- Stripping off old antifouling coating, if necessary down to bare metal or patch
- (Hull repairs, if necessary)
- Cost of primer and antifouling paint
- Application of primer and antifouling coatings
- Tug and pilot to exit dry dock
- Opportunity costs/lost revenue while in dry dock
- Dry docking charges
- Inspection costs

The main factors affecting the cost of antifoulant coating are:

- Size of the ship
- Dry dock costs
- The make and type of antifouling coating

The introduction of mechanically resistant coatings (e.g. Hydrex NV, 2006) changes the dynamics of hull protection. These coatings are biocide-free and therefore foul relatively quickly but are designed to be cleaned regularly underwater by divers using special rotating brushes that also 'condition' the surface to provide optimal smoothness and therefore best fuel efficiency. It is usually recommended that hulls protected with other types of antifouling coatings should not be cleaned in-water because of the risks of releasing a pulse of biocide into the water, or if it is necessary to clean them, they be cleaned only very lightly. The advantages of a mechanically resistant coating are first that it does not release biocide into the marine environment and second that it requires infrequent repair. The disadvantages are that regular and frequent in-water cleaning by diver can be expensive and presumably not available at every port, and unless the fouling debris removed by the divers is collected with a high degree of precision, there are high risks of dislodging the fouling organisms, which may become established near the vessel. These issues are discussed at length by Hopkins and Forrest (Hopkins and Forrest, 2008) and Hopkins et al. (2009). The Australian anti-fouling and in-water cleaning guidelines (DAFF et al., 2013) provide for a more flexible approach to biofouling management than the outright ban on in-water cleaning that the previous version of the guidelines gave (ANZECC, 2000), which makes biocide-free coatings viable alternatives to biocide-loaded coatings.

Biofouling is not restricted to external surfaces of ships' hulls and niche areas (Australian Shipowners Association, 2007). Fouling of internal piping also occurs, which can cause major problems for sea water inlets and outlets, cooling systems and other pipe work. Many vessels have marine growth prevention systems (MGPS) installed to control this form of biofouling. There are many different systems using many different forms of treatment, including biocide release, ozone and thermal or osmotic (freshwater) shock. Grandison et al. (2012) review the MGPS in use in the Royal Australian Navy and a selection of commercial off-the-shelf systems and document the advantages and disadvantages of each.

Glossary

Allee effect A phenomenon in biology where a decline in individual fitness at a low population density or an enforced low population density can result in the population becoming extinct. In the marine environment, where most organisms are broadcast spawners, low population density can result in failure to reproduce.

Ballast water Water (including sediment that is or has been contained in water) held in tanks and cargo holds of ships to increase stability and manoeuvrability during transit.

Biofouling The attachment of marine organisms to any part of a vessel, or any equipment attached to or on board the vessel, aquaculture equipment, mooring devices and the like.

Bulk carrier A ship designed to carry cargo in bulk. Bulk carriers are divided into a number of class sizes e.g. Handysize, Handymax, Panamax and Capesize (q.v.).

Catadromous Organisms that live in fresh water but migrate to the sea to breed.

Classification society Classification societies establish and maintain standards for the construction and continued safe operation of ships and structures offshore. Examples include Lloyds, Bureau Veritas and American Bureau of Shipping.

Cryptogenic A species whose geographic origins (i.e. whether they are native or non-indigenous) are uncertain.

Capesize A size class of bulk carrier. These are the largest bulk carriers which are too big to go through either the Suez or Panama Canals and must therefore sail around either Cape Horn or Cape of Good Hope to transit between Pacific, Atlantic or Indian oceans. Their deadweight tonnage range is from about 80,000–300,000 tonnes or greater.

CCIMPE Consultative Committee on Introduced Marine Pest Emergencies

Deadweight tonnage A measure of the total weight a ship can safely carry and includes fuel, ballast water fresh water, provisions and crew as well as cargo.

Dry dock A large dock built so that the base is below sea level and with watertight gates to allow dry access to a ship's hull for inspection, repairs and re-painting, etc.

FPSO Floating Production, Storage and Offloading. A large oil tanker moored semi-permanently above an offshore oil field to upload and

	store oil ready for collection by oil tankers.
Handymax	A size class of bulk carriers with a deadweight tonnage range of about 35,000–60,000 tonnes.
Handysize	The smallest size class of bulk carriers with a deadweight tonnage range of about 10,000–35,000 tonnes.
IMO	International Maritime Organization. An agency of the United Nations with responsibility for the safety and security of shipping and the prevention of marine pollution by ships.
IMS	Invasive marine species.
MODU	Mobile Offshore Drilling Unit. A mobile oil rig.
Niche area	An area in a ship's hull where prevention of biofouling is difficult. Examples include sea chests (q.v.), bow and stern thrusters, sacrificial anodes and rudder stocks.
National Control Plans	Control plans for a number (currently six) of invasive marine species established in Australia. Developed under the 'Ongoing management and control' component of the National System (q.v.).
National System	National System for the Prevention and Management of Marine Pest Incursions. Established in Australia in 2000.
Newcastlemax	The maximum dimensions of a bulk carrier that can be accepted by the port of Newcastle, NSW.
NIMS	Non-indigenous marine species.
Panamax	A size class of bulk carriers with a deadweight tonnage range of about 60,000–80,000 tonnes.
ROV	Remotely-operated vehicle. Usually deployed underwater for a range of tasks.
Sea chest	An inset space in a ship's hull often for inlet and outlet pipes and usually covered with a grate.
VLCC	Very Large Crude Carrier. A very large type of oil tanker.
Wozmax	A sub-category of Capesize vessels. The maximum dimensions that can be accepted by the Western Australian (WOZ) ports of Port Hedland, Port Walcott and Dampier.

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