Risk Analysis:
Vessel Biofouling

ISBN 978-0-478-37548-0 (print)
ISBN 978-0-478-37549-7 (online)

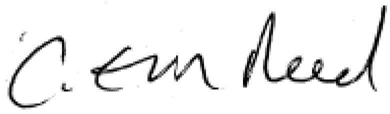
15 February 2011

Ministry of Agriculture and Forestry
Te Manatū Ahuwhenua, Ngāherehere

Risk Analysis: Vessel Biofouling

15 February 2011

Approved for general release



Christine Reed
Manager, Risk Analysis
Ministry of Agriculture and Forestry

Requests for further copies should be directed to:

Publication Adviser
MAF Information Bureau
P O Box 2526
WELLINGTON

Telephone: 0800 00 83 33
Facsimile: 04-894 0300

This publication is also available on the MAF website at
http://www.biosecurity.govt.nz/regs/imports/ihs/risk

Contributors to this risk analysis

1. Primary author/s

Dr Andrew Bell	Senior Adviser Risk Analysis, Marine	MAF Biosecurity New Zealand Wellington
Simon Phillips	Adviser Risk Analysis, Marine	MAF Biosecurity New Zealand Wellington
Dr Eugene Georgiades	Senior Adviser Risk Analysis, Marine	MAF Biosecurity New Zealand Wellington
Dr Daniel Kluza	Senior Adviser Risk Analysis, Marine	MAF Biosecurity New Zealand Wellington

2. Secondary contributors

Dr Christopher Denny	Adviser Border Standards	MAF Biosecurity New Zealand Wellington

3. External peer review

John Lewis
Principal Marine Consultant
ES Link Services Pty Ltd
Melbourne, Victoria, Australia

Richard Piola
Senior Scientist
Cawthron Institute
Nelson, New Zealand

The draft risk analysis has also been internally reviewed by:

- Liz Jones (Border Standards);
- Justin McDonald (Post-Clearance);
- Melanie Newfield (Risk Analysis);
- Howard Pharo (Risk Analysis);
- Sandy Toy (Risk Analysis).

The contribution of all the reviewers is gratefully acknowledged.

Contents

Page

Executive summary

The purpose of this risk analysis is to assess the biosecurity risks associated with biofouling organisms on the hulls of vessels arriving in New Zealand from international waters.

Species within twelve of the twenty broad taxonomic groups assessed were determined to present non-negligible risks to New Zealand's core values and for which risk management measures can be justified. A common feature among these groups is that they are all macro-fouling organisms with either the potential to significantly modify ecosystem structure and function, or the potential to upon impact core economic values, such as aquaculture.

Based on the outcome of this risk assessment and recognising the need to rapidly clear vessels at the border, it is proposed that any macro-organisms found on the hull of an arriving international vessel are treated as risk organisms. This decision is based on the difficulty in identifying marine organisms *in-situ,* and the association of macro-fouling organisms with the introduction of non-indigenous species. As a result, the presence of macro-fouling organisms can be considered to indicate the existence of a biosecurity risk which needs to be managed.

Introduction

Biofouling is a process where organisms accumulate on a clean surface that has been immersed in the marine environment. Biofouling on vessel hulls thus provides a means by which organisms can be transferred to new locations. Biocidal antifouling paints, developed in the 1970's, proved very effective against fouling organisms (in particular, those paints containing the active ingredient tributyltin). Therefore, the risk of introduction of non-indigenous species by the vessel biofouling pathway was thought to be mitigated. However, there is an increasing amount of evidence/research indicating that this is still an important entry pathway. Vessel biofouling has been associated with the introduction of non-indigenous organisms in many parts of the world, including New Zealand. For example, more than 65% of non-indigenous marine species in the waters of New Zealand, Hawai'i (USA), and Port Phillip Bay (Australia), were likely translocated as biofouling. In addition, the following factors have provided further opportunities for the transfer of non-indigenous species via vessel biofouling:

- Increased shipping volumes from international trade mean there is an increase in the volume of non-indigneous species entering New Zealand (i.e. increased propagule pressure);
- The opening of new trade routes to New Zealand mean there is the prospect for biofouling organisms, not previously encountered, entering and establishing in New Zealand;
- The increased transit speeds of commercial vessels can lead to increased arrival of viable non-indigenous species as biofouling organisms are subjected to translocation stresses for shorter periods.

To better understand and manage vessel biofouling risks in a New Zealand context, MAFBNZ commissioned a multi-year research survey of international vessels arriving in New Zealand. The objectives of the study were to determine:

- the identity (species), origin (native, non-indigenous, unknown) and extent of biofouling occurrence on vessels;

- the relationship between the presence of non-indigenous species and the amount/extent of biofouling on a vessel;
- the factors that influence the presence of non-indigenous species and the amount of biofouling on vessels (e.g., vessel maintenance regime, voyage history).

Findings from the MAFBNZ research programme demonstrate that:

- All major vessel types (recreational, passenger, fishing and commercial merchant) are likely to have some associated biofouling;
- Of 187 species identified in the study, over 66% were non-indigenous to New Zealand; of these species, 73% are not yet established in New Zealand;
- The greater the amount of biofouling on a vessel, the higher the number of non-indigenous species present;
- Biofouling organisms were predominantly arthropods (mostly barnacles), tube-forming marine worms, bryozoans, bivalve molluscs and macroalgae;
- Biofouling organisms generally accumulate in niche areas[1], where prevention of biofouling is difficult, or may not be considered important;
- Niche areas (including seachests and internal seawater systems) pose a substantial biosecurity risk despite accounting for a relatively small proportion of the hull.

Although not all non-indigenous biofouling species will necessarily pose a threat to New Zealand's core values, it is difficult to predict the identity and impacts of future invasive species. The introduction of some non-indigenous species to new locations can have far-ranging impacts on the marine environment and the people reliant upon it. For example, worldwide, introductions of non-indigenous marine species have impacted upon the following environmental, economic and socio-cultural values:

- Direct impacts to environmental values by non-indigenous species include the recession of kelp forests. For example, in the Gulf of Maine, USA, the non-indigenous bryozoan *Membranipora membranacea* have formed colonies on indigenous kelp fronds, causing them to become brittle and break;
- Direct economic impacts by non-indigenous species can affect both maritime and non-maritime industries. The green mussel (*Perna viridis*) was first identified in Tampa Bay, USA, in 1999. Since then dense assemblages of these bivalves now clog sea water intakes of power stations and desalination plants, weigh down navigational buoys and foul vessel hulls and engines;
- Further, in Sweden the economic impact of the bay barnacle *Balanus improvisus*, calculated as fouling control costs, is estimated at 166-418 million Swedish krona per year, of which recreational vessels accounted for approximately 74%;
- Sea squirts such as *Ciona intestinalis*, *Styela clava* and *Didemnum vexillum* have become significant fouling pests of shellfish aquaculture. In Nova Scotia and Prince Edward Island, Canada, non-indigenous sea squirts compete with stock, causing a reduction in size and condition and increased mortality. Sea squirt fouling can increase labour and processing costs; for example, *D. vexillum* fouling of mussel lines in the Marlborough Sounds costs growers an estimated $2.2million per annum;

[1]Niche areas are places on a vessel's hull where protection from anti-fouling paint degrades quickly, areas that are not coated in antifouling paint, and recesses that are protected from the drag created by the boat moving through the water. Niche areas include propellers, rudder shafts, bow thrusters, seachests and dry-docking support strips.

- In New Zealand, non-indigenous tubeworm species *Polydora websteri* and *P. hoplura* have had negative effects on oyster aquaculture.
- Commercial fisheries can be indirectly affected by non-indigenous species. The Asian clam *Corbula amurensis*[2], introduced to San Francisco Bay, has altered the food chain to a point where commercially harvested anchovies no longer occur in local waters;
- Non-indigenous species may also impact on economic and social values as, in the State of Hawai'i, the island of Maui loses US$20 million of tourism revenue annually due to the fouling of beaches by non-indigenous algae;
- Recreational fisheries are also vulnerable to non-indigenous species. For example, the introduction of the clam *Ruditapes philippinarum*[2] to Venice, Italy, has been linked to reduced fishing catch rates, and fishers in turn have changed their angling habits.

New Zealand receives, on average, over 3,000 international vessel arrivals annually and is reliant on vessels to move 98% of traded goods. New Zealand has built a reputation for marine service provision, which has significant economic benefit for the country. Vessels arrive throughout the year, with a peak in activity from October to December due to the arrival of recreational vessels.

Different vessel types have been associated with different levels of risk based upon their physical characteristics, maintenance history, voyage characteristics and accumulation of biofouling. For example, recreational vessels have been identified as being high risk due to factors such as long lay-up periods, slow and itinerant voyages, and the marginal benefit of in-service maintenance to the owner. By contrast, commercial vessels, such as container vessels, have rapid turn around times and significant efficiency incentives to have clean hulls so potentially pose a lesser biosecurity risk, however, these vessels have a large surface area of niche areas, such as seachests, that are particularly vulnerable to biofouling.

Current risk management

There are as yet no mandatory requirements in any country to control biofouling. As a result, biofouling risk management has been predominantly coincident with vessels using anti-fouling systems and in-service maintenance for their own purposes. In recent years the use of these measures for preventing the introduction of non-indigenous species has been encouraged by MAFBNZ communication campaigns. However, as the benefits of biofouling management are considered to be marginal by some vessel owners, the biofouling load is variable between vessels. In addition, there are niche areas on the hull, such as seachests, where fouling can accumulate without noticeably impacting performance. As a consequence, even apparently well-maintained vessels may have biosecurity risks that require mitigation.

The risks associated with biofouling are beginning to be addressed at national and international levels. Australia is considering the implementation of biofouling management requirements for all international vessel arrivals, and the International Maritime Organisation is investigating the development of guidelines containing 'International Measures for Minimizing the Transfer of Invasive Aquatic Species through Biofouling of Ships". However, international regulation is unlikely for at least another 5 years.

[2]Introductions of this species have occurred via ballast water. Although not currently known as a vessel biofouling species, the environmental impacts of this organism are representative of the potential risks posed by hull fouling bivalve species.

The ongoing risks of vessel biofouling are of immediate concern to MAFBNZ given that the marine environment is a key part of many of New Zealand's economic, social and cultural values, which include aquaculture, commercial fisheries, tourism, recreational activities and biodiversity:

- New Zealand's marine ecosystems and species are highly diverse, and it is estimated that as much as 80% of the country's native biodiversity occurs in the sea. Species found only in New Zealand account for 44% of our marine biodiversity, a very high number that distinguishes New Zealand as a global hotspot for marine biodiversity;
- The commercial value of New Zealand's wild and farmed fisheries is $1.2–1.5 billion annually;
- The majority of New Zealanders live within 50 kilometres of the coastline, and many people swim, fish, and gather seafood from the coast;
- Māori have a close cultural relationship with the ocean. It is regarded as a tāonga that is integral to their culture and identity. The sea is important to tangata whenua as a source of food, and the mana of hapu and iwi is still closely linked to their ability to provide hospitality to visitors through plentiful kaimoana. The sea is also an important part of Māori spirituality and mythology.

It is clear that New Zealand's coastal environments[3] are particularly important with respect to economic, social and cultural values, yet these areas are also the marine habitats most likely to be impacted upon by non-indigenous species. Activities that have the potential to introduce non-indigenous species are concentrated on coastal areas (e.g., shipping and boating). In addition, human impacts on these coastal environments (e.g., pollution and sedimentation) can create conditions that favour the establishment of non-indigenous species.

Methodology

The risk analysis was undertaken following the MAFBNZ risk analysis procedures, which are consistent with international standards for risk analysis. Risk management options, from published literature, are detailed with an analysis of their likely efficacy against identified hazards.

In excess of 2,000 species have been associated with the fouling of vessel hulls, therefore in order to keep the analysis manageable, twenty broad taxonomic groups were analysed. For each group an initial hazard identification assessment determined whether a risk analysis was required. For groups identified as potential hazards, the following factors were assessed sequentially with a determination of non-negligible or negligible for each: the likelihood of entry into New Zealand, the likelihood of establishment in New Zealand, and potential consequences of establishment. The analysis of each group concluded with the risk estimation and the assessment of potential management options.

Risk assessment results

Species within twelve of the twenty broad taxonomic groups assessed were determined to present non-negligible risks to New Zealand's core values and for which risk management measures may be considered. A common feature among these groups is that they are all macro-fouling organisms with either the potential to significantly modify ecosystem structure

[3]The area within New Zealand's territorial sea (extending 12 nautical miles from the coast) and/or the extent of the continental shelf (a depth profile of 250 m).

and function, or the potential to impact upon core economic values, such as aquaculture. While it is difficult to forecast the exact nature of impacts, it is known that introduced marine species can significantly alter local environments leading to impacts on ecosystem services. Economic, environmental, social and cultural values at risk from these impacts include aquaculture, fisheries, tāonga species, iconic habitats and recreational uses of the coastal environment, respectively.

The groups identified as posing a non-negligible risk to New Zealand are:

- Amphipods and Isopods;
- Barnacles;
- Bivalves;
- Bristleworms;
- Bryozoans;
- Crabs;
- Echinoderms;
- Flatworms;
- Gastropods;
- Hydroids;
- Macroalgae;
- Sea squirts.

Risk management options

The most effective way of eliminating the entry and establishment of non-indigenous species into New Zealand would be to close the border, however, this is not a feasible option. Nor is it possible to determine an appropriate level of tolerance for vessel biofouling beyond what is practically achievable. Preventative measures will reduce risks prior to a vessels arrival however, these measures alone may not provide an adequate level of mitigation due to the difficulties associated with fouling prevention.

Given the broad range of organisms that have been identified as biofoulers and their potential to be associated with each other in fouling assemblages, generic, broad-spectrum treatments are likely to be most effective (i.e. it is rare that biofouling assemblages are composed of an individual species). Known treatment options range from preventative measures, such as anti-fouling systems, to reactive containment measures, such as in-water encapsulation or land-based cleaning. The practicalities of each option vary with vessel type, size and operating conditions. As the individual options assessed are not expected to fully mitigate the risk, a combination of treatments may be required.

The options for eradicating or managing an established species in the marine environment are limited, difficult to perform and expensive. For example, the eradication of the black striped mussel from three marinas in the Australian Northern Territory, cost in excess of $AU2.2 million and required the use of a biocide to kill all life in the marinas. In New Zealand, incursion responses to the sea squirt *S. clava* and the Mediterranean fanworm (*Sabella spallanzanii*) have cost $2.2 million and $1 million, respectively, with a further $1.5 million earmarked for continuing *Sabella* control. Given the ongoing expenses associated with response and control efforts, preventing the establishment of non-indigenous species is preferable to responding to the discovery of a new incursion.

The identification of marine organisms *in-situ* can be difficult and even expert taxonomic identification in the laboratory can be challenging and time-consuming. Therefore, based on the outcome of this risk assessment and recognising the need to rapidly clear vessels at the border, it is proposed that any macro-organisms found on the hull of an arriving international vessel are treated as risk organisms. This is consistent with the current anti-fouling system technology which seeks to limit biofouling to no more than a slime layer.

Given that fouling is most likely to accumulate in niche areas, it is suggested that, where equipment is available, inspection for macro-fouling entails a rapid visual inspection of the hull from the waterline followed by an underwater inspection of the stern area focusing on any grates and appendages such as the rudder, propeller and driveshaft. The presence of macro-fouling organisms can be considered to indicate the existence of a biosecurity risk which needs to be managed.

Current understanding of the efficacy of inspection to characterise vessel biofouling is limited. As a consequence, the further development and validation of inspection decisions would enhance this understanding.

Collection of additional information on vessel characteristics will likely aid future inspection efficacy. At present there are no clear indicators of biofouling risk, such as voyage history or maintenance regime, for international vessels entering New Zealand.

To support the development of vessel risk profiles, consideration should be given to require vessels to supply verifiable documentation of their voyage history, maintenance regime and/or biofouling management plan. A communications plan informing vessel owners/masters of the benefits of keeping accurate records would support this approach.

Definitions

Anti-fouling system: a coating, paint, surface treatment, surface, or device that is used on a vessel to control or prevent the attachment of organisms.

Biofouling: the accumulation of aquatic organisms on surfaces immersed in, or exposed to, the aquatic environment.

Biosecurity: the exclusion, eradication or effective management of risks posed by pests and diseases to the economy, environment and human health.

Consequence: the adverse effects or harm as a result of entry and establishment of a hazard, which cause environmental, economic and/or socio-cultural values to be degraded in the short or long term.

Establishment: perpetuation into the foreseeable future of organisms within an area of entry.

Hazard: any disease or organism that has the potential to produce adverse consequences.

Inspector: a person who is appointed as an inspector under section 103 of the Biosecurity Act (1993).

MAF: the Ministry of Agriculture and Forestry.

Negligible: to be so small or insignificant as not to be worth considering.

Niche areas: areas on a ship that are susceptible to biofouling due to, different hydrodynamic forces, susceptibility to coating system wear or damage, or being inadequately, or not, painted e.g. seachests, bow thrusters, propeller shafts, inlet gratings, dry dock support strips etc.

Risk: the likelihood of the occurrence and the likely magnitude of the consequences of an adverse event.

Transitional facility: any place approved as a transitional facility in accordance with section 39 of the Biosecurity Act (1993) for the purpose of inspection, storage, treatment, quarantine, holding or destruction of specified types of un-cleared goods; or part of a port declared to be a transitional facility in accordance with section 39.

Vessel: a mobile structure of any type whatsoever operating in the marine environment and includes floating craft, fixed or floating platforms, and floating production storage and off-loading units (FPSOs).

1. Introduction

This risk analysis assesses the biosecurity risks associated with vessel biofouling. These analyses will inform any requirements for incoming vessels that may be developed under the Biosecurity Act (1993). This analysis applies to all vessel types.

1.1. BACKGROUND

1.1.1. The biology of marine biofouling

Any surface immersed in seawater will rapidly develop a biofilm. A conditioning film of organic material will adhere within minutes of immersion and is rapidly colonised by single celled organisms such as bacteria, microalgae and cyanobacteria (Wahl 1989, Cooksey & Wigglesworth-Cooksey 1995, Callow & Callow 2002). These unicellular recruits exude extrapolymeric substances (EPS) which give the biofilm its characteristic slimy texture, and thus its colloquial description of "slime layer". The biofilm is also referred to as primary or simple fouling as it is the first biological phase in the fouling process and has little structural complexity. Multicellular organisms, such as macroalgae (where light permits), barnacles, bryozoa and tubeworms, subsequently settle to create a more complex fouling cover. This settlement may be facilitated by the biofilm as the bacteria, microalgae and/or EPS are attractive as food, provide an attachment substrate, or elicit a chemotactic response from, recruits of macro-fouling organisms (Lam et al. 2003, Qian et al. 2003, Qian et al. 2007). However, the biofilm is not essential for macro-organism recruitment. Spores of *Ulva* spp. for example, have been found to recruit directly to artificial surfaces that have been "cleaned" prior to exposure (Joint et al. 2000).

The complexity of biofouling assemblages increases as macro-fouling develops, this creates habitat for other organisms, including mobile species such as isopods and crabs. If left undisturbed the fouling assemblage will continue to develop toward a steady state. On artificial substrates this may not equate with the fouling assemblages seen in neighbouring natural habitats, such as rocky reefs, as the nature of the underlying substrate can influence recruitment and succession processes (Glasby 1999). In addition, the abundance and availability of potential recruits, inter- and intra-specific competition between settled organisms and the presence of predators can all influence assemblage structure. For example, Figure 1.1 shows the fouling assemblage found on a marker buoy from Wellington harbour which is heavily fouled by the stalked ascidian *Pyura pachydermatina*. In the Wellington region, *P. pachydermatina* is common but is usually found as isolated individuals or small groups at low to moderate densities. Factors which may have influenced their dominance in this particular case include the aspect of the buoy in relation to light, water depth and currents, its free floating state, the anti-fouling paint applied (and its condition), the season in which the buoy was deployed, the combination of materials used to construct the buoy, and the design of the buoy itself.

Figure 1.1 *Pyura pachydermatina* fouling on a marker buoy removed from Wellington Harbour (Photo MAFBNZ).

This propensity for biofouling to occur in the marine environment has led to significant research effort being expended on management solutions. The cost of managing biofouling varies and often no single management option is completely effective. In addition, there is an apparent "arms race" occurring between biofouling organisms and the introduction management measures, for example, some common biofouling species have developed biocide resistance (e.g. Hoare et al. 1995 a & b, Cassé & Swain 2006, Dafforn et al. 2008). Moreover, the benefits of, and therefore the imperative for, biofouling management differs. This, therefore, leads to different levels of management on artificial structures such as vessels, marina pontoons and wharf piles. Mobile structures increase the likelihood of biosecurity risk due to their ability to transfer fouling organisms beyond their natural dispersal limits. The likelihood of this risk will vary depending upon the effort expended on biofouling management by the structure's owner/operator.

1.1.2. The impacts of translocated fouling organisms

The translocation of fouling organisms has led to significant impacts in aquatic environments where introductions have been made. A global acknowledgement of the problem can be inferred from the comprehensive research review presented to the International Maritime Organization by the correspondence group on the "Development of International Measures for Minimizing the Transfer of Invasive Aquatic Species through Biofouling of Ships" (IMO 2008).

Biofouling species, both native and non-indigenous[4], can impact upon maritime infrastructure. For example, biofouling on vessels and other marine structures can interfere with operations, impose increased loading, accelerate corrosion and cause navigation buoys to sink. For vessels, the most significant impact of biofouling is on performance, as the hydrodynamic drag necessitates the use of more power and fuel to move a ship through the water (Lewis 1998). Many ships and coastal industries that use piped seawater for engine and equipment cooling, fire fighting, and potable water generation are impacted by biofouling within the system components which reduce efficiency, or cause failures (Jenner et al. 1998). The introduction of non-indigenous species can significantly increase these impacts and, as a result, the operating costs of maritime infrastructure.

Although biofouling has always been a problem for mariners, new biofouling-mediated introductions continue to pose a threat as ships ply interoceanic trade routes faster and more frequently. Direct economic impacts by non-indigenous species can affect both maritime and non-maritime industries. The green mussel (*Perna viridis*) was first identified in Tampa Bay, USA, in 1999. Since then dense assemblages of these bivalves now clog sea water intakes of power stations and desalination plants, weigh down navigational buoys and foul vessel hulls and engines (Baker et al. 2004 *In* Mitchem et al. 2007). In addition, the ability of some species, for example the tubeworms *Hydroides elegans* and *H. sanctaecrucis,* to now colonise the most common copper-based anti-fouling coatings indicates that biofouling management must continually evolve (Lewis & Smith 1991, Lewis et al. 2006).

Non-indigenous species can impact the environment into which they are introduced by dominating benthic habitats, disturbing native communities, and displacing local species (Schwindt et al. 2001, Branch et al. 2008, Hayward et al. 2008, Reise et al. 2008). This is usually due to the species life history characteristics such as high reproductive output, mass settlement and rapid growth rates (Kado 2003, Lewis 2004, Casas et al. 2008). In addition, the introduced species are potentially freed from the limitations of predation, parasites and diseases.

Disturbed or bare areas in the receiving environment (for example, in ports physical disturbance and pollution can create colonisable spaces) provide sites for initial colonisation and subsequent wider dispersal (Tyrrell & Byers 2007). In addition to physical monopolisation of space, invasive biofouling species can impact native species by shading, sweeping, smothering or consuming larvae and juveniles (Wyatt et al. 2005, Gribben & Wright 2006). Some non-indigenous species can alter the substrates or habitat architecture thereby indirectly changing the physical conditions of the habitat and affecting other organisms (Wallentinus & Nyberg 2007, Sousa et al. 2009). For example, in Germany there has been a shift from native mussels to alien oysters (Kochmann et al. 2008), in Australia the physical structure of Port Philip Bay has been modified by the European fanworm *Sabella spallanzanii* (O'Brien et al. 2006), and in North America the Chinese mitten crab *Eriocheir sinensis* has eroded river and lake embankments (Veldhuizen & Stanish 1999).

Examples of species, likely introduced by hull biofouling, impacting aquaculture include the ascidians *Didemnum vexillum* (Sinner & Coutts 2003, Coutts & Sinner 2004), *Ciona intestinalis*

[4] Once translocated and established in an environment beyond their natural range, species are referred to as non-indigenous regardless of their impact. This term is used throughout the document when not quoting a referenced source. However, it is noted that this terminology is not consistent in the literature (Lockwood et al. 2007). As a consequence, when referring to published literature, the terminology used in that literature will be used to ensure we are not misrepresenting their work.

(Robinson et al. 2006, Hawes et al. 2007) and *Styela clava* (Bourque et al. 2003), the algae *Colpomenia peregrina* (Ribera & Boudouresque 1995), the serpulid tubeworm *Ficopomatus enigmaticus* (Read & Gordon 1991), and the bivalves *Mytilus galloprovincialis* and *Crassostrea gigas* (Dinamani 1971, Apte et al. 2000). Wild fisheries have also been impacted by the introduction of biofouling species including the bryozoans *Membraniporopsis tubigera* and *Biflustra grandicella* (Gordon et al. 2006) and the ascidian *Didemnum* sp. (Valentine et al. 2007).

In general, the impacts to aquaculture include the loss of product, decreased product yield, decreased productivity and decreased handling efficiencies. For example, in Nova Scotia (Canada), the sea squirt *C. intestinalis* has caused up to 50% mortality to blue mussels (*Mytilus edulis*) and incurs a loss of up to C$1.86 for every 1 kg of sea squirt per m^2 (Daigle & Herbinger 2009). The clubbed tunicate (*S. clava*) has also been recorded to cause major economic impacts due to resource competition and smothering resulting in reduced mussel production in Prince Edward Island (Canada) (Davis & Davis 2009). In addition to decreasing stock marketability and yields, sea squirt fouling can increase labour and processing costs, for example, for mussel aquaculture in Prince Edward Island, Canada, these additional costs were conservatively estimated at 19% per pound to market price (Thompson & McNair 2004).

1.1.3. Vessels as a vector

Vessels have long been identified as vectors for the translocation of marine species (Chilton 1910, Allen 1953, Skerman 1960, Rainer 1995). However, the risks associated with the translocation of non-indigenous species in the marine environment were not widely recognised until the latter part of the 20th century (e.g. Carlton & Geller 1993). Until the beginning of the 21st century ballast water was generally considered the most significant mechanism for non-indigenous species translocations. This was because the advent of effective biocidal anti-fouling paints (in particular those tributyltin-based) was thought to have reduced biofouling to a negligible level (Lewis 2004) and the risk of ballast water successfully translocating an entire ecosystem was thought to be extreme (Carlton et al. 1995, Gollasch et al. 2002, Minton et al. 2005). As a consequence, mitigation of ballast water associated biosecurity risk has received attention both domestically (Import Health Standard for Ships' Ballast Water from all Countries, MAFBNZ 2005) and internationally (IMO Ballast Water Management Convention 2004).

Since the 1990's, a growing body of research has demonstrated that vessel biofouling has continued to be a significant vector for non-indigenous species translocations (James & Hayden 2000, Ruiz et al. 2000, Hewitt et al. 2004, Lewis et al. 2006). In New Zealand, Cranfield et al. (1998) determined that between 69% and 90% of known introduced marine organisms were probably associated with biofouling as a means of introduction and, more recently, Kospartov et al. (in press) has refined this figure as up to 87%. Similar assessments in other countries have identified that 70% of marine species introduced to Hawai'i (Eldredge & Carlton 2002), 70% of non-native coastal marine species in North America (Fofonoff et al. 2003), 78% of non-indigenous species in Port Phillip Bay (Hewitt et al. 2004), and 42% of unintentional introductions of marine species to Japan (Otani 2006), are likely to have been introduced by hull fouling.

The type and quantity of vessel biofouling is known to vary with vessel type, maintenance regime, usage and location (Stubbings 1947, Skerman 1960, Rainer 1995, James & Hayden 2000, Lewis 2004). As a consequence, the biosecurity risk posed by hull biofouling can be variable, not only between vessels, but also for a vessel over time and location. However, there appear to be some general differences in factors between different vessel types which allow a division of vessel types to be made for biosecurity purposes. For example, in a review of biofouling entering Australia Hilliard et al. (2006) noted that recreational yachts had very different biosecurity risk profiles to illegal vessels apprehended in Australian waters and, thus, require separate treatment. Similarly, James & Hayden (2000) noted considerable differences in the fouling of recreational yachts relative to cargo vessels.

MAFBNZ commissioned research on the biofouling of international vessels arriving in New Zealand, which has confirmed that the incidence of biofouling is difficult to predict being a complex combination of events (Inglis et al. in prep.). However, there do appear to be some factors that influence the extent of biofouling more than others (Inglis et al. in prep.). The key factors that predicted biofouling presence in this study were those related to vessel maintenance, although other factors of similar predictive value differed between vessel classes. The other factors influencing the prediction of fouling on commercial and passenger vessels were related to design parameters, such as tonnage. However, for recreational vessels factors related to voyage history, such as time in port, contributed more to the predictive ability of the model (Inglis et al. in prep.).

On average New Zealand receives 3,132 ± 80 (mean ± s.d.) vessel arrivals annually (NZCS 2008). Arriving vessels range in size from small (averaging 15 tons) yachts to large (>100,000 tons) tankers. Commercial vessels account for most of the vessel traffic (Table 1.1).

Table 1.1. Annual arrivals by vessel type (NZCS 2008). Annual average (± 1 s.d.) is for the period 2003 to 2007.

Vessel Class		2003	2004	2005	2006	2007	Annual Average (± 1 s.d.)
Commercial vessels	Container	1080	1017	1038	1204	1205	1109 ± 90
	Bulk	485	439	418	426	408	435 ± 30
	General Cargo	321	265	227	184	206	241 ± 54
	Tanker	263	274	253	240	295	265 ± 21
	Other	273	324	317	243	226	277 ± 44
Cruise vessels		38	29	44	56	72	48 ± 17
Fishing vessels		96	72	78	67	75	78 ± 11
Recreational vessels		668	633	660	635	657	651 ± 16
Slow moving and/or specialist craft		17	23	21	43	45	30 ± 13

For the purposes of risk analysis MAFBNZ has defined five classes of vessel:

(1) recreational vessels;
(2) commercial vessels;
(3) cruise vessels;
(4) fishing vessels;
(5) slow moving and/or specialist craft (including Naval vessels).

1.1.4. The New Zealand marine environment

Terrestrial New Zealand is a collection of islands extending over twenty two degrees of latitude. By contrast, New Zealand's marine environment, as defined by the Exclusive Economic Zone (EEZ), is the fifth largest in the world stretching over thirty degrees of latitude and encompasses approximately 4.2 million km^2 (about 14 times the terrestrial landmass) (Gordon 2009). This area contains a diverse range of habitats, biodiversity and ecosystems.

New Zealand's marine environment has been identified as containing 80% of New Zealand's biodiversity. The north-eastern North Island is well connected to subtropical Australian waters, however, the rest of New Zealand is relatively isolated, leading to high levels of endemism (Gordon 2009).

The currents around New Zealand can be coarsely described as a flow of subtropical water from the Tasman Sea circulating around New Zealand's main islands before continuing eastwards. However, with a similarly eastward flow of subantarctic water to the south, the annual average sea surface temperature ranges from 18°C in the northeast of the North Island to 11°C in the southeast of the South Island. The combination of current and temperature environments has led to biogeographic regionalisation (Morton 2004) dividing the north-eastern North Island (Auporian province) from the remaining North and entire South Islands (Cookian province). The significant difference in these two provinces is derived from the influence of the East Australian Current, which flows down into the Auporian province, resulting in warmer waters and a migration of tropical species, many of which have their southern limit off eastern Northland (Morton 2004). In addition, New Zealand is exposed to large storm systems which, combined with the current flows, lead to well mixed coastal waters supporting high productivity (Laing & Chiswell 2003).

For the purpose of this risk analysis New Zealand's marine environment can be refined to the territorial sea which is the current jurisdiction of the Biosecurity Act 1993. Further limitation to the territorial sea around the dominant land masses (North Island and South Island) can be made as these land masses contain all of the official ports of first entry which are exposed to the greatest likelihood of entry and establishment and present the hubs from which any newly established species will disperse. This zone is still extensive with approximately 5,000 km of coastline covering environments ranging from pristine to highly modified, sheltered to exposed, and estuarine to oceanic.

1.2. SCOPE

This risk analysis considers the biosecurity risks from general marine biofouling on the hulls of vessels arriving from international waters. Biofouling may occur on all submerged vessel surfaces including internal niches, such as seachests and bow thrusters.

Organisms which are found exclusively in ballast water, ballast tanks, holds, anchor wells, bait wells, heat exchangers or any structures located above the waterline have been excluded from the scope of this document. Diseases and parasites associated with biofouling organisms have also been excluded due to the need for further scientific research confirming the extent and effects of the transfer of diseases and parasites via non-indigenous marine organisms (Vignon & Sasal 2010), the high diversity of fouling organisms and the diseases and parasites that they may carry, and the time constraints associated with this risk analysis.

An analysis of the options available to manage biofouling is included, however, no specific recommendations are supplied. Different vessel classes may have different risks thus analyses specific to each class will be appended to this analysis as they are completed.

1.3. REFERENCES

Allen, F E (1953) Distribution of marine invertebrates by ships. *Australian Journal of Marine and Freshwater Research* 1: 106-109.

Apte, S; Holland, B S; Godwin, L S; Gardner, J P A (2000) Jumping ship: a stepping stone event mediating transfer of a non-indigenous species via a potentially unsuitable environment. *Biological Invasions* 2: 75-79.

Ashton, G; Boos, K; Shucksmith, R; Cook, E (2006) Risk assessment of hull fouling as a vector for marine non-natives in Scotland. *Aquatic Invasions* 1: 214-218.

Bishop, M W H (1947) Establishment of an immigrant barnacle in British coastal waters. *Nature* 159: 501.

Blum, J C; Chang, A L; Liljesthröm, M; Schenk, M E; Steinberg, M K; Ruiz, G M (2007) The non-native solitary ascidian *Ciona intestinalis* (L.) depresses species richness. *Journal of Experimental Marine Biology and Ecology* 342: 5-14.

Bourque, D; Landry, T; Davidson, J; McNair, N (2003) Impact of an invasive tunicate in Atlantic Canada: Recruitment and competition. *Journal of Shellfish Research* 22: 320.

Branch, G M; Odendaal, F; Robinson, T B (2008) Long-term monitoring of the arrival, expansion and effects of the alien mussel *Mytilus galloprovincialis* relative to wave action. *Marine Ecology Progress Series* 370: 171-183.

Callow, M E; Callow, J A (2002) Marine biofouling: a sticky problem. *Biologist* 49: 1-5.

Carlton, J T; Geller, J B (1993) Ecological roulette: the global transport and invasion of non-indigenous marine organisms. *Science* 261: 78-82.

Carlton, J T; Reid, D M; van Leeuwen, H (1995) *Shipping study – The role of shipping in the introduction of non-indigenous aquatic organisms to the coastal waters of the United States (other than the Great Lakes) and an analysis of control options.* The National Sea Grant Program/Connecticut Sea Grant Project R/ES-6 No. CG-D-11-95.

Casas, G N; Piriz, M L; Parodi, E R (2008) Population features of the invasive kelp *Undaria pinnatifida* (Phaeophyceae: Laminariales) in Neuvo Gulf (Patagonia, Argentina). *Journal of the Marine Biological Association of the United Kingdom* 88(1): 21-28.

Cassé, F; Swain, G W (2006) The development of micro-fouling on four commercial anti-fouling coatings under static and dynamic immersion. *International Biodeterioration and Biodegradation* 57: 179-185

Chilton, C (1910) Dispersal of marine crustacea by means of ships. *Transactions of the New Zealand Institute* 43: 131-133.

Claudi, R; Mackie, G L (1994) *Practical manual for zebra mussel monitoring and control*. Lewis Publishers, Ann Arbor.

Cooksey, K E; Wigglesworth-Cooksey, B (1995) Adhesion of bacteria and diatoms to surfaces in the sea: a review. *Aquatic Microbial Ecology* 9: 87-96.

Coutts, A D M; Sinner, J (2004) *An updated benefit-cost analysis of management options for* Didemnum vexillum *in Queen Charlotte Sound*. Cawthron Report No. 925. July 2004.

Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) *Adventive marine species in New Zealand*. NIWA Technical Report 34.

Dafforn, K A; Glasby, T M; Johnston, E L (2008) Differential effects of tributyltin and copper antifoulants on recruitment of non-indigenous species. *Biofouling* 24: 23-33.

Daigle, R M; Herbinger, C M (2009) Ecological interactions between the vase tunicate (*Ciona intestinalis*) and the farmed blue mussel (*Mytilus edulis*) in Nova Scotia, Canada. *Aquatic Invasions* 4(1): 177-187.

Davis, M H; Davis, M E (2009) *Styela clava* – a new threat to the Mediterranean shellfish industry? *Journal of Aquatic Invasions* 4(1): 283-289.

Dinamani, P (1971) Occurrence of the Japanese oyster, *Crassostrea gigas* (Thunberg), in northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 5: 352-357.

Eldredge, L G; Carlton, J T (2002) Hawai'ian marine bioinvasions: a preliminary assessment. *Pacific Science* 56: 211-212.

Floerl, O (2002) *Intracoastal spread of fouling organisms by recreational yachts*. PhD thesis, James Cook University, Townsville.

Fofonoff, P W; Ruiz, G M; Steves, B; Carlton, J T (2003) In ships or on ships? Mechanisms of transfer and invasion for non-native species to the coasts of North America. In Ruiz, G M; Carlton, J T (Eds) *Invasive Species: Vectors and Management Strategies*. Island Press, Washington pp 152-182.

Glasby, T M (1999) Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney, Australia. *Estuarine, Coastal and Shelf Science* 48: 281-290.

Gollasch, S; MacDonald, E; Belson, S; Botnen, H; Christensen, J T; Hamer, J P; Houvenaghel, G; Jelmert, A; Lucas, I; Masson, D; McCollin, T; Olenin, S; Persson, A; Wallentinus, I (2002) Life in ballast tanks. In Leppäkoski, E; Gollasch, S; Olenin, S (Eds) *Aquatic Invasive Species of Europe – Distribution, Impacts and Management.* Kluwer Academic Publishers, London.

Gordon, D P (Ed) (2009) New Zealand inventory of biodiversity. Volume 1. Kingdom Animalia: Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press, Christchurch, New Zealand.

Gordon, D P; Ramalho, L V; Taylor, P D (2006) An unreported invasive bryozoan that can affect livelihoods - *Membraniporopsis tubigera* in New Zealand and Brazil. *Bulletin of Marine Science* 78: 331-342.

Grant, W S; Cherry, M I (1985) *Mytilus galloprovincialis* in southern Africa. *Journal of Experimental Marine Biology and Ecology* 90: 179-191.

Gribben, P E; Wright, J T (2006) Sublethal effects on reproduction in native fauna: are females more vulnerable to biological invasion. *Oecologia* 149: 352-361.

Hayes, K; Gunasekera, R; Patil, J; Sliwa, C; Migus, S; McEnnulty, F; Dunstan, P; Green, M; Sutton, C (2007) *Empirical validation: Small vessel translocation of key threatening species Stage II —* Undaria pinnatifida*.* Department of Environment and Water Resources, Canberra.

Hayward, B W; Grenfell, H R; Sabaa, A T; Morley, M S (2008) Ecological impact of the introduction to New Zealand of Asian date mussels and cordgrass – the foraminiferal, ostracod and molluscan record. *Estuaries and Coasts* 31: 941-959.

Herborg, L-M; Rushton, S P; Clare, A S; Bentley, M G (2005) The invasion of the Chinese mitten crab (*Eriocheir sinensis*) in the United Kingdom and its comparison to continental Europe. *Biological Invasions* 7: 959–968.

Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B; Boyd, S; Cohen, B F; Currie, D R; Gomon, M F; Keough, M J; Lewis, J A; Lockett, M M; Mays, N; MacArthur, M A; O'Hara, T D; Poore, G C B; Ross, D J; Storey, M J; Watson, J E; Wilson, R S (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology* 144: 183-202.

Hicks, D W; Tunnell, J W (1995) Ecological notes and patterns of dispersal in the recently introduced mussel *Perna perna* (Linne, 1758) in the Gulf of Mexico. *American Malacological Bulletin* 11: 203-206.

Hilliard R; Polglaze, J; LeProvost, I (2006) *Review and evaluation of the biofouling protocol for vessels less than 25 m in length.* Australian Quarantine and Inspection Service, Canberra.

Hoare, K; Beaumont, A R; Davenport, J (1995a) Variation among populations in the resistance of *Mytilus edulis* embryos to copper: adaptation to pollution? *Marine Ecology Progress Series* 120: 155-161.

Hoare, K; Davenport, J; Beaumont, A R (1995b) Effects of exposure and previous exposure to copper on growth of veliger larvae and survivorship of *Mytilus edulis* juveniles. *Marine Ecology Progress Series* 120: 163-168.

Howes, S; Herbinger, C M; Darnell, P; Vercaemer, B (2007) Spatial and temporal patterns of recruitment of the tunicate *Ciona intestinalis* on a mussel farm in Nova Scotia, Canada. *Journal of Experimental Marine Biology and Ecology* 342: 85-92.

IMO (2009) *Development of international measures for minimizing the transfer of invasive aquatic species through biofouling of ships. Research review – biofouling as a mechanism for invasive aquatic species transfer*. Sub-committee on Bulk Liquids and Gases, BLG13/INF.3 International Maritime Organization, London.

James, P; Hayden B (2000) *The potential for the introduction of exotic species by vessel hull fouling: a preliminary study*. NIWA client report WLG00/51

Jenner, H A; Whitehouse, J W; Taylor, C J L; Khalanski, M (1998) Cooling water management in European power stations. Biology and control of fouling. *Hydroécologie Appliquée* 10.

Joint, I; Callow, M E; Callow, J A; Clarke, K R (2000) The attachment of *Enteromorpha* zoospores to a bacterial biofilm assemblage. *Biofouling* 16: 151-158.

Kado, R (2003) Invasion of Japanese shores by the NE Pacific barnacle *Balanus glandula* and its ecological and biogeographical impact. *Marine Ecology Progress Series* 249: 199-206.

Kinloch, M; Summerson, R; Curran, D (2003) *Domestic vessel movements and the spread of marine pests: Risk and management approaches.* Department of Agriculture, Fisheries and Forestry, Canberra.

Kochmann, J; Buschbaum, C; Volkenborn, N; Reise, K (2008) Shift from native mussels to alien oysters: Differential effects of ecosystem engineers. *Journal of Experimental Marine Biology and Ecology* 364: 1-10.

Laing, A; Chiswell, S (2003) The Ocean Medium. In Andrew, N; Francis, M (Eds) *The living reef: the ecology of New Zealand's rocky reefs*. Craig Cotton Publishing, Nelson, New Zealand.

Lam, C; Harder, T; Qian, P-Y (2003) Induction of larval settlement in the polychaete *Hydroides elegans* by surface-associated settlement cues of marine benthic diatoms. *Marine Ecology Progress Series* 263: 225-237.

Lewis, J A (1998) Marine biofouling and its prevention on underwater surfaces. *Materials Forum* 22: 41-61.

Lewis, J A (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research.* Ballast Water Research Series Report No. 14 Department of Agriculture, Fisheries and Forestry, Canberra.

Lewis, J A; Smith, B S (1991) Hydroides settlement in Sydney Harbour (Australia) and its control in sea-water cooling systems. In Rossmoore, H W (Ed), *Biodeterioration and Biodegradation 8*. Elsevier Applied Science, London; pp 464-466.

Lewis, J A; Watson, C; ten Hove, H A (2006) Establishment of the Caribbean serpulid tubeworm *Hydroides sanctaecrucis* Kroyer Morch, 1863, in northern Australia. *Biological Invasions* 8: 665-671.

McFadden, A; Rowden, T; Gould, B (2007) Response to a marine incursion of *Styela clava*. *Surveillance* 34: 4-8.

Minchin, D (1996) Management of the introduction and transfer of marine molluscs. *Aquatic Conservation* 6: 229-244.

Minchin, D (2007) Rapid coastal survey for targeted alien species associated with floating pontoons in Ireland. *Aquatic Invasions* 2: 63-70.

Minchin, D (2007) A checklist of alien and cryptogenic aquatic species in Ireland. *Aquatic Invasions* 2: 341-366.

Minchin, D; Floerl, O; Savini, D; Occhipinti-Ambrogi, A (2006) Chapter 6: Small craft and the spread of exotic species. In Davenport, J; Davenport, J L (Eds) *The ecology of transportation: Managing mobility for the environment.* Springer, The Netherlands.

Minton, M S; Verling, E; Miller, A W; Ruiz, G M (2005) Reducing propagule supply and coastal invasions via ships: effects of emerging strategies. *Frontiers in Ecology and the Environment* 3: 304-308.

Mitchem, E L: Fajans, J S; Baker, S M (2007). Contrasting response of two native crustaceans to non-indigenous prey, the green mussel, *Perna viridis*. *Florida Scientist* 70(2), 180-188.

Morton, J (2004) *Seashore ecology of New Zealand and the Pacific*. David Bateman, Auckland, New Zealand.

NZCS (2008) International vessel arrival and departure information 2000 – 2007. Supplied by the intelligence section of the New Zealand Customs Service in March 2008.

O'Brien, A L; Ross, D J; Keough, M J (2006) Effects of the physical structure of an introduced polychaete, *Sabella spallanzanii* on macrofauna in soft sediments. *Marine and Freshwater Research* 57: 363-371.

Otani, M (2006) Important vectors for marine organisms unintentionally introduced to Japanese waters. In Koike, F; Clout, M N; Kawamichi, M; De Poorter, M; Iwatsuki, K (Eds) *Assessment and control of biological invasion risks*. IUCN, Switzerland; pp 92-103.

Pimentel, D; Zuniga, R; Morrison, D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* 52: 273-288.

Pollux, B; Minchin, D; Van der Velde, G; Van Alen, T; Moon-Van Der Staay, S Y; Hackstein, J (2003) Zebra mussels (*Dreissena polymorpha*) in Ireland, AFLP-fingerprinting and boat traffic both indicate an origin from Britain. *Freshwater Biology* 48: 1127-1139.

Power, A J; Walker, R L; Payne, K; Hurley, D (2004) First occurrence of the non-indigenous green mussel, *Perna viridis* in coastal Georgia, United States*. Journal of Shellfish Research* 23: 741-744.

Qian, P-Y; Lau, S C K; Dahms, H-U; Dobretsov, S; Harder, T (2007) Marine biofilms as mediators of colonization by marine microorganisms: Implications for anti-fouling and aquaculture. *Marine Biotechnology* 9: 399-410.

Qian, P-Y; Thiyagarajan, V; Lau, S C K; Cheung, S C K (2003) Relationship between bacterial community profile in biofilm and attachment of the acorn barnacle *Balanus amphitrite*. *Aquatic Microbial Ecology* 33: 225-237.

Rainer, S F (1995) *Potential for the introduction and translocation of exotic species by hull fouling: A preliminary assessment.* Centre for Research on Introduced Marine Pests, Technical Report 1. CSIRO Australia.

Rajagopal, S; Venugopalan, V P; van der Velde, G; Jenner, H A (2006) Greening of the coasts: a review of the *Perna viridis* success story. *Aquatic Ecology* 40: 273-297.

Read, G B; Gordon, D P (1991) Adventive occurrence of the fouling serpulid *Ficopomatus enigmaticus* (Polychaeta) in New Zealand. *New Zealand Journal of Marine and Freshwater Research* 25: 269-273.

Reise, K; Herre, E; Sturm, M (2008) Mudflat biota since 1930: change beyond return? *Helgoland Marine Research* 62: 13-22.

Ribera, M A; Boudouresque, C F (1995) Introduced marine plants, with special reference to macroalgae: mechanisms and impact. *Progress in Phycological Research* 11: 187-268.

Robinson, T B; Griffiths, C L; McQuaid, C D; Rius, M (2005) Marine alien species of South Africa — status and impacts. *African Journal of Marine Science* 27: 297-306.

Ruiz, G M; Fofonoff, P W; Carlton, J T; Wonham, M J; Hines, A H (2000) Invasion of coastal marine communities in North America: apparent patterns, processes and biases. *Annual Review of Ecology and Systematics* 31: 481-531.

Russell, L K; Hepburn, C D; Hurd, C L; Stuart, M D (2008) The expanding range of *Undaria pinnatifida* in southern New Zealand: distribution, dispersal mechanisms and the invasion of wave-exposed environments. *Biological Invasions* 10: 103-115.

Schaffelke, B; Hewitt, C L (2007) Impacts of introduced seaweeds. *Botanica Marina* 50: 397-417.

Schwindt, E; Bortolus, A; Iribarne, O O (2001) Invasion of reef-builder polychaete: direct and indirect impacts on the native community structure. *Biological Invasions* 3: 137-149.

Sinner, J; Coutts, A D M (2003) *Benefit-cost analysis of management options for* Didemnum vexillum *in Shakespeare Bay*. Cawthron Report No.924.

Skerman, T M (1960) Ship fouling in New Zealand waters: A survey of marine fouling organisms from vessels of the coastal and overseas trade. *New Zealand Journal of Science* 3: 620-648.

Smith, J E; Hunter, C L; Smith, C M (2002) Distribution and reproductive characteristics of non-indigenous and invasive marine algae in the Hawai'ian Islands. *Pacific Science* 56: 299-315.

Sousa, R; Gutierrez, J L; Aldridge, D C (2009) Non-indigenous invasive bivalves as ecosystem engineers. *Biological Invasions* 11: 2367-2385.

Stubbings, H G (1947) Biology of the underwater fouling of ships' hulls. *Journal of Naval Engineering* 1: 41-50.

Thompson, R; MacNair, N (2004) *An overview of the clubbed tunicate (*Styela clava*) in Prince Edward Island*. PEI Department of Agriculture, Fisheries, Aquaculture and Forestry Technical Report 234

Tyrell, M C; Byers, J E (2007) Do artificial substrates favour non-indigenous fouling species over native species? *Journal of Experimental Marine Biology and Ecology* 342: 54-60.

Valentine, P C; Collie, J S; Reid, R N; Asch, R G; Guida, V G; Blackwood, D S (2007) The occurrence of the colonial ascidian *Didemnum* sp. on Georges Bank gravel habitat - Ecological observations and potential effects on groundfish and scallop fisheries. *Journal of Experimental Marine Biology and Ecology* 342: 179-181.

Veldhuizen, T C; Stanish, S (1999) *Overview of the life history, distribution, abundance and impacts of the Chinese mitten crab,* Eriocheir sinensis. Department of Water Resources, Environmental Services Office, Sacramento, California.

Vignon, M; Sasal, P (2010) Fish introduction and parasites in marine ecosystems: the need for information. *Environmental Biology of Fishes* 87:1-8.

Wahl, M (1989) Marine epibiosis. I. Fouling and anti-fouling: some basic aspects. *Marine Ecology Progress Series* 58: 175-189.

Wallentius, I; Nyberg, C D (2007) Introduced marine organisms as habitat modifiers. *Marine Pollution Bulletin* 55: 323-332.

Willan R C; Russell, B C; Murfet, N B; Moore, K L; McEnnulty, F R; Horner, S K; Hewitt, C L; Dally, G M; Campbell, M L; Bourke, S T (2000) Outbreak of *Mytilopsis sallei* (Recluz, 1849)(Bivalvia:Dreissenidae) in Australia. *Molluscan Research* 20(2): 25-30.

Wyatt, A S J; Hewitt, C L; Walker, D I; Ward, T J (2005) Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment, *Diversity and Distributions* 11(1): 33-44.

2. Methodology

The methodology used in this risk analysis was adapted from the MAFBNZ risk analysis procedures (MAFBNZ 2006). These procedures provide a framework which is compliant with the requirements of the World Trade Organisation agreement on the application of sanitary and phytosanitary measures (WTO 1995) and the Biosecurity Act (1993). In this risk analysis, the risk management options are presented prior to the risk assessment of taxonomic groupings, as hull biofouling assemblages are typically managed as a whole rather than being organism/group specific. The hazard identification and risk analysis sections assess the biosecurity risk with each step contingent upon a determination of non-negligible likelihood/consequence in the previous one. The risk analysis conducted is based on currently available information, however, it may be revised as new information becomes accessible.

2.1. RISK MANAGEMENT OPTIONS

Risk management options, in the context of this risk analysis, are techniques used to control or eliminate the accumulation of biofouling either for biosecurity or typical vessel management needs. Published control methods are reported along with an assessment of the efficacy and feasibility of application. However, none of the risk management options are specifically recommended as per the revision of risks analysis procedures (10/10/2007).

2.2. HAZARD IDENTIFICATION AND SCOPING

Biosecurity hazards are organisms which could cause unwanted harm to New Zealand's environmental, economic, social and cultural values. For some commodities, extensive literature and formal lists of associated organisms are available. However, no such list exists for species associated with vessel biofouling and the information on species distributions present in international environments, including ports and marinas, is limited. To construct an initial list of potential hazard organisms a review of all species known to be associated with hull fouling was initiated.

In reviewing known biofouling species it immediately became apparent that a large number (>2,000) of species have been associated with vessel hulls (Anderson et al. 2003, Minchin et al. 2006). As a result, a grouping approach was adopted to make the analysis more manageable. Twenty broad taxonomic groupings, which were loosely based on similarities in traits, were deemed appropriate for the risk assessment (Table 2.1).

Table 2.1. The organism groups selected as representing biofouling organisms. Taxonomy indicates the resolution at which the group was examined. Taxonomy follows Species 2000 (Bisby et al. 2009).

Organism Group	Kingdom	Phylum	Class	Order
Amphipods and Isopods	Animalia	Arthropoda	Malacostraca	Amphipoda
				Isopoda
Bacteria and Viruses	Bacteria			
	Viruses			
Barnacles	Animalia	Arthropoda	Maxillopoda	Sessilia
Bivalves	Animalia	Mollusca	Bivalvia	
Bristleworms	Animalia	Annelida	Polychaeta	
Bryozoa	Animalia	Bryozoa		

Organism Group	Kingdom	Phylum	Class	Order
Crabs	Animalia	Arthropoda	Malacostraca	Decapoda
Echinoderms	Animalia	Echinodermata		
Flatworms	Animalia	Platyhelminthes		
Gastropods and Chitons	Animalia	Mollusca	Polyplacophora	
			Gastropoda	
Hydroids	Animalia	Cnidaria	Hydrozoa	
Macroalgae	Chromista	Ochrophyta		
	Plantae	Chlorophyta		
		Rhodophyta		
Microalgae	Chromista			
	Plantae	Bacillariophyta		
Other Arthropods	Animalia	Arthropoda	Merostomata	
			Malacostraca	
			Ostracoda	
			Pycnogonida	
Peanut worms	Animalia	Sipuncula		
Round worms	Animalia	Nematoda		
Ribbon worms	Animalia	Nemertea		
Sea anemones and corals	Animalia	Cnidaria	Anthozoa	
Sea squirts	Animalia	Tunicata	Ascidiacea	
Sponges	Animalia	Porifera		

Hazard identifications were initially conducted for each of the twenty taxonomic groupings. This determined whether species within the groupings represented a potential hazard to New Zealand via the vessel biofouling pathway. The hazard assessment addressed the following areas of interest in paragraph order:

- broad group taxonomy, number of species and general habitats;
- life-cycle(s);
- ecology and importance;
- known introduced species and their impacts.

From the above assessment, species within the taxonomic group were determined to be a potential hazard if they met the following criteria:

- are known components of marine biofouling assemblages[5];
- are known to have been introduced to new locations;
- have known impacts on core values.

If these criteria were met, then a risk assessment was performed.

2.3. RISK ASSESSMENT

Risk assessment is the evaluation of the likelihood and consequences to environmental, economic, social and cultural values. The aim of the risk assessment was to identify hazards which presented an unacceptable level of risk, for which risk management options would be required. The overall risk estimation was based on three inter-related evaluations:

- the entry assessment;
- the establishment assessment;
- the consequence assessment.

[2] Note: The criteria "are known components of marine biofouling assemblages" *does not specifically* indicate that members within the group are known to foul vessel hulls. This criteria was designed to establish whether or not species within the group are part of biofouling communities in the general marine environment. From this pool of general marine fouling organisms, a smaller group may be capable of fouling vessel hulls. The presence/absence of this smaller group was, where necessary, covered by the entry assessment.

As previously stated, border interception data and in place biosecurity measures support the risk assessments of many commodity pathways. As a result, risk descriptors may be used to indicate a relative level of likelihood, consequence, or risk. However, in the case of vessel hull biofouling, no border interception data and no specific measures exist, therefore, general information regarding organism life-histories and literature specific to the subject of biofouling were used for the risk assessment. Uncertainties and assumptions encountered throughout the risk assessment were identified within the appropriate section.

The entry assessment addressed the following areas of concern:

- occurrence on pathways of vessels entering New Zealand;
- association with vessel biofouling;
    - associations with other taxa;
    - antifouling resistance;
- oceanic survival/tolerance;
- recent records of entry into New Zealand.

From this information the determination of likelihood of entry was made based on the following criteria:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation.

If these criteria were met, it was determined that there was a non-negligible risk of species within the taxonomic grouping entering New Zealand via hull biofouling. As a result, an establishment assessment was performed.

The establishment assessment addressed the following areas of concern:

- suitability of the New Zealand environment;
- reproductive characteristics;
    - dispersal characteristics;
    - propagule pressure;
- records of recent establishment.

From this information the determination of likelihood of establishment was made based on the following criteria:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming established populations in New Zealand;
- have been recently recorded as established in New Zealand.

If these criteria were met, it was determined that there was a non-negligible risk of species within the taxonomic grouping establishing in New Zealand following entry via hull biofouling. As a result, a consequence assessment was performed.

The consequence assessment summarised known existing and potential impacts to the four core values designated by MAFBNZ New Zealand. As New Zealand data regarding the impacts of non-indigenous species are relatively scarce, international impact data was sourced and related back to New Zealand conditions where relevant. The examples of impacts within the consequence assessments, although not necessarily attributed to organisms introduced to those areas via vessel hull fouling, are representative of the potential consequences posed by species within the broad taxonomic groups that may be translocated via vessel hulls.

The risk estimation summarised the entry, establishment and consequence assessments to conclude whether or not organisms within the taxonomic group presented a non-negligible risk to New Zealand's core values. Possible risk mitigation measures were outlined when the outcome of the risk estimation was non-negligible.

A precautionary approach to the application of risk management measures was adopted when there was significant uncertainty in the estimated risk. In these circumstances, the measures proposed were consistent with those groups where equivalent uncertainties exist and should be subject to review as soon as additional information becomes available.

2.4. REFERENCES

Anderson, C; Atlar, M; Callow, M; Candries, M; Milne, A; Townsin, R L (2003) The development of foul-release coatings for seagoing vessels. *Proceedings of the Institute of Marine Engineering, Science and Technology Part B: Journal of Marine Design and Operations* 4: 11-23.

Awad, A; Clarke, C; Greyling, L; Hilliard, R; Polglaze, J; Raaymakers, S (2003) *Ballast water risk assessment, Port of Saldanha Bay, Republic of South Africa, November 2003: Final report*. Globallast Monograph Series No. 13; IMO, London.

Bisby, F A; Roskov, Y R; Orrell, T M; Nicolson, D; Paglinawan, L E; Bailly, N; Kirk, P M; Bourgoin, T; Baillargeon, G (Eds) (2009). *Species 2000 & ITIS Catalogue of Life: 2009 Annual Checklist*. Digital resource at www.catalogueoflife.org/annual-checklist/2009/. Species 2000: Reading, UK.

Coutts, A D M (2002) *A biosecurity investigation of a barge in the Marlborough Sounds*. Cawthron Report 744. July 2002.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kelleher, G; Bleakley, C; Wells, S (Eds) (1995) *A global representative system of marine protected areas*. Vol I – IV. The World Bank; Washington DC, USA.

MAFBNZ (2006) *Risk analysis procedures. Version 1.* MAF Biosecurity New Zealand, Wellington.

Minchin, D; Floerl, O; Savini, D; Occhipinti-Ambrogi, A (2006) *Chapter 6: Small craft and the spread of exotic species*. In Davenport, J; Davenport, J L (Eds) The ecology of transportation: Managing mobility for the environment. Springer; The Netherlands.

NZCS (2008) International vessel arrival and departure information 2000 – 2007. Supplied by the intelligence section of New Zealand Customs Service in March 2008.

WTO (1995) *Agreement on the application of sanitary and phytosanitary measures*. World Trade Organisation; Geneva.

3. Risk management options

Vessel biofouling poses a biosecurity risk through the introduction of non-indigenous species which may have negative impacts on New Zealand's environmental, social, cultural or economic values. The most effective way of eliminating such risk would be to close the border. However, this is not a feasible option because:

- New Zealand is reliant on shipping for 98% of international trade by volume (Statistics New Zealand 2006);
- there is a significant additional economic benefit associated with vessel arrivals, including:
    - tourism;
    - refitting;
    - resupply;
    - port services;
- New Zealand has international obligations to provide a port of refuge to vessels in distress or attempting to escape conditions which may endanger life at sea;
- there is likely to be significant discontent from both domestic and international stakeholders if freedom of movement is curtailed for little obvious gain.

Given the continual arrival of vessels, it is inevitable that non-indigenous species associated with biofouling on vessel hulls will enter New Zealand.

Due to a lack of specific understanding of the biology of most biofouling species and the level of consequence they pose, it is not currently possible to determine an appropriate level of tolerance for vessel biofouling beyond what is practically achievable. As a result, generic options available to minimise or eliminate biofouling to an acceptable level have been identified. No analysis of the risk management objective or treatment preference(s) of any of the management measures is provided.

Risk management can be subdivided into preventative measures and border treatments. Preventative measures will reduce likelihood of translocation prior to a vessels' arrival, however, this option alone may not provide an adequate level of risk mitigation due to the difficulties of fouling prevention. The practical point of intervention for vessel biofouling is where the fouling exceeds a slime layer and becomes complex. The rationale for this is:

- the creation of a slime layer (bacteria, viruses, microalgae) occurs rapidly on immersion and, aside from continuous cleaning, there is no effective preventative technology;
- effective technology to prevent macro-fouling is available and commonly used;
- the greater the density and diversity of fouling, the more likely it is that non-indigenous species will be translocated;
- the biogeography of micro-fouling species is still widely debated.

The potential treatment options are described along with an analysis of their possible efficacy against identified hazards and any residual biosecurity risks. The use of measures to mitigate the risk from specific organism groupings is discussed, where applicable, in the specific risk management sections of those groupings.

3.1. PREVENTION

3.1.1. Vessel maintenance

The impact of biofouling on the operational performance of vessels is well known (Redfield & Ketchum 1952, Callow & Callow 2002). As a consequence, most vessel owners make active attempts to minimise hull biofouling, with the most common management measure being the use of anti-fouling paint. Anti-fouling paints can be coarsely classified into hard, ablative, self-polishing and foul-release categories (Inglis et al. 2008). All but the foul-release paints use biocides, such as copper, to deter fouling, however, the effectiveness of these systems can vary due to:

- the stochastic nature of biofouling (Wahl 1981);
- the presence or absence of species tolerant to the anti-fouling systems (e.g. copper tolerant bivalves (Hoare et al. 1995 a, b) and algae (Russell & Morris 1970, Reed & Moffat 1983));
- the type, age and condition of the anti-fouling coating (Piola & Johnston 2008);
- voyage and vessel characteristics (Lewis 2004).

The use of in-water cleaning to maintain performance, enhance aesthetics and attempt to extend the life of the anti-fouling system has been found to be common for recreational vessels but uncommon for commercial vessels (Inglis et al. in prep.). By contrast, Bohlander (2009) noted that it is common for commercial vessels to complete in-service maintenance procedures such as propeller polishing and, where biofouling is thought to be problematic, in-water cleaning. In many jurisdictions it is difficult to undertake in-water cleaning due to concerns regarding non-indigenous species introductions and the entry of excessive concentrations of anti-fouling biocides into the environment. However, there is a developing understanding that in-service maintenance practices, such as in-water cleaning, could have more beneficial outcomes than risks. For example, a review of the ANZECC code of practice for in-water hull cleaning is currently being undertaken to examine under what circumstances the benefits may exceed the risks.

For most vessel owners the efficacy of an anti-fouling system and any in-service maintenance program is measured in terms of vessel performance. While this almost certainly reduces the potential biofouling load, fouling may accumulate where it is considered to not significantly impact on performance and/or cannot easily be treated, such as the interiors of seachests. From a biosecurity perspective, the presence of biofouling is of concern regardless of the location.

The MAFBNZ commissioned research on vessel biofouling shows that fouled recreational vessels were most likely to have substantial macro-fouling in niche areas, such as the propeller and shaft, while commercial vessels accumulated most fouling around seachest grates and bow thrusters (Inglis et al. in prep.). These findings are similar to those of the Australian Shipowners Association (ASA 2006) which noted that the anti-fouling systems in niche areas are likely to be compromised or absent. In addition, water flows within niche areas may not be laminar and thus create a sanctuary from biocides and physical forces which may otherwise control fouling (ASA 2006). Therefore, even if attention is paid to niche areas during in-water cleaning, they may be more susceptible to reinfestation as the deterrents to settling organisms may be either compromised or absent.

3.2. MITIGATION AT THE BORDER

While it is likely that a combination of an effective anti-fouling system and good vessel husbandry will reduce biofouling risk, there is likely to be a latent risk associated with niche areas. Given that some vessels will not be as well maintained as they could be, it is necessary to consider a range of options to mitigate biofouling when the vessel is inspected at the border.

At present, all vessels are inspected predominantly for hazard organisms or risk goods carried on or in the vessel, however, hull biofouling is not generally considered as a primary option. Inspectors do have the ability to demand that a vessel be hauled out and cleaned if they suspect that the biofouling is significant enough to constitute a likely risk to New Zealand. However, the only method available for the inspector to judge the level of fouling is observation of the hull from the dock or a pontoon. This method has obvious difficulties for the determination of fouling within vessel niche areas. Trials on the use of underwater pole-cameras to examine the biofouling on recreational vessel hulls are currently underway.

3.2.1. Inspection

Identifying marine organisms *in-situ* can be difficult and even expert taxonomic identification in the laboratory can be challenging and time-consuming. In addition, the introduction of non-indigenous species has been associated with the presence of macro-fouling organisms (Inglis et al. in prep.). Based on the outcome of this risk assessment and recognising the need to rapidly clear vessels at the border, it is proposed that any macro-organisms found on the hull of an arriving international vessel are treated as risk organisms. This is consistent with the current preventative anti-fouling system technology which seeks to limit biofouling to no more than a slime layer.

It is often difficult to detect the presence of fouling organisms from the surface due to factors such as water clarity, wharf access and accessibility of niche areas. While the presence of waterline fouling indicates more substantial fouling below the waterline, the absence of waterline fouling does not indicate the absence of substantial fouling elsewhere on the hull (IMO 2007). Given that fouling is most likely to accumulate in niche areas, it is suggested that, where equipment is available, inspection for macro-fouling entails a rapid visual inspection of the hull from the waterline followed by an underwater inspection of the stern area focusing on any grates and appendages such as the rudder, propeller and driveshaft. The presence of macro-fouling organisms can be considered to indicate the existence of a biosecurity risk which needs to be managed.

Current understanding of the efficacy of inspection to characterise vessel biofouling is limited. As a consequence, validation of inspection decisions needs further development. This can be achieved by auditing the initial vessel inspection decision at the border against a thorough inspection of biofouling (Hilliard et al. 2006). The auditing process should include a random inspection of vessels deemed to pose a negligible risk as well as all vessels directed to treat their biofouling (Hilliard et al. 2006).

The collection of additional information on vessel characteristics will likely aid future inspection efficacy. At present there are no clear indicators of biofouling risk, such as voyage history or maintenance regime, for international vessels entering New Zealand. One reason for this may have been the relatively small sample size of the MAFBNZ commissioned research on vessel biofouling which targeted 15% of incoming vessels. It may be that with further information better predictors of risk will become evident. To support the development of vessel risk profiles, consideration should be given to the requirement of vessels to supply verifiable documentation of their voyage history, maintenance regime and/or biofouling management plan. A communications plan informing vessel owners/masters of the benefits of keeping accurate records may be put in place to support this approach.

3.2.2. In-water treatments

In-water treatments refer to those treatments carried out while the vessel remains in the water. This may occur at the quarantine wharf or require moving the vessel to a specific location designated for such activity.

3.2.2.1. Removal by hand

In-water removal of biofouling organisms by hand is common on smaller vessels and is generally carried out by a diver using tools, such as paint scrapers (Floerl et al. 2005c). This is an effective method for removing some organisms, especially where they are in isolated patches and can be contained with a low risk of viable propagules being released. For example, this approach was successfully used in the Lyttelton port in response to the entry of *Sabella spallanzanii*.

While removal by hand may be effective in removing visible organisms, it may not eliminate the risk. For example, Floerl et al. (2005c) found that 72% of organisms removed from vessel hulls by hand scraping remained viable after removal. Similarly, Woods et al. (2007) recorded 72% (summer) and 66% (winter) of removed organisms (excluding tubiculous polychaetes) as still viable one hour after in-water hand scraping. As a consequence, to be effective as a biosecurity treatment option, hand removal must contain all removed organisms and any potential propagules. This is most likely to be possible where the organisms in question are large and easily removed. However, the success of this procedure may be reduced as fouling assemblages:

- become more complex;
- contain organisms likely to release propagules during removal;
- contain a larger proportion of organisms strongly affixed to the surface;
- are present in difficult to access areas.

3.2.2.2. Mechanical cleaning

Mechanical in-water cleaning technologies range from water blasters to automated rotating brush systems. Generally, these systems are used on larger commercial vessels due to factors including:

- expense (both to purchase and operate);
- availability (often only a single operator in a region);
- potential to damage the hull (in particular composite hulls where fibre damage or delamination may be induced);
- ineffectiveness over curved surfaces (most mechanical systems are designed to cover the hull of large vessels where most panels are flat or gently curved).

Rotating brushes are the most common mechanical cleaning systems and have typically been developed for naval or commercial vessels. However, as yet, none of the in-water systems currently available capture 100% of debris removed during the cleaning process (Bohlander 2009). Moreover, they are not usually effective at cleaning the entire hull due to an inability to conform to curves and/or access areas with appendages, such as around the vessels stern (Floerl et al. 2005c, Hopkins et al. in press). In-water water blasting could avoid some of these issues, however, the equipment is specialised, difficult to obtain and difficult to use safely. It has also been found that a significant proportion of organisms dislodged from the hull by in-water water blasting will survive (MAFBNZ 2009).

3.2.2.3. Encapsulation

This method uses an impervious material to cover a vessel hull in order to reduce the water volume surrounding the hull thereby creating toxic conditions for the fouling organisms. Several studies have found this to be an effective method of treating fouling organisms (McEnnulty et al. 2001, Coutts & Forrest 2007). There are two methods commonly used for encapsulation. The first is where an appropriate material (commonly silage wrap) is wrapped around the vessel to provide a temporary, single use, cover. The second method is where a shroud, suspended beneath inflatable pontoons, is drawn around the vessel. Such systems are usually custom built and reusable (e.g. http://www.seapen.com.au). In both cases the water between the encapsulation and the hull is evacuated to expedite the process of attaining anoxia. It is possible to further accelerate the process by the addition of freshwater or chemicals, such as acetic acid or hypochlorite.

The chief benefits of encapsulation are the indiscriminate killing of fouling organisms such as those in hard to access niche areas, including internal seawater systems, and the ability to rapidly deploy in most locations. The deployment of encapsulation technology can be especially rapid where reusable systems are available (i.e. manoeuvre the vessel into the shroud). The availability of equipment and personnel to deploy an encapsulation method will determine the feasibility of use in most locations.

The effective use of encapsulation as a treatment option requires further development in order to be able to specify the following:

- effective kill times;
- effective dosages for additives;
- the appropriate use of any additives (environmental and anti-fouling system concerns);
- safe containment and disposal of debris;
- the latent risk of vessel fouling[6] (e.g. barnacle tests).

3.2.2.4. Heat treatment

Heat has been used as a biofouling control measure for many different applications including the internal piping systems of power plants (Rajagopal et al. 1994). To be effective, heat treatments need to achieve temperatures exceeding 45°C throughout the entire fouling assemblage for an appropriate period of time[7] (Wallentinus 1999, McEnnulty et al. 2001). However, for effective killing the heat must be evenly applied

[6] While fouling organisms are killed during treatment their calcareous tests can remain attached to the vessel hull. Once the encapsulation is removed and the hull is exposed to open sea-water again the tests provide an attachment site for fouling organisms and facilitate the growth of biofouling. The latent risk comes from larvae, spores, and organisms present in the area settling upon or sheltering in the tests and being subsequently translocated. The presence of vacated tests or dead fouling structures can be an important vector for the secondary spread of established non-indigenous species.

[7] As the temperature increases the exposure time needed to kill respective organisms decreases. For example, to make cysts of the dinoflagellate *Gymnodinium catenatum* inactive they can be exposed to water of 38-40°C for 120 seconds or water of 45-47°C for 30 seconds (Bolch & Halegraeff 1993 *In* McEnnulty et al. 2001).

which is difficult for dense assemblages. Difficulties may also arise due to the large thermal buffering capacity of seawater, the nature of the substrate and the nature of the fouling itself. Whilst engineering solutions can overcome some of these difficulties, uniform application over dense assemblages is difficult to manage.

On vessel hulls, the only reported use of heat treatment was for the removal of an *Undaria pinnatifida* infestation on a sunken vessel (Wooten et al. 2004). While successful, this method was found to be resource intensive and expensive. A developing technology is the use of heat treatment for in-service maintenance. That is, heat treatment applications are made to eliminate newly settled fouling organisms to prevent the establishment of complex fouling assemblages (http://www.commercialdiving.com.au).

3.2.2.5. Freshwater

Immersion in freshwater has been suggested as a biofouling treatment by, for example, the sailing of a vessel into a freshwater river environment. However, due to the risks of organisms releasing or spawning while being moved from marine to fresh waters, this method appears to be more appropriate for pre-departure, i.e. a vessel deemed clean on arrival but needing to ensure it is clean before arriving at the next destination. One instance of this method has been documented as a pre-departure treatment (Brock et al. 1999). While it achieved a significant reduction in the biofouling biomass, it failed to stop the introduction of a new species at its destination (Apte et al. 2000). Factors likely to have reduced success include the water not being entirely fresh (salinity 6-10 ppt) and the short exposure period (9 days). Freshwater may be a useful treatment option prior to departure if the exposure salinity and time are appropriate, however, it should be noted that many fouling species have wide environmental tolerances so an extended period may be required. The efficacy of freshwater treatment could be enhanced if used in combination with other methods, such as scraping to reduce the bulk of the hull fouling. Freshwater exposure has the significant benefit of being able to reach all niche areas including internal seawater systems.

In New Zealand the use of freshwater as a treatment option, without encapsulation, faces some constraints. While many New Zealand ports are located within estuaries the navigable portion of the estuary is typically saline (>10 ppt) and often not appropriate for larger vessels because of physical (e.g. depth) and/or use (e.g. aquaculture) constraints. Westport is currently the only port of first arrival likely to meet the requisite conditions of freshwater (<5 ppt) in the port area. Creating a freshwater lock in port environments may be possible for some locations, however, this option may be extremely costly in comparison with other hull fouling treatments.

3.2.3. Land based treatments

The removal of vessels from the water can be achieved by methods ranging from travel lifts to dry docks. In New Zealand such facilities are generally only available for smaller vessels with the two major dry dock facilities being in Auckland (180 m length, 24 m beam) and Lyttelton (130 m length, 14 m beam).

The process of removing a vessel from the water can have an associated likelihood of entry as organisms can be released into the receiving environment during vessel removal. This is especially so when travel- or synchro-lifts or slipways are used. However, unlike in-water cleaning, haul outs occur at discrete locations allowing targeted monitoring and mitigation of possible biosecurity risks. Moreover, most facilities are located near significant ports (e.g. Ashby's shipyard in Opua is only a few hundred metres from the quarantine wharf) that are subject to biosecurity surveillance programs.

Land based treatments may result in the release and subsequent establishment of non-indigenous organisms by allowing viable material in the effluent stream back into the sea (McClary & Nelligan 2001, Floerl et al. 2005c, Woods et al. 2006). Although most boatyards have coarse gratings to capture large debris and paint chips, this will not stop the release of smaller viable propagules such as gametes. At present there are no MAFBNZ certified boatyards, however, it is likely that, in future, boatyards performing waterblasting services for biosecurity purposes will have to comply with the requirements of the MAFBNZ transitional facility standard. For those vessels that can be removed from the water there are three methods with which biofouling can be treated.

3.2.3.1. Scraping

Tools ranging from a hand scraper to a shovel can be effective in rapidly removing large scale fouling from a vessel hull. Scraping is commonly used on vessels that require fouling removal prior to being placed on a hard stand for maintenance. The efficacy of this treatment is largely dependant upon the operator and the types of tools used. Having a vessel out of the water makes it easier for the operator to observe and remove fouling. However, removing some types of encrusting organisms with a scraper can lead to hull damage, such as gouging. Moreover, scraping is likely to remove only easily visible fouling, missing small but viable fragments and potentially leaving settlement substrates, such as remnant barnacle tests (Floerl et al. 2005b).

The key benefits of scraping are that, following haulout and inspection, scraping can be completed rapidly with tools which are readily accessible[8]. Assuming facilities are available, a moderately fouled (<20% of the hull covered in macro-fouling) vessel could be returned to the water in less than an hour.

Removing fouling on land has been shown to reduce the proportion of viable organisms in the debris relative to in-water cleaning (Floerl et al. 2005c) due to the influences of desiccation and trampling. However, there is a latent risk associated with viable organisms in the debris, therefore, it must be ensured that the debris does not return to the water. This is a requirement for any facility complying with the MAFBNZ transitional facility standard.

3.2.3.2. Waterblasting

Waterblasting is a common technique for fouling removal prior to further maintenance. Waterblasting entails spraying water under pressure (2,000 psi or greater) from a lance, often with the water exiting in a triangular, scraper-like pattern. This system enables the operator to systematically clear the hull of fouling with less physical effort, more likelihood of removing microscopic fouling, and less likelihood of damaging the hull. However, some anti-fouling systems can be adversely affected by waterblasting. The efficacy of this treatment is dependant upon the diligence of the operator.

For recreational vessels it is possible to determine the time and cost associated with standard waterblasting. Ashby's boatyard in Opua, for example, advertise a service for

[8] Compared to other mitigation options such as rotating brushes, encapsulation, and heat treatment, which are still very much in the process of development and require specialised equipement, all that scraping requires is often a common paint scraper.

removing a vessel from the water on the travel lift, waterblasting the hull and returning it to the water for less than $600.00 for a 20.3m vessel (http://www.ashbyboats.co.nz accessed 10/03/2009). However, the costs will be substantially greater for vessels requiring more specialised haul out facilities or dry dock space.

To ensure waterblasting does not result in the release of organisms into the environment, the debris removed from the hull and the liquid effluent need to be contained and filtered.

3.2.3.3. Desiccation

Desiccation, or air-drying, is a technique where the vessel is hauled out and left on a hard stand until all fouling is rendered non-viable. Although desiccation is most commonly used to treat fouling on aquaculture and fishing equipment, this technique has been recognised as having potential for controlling vessel biofouling in the biosecurity context (Forrest et al. 2004, Coutts & Forrest 2005, Hilliard et al. 2006). The benefits of this technique are generally considered to be that it is easy to perform and does not require the use of any environmentally damaging additives.

The drawbacks of dessication, however, are considerable. Hilliard et al. (2006) found that "there is no set time which can be considered reliable for applying to all parts of a hull" and thus recommended that the minimum period be 21 days, with any reduction requiring expert analysis of the fouling on a case-by-case basis. Given the time and conditions likely to be required for effective dessication it is unlikely that it will be as cost effective as other management options.

In addition, the vessel location during dessication would need to be away from any sources of moisture, including rainfall, sea spray and high humidity. These criteria are difficult to achieve at most hard stand facilities. Moreover, there will be issues of odour as the fouling rots and this would need to be managed to comply with local regulations.

Determining organism viability so a vessel can be declared risk free is also difficult. Although a fouling organism may not survive the desiccation process, it could still release viable propagules once returned to the water, e.g. *Undaria pinnatifida* (Saito 1975). In addition, the presence of dead material on the hull provides an excellent settlement surface for biofouling reinfestation of the hull.

3.3. REFERENCES

ASA (2006) *Assessment of introduced marine pest risks associated with niche areas in commercial shipping*. Australian Shipowners Association, Melbourne, Australia.

Aquenal (2008) *IMProtector: A quarantine and treatment system for apprehended illegal foreign fishing vessels*. Aquenal Pty Ltd; Tasmania, Australia.

Bax, N; Hayes, K; Marshall, A; Parry, D; Thresher, R (2002) Man-made marinas as sheltered islands for alien marine organisms: Establishment and eradication of an alien invasive marine species. In Veitch, C R; Clout, M N (Eds) *Turning the Tide: The Eradication of Invasive Species*. IUCN SSC Invasive Species Specialist Group; Auckland, New Zealand; pp 26-39.

Bohlander, J (2009) *Review of options for in-water cleaning of ships*. MAF Biosecurity New Zealand Technical Paper No: 2009/42. http://www.biosecurity.govt.nz/files/pests/salt-freshwater/options-for-in-water-cleaning-of-ships.pdf

Brock, R; Bailey-Brock, J H; Goody, J (1999) A case study of efficacy of freshwater immersion in controlling introduction of alien marine fouling communities: the USS Missouri. *Pacific Science* 53: 223-231.

Callow, M E; Callow, J A (2002) Marine biofouling: a sticky problem. *Biologist* 49(1): 1-5.

Coutts, A D M (2002) *A biosecurity investigation of a barge in the Marlborough Sounds*. Cawthron Report No 744. Cawthron Institute; Nelson, New Zealand.

Coutts, A D M; Forrest, B M (2007) Development and application of tools for incursion response: lessons learned from the management of a potential marine pest. *Journal of Experimental Marine Biology and Ecology* 342: 154-162.

Creese, R G; Davis, A R; Glasby, T M (2004) *Eradicating and preventing the spread of the invasive alga* Caulerpa taxifolia *in NSW*. NSW Fisheries Final Report No. 64, NSW Fisheries; Cronulla, New South Wales, Australia.

Davidson, I C; McCann, L D; Sytsma, M D; Ruiz, G M (2008) Interrupting a multi-species bioinvasion vector: The efficacy of in-water cleaning for removing biofouling on obsolete vessels. *Marine Pollution Bulletin* 56: 1538-1544.

Denny, C M (2007) *In-situ plastic encapsulation of the NZHMS "Canterbury" frigate: a trial of a response tool for marine fouling pests*. Cawthron Report No 1271. Cawthron Institute; Nelson, New Zealand.

Ferguson, R (2000) *The effectiveness of Australia's response to the black striped mussel incursion in Darwin, Australia*. A report of the marine pest incursion management workshop, Darwin.

Floerl, O; Inglis, G J; Hayden, B (2005a) A risk-based predictive tool to prevent accidental introductions of non-indigenous marine species. *Environmental Management* 35: 765-778.

Floerl, O; Inglis, G J; Marsh, H M (2005b) Selectivity in vector management: an investigation of the effectiveness of measures used to prevent transport of non-indigenous species. *Biological Invasions* 7: 459-475.

Floerl, O; Norton, N; Inglis, G; Hayden, B; Middleton, C; Smith, M; Alcock, N; Fitridge, I (2005c) *Efficacy of hull cleaning operations in containing biological material I. Risk assessment*. MAF Biosecurity New Zealand Technical Paper 08/12, MAF Biosecurity New Zealand Wellington.

Forrest, B M; Blakemore, K A (2006) Evaluation of treatments to reduce the spread of a marine plant pest with aquaculture transfers. *Aquaculture* 257: 333-345.

GISP (2008) *Marine Biofouling: An assessment of risks and management initiatives*. Global Invasive Species Programme and the UNEP Regional Seas Programme.

Godwin, L (2004) Development of an initial framework for the management of hull fouling as a marine invasive species transport mechanism. Proceedings of a workshop on current issues and potential management strategies. Feb. 12 – 13, Honolulu, Hawai'i.

Hay, C H (1990) The dispersal of sporophytes of *Undaria pinnatifida* by coastal shipping in New Zealand, and implications for further dispersal of *Undaria* in France. *British Phycological Journal* 25: 301-313

Hewitt, C L; Campbell, M L (2007) Mechanisms for the prevention of marine bioinvasions for better biosecurity. *Marine Pollution Bulletin* 55: 395–401

Hilliard, R; Polglaze, J; LeProvost, I (2006) *Review and evaluation of the biofouling protocol for vessels less than 25m in length.* Australian Quarantine and Inspection Service, Canberra

Holm, E R; Haslbeck, E G; Horinek, A A (2003) Evaluation of brushes for removal of fouling from fouling- release surfaces, using a hydraulic cleaning device. *Biofouling* 19: 297–305.

Hopkins, G A; Forrest, B M (2008) Management options for vessel hull fouling: an overview of risks posed by in-water cleaning. *ICES Journal of Marine Science* 65: 811-815.

Hopkins, G A; Forest, B M; Coutts, A (In Press) *Determining the efficacy of incursion response tools: rotating brush technology (coupled with suction capability)*. Biosecurity New Zealand Technical Paper, MAF Biosecurity New Zealand, Wellington, New Zealand

Inglis, G; Ponder-Sutton, A; Unwin, M; Floerl, O (2008) *Vessel biofouling as a vector for the introduction of non-indigenous marine species to New Zealand: Management Tools*. Final report for MAF Biosecurity New Zealand Research Project ZBS2004/04. MAF Biosecurity New Zealand, Wellington

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kalumuck, K M; Chahine, G L; Frederick, G S; Aley, P D (1997) Development of a DynaJet™ cavitating water jet cleaning tool for underwater marine fouling removal. 9th American Waterjet Conference

Lewis, J A (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research.* Ballast Water Research Series Report No. 14 Department of Agriculture, Fisheries and Forestry, Canberra

MAFBNZ (2009) *Perna perna* http://www.biosecurity.govt.nz/pests/perna-perna Accessed 20 April 2009.

McClary, D; Nelligan, R (2001) Alternative biosecurity management tools for vector threats: technical guidelines for acceptable hull cleaning facilities. Ministry of Fisheries Project Report ZBS2000-03, Ministry of Fisheries, Wellington, New Zealand.

McEnnulty, F R; Bax, N J; Schaffelke, B; Campbell, M L (2001) *A review of rapid response options for the control of ABWMAC listed introduced marine pest species and related taxa in Australian waters*. Centre for Research on Introduced Marine Pests, CSIRO Marine Research, Hobart.

Piola, R; Johnston, E (2008) The potential for translocation of marine species via small-scale disruptions to anti-fouling surfaces. *Biofouling* 24: 145-155.

Rajagopal, S; Azariah, J; Nair, K V K (1994) Heat treatment as a fouling control method for Indian coastal power plants. In Thompson, M F; Nagabhushanam, R; Sarojini, R; Fingerman, M (Eds) *Recent developments in biofouling control.* Oxford and IBH; New Delhi, India; pp 391-396.

Redfield, A C; Ketchum, B H (1952) *Ship resistance*. In Redfield, A C; Ketchum, B H (Eds) Marine Fouling and Its Prevention. U.S. Naval Institute; Annapolis, USA.

Reed, R H; Moffat, L (1983) Copper toxicity and copper tolerance in *Enteromorpha compressa* (L.) Grev. *Journal of Experimental Marine Biology and Ecology* 69: 85-103.

Russell, G; Morris, O P (1970) Copper tolerance in the marine fouling alga *Ectocarpus* siliceous. *Nature* 228: 288-289.

Saito, Y (1975) *Undaria*. In Tokida, J; Hirose, H (Eds) *Advance of phycology in Japan*. Dr. W. Junk b.v.; The Hague; pp.304-320;

Statistics New Zealand (2006) *New Zealand's marine economy 1997 – 2002*. Statistics New Zealand Environmental Series. Statistics New Zealand, Wellington.

Ten Hallers-Tjabbes, C C (2007) *Environmentally sound and safe removal of harmful anti-fouling systems and of cleaning of ships' hulls.* SAFEMED Project Task 3.8. Final

Report prepared under the Project EUROMED Cooperation on Maritime Safety and Prevention of Pollution from Ships SAFEMED

Wallentinus, I (1999) Exotics across the ocean - *Undaria pinnatifida* (Harvey) Suringar, 1872. In Gollasch, S; Minchin, D; Rosenthal, H; Voigt, M (Eds) *Exotics across the ocean: Case histories on introduced species: their biology, distribution, range expansion and impact*. Institute for Marine Science, University of Kiel, Germany pp.11-19

Woods, C; Floerl, O; Fitridge, I; Johnston, O; Robinson, K; Rupp, D; Davey, N; Rush, N; Smith, M (2007) *Efficacy of hull cleaning operations in containing biological material. II. Seasonal variability*. MAF Biosecurity New Zealand Technical Paper Series 08/11. MAF Biosecurity New Zealand, Wellington

Wotton, D M; Hewitt, C L (2004) Marine biosecurity post-border management: developing incursion response systems for New Zealand. *New Zealand Journal of Marine and Freshwater Research* 38: 553-559.

Wotton, D M; O'Brien, C; Stuart, M D; Fergus, D J (2004) Eradication success down under: heat treatment of a sunken trawler to kill the invasive seaweed *Undaria pinnatifida*. *Marine Pollution Bulletin* 49: 844-849.

4. Risk assessment

4.1. AMPHIPODS AND ISOPODS (ORDER: AMPHIPODA AND ISOPODA)

4.1.1. Hazard Identification

Amphipods and isopods are typically small (1-28 mm long) crustaceans that are widely distributed, diverse, abundant, and important components of aquatic ecosystems (Ruppert et al. 2004). The order Amphipoda covers over 6,000 species while the order Isopoda has approximately 4,000 species (Ruppert et al. 2004). Although most species inhabit marine environments, amphipods and isopods are also found in estuarine and freshwater habitats. In addition, both orders have terrestrial species. The morphological distinction between amphipods and isopods is that amphipods are compressed laterally, giving them a shrimp-like appearance, whilst isopods are dorsoventrally flattened.

The amphipod and isopod life-cycle typically involves a short life-span (~1 year), with the females spawning multiple times (Ruppert et al. 2004, Buschbaum & Gutow 2005). Members of both orders brood their eggs, clutch sizes ranging from 2–750 eggs, from which the young hatch as miniature adults (Ruppert et al. 2004). These organisms do not disperse widely outside of their native ranges and their introduction to new environments has been linked to either hull fouling or rafting[9] (Talley et al. 2001, Buschbaum & Gutow 2005, Davidson et al. 2008, Montelli & Lewis 2008).

Ecologically, amphipods and isopods play an important role because they often exhibit high densities and diversity (Ruppert et al. 2004, Davidson et al. 2008). Though generally omnivorous detritus feeders or scavengers, both orders contain predatory and parasitic species (Ruppert et al. 2004, Montelli & Lewis 2008). As predators and micro-grazers, some species can influence the size distribution, composition, and population dynamics of other epifaunal, planktonic, and algal assemblages (Hooff & Bollens 2004, Marion et al. 2008, Orav-Kotta et al. 2009). Amphipods and isopods are also an important food source for larger animals (Marion et al. 2008). These organisms commonly live in association with habitat forming fouling organisms, such as algae, barnacles, and bryozoans (Ruppert et al. 2004, Montelli & Lewis 2008, Guerra-Garcia & Tierno de Figueroa 2009).

Non-indigenous species of amphipods and isopods, for example, *Caprella mutica*, *Gammarus tigrinus*, *Sphaeroma quoianum* and *S. walkeri,* are known to modify habitat, alter food webs, and regulate community structure (Cook et al. 2007, Marion et al. 2008, Packalen et al. 2008, Piscart et al. 2008, Orav-Kotta et al. 2009). Some isopods, such as members of the family Limnoriidae, can cause severe impacts by their boring behaviour and thus affect marine plants, lateral banks and wooden structures (Borges et al. 2008, Brearley et al. 2008, Davidson et al. 2008). By contrast, amphipods and isopods have been observed to occur in high densities and abundances with no impacts identified. For example, *S. walkeri* is noted to occur in high densities on aquaculture cages with no apparent impacts to the cultured organisms (Mak et al. 1985).

Uncertainties regarding the risk presented by amphipods and isopods include a taxonomy that is currently poorly known, and limited records of distributions, biology or ecology (Hewitt et al. 1999, Chapman & Carlton 1991, Chapman & Carlton 1994, Bollens et al. 2002, Montelli

[9] Organisms or eggs can attach to floating debris (organic or inorganic) and be carried to new lovcations by currents.

& Lewis 2008). As a result, it is difficult to determine the extent of amphipod and isopod introductions, impacts, local resilience, and importance to ecosystem structure and function beyond the few studied species.

Given that amphipods and isopods:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that amphipod and isopod species represent a potential hazard via vessel biofouling.

4.1.2. Risk Assessment

4.1.2.1. Entry assessment

Amphipods and isopods are common and widely distributed in soft and hard substrate communities and are reported to occupy sandy beach, rocky shore, continental shelf and continental slope environments (Ruppert et al. 2004, Montelli & Lewis 2008). In the coastal environment they are often associated with the natural and artificial substrates of ports and marinas. Their mobile nature and swimming ability enables them to move from benthic environments onto vessel hulls.

As components of hull fouling assemblages, amphipods and isopods are often found living on or amongst other hull fouling species which provide food and/or shelter (Buschbaum & Gutow 2005, Martinez & Adarraga 2008, Montelli & Lewis 2008). The associations between amphipods, isopods and other hull fouling species can be specific. For example, Montelli & Lewis (2008) found isopods were mostly associated with acorn barnacles while caprellid amphipods were mostly associated with filamentous algae. However, tube building amphipods have been observed to be the only macroscopic organisms living on anti-fouling paints under specific test conditions (Lewis 2004). Similarly, boring isopods are likely to be associated with wooden hulls regardless of the presence of other fouling species. While no longer common, there remains a small but consistent influx of wooden hulled recreational vessels into New Zealand[10].

Amphipods and isopods are known to survive translocation on the hulls of vessels. For example, Montelli & Lewis (2008) reported that many species have been translocated on the hulls of Australian Naval vessels from both temperate and tropical waters. In the MAFBNZ commissioned research on biofouling Inglis et al. (in prep.) found that, of international vessels entering New Zealand, 21% had amphipod and 5% had isopod fouling the hull, respectively. Of those species identified, five amphipods and six isopods were not known to be established in New Zealand. These examples show that some hull fouling amphipods and isopods can survive transport from international locations.

Given that particular amphipod and isopod species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous amphipod and isopod species into New Zealand via vessel biofouling is non-negligible.

[10] 11 recreational vessels with wooden hulls, from a sampling total of 182 recreational vessels, were recorded over two years by Inglis et al. (in prep.).

4.1.2.2. Establishment assessment

As amphipods and isopods are capable swimmers they could, given the right stimulus, migrate from a vessel hull to the receiving environment. As some species can have high site fidelity, the stimulus for disembarking may need to be quite strong, for example, the physical dislodgement of associated fouling. However, amphipods and isopods, in general, are indiscriminate regarding the substratum and fouling community they inhabit. As a result, establishment of these organisms is more likely to be influenced by physical conditions (with temperature a known limitation) than substrate type (Ingolfsson & Agnarsson 2003, Marion et al. 2008, Montelli & Lewis 2008).

Port environments are usually sheltered locations with high detritus content, providing a suitable habitat for many species of amphipod and isopod. As observed with the establishment of *C. mutica* in Europe, ports and artificial hard substrates may act as initial points for founding populations (Buschbaum & Gutow 2005). After establishment, it is apparent that amphipod and isopod species can subsequently spread throughout the local estuary or immediate coastal area (Cook et al. 2007, Davidson et al. 2008, Marion et al. 2008)

The establishment of certain species may depend upon the fouling assemblages already present due to their specific associations with a particular form or species of macro-fouling e.g. caprellid amphipods and filamentous algae (Montelli & Lewis 2008). However, appropriate habitats, to support newly arrived individuals, are likely to be common in New Zealand port environments.

Once relocated, the ability of amphipods and isopods to establish sustaining populations is supported by their life history characteristics. Specifically, the ability to spawn multiple times with each event potentially producing hundreds of offspring (Ruppert et al. 2004, Cook et al. 2007). For example, the invasion success of *G. tigrinus* may be dependent on the higher reproductive capacity*,* with regard to wider temperature and salinity ranges, compared to the native species (Pinkster et al. 1977 *In* Grabowski et al 2007, Savage 1982 *In* Grabowski et al 2007). In addition, *G. tigrinus* has the ability to thrive in heavily polluted waters relative to native species (Pinkster et al. 1977 *In* Grabowski et al 2007).

Known introductions of amphipods and isopods have occurred in Australia (Poore & Storey 1999) and New Zealand (Kospartov et al. in press). Of the 17 non-indigenous amphipods and 10 non-indigenous isopods recorded in New Zealand, 11 amphipods and two isopods are established and all are thought to have been vectored by vessel biofouling (Kospartov et al. in press).

Given that particular amphipod and isopod species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous amphipod and isopod species in New Zealand via vessel biofouling is non-negligible.

4.1.2.3. Consequence assessment

The consequences of amphipods and isopods establishing in New Zealand could be wide ranging. High abundances of amphipod and isopod species have been shown to affect zooplankton, benthic habitats and communities. For example, *G. tigrinis,* introduced to

Finland, has outcompeted and replaced native herbivorous *Gammarus* species (Packalen et al. 2008). Similarly, *G. tigrinus* has the potential to outcompete the native *G. salinus* within *Pilayella littoralis* dominated macrophyte beds as, in laboratory experiments, the presence of *G. tigrinus* within *Pilayella littoralis* induced a higher mortality of *G. salinus* (Orav-Kotta et al. 2009). Therefore, the recent blooms of *P. littoralis* are expected to favour the formation of *G. tigrinus* dominated communities in the coastal range of the northern Baltic Sea. The removal of native amphipods and isopods as intermediaries in the food chain may have flow on effects for the organisms dependant upon them.

Boring isopods have relatively well known impacts on marine structures and habitat. For example, *S. terebrans* bores into mangroves and *S. quoianum,* a wood borer in its native Australian range, bores into banks in North America, accelerating erosion in estuaries (Talley et al. 2001, Davidson et al. 2008). Talley et al. (2001) observed a correlation between the rate of lateral bank loss and density of *Sphaeroma* spp. burrows, whereby *Sphaeroma* spp. could increase the rate of sediment loss by as much as 240%. In some areas where *Sphaeroma* spp. is abundant, lateral erosion exceeds one metre per year (Talley et al. 2001). Erosion of banks may increase the sedimentation of estuaries leading to light attenuation, reduced water quality and the smothering of benthic organisms.

The taxonomy of amphipods and isopods is poorly known, and there are few records of distributions, biology or ecology (Chapman & Carlton 1991, Chapman & Carlton 1994, Hewitt et al. 1999, Bollens et al. 2002, Montelli & Lewis 2008). As a consequence, it is difficult to determine the extent of amphipod and isopod introductions, impacts, local resilience, and importance to ecosystem structure and function beyond the few studied species.

4.1.3. Risk Estimation

Given that particular amphipod and isopod species:
- have the potential to enter New Zealand via vessel biofouling;
- are capable of forming establishing populations in New Zealand;
- have recently established populations within New Zealand;
- are known for their invasive behaviours;
- have known recorded impacts to core values;

it is concluded that the import of non-indigenous amphipod and isopod species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.1.4. Risk Management

A generic approach is suitable to managing the risk associated with amphipods and isopods (Section 3). However, the mobility of amphipods and isopods presents further management difficulties, and as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape. The fact that amphipods can also form tubes on vessel hulls devoid of macro-fouling (Montelli & Lewis 2008) should also be considered during inspections.

4.1.5. References

Berezina, N A (2007) Invasions of alien amphipods (Amphipoda: Gammaridea) in aquatic ecosystems of North-Western Russia: pathways and consequences. *Hydrobiologia* 590: 15-29.

Bollache, L; Dick, J T A; Farnsworth, K D; Montgomery, W I (2008) Comparison of the functional responses of invasive and native amphipods. *Biology Letters* 4: 166-169.

Bollens, S M; Cordell, J R; Avent, S; Hooff, R (2002) Zooplankton invasions: a brief review, plus two case studies from the northeast Pacific Ocean. *Hydrobiologia* 480: 87-110.

Borges, L M S; Cragg, S M; Bergot, J; Williams, J R; Shayler, B; Sawyer, G S (2008) Laboratory screening of tropical hardwoods for natural resistance to the marine borer *Limnoria quadripunctat*a: the role of leachable and non-leachable factors. *Holzforschung* 62: 99-111.

Brearley, A; Kendrick, G A; Walker, D I (2008) How does burrowing by the isopod *Limnoria agrostisa* (Crustacea: Limnoriidae) affect the leaf canopy of southern Australian seagrass *Amphibolus graffithii*? *Marine Biology* 156: 65-77.

Buschbaum, C; Gutow, L (2008) Mass occurrence of an introduced crustacean (*Caprella cf. mutica*) in the south-eastern North Sea. *Helgoland Marine Research* 59: 252-253.

Chapman, J W; Carlton, J T (1991) A test of criteria for introduced species: the global invasion by the isopod *Synodotia laevidorsalis* (Miers, 1881). *Journal of Crustacean Biology* 11: 386-400.

Chapman, J W; Carlton, J T (1994) Predicted discoveries of the introduced isopod *Synidotea laevidorslis* (Miers, 1881). *Journal of Crustacean Biology* 14: 700-714.

Cook, E J; Jahnke, M; Kerckhof, F; Minchin, D; Faasse, M; Boos, K; Ashton, G (2007) European expansion of the introduced amphipod *Caprella mutica* Schurin 1935. *Aquatic Invasion* 2: 411-421.

Cordell, J R; Morrison, S M (1996) The invasive Asian copepod *Pseudodiaptomus inopinus* in Oregon, Washington, and British Columbia estuaries. *Estuaries* 19: 629-638.

Cordell, J R; Rasmussen, M; Bollens, S M (2007) Biology of the introduced copepod *Pseudodiaptomus inopinus* in a northeast Pacific estuary. *Marine Ecology Progress Series* 333: 213-227.

Davidson, T M; Hewitt, C L; Campbell, M (2008) Density, distribution and habitat use among native and introduced populations of the Australasian burrowing isopod *Sphaeroma quoianum*. *Biological Invasions* 10: 399-410.

Guerra-Garcia, J M; Tierno de Figueroa, J M (2009) What do caprellids (Crustacea: Amphipoda) feed on? *Marine Biology* 156: 1881-1890.

Gewant, D S; Bollens, S M (2005) Macrozooplankton and micronekton of the lower San Francisco estuary: seasonal, interannual, and regional variation in relation to environmental conditions. *Estuaries* 28: 473-485.

Grabowski, M; Jażdżewski, K; Konopacka, A (2007) Alien Crustacea in Polish waters – Amphipoda. *Aquatic Invasions* 2(1): 25-38.

Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B (1999) *Marine biological invasions of Port Phillip Bay, Victoria*. Centre for Research on Introduced Marine Pests Technical Report No. 20. CSIRO Marine Research; Hobart, Australia.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Ingolfsson, A; Agnarsson, I (2008) Amphipods and isopods in the rocky intertidal: dispersal and movements during high tide. *Marine Biology* 143: 859-866.

Kelly, D W; Muirhead, J R; Heath, D D; MacIsaac, H J (2006) Contrasting patterns in genetic diversity following multiple invasions of fresh and brackish waters. *Molecular Ecology* 15: 3641-3653.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Lewis, J (2001) Hull fouling as a vector for the translocation of marine organisms. Report 1: Hull Fouling Research. Ballast *Water Research Series Report* No. 14. Department of Agriculture, Fisheries and Forestry; Canberra, Australia.

NIMPIS (2010) *Sphaeroma walkeri* species summary. In Hewitt, C L; Martin, R B; Sliwa, C; McEnnulty, F R; Murphy, N E; Jones, T; Cooper, S (Eds) National Introduced Marine Pest Information System. http://crimp.marine.csiro.au/nimpis - accessed 17 June 2008.

Mak, P M S; Huang, Z G; Morton, B S (1985) *Sphaeroma walkeri* Stebbing (Isopoda, Sphaeromatidae) introduced into and established in Hong Kong. *Crustaceana* 49: 75-82.

Marion, A; Harvey, M (2008) Feeding ecology and predation impact of the recently established amphipod, *Themisto libellula*, in the St. Lawrence marine system, Canada. *Marine Ecology Progress Series* 373: 53-70.

Martinez, J; Adarraga, I (2008) First record of invasive caprellid *Caprella scuara* Templeton, 1836 sensu lato (Crustacea: Amphipoda: Caprellidae) from the Iberian Peninsula. *Aquatic Invasions* 3: 165-171.

Montelli, L; Lewis, J (2008) *Survey of Biofouling on Australian Navy Ships: Crustacea; Isopoda and Amphipoda; Caprellidea.* DSTO Defence Science and Technology Organisation; Victoria, Australia.

Morton, J (2004) *Seashore Ecology of New Zealand and the Pacific*. David Bateman Ltd; Auckland, New Zealand.

Orav-Kotta, H; Kotta, J; Herkul, K; Kotta, I; Paalme, T (2009) Seasonal variability in the grazing potential of the invasive amphipod *Gammarus tigrinus* and the native amphipod *Gammarus salinus* (Amphipoda: Crustacea) in the northern Baltic Sea. *Biological Invasions* 11: 597-608.

Packalen, A; Korpinen, S; Lehtonen, K K (2008) The invasive amphipod species *Gammarus tigrinus* (Sexton 1939) can rapidly change littoral communities in the Gulf of Finland (Baltic Sea). *Aquatic Invasions* 3: 405-412.

Poore, G C B; Storey, M (1999) Soft sediment crustacea of Port Phillip Bay. In, Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B (Eds) *Marine Biological Invasions of Port Phillip Bay, Victoria*. Centre for Research on Introduced Marine Pests Technical Report 20. CSIRO, Marine Research, Hobart, Australia.

Piscart, C; Maazouzi, C; Marmonier, P (2008) Range expansion of the North American alien amphipod *Gammarus tigrinus* Sexton 1939 (Crustacea: Gammaridae) in Brittany, France. *Aquatic Invasion*s 3: 449-453.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Talley, T S; Crooks, J A; Levin, L A (2001) Habitat utilization and alteration by the invasive burrowing isopod, *Sphaeroma quoyanum*, in California salt marshes. Marine Biology 138: 561-573.

4.2. BACTERIA AND VIRUSES

4.2.1. Hazard identification

Bacteria are well known components of the biofilms which rapidly develop on submerged surfaces (Skerman 1956, ZoBell 1959, Wahl 1989, Cooksey & Wigglesworth-Cooksey 1995, Callow & Callow 2002). Bacteria in biofilms may be a single species or a complex of species and are known to function cooperatively providing the community with a level of protection against predation, stressors (such as salinity, temperature or ultraviolet shock) and sheer forces (Davey & O'Toole 2000). Moreover, biofilms have been observed to modify their structure to conform to changing external stressors such as current flows (Purevdorj et al. 2002, Hall-Stoodley et al. 2004).

While in the past there has been a view that marine bacterial species are omnipresent, improved identification techniques have begun to show a distinct bacterial biogeography (Martiny et al. 2006, Ramette & Tiedje 2007). Furhman et al. (2008), for example, have reported "geographic patterns of [marine bacterial] species diversity that are similar to those seen in other organisms" noting a strong correlation between species richness, latitude and temperature. Although such observations are recent, they suggest that bacteria could be translocated, via anthropogenic vectors, beyond their natural limits with similar consequences to other invasive organisms. Indeed this has been observed in the context of ballast water translocation (Drake et al. 2001). However, despite its volume, ballast water distributes primarily pelagic bacterial species whereas the greatest numbers and diversity of bacteria are those attached to surfaces as biofilm (Davey & O'Toole 2000).

Potentially pathogenic species found in marine biofilms include *Vibrio parahaemolyticus, Pseudomonas aeruginosa* and *P. putrefaciens* (Zambon et al. 1984, Lee et al. 2003, Railkin 2004). However, research on the biofilms associated with vessel hulls has tended to focus on its association with biofouling macro-species (Cassé & Swain 2006). Similarly, few investigations of biofilms on natural substrates in the marine environment have considered pathogens. However, it has been confirmed that pathogens are associated with biofilms and that pathogens have the ability to exit biofilms to facilitate cell distribution (Boyd & Chakrabarty 1995, Decho 2000, Hall-Stoodley et al. 2004). Pathogenic bacteria have been isolated from the biofilm of ballast water tanks (Drake et al. 2005), similar to the findings of more intensively researched systems, e.g. municipal drinking water supplies (Walker et al. 1995). Virus-like particles have also been reported from ballast water tank biofilms (Drake et al. 2005) and it is likely that they would be associated with hull biofouling. However, to date, there have been no known bacterial or viral outbreaks attributed to biofouling.

The MAFBNZ commissioned research on biofouling showed the presence of cyanobacteria in some samples of passenger and commercial vessel biofouling, and no viruses (Inglis et al. in prep.). However, it is noted that the sampling methods focused on macro-fouling organisms rather than micro-organisms. A knowledge gap needs to be addressed regarding the potential of pathogenic bacteria and viruses to be translocated beyond their natural geographic ranges. Initial investigations of biofouling related pathogens are currently being conducted by the MAFBNZ Wallaceville National Centre for Biosecurity and Infectious Diseases.

Given that bacteria and viruses;

- are well documented components of biofilms;
- are likely to be present on all vessel hulls;

- of human pathogenic form are known to be present in biofilms though have yet to be specifically associated with vessel hull fouling;
- have an unknown potential to establish and become problematic in New Zealand's marine environment;

it is concluded that bacteria and virus species represent a potential hazard via vessel biofouling.

As current anti-fouling system technologies only have the capacity to control biofouling to the biofilm (slime layer) level, the prevention of the entry of non-indigenous bacteria and viruses via vessel hull fouling cannot be practically achieved. As a consequence, further research is required to confirm the presence and translocation potential of potentially hazardous bacteria and viruses before the development and implementation of biofilm control measures can be justified specifically from the biosecurity perspective.

4.2.2. References

Boyd, A; Chakrabarty, A M (1995) *Pseudomonas aeruginosa* biofilms: role of the alginate exopolysaccharide. *Journal of Industrial Microbiology and Biotechnology* 15: 162-168.

Callow, M E; Callow, J A (2002) Marine biofouling: a sticky problem. *Biologist* 49: 1-5.

Cassé, F; Swain, G W (2006) The development of micro-fouling on four commercial anti-fouling coatings under static and dynamic immersion. *International Biodeterioration and Biodegradation* 57: 179-185.

Cooksey, K E; Wigglesworth-Cooksey, B (1995) Adhesion of bacteria and diatoms to surfaces in the sea: a review. *Aquatic Microbial Ecology* 9: 87-96.

Davey, M E; O'Toole, G A (2000) Microbial biofilms: from ecology to molecular genetics. *Microbiology and Molecular Biology Reviews* 64: 847-867.

Decho, A W (2000) Microbial biofilms in intertidal systems: an overview. *Continental Shelf Research* 20: 1257-1273.

Drake, L A; Choi, K-H; Ruiz, G M; Dobbs, F C (2001) Global redistribution of bacterioplankton and virioplankton communities. *Biological Invasions* 3: 193-199.

Drake, L A; Meyer, A E; Forsberg, R L; Baier, R E; Doblin, M A; Heinemann, S; Johnson, W P; Koch, M; Rublee, P A; Dobbs, F C (2005) Potential invasion of microorganisms and pathogens via 'interior hull fouling': biofilms inside ballast water tanks. *Biological Invasions* 7: 969-982.

Fuhrman, J A; Steele, J A; Hewson, I; Schwalbach, M S; Brown, M V; Green, J L; Brown, J H (2008) A latitudinal diversity gradient in planktonic marine bacteria. *Proceedings of the National Academy of Sciences* 105: 7774-7778.

Hall-Stoodley, L; Costerton, J W; Stoodley, P (2004) Bacterial biofilms: from the natural environment to infectious diseases. *Nature Reviews Microbiology* 2: 95-108.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Martiny, J B H; Bohannan, B J M; Brown, J H; Colwell, R K; Fuhrman, J A; Green, J L; Horner-Devine, M C; Kane, M; Krumins, J A; Kuske, C R; Morin, P J; Naeem, S; Øvreas, L; Reysenbach, A-L; Smith, V H; Staley, J T (2006) Microbial biogeography: putting microorganisms on the map. *Nature Reviews Microbiology* 4: 102-112.

Purevdorj, B; Costerton, J W; Stoodley, P (2002) Influence of hydrodynamics and cell signalling on the structure and behaviour of *Pseudomonas aeruginosa* biofilms. *Applied and Environmental Microbiology* 68: 4457-4464.

Railkin, A I (2004) Marine biofouling: colonization processes and defences. CRC Press, Boca Raton.

Rammette, A; Tiedje, J M (2007) Biogeography: An emerging cornerstone for understanding prokaryotic diversity, ecology, and evolution. *Microbial Ecology* 53: 197-207.

Skerman, T M (1956) The nature and development of primary films on surfaces submerged in the sea. *New Zealand Journal of Science and Technology* 38B: 44-57.

Wahl, M (1989) Marine epibiosis. I. Fouling and anti-fouling: some basic aspects. *Marine Ecology Progress Series* 58: 175-189.

Walker, J T; Mackerness, C W; Rogers, J; Keevil, C W (1995) Heterogeneous mosaic biofilm – A haven for waterborne pathogens. In Lappin-Scott, H M; Costerton, J W (Eds) *Microbial Biofilms*. Cambridge University Press; Cambridge, United Kingdom.

Zambon, J J; Huber, P S; Meyer, A E; Slots, J; Fornalik, M S; Baier, R E (1984) In situ identification of bacterial species in marine micro-fouling films by using an immunofluorescence technique. *Applied and Environmental Microbiology* 48: 1214-1220.

ZoBell, C E (1959) Introduction to marine microbiology. In Skerman, T M. Contributions to marine microbiology. *New Zealand Department of Scientific and Industrial Research Information Series* 22: 7-20.

4.3. BARNACLES (ORDER: SESSILIA)

4.3.1. Hazard Identification

Barnacles are the common name for the group of arthropods classified within the class Maxillopoda. However, for simplicity, within this hazard identification and risk assessment the term "barnacle" will be used to refer to the order Sessilia within the Maxillopoda. The order Sessilia is composed of several families; the Verrucidae are a deep water family, while the remaining families make up a group commonly referred to as Balanomorpha (Species 2000, Perez-Losada et al. 2004). Although the order Pedunculata (goose or stalked barnacles) are also commonly known as barnacles these are found most often in the pelagic environment (Barnes et al. 2004, Lewis 2004) and are therefore not considered in this assessment.

Adult epibenthic barnacles live permanently attached to the surface of both natural and artificial hard substrates and disperse via a planktonic larval phase (Skerman 1960, Barnes et al. 2004, Lawson et al. 2004). Usually found in conspecific clumps, barnacles reproduce sexually and must be located within penis reach to achieve a self sustaining population (Barnes & Crisp 1956, Kent et al. 2003, Aldred & Clare 2008). Larvae are released into the plankton and develop over a period from days to weeks before settlement (Callow & Callow 2002). Attachment is head-down and the larval carapace, which encloses the entire body, persists and becomes a shell of calcareous plates. The top plates open to allow the thorax to be extended for filter feeding (Miller 2007). The adhesive used in cementation and exploration contains chemical cues that induce the settlement of other barnacle larvae (Callow & Callow 2002).

Barnacles inhabit intertidal and shallow coastal waters and are important as ecosystem engineers. The thickly calcified shell and valves protect against predation, desiccation and sheer stresses (Foster 1971, Ruppert et al. 2004). They grow cemented directly to a surface from which it takes considerable force to remove (Crisp et al. 1985). As a result, barnacles are able to live in stressful environments, such as the rocky intertidal, with little competition for space and therefore can occur in high densities (Ruppert et al. 2004, Miller 2007). Due to their dominance of space and hard structures, barnacles can provide shelter and attachment points for an assortment of other organisms (Leppakoski & Olenin 2000).

Barnacles are well known nuisance foulers of vessel hulls and other artificial structures (Railkin 2004, Sakaguchi & Namihira 2006). As the speed and fuel efficiency of a barnacle-fouled vessel may be reduced by up to 30%, extensive research has investigated their control (Railkin 2004, Aldred & Clare 2008). The cementation of barnacles to artificial surfaces means that removal can be a difficult and time consuming task, often leaving residue or damaging the surface of the anti-fouling system (Thiyagarajan et al. 1997, South Carolina Department of Natural Resources 2006). This residue and damage can provide a surface on to which other fouling species can attach.

Introduced globally, non-indigenous species of barnacles can modify community structures by excluding native communities of barnacles, mussels, or macroalgae (Kado 2003, Lawson et al. 2004, Watson et al. 2005, Geller et al. 2008). Invasive barnacles have also covered previously bare space creating novel habitat in Hawai'i and eastern South America (Zabin & Altieri 2007, Laird & Griffiths 2008). Such community and habitat modification has also affected mobile intertidal species, such as the native Hawai'ian limpet *Siphonaria normalis* (Zabin & Altieri 2007). Non-indigenous barnacles dominate sections of the coast in various countries, such as Ireland and Japan, and have restricted the habitats of native barnacles (Kado 2003, Lawson et al. 2004, Watson et al. 2004).

Given that barnacles:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that barnacle species represent a potential hazard via vessel biofouling.

4.3.2. Risk Assessment

4.3.2.1. Entry assessment

Barnacles occur in coastal environments, on both natural and artificial hard substrates, from latitudes 57°S to 70°N (Barnes et al. 2004). Some species inhabit port and harbour environments, thus placing them in close proximity to vessels. Vessel hulls present an ideal settlement option for barnacle larvae as they are attracted to open hard surfaces that generally free of other biofouling, and to other barnacles (Olivier et al. 2000, Anderson et al. 2003, Ruppert et al. 2004). Moreover, the sheltered nature of ports and harbours retain pelagic larvae, thus increasing the likelihood of settlement onto vessels (Floerl et al. 2003, Floerl et al. 2005, Davidson et al. 2008).

Barnacles are highly prevalent hull fouling organisms (Wood & Allen 1958, Christie & Dalley 1987 *In* Floerl & Inglis 2003, Aldred & Clare 2008, Hung et al. 2008). As such, biocidal anti-fouling paints have been developed to have high efficacy against barnacles by altering the attachment characteristics of metamorphosing larvae thus causing juveniles to detach more easily (Crisp & Austin 1960, Floerl et al. 2005). However, if a juvenile or adult barnacle is removed any basal membrane or cement remaining will act as an attractant to other settling larvae (Crisp 1961, Crisp & Meadows 1962). In addition, settlement on these remnants provides protection from anti-fouling biocides. Thus improper cleaning of hulls may facilitate the settlement of barnacles by leaving remnant tests or small-scale disruptions to the anti-fouling system (Floerl et al. 2005, Piola & Johnston 2008).

Barnacles are habitat structuring organisms due to their gregarious settling habit. Thus, as hull fouling, barnacles may aid the recruitment of other fouling organisms by providing protection from sheer forces and predation (Railkin et al. 1985, Leppakoski & Olenin 2000, Minchin & Gollasch 2003).

Barnacles are tolerant to environmental stressors, aiding their survival during translocation. For example, the temperate barnacles *Austrominius modestus* and *Balanus glandula* have been translocated between hemispheres indicating survival through the thermal barrier of the equator (O'Riordon & Ramsay 1999, Lepakoski & Olenin 2000, Geller et al. 2008, Simon-Blecher et al. 2008). Similarly the spread of tropical barnacles, such as *Chthamalus proteus*, indicates that they can withstand passages through cold or freshwater environments (Zardus & Hadfield 2005).

Hull fouling is the primary mechanism by which barnacles have been known to enter New Zealand (Kospartov et al. in press). In the MAFBNZ commissioned research on biofouling, Inglis et al. (in prep.) found that 60% of sampled vessels entering New Zealand had barnacle fouling. In addition, 42 of the 45 species recorded were not established in New Zealand (Inglis et al. in prep.). Similarly, Otani et al. (2007) found 22 species of barnacles on bulk carriers in Osaka Bay, Japan, of which 14 had not been previously recorded in the bay.

Genetic analysis of the widely, and recently, distributed *Megabalanus coccopoma*, *M. rosa* and *M. volcano* suggest that shipping was the translocation vector (Yamaguchi et al. 2009).

Given that particular barnacle species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous barnacle species into New Zealand via vessel biofouling is non-negligible.

4.3.2.2. Establishment assessment

It is likely that non-indigenous barnacles will find a suitable habitat in New Zealand given that approximately half the shoreline is composed of hard substrata (Carter & Saunders 2003). In addition, artificial substrates, such as provided by ports, yield favourable conditions for the establishment of sustaining populations (Sakaguchi & Namihira 2006, Weigemann 2008).

Barnacles are most likely to establish in a new environment through the release of larvae. Barnacles release large numbers of planktonic larvae which can be widely dispersed prior to attaching to a suitable substrate (Jarrett 1997, Thiyagarajan 2002a, Thiyagarajan 2002b, Thiyagarajan 2003). Larval release generally occurs during summer months, however, some species, such *Elminius modestus*, release larvae year-round (Skerman 1960).

For a barnacle population to be reproductive it must reach critical density, i.e. individuals of the same species must be able to reach each other for copulation to occur (Barnes & Crisp 1956, Kent et al. 2003, Aldred & Clare 2008). As a result, newly settled barnacles must be aggregated to create a sustainable population. While barnacles have been known to survive dislocation from a hull (Woods et al. 2007), a debris field with live barnacles is not likely to support a reproductive population due to continual disturbance. However, in areas of restricted flow, such as sheltered harbours and ports, larvae can be retained increasing the number of recruits and, therefore, the likelihood of gregarious and successful settlement (Floerl et al. 2003, Floerl et al. 2005). Furthermore, the presence of conspecific adult barnacles, or their remnant tests, can be an important settlement cue (Jeffery 2002, NIMPIS 2010). The ability of barnacles to settle on artificial substrates in exposed areas not tolerated by other taxa means they can form beachhead populations from which they can be spread. For example, *Austrominius modestus* inhabits sheltered locations and has been spread around the Irish coastline through vessel transport (Allen et al. 2006).

Water temperature is a critical factor to the success of non-indigenous barnacle species as development, settlement, and metamorphosis of larvae all depend upon this factor (Watson et al. 2005, Weigemann 2008). For example, although *Amphibalanus amphitrite* larvae have been found in German waters they have not established on the shore because it is too cold for the larvae to fully develop (Wiegemann 2008). Similarly, in Peter the Great Bay, Sea of Japan, cold waters not only restrict the reproductive window of *A. amphitrite* but also result in winter mortalities (Zvyagintsev & Korn 2003). Barnacles that establish in New Zealand are likely to have subtropical or temperate distributions. For example, in the MAFBNZ commissioned research on biofouling, approximately half of the non-indigenous barnacle species recorded entering New Zealand on recreational vessels had native distributions in temperate environments (Inglis et al. in prep.). Therefore, it is likely that these species will find suitable conditions for establishment. Kospartov et al. (in press) found that six non-indigenous barnacles have recently established in New Zealand. Of those, the two *Austromegabalanus* species are noted to have established since 2005.

Given that particular barnacle species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous barnacle species in New Zealand via vessel biofouling is non-negligible.

4.3.2.3. Consequence assessment

The consequences of barnacles are increased maintenance and transport costs, threats to local biodiversity, and the facilitation of the entry of other non-indigenous organisms.

The further introduction of non-indigenous hull fouling barnacles to New Zealand will impact maintenance of hulls and fuel efficiency, especially given that some are very large, such as, *Balanus* cf *rostatus*, *Perforatus perforatus*, and *Austrobalanus imperator*. For example, in Sweden the economic impact of the bay barnacle *B. improvisus*, calculated as fouling control costs, is estimated at 166-418 million Swedish krona per year, of which recreational vessels account for approximately 74% (Gren et al. 2007). Although not all these costs were specifically due to *B. improvisus*, this organism is considered the most problematic fouling organism in Swedish waters. The environmental costs caused by toxic compounds used in anti-fouling paints are not accounted for in this cost estimate (Gren et al. 2007).

Non-indigenous barnacles occupying natural or artificial substrata in the intertidal zone may, through competition or the presence of an open niche, alter the local marine community. Impacts may include reduction or elimination of local barnacles, decreased mobility for organisms such as limpets, or elimination of organisms such as mussels and algae.
For example, the Hawai'ian archipelago was generally without high cover of intertidal barnacles until the recent invasion by *Chthamalus proteus*. *C. proteus* has become highly abundant in the low-energy environments of the island of Oahu, whereby, cover of *C. proteus* can extend from the high to low intertidal and reach up to 100% cover in bays on the windward side. Field surveys and experimental results suggest that the creation of suboptimal grazing conditions by barnacle recruitment has resulted in limpets, *Siphonaria normalis,* moving to patches with lower numbers of barnacles (Zabin & Altieri 2007). In Argentina, the introduced *B. glandula,* is presently distributed over 2900 km, with an estimated dispersion rate of 244 km per year (Swindt 2007). In most intertidals of Argentina *B. glandula* has formed dense monospecific belts devoid of competition from other barnacle species (Swindt 2007).

As habitat providers and common foulers that deposit calcareous shells, barnacles have the potential to facilitate many hull fouling groups. Both mobile and attached organisms, such as crabs, macroalgae, ostracods, copepods, and juvenile bivalves have been associated with barnacle hull fouling (Railkin et al. 1985, Leppakoski & Olenin 2000, Minchin & Gollasch 2003). The consequences of these introductions are discussed within the respective group risk assessments.

4.3.3. Risk Estimation

Given that particular barnacle species:

- have been frequently recorded entering New Zealand via vessel fouling;
- have records of recent establishment in New Zealand;
- have known invasive behaviour;

- have been introduced globally via vessel fouling with, in some cases, significant impacts;
- can be ecosystem engineers;
- can facilitate the entry and establishment of a variety of other non-indigenous organisms;
- are likely to impact a range New Zealand's core values, such as, economic and environmental values;

it is concluded that the import of non-indigenous barnacle species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.3.4. Risk Management

The treatment of hulls to manage the risks posed by barnacles is covered by the generic options (Section 3). An effective management of barnacle risk is their removal from vessel hulls, as the removal process is likely to result in barnacle mortality. By contrast, application of *in-situ* techniques, such as encapsulation, are challenging due to the high tolerance of barnacles to adverse conditions. The choice of removal technique should consider the complete removal of the test, including cement, and the containment of potential larval and associated fouling releases.

4.3.5. References

Aldred, N; Clare, A S (2008) The adhesive strategies of cyprids and development of barnacle-resistant marine coatings. *Biofouling* 24(5): 351-363.

Barnes, H; Crisp, D J (1956) Evidence of self-fertilization in certain species of barnacles. *Journal of the Marine Biological Association of the United Kingdom* 35: 631–639.

Barnes, D K A; Warren, N L; Webb, K; Phalan, B; Reid, K (2004) Polar pedunculate barnacles piggy-back on pycnogona, penguins, pinniped seals and plastics. *Marine Ecology Progress Series* 284: 305-310.

Bertness, M D; Gaines, S D; Yeh, S M (1998) Making mountains out of barnacles: the dynamics of acorn barnacle hummocking. *Ecology* 74(4): 1382-1394.

Callow, M E; Callow, J A (2002) Marine biofouling: a sticky problem. *Biologist* 49(1): 1-5.

Carter, L; Saunders, H (2003) Rocky Reefs. In Andrew N; Francis M (Ed) *The Living Reef – the ecology of New Zealand's rocky reefs*. Craig Potton Publishing; Nelson, New Zealand; pp 18-23.

Crisp, D J (1961) Territorial behaviour in barnacle settlement. *Journal of Experimental Biology* 38(2): 429.

Crisp, D J; Austin, A P (1960) The action of copper in anti-fouling paints. *Annals of Applied Biology* 48: 787-799.

Crisp, D J; Meadows, P S (1962) The chemical basis of gregariousness in cirripedes. *Proceedings of the Royal Society of London Series B*. 158: 364.

Crisp, D J; Walker, G; Yule, A B (1985) Adhesion and substrate choice in mussels and barnacles. *Journal of Colloid and Interface Science* 104(1): 40-50.

Floerl, O; Inglis, G J (2003) Boat harbour design can exacerbate hull fouling. *Austral Ecology*. 28: 116-127.

Floerl, O; Inglis, G J; Marsh, H M (2005) Selectivity in vector management: an investigation of the effectiveness of measures used to prevent transport of Non-indigenous species. *Biological Invasions* 7: 459-475.

Foster, B A (1971) Desiccation as a factor in the intertidal zonation of barnacles. *Marine Biology* 8: 12-29.

Geller, J; Sotka, E E; Kado, R; Palumbi, S R; Schwindt, E (2008) Sources of invasion of a northeastern Pacific acorn barnacle, *Balanus glandula*, in Japan and Argentina. *Marine Ecology Progress Series* 358: 211-218.

Gren, I-M; Isaacs, L; Carlsson, M (2007) *Calculations of Costs of Invasive Alien Species in Sweden—Technical Report. Working paper no. 7,.* Department of Economics, Swedish University of Agricultural Sciences. Uppsala, Sweden.

Gulf States Marine Fisheries Commission (2005) *Balanus trigonus* fact sheet. http://nis.gsmfc.org/nis_factsheet2.php?toc_id=121 - accessed 21 May 2008.

Hung, O S; Thiyagarajan, V; Qian, P Y (2008) Preferential attachment of barnacle larvae to natural multi-species biofilms: Does surface wetability matter? *Journal of Experimental Marine Biology and Ecology* 361: 36-41.

ICES (2004) Report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO). 25-26 March 2004; Cesenatico, Italy.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Jarrett, J N (1997) Temporal variation in substratum specificity of *Semibalanus balanoides* (Linnaeus) cyprids. *Journal of Experimental Marine Biology and Ecology* 211: 103-114.

Jeffery, C J (2002) New settlers and recruits do not enhance settlement of a gregarious intertidal barnacle in New South Wales. *Journal of Experimental Marine Biology and Ecology* 275:131-146.

Kado, R (2003) Invasion of Japanese shores by the NE Pacific barnacle *Balanus glandula* and its ecological and biogeographical impact. *Marine Ecology Progress Series* 249: 199-206.

Kent, A; Hawkins, S J; Patrick Doncaster, C (2003) Population consequences of mutual attraction between settling and adult barnacles. *Journal of Animal Ecology* 72: 941-952.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Laird, M C; Griffiths, C L (2008) Present distribution and abundance of the introduced barnacle *Balanus glandula* in South Africa. *African Journal of Marine Science* 30(1): 93-100.

Lawson, J; Davenport, J; Whitaker, A (2004) Barnacle distribution in Lough Hyne Marine Nature Reserve: a new baseline and an account of invasion by the introduced Australasian species *Elminius modestus* Darwin**.** *Estuarine, Coastal and Shelf Science* 60 (4): 729-735.

Leppakoski, E; Olenin, S (2000) Non-native species and rates of spread: lessons from the brackish Baltic Sea. *Biological Invasions* 2: 151-163.

Lewis, J (2004) *Hull Fouling as a Vector for the Translocation of Marine Organisms. Report 1: Hull fouling research.* Ballast Water Research Series Report No. 14. Dept of Agriculture, Fisheries and Forestry; Canberra, Australia; 129pp.

Lohrer, A M; Whitlatch, R B; Wada, K; Fukui, Y (2000) Home and Away: comparisons of resource utilization by a marine species in native and invaded habitats. *Biological Invasions* 2: 41-57.

Lopez, D A; Lopez, B A; Burgos, I C; Arriagada, S E; Gonzalez, M L (2007) Consequences of base modification in hummocks of the barnacle *Austromegabalanus psittacus*. New *Zealand Journal of Marine and Freshwater Research* 41: 291-298.

Miller, L (2007) Feeding in extreme flows: behaviour compensates for mechanical constraints in barnacle cirri. *Marine Ecology Progress Series* 349: 227-234.

Minchin, D; Gollasch, S (2003) Fouling and ships' hulls: how changing circumstances and spawning events may result in the spread of exotic species. *Biofouling* 19 (supplement): 111-122.

NIMPIS (2010). Megabalanus rosa reproduction and habitat. National Introduced Marine Pest Information System. Web publication http://www.marinepests.gov.au/nimpis. Date of access: 30/08/2009

O'Riordan, R M; Ramsay, N F (1999) The current distribution and abundance of the Australasian barnacle *Elminius modestus* in Portugal. *Journal of the Marine Biological Association of the United Kingdom* 79: 937-939.

Otani, M; Oumi, T; Uwai, S; Hanyuda, T; Prabowo, R E; Yamaguchi, T; Kawai, H (2007) Occurrence and diversity of barnacles on international ships visiting Osaka Bay, Japan, and the risk of their introduction. *Biofouling* 23: 277-286.

Perez-Losada, M; Hoeg, J T; Crandall, K A (2004) Unravelling the Evolutionary Radiation of the Thoracican Barnacles Using Molecular and Morphological Evidence: A Comparison of the Several Divergence Time Estimation Approaches. *Systematic Biology* 53(2): 244-264.

Piola, R F; Johnston, E L (2008) The potential for translocation of marine species via small-scale disruptions to anti-fouling surfaces. *Biofouling* 24(3): 145-155.

Railkin, A I (2004) *Marine Biofouling: Colonization Processes and Defences*. CRC Press; 303pp

Railkin, A I; Makarov, V N; Shoshina, E V (1985) The influence of substrate on settlement and attachment of *Laminaria saccharina* zoospores. *Biologija Morya* 1: 37

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Sakaguchi, I; Namihira, T (2006) Settlement and growth of barnacles at a power station in Okinawa. *Sessile Organisms* 23(1): 17-23.

Schwindt, E (2007) The invasion of the acorn barnacle *Balanus glandula* in the south-western Atlantic 40 years later. *Journal of the Marine Biological Association of the United Kingdom* 87:1219-1225.

Simon-Blecher, N; Granevitze, Z; Achituv, Y (2008) Balanus glandula: from North-West America to the west coast of South Africa. *African Journal of Marine Science* 30(1): 85-92.

Skerman, TM (1960) Ship-fouling in New Zealand waters: a survey of marine fouling organisms of the coastal and overseas trades. *New Zealand Journal of Science* 3: 620-648.

South Carolina Department of Natural Resources (2006) News release #06–342: Pesky exotic barnacle found in Charleston Harbor estuary. Accessed 23 May 2008.

Species 2000. *Catalogue of Life: Dynamic Checklist*. http://www.catalogueoflife.org/dynamic-checklist/browse_taxa.php. Accessed 3 March 2009.

Thiyagarajan, V; Venugopalan, V P; Subramoniam, T; Nair, V K (1997) Macro-fouling in the cooling water conduits of a coastal power station. *Indian Journal of Marine Sciences* 26(3): 305-308.

Thiyagarajan, V; Harder, T; Qian, P Y (2002a) Effect of the physiological condition of cyprids and laboratory-mimicked seasonal conditions on the metamorphic success of *Balanus amphitrite* Darwin. *Journal of experimental Marine Biology and Ecology* 274: 65-74.

Thiyagarajan, V; Harder, T; Qian, P Y (2002b) Relationship between cyprid energy reserves and metamorphosis in the barnacle *Balanus amphitrite* Darwin (Cirripedia; Thoracica). *Journal of experimental Marine Biology and Ecology* 280: 79-93.

Thiyagarajan, V; Harder, T; Qian, P Y (2003) Combined effect of temperature and salinity on larval development and attachment of the subtidal barnacle *Balanus trigonus* Darwin. *Journal of experimental Marine Biology and Ecology* 287: 223-236.

Watson, D I; O'Riordan, R M; Barnes, D K A; Cross, T (2005) Temporal and spatial variability in the recruitment of barnacles and the local dominance of *Elminius* *modestus* Darwin in SW Ireland. *Estuarine, Coastal and Shelf Science* 63(1-2): 119-131.

Wood, E J F; Allen, F E (1958) Common marine fouling organisms of Australian waters. Department of the Navy, Navy Office; Melbourne, Australia; 23 pp.

Wiegemann, M (2008) Wild cyprids metamorphosing in vitro reveal the presence of *Balanus amphitrite* Darwin, 1854 in the German Bight basin. *Aquatic Invasions* 3(2): 235-238.

Yamaguchi, T; Prabowo, R E; Ohshiro, Y; Shimono, T; Jones, D; Kawai, H; Otani, M; Oshino, A; Inagawa, S; Akaya, T; Tamura, I (2009) The introduction to Japan of the Titan barnacle, *Megabalanus coccopoma* (Darwin, 1854) (Cirripedia: Balanomorpha) and the role of shipping in its translocation. *Biofouling* 25: 325-333.

Zabin, C J; Altieri, A (2007) A Hawai'ian limpet facilitates recruitment of a competitively dominant barnacle. *Marine Ecology Progress Series* 337: 175-185.

Zabin, C J; Zardus, J; Pitombo, F B; Fread, V; Hadfield, M G (2007) A tale of three seas: consistency of natural history traits in a Caribbean-Atlantic barnacle introduced to Hawai'i. *Biological Invasions* 9: 523-544.

Zardus, J D; Hadfield, M G (2005) Multiple origins and incursions of the Atlantic barnacle *Chthamalus proteus* in the Pacific. *Molecular Ecology* 14: 3719-3733.

Zvyagintsev, A Y; Korn, O M (2003) Life history of the barnacle *Balanus amphitrite* Darwin and its role in fouling communities of Peter the Great Bay, Sea of Japan. *Russian Journal of Marine Biology* 29(1): 41-48.

4.4. BIVALVE MOLLUSCS (CLASS: BIVALVIA)

4.4.1. Hazard Identification

The class Bivalvia are molluscs characterised by the presence of two calcareous valves which enclose soft body parts. Bivalves are found in aquatic environments, ranging from freshwater to marine, from the intertidal to abyssal depths. In general, bivalves live in benthic environments with either an epifaunal or infaunal habit. The majority of bivalve species have limited mobility and therefore are largely sedentary and suspension feeding. Bivalves are often important ecosystem engineers providing, for example, benthic pelagic coupling and habitat structure. This in turn influences local community diversity, abundance and carrying capacity in both the benthic and pelagic environments (Creese et al. 1997, Newell 2004, Porter et al. 2004, Ruesink et al. 2004, Chapman et al. 2005, Norling & Kautsky 2007). For example, a clump of mussels may accumulate sediments in their byssal threads which provide habitat for species such as polychaetes. These mussels may also provide nutrients through the deposition of faecal and pseudofaecal pellets which will sustain associated organisms (Kochmann et al. 2008).

Many bivalve species are of high economic and social importance as food species. Naturally occurring populations, enhanced populations and aquaculture all contribute to produce bivalves for human consumption. Bivalve species which are commonly cultured demonstrate characteristics of rapid growth, high fecundity and dominant occupation of space (e.g. *Mytilus* spp. Bayne 1976). These characteristics are particularly common to oysters and mussels that act as ecosystem engineers. However, these characteristics also give many bivalve species the potential to be highly invasive.

Known invasive marine species include the mussels *Perna perna* (Hicks & Tunnell 1995), *Perna viridis* (Rajagopal et al. 2006), *Mytilus galloprovincialis* (Hanekom 2008), *Mytilopsis sallei* (Galil & Bogi 2008) and the oyster *Crassostrea gigas*. Invasions by these species have been reported to have significant impacts on ecosystem structure and function (Bownes & McQuaid 2006, Ruesink et al. 2006, Kochmann et al. 2008). Invasive bivalves have also had significant economic impacts through modifying fisheries (e.g. mussel culture in South Africa (Robinson et al. 2005), oyster culture in New Zealand (Dinamani 1991) and fouling infrastructure (e.g. *Perna viridis* in India (Rajagopal et al. 1991)).

Bivalves are well known components of vessel biofouling (Skerman 1960, Lewis 2004, Farrapeira et al. 2007). Bivalves, notably *Mytilid* species, have been frequently reported to be the dominant fouling macro-organisms on vessel hulls (Apte et al. 2000, Gollasch 2002, Farrapeira et al. 2007, Lee & Chown 2007). Recreational vessels have been implicated in at least two significant bivalve incursions in Australia with *P. viridis* in tropical far north Queensland (Stafford et al. 2007) and the establishment of *M. sallei* in Darwin harbour (Willan et al. 2000). Differences exist between vessel type and the likely bivalve fouling assemblages, with recreational vessels more likely to be fouled with oysters and commercial vessels with mussels (James & Hayden 2000, Inglis et al. in prep.).

Most temperate bivalve species known to be pests (e.g. *C. gigas*, *M. galloprovincialis* and *Musculista senhousia*) have previously been found in New Zealand (Dinamani 1971, Creese et al. 1997, Hayward et al. 1999b, Gardner 2004). Although *P. viridis* and *M. sallei* have established in various global locations, including eastern Australia, they are not yet established in New Zealand. *P. perna,* which is native to South Africa and a known pest in central and southern America, has been introduced to New Zealand but not yet known to have established (MAFBNZ 2009). *Dreissena polymorpha* is also a high profile, biofouling, pest bivalve, however, it is restricted to freshwater environments and as a consequence is unlikely to reach New Zealand as vessel biofouling.

Given that bivalves:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that bivalve species represent a potential hazard via vessel biofouling.

4.4.2. Risk Assessment

4.4.2.1. Entry assessment

Typical fouling representatives of Bivalvia are often dominant components of intertidal and shallow subtidal coastal ecosystems. Most bivalves produce large quantities of gametes, which have a planktonic larval stage enabling dispersal beyond the parent location. Therefore, propagules of bivalve species with the potential to foul artificial structures are likely to be present in the majority coastal waters.

Anti-fouling systems can be effective in killing, or deterring the settlement of, bivalve larvae. However, anti-fouling systems have been implicated in selecting specifically for non-indigenous species (Dafforn et al. 2008) which have resistance to the active ingredients. For example, bivalve species found in polluted environments, such as ports, are likely to have developed a level of resistance to copper compounds (Hoare et al. 1995 a, b). As a result, vessels exposed to competent bivalve larvae in port/marina environments could become fouled regardless of the quality of their anti-fouling system.

Source bivalve populations may thrive due to their ability to survive extended periods in conditions unsuitable for potential competing species. For example, many marinas are situated in estuarine locations which can often harbour wide environmental ranges. Apte et al. (2000) described the spawning of *M. galloprovincialis* on the hull of a naval vessel following its arrival to Hawai'i despite the vessel having spent 9 days moored in the Columbia river as freshwater treatment (salinity 2-10 ppt) prior to its departure.

Whether attached by byssus or cemented directly to the surface, once adhered to a surface bivalves can resist substantial dislodgement forces. However, many vessel owners/operators are likely to remove obvious macro-fouling to minimise performance penalties. As bivalve fouling is likely to be conspicuous on, and consequently removed from, the open hull, bivalves attached to vessels en-route are likely to be in locations not easily inspected and/or cleaned such as the propeller, drive shaft or sacrificial anodes. Bivalves are well known to survive travel over considerable distances with the widespread distribution of *M. galloprovincialis* attributed primarily to transport as biofouling. Internal seawater systems, including seachests, are also likely locations for bivalve fouling (Lewis & Coutts 2009).

Given that particular bivalve species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous bivalve species into New Zealand via vessel biofouling is non-negligible.

4.4.2.2. Establishment assessment

As bivalves are likely to be stimulated to spawn due to environmental stressors such as thermal shock (e.g. Apte et al. 2000), the highest risk areas are likely to be the first ports of call vessels make following arrival in the coastal zone. The ports of first arrival in New Zealand provide a range of natural and artificial habitats that include hard to soft and natural to artificial substrates. Temperature ranges are generally warm temperate in the north and cool temperate in the south, although most ports are exposed to significant fluctuations in environmental conditions due to their estuarine location. These conditions are likely to be conducive to successful spawning, larval development and settlement for bivalve species arriving from the majority of potential donor ports. However, only the northern most ports (north of Auckland) are likely to provide conditions where tropical species, such as *P. viridis*, could establish persistent populations.

Areas to the north of the North Island and at the top of the South Island provide conditions which are conducive to bivalve aquaculture. Neighbouring major shipping routes, these areas are also conducive to the establishment of non-indigenous bivalves. Whilst farmers may identify and remove novel species, the cryptic morphology of invasive bivalves compared to native species may mean a new incursion is undetected by farmers. Past incursions have also shown that aquaculture areas can perpetuate non-target invasive species which can then be vectored by stock and equipment movements, for example, *Didemnum vexillum, Styela clava* and *Eudistoma elongatum*.

On arrival to New Zealand, non-indigenous bivalve species could establish a population by relocation from the vessel to the receiving environment. Most adult bivalves are unlikely to actively relocate from a vessel hull, however, they are known to survive physical removal (MAFBNZ 2009). This is especially evident in species that attach by byssus. Species without byssus, such as Pacific oysters, are also known to survive physical removal and live in habitat that would otherwise be unsuitable. However, such species are vulnerable to damage of the attachment surface which exposes the soft body to damage, disease and predation.

Bivalves may also establish in the receiving environment by gamete release (Apte et al. 2000). Bivalves produce large numbers of gametes (in some cases millions of eggs per female), usually released in a synchronous broadcast spawn. For broadcast spawning species to successfully reproduce they would need the following conditions to be met: the presence of at least one bivalve of each sex, the bivalves to be in a reproductively mature condition, to be close enough together for spawned eggs and sperm to fertilise, and to be stimulated to spawn synchronously (e.g. oysters are known to mutually stimulate spawning (Pauley 1988)).

The likelihood of the above conditions being met is high, as bivalves are generally gregarious settlers forming clumps rather than existing as isolated individuals. The existence of bivalves in clumps will enable mature adults to detect chemosensory spawning signals leading to greater likelihood of spawning success. Although anti-fouling compounds can reduce the condition of adult oysters, such as *C. gigas*, the viability and growth of embryos or larvae is unaffected (His & Robert 1987). In addition, physical damage, such as that sustained during removal, may result in the release of gametes. Further, New Zealand ports are generally located in highly productive estuarine waters.

Bivalve larvae are usually free swimming with the planktonic stage lasting from days to months. Thus larvae may move well beyond the environment in which they were released. In the case of the port or marina environment, this may considerably enhance the chances of larvae finding a suitable habitat for colonisation. While mortality of the larvae can be high (>90%) the vast quantities of larvae released mean that a single breeding pair can still establish a founder population.

Some bivalve species brood larvae by inhaling sperm and hold the fertilised eggs until they are released as the swimming veliger stage. Therefore, a single brooding female, fertilised in a previous port, could effectively establish a new population.

Given that particular bivalve species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous bivalve species in New Zealand via vessel biofouling is non-negligible.

4.4.2.3. Consequence assessment

An incursion of a non-indigenous bivalve could have significant impacts on economic, environmental, social and cultural values. Key to this is the ecosystem engineering nature of many bivalve species, i.e. the modification of habitat for their own benefit (Ruesink et al. 2006). Reported ecosystem impacts of bivalves include the alteration of water quality (Porter et al. 2004), displacement of native species (e.g. *M. galloprovincialis* in South Africa) (Hanekom 2008), the modification of community structure (Chapman et al. 2005) and the modification of ecosystem function (Norling & Kautsky 2007). Bivalves can exert both top-down and bottom-up influences over ecosystem function by filtering phytoplankton and providing (or removing) nutrients from the water column or substrate. However, an overabundance of bivalves can lead to deoxygenation of sediments, inhibition of nitrification/denitrification processes, release of phosphorus into the water column and build-up of hydrogen sulphide in sediments (Newell 2004).

The Asian clam, *Corbula amurensis,* was introduced into the northern San Francisco Bay in 1986. Within two years the clam predominated (>95%) in both total number of individuals (>25,000 individuals per m^2) and biomass and had halted the re-establishment of dry-period estuarine species (Nicholls et al. 1990). As a result, the benthic community structure and dynamics of this region of San Francisco Bay have been irreversibly altered. For example, filtering of the water column by the large clam populations has eliminated summer-long phytoplankton blooms since 1987 (Kimmerer 2006). Such changes to primary productivity has resulted in varying responses, for example, mysid abundance declined sharply, and populations of three species of estuarine-dependent fish declined, however, other species have maintained their abundance (Kimmerer 2002 *In* Kimmerer 2006). In addition, the altered food chains have halted the commercial harvesting of anchovies in local waters (Kimmerer 2006).

In Australia, the non-indigenous clam *Corbula gibba* to Port Phillip Bay (Australia) has reached densities of up to 2600 per m^2 and dominates deeper subtidal assemblages (Currie & Parry 1996 *In* Talman and Keough 2001). It has been shown in field experiments that ambient densities of *C. gibba* can significantly affect the size and growth of juveniles of the commercial scallop *Pecten fumatus* via resource competition (Talman and Keough 2001).

In the USA, the green mussel (*P. viridis*) was first identified in Tampa Bay, Florida, in 1999. Since then dense assemblages of these bivalves now clog sea water intakes of power stations and desalination plants, weigh down navigational buoys and foul vessel hulls and engines (Baker et al. 2004 *In* Mitchem et al. 2007).

Recreational fisheries are also vulnerable to non-indigenous bivalve species. For example, the introduction of the clam *Ruditapes philippinarum* to Venice, Italy, has been linked to reduced fishing catch rates, and fishers in turn have changed their angling habits (Albernini et al. 2007).

The impacts of introduced bivalves in New Zealand have generally been poorly described. For example, the Pacific oyster (*C. gigas*) displaced the native oyster (*C. glomerata*) from a broad distribution in the intertidal to a rare occurrence in the high intertidal. However, the implications of this shift on ecosystem structure and function are unknown. In economic terms, the Pacific oyster has become the only commercially cultured oyster in New Zealand because its rapid growth and high fecundity lead it to out-compete native oysters on growing racks. While this has provided New Zealand oyster farmers with an easily marketable species it has also meant they have lost the opportunity to cultivate and market a niche product as has been successfully demonstrated with the New Zealand green lipped mussel (*Perna canaliculus*). Equally northern Māori have been deprived of a toanga species.

The introduction of *M. senhousia* to New Zealand has resulted in significant alterations in benthic community structure, composition and abundance associated with the ephemeral *M. senhousia* mats (Creese et al. 1997, Hayward et al. 1999a). In the Waitemata harbour, not only can *M. senhousia* produce substantial mats, but they also remain in lower densities throughout the harbour, contributing to changes in the benthic species composition (Lohrer et al. 2008). The impact of *M. senhousia* could be both beneficial, by providing an abundant food source to higher trophic levels, and detrimental, by reducing the diversity of native species upon which ecosystem resilience may have been reliant.

New Zealand has a substantial aquaculture industry heavily reliant on the culture of the native mussel *P. canaliculus*. This could be at risk with the introduction of a competing bivalve species. Given the reliance upon a single species, the New Zealand industry could be vulnerable if a successfully introduced mussel caused a reduction in productivity similar to that already caused by the blue mussel (*M. galloprovincialis*). The success of *P. canaliculus* as an aquaculture species can be linked to qualities which make it a potential invasive in other countries (Pollard & Hutchings 1990). These features could provide *P. canaliculus* with a substantial capacity to resist invasive mussels, however, this may be limited to certain environments as noted for *P. perna* resisting *M. galloprovincialis* in South Africa (Nicastro et al. 2008). In addition to substantial economic effects, any loss or reduction in abundance of *P. canaliculus* will also have impacts on cultural and social values as it is a significant target species for recreational and customary harvesters.

Bivalves may facilitate the import of associated nuisance flora and fauna that are not established in New Zealand. For example, fouling species on mussel shells may reduce the profit margin for in-shell products due to the need for additional processing. Other associated species could foul aquaculture infrastructure or wild fishery habitats impacting of productivity, accessibility and/or desirability of harvests.

4.4.3. Risk estimation

Given that particular bivalve species:

- are commonly associated with hull fouling;
- have been introduced and established globally via vessel fouling;
- have known invasive behaviour;
- can be ecosystem engineers;
- can facilitate the entry and establishment of a variety of other non-indigenous organisms;
- have been found to cause substantive change to native ecosystems globally, including New Zealand;
- may negatively impact green lipped mussel aquaculture which accounts for the majority of New Zealand's aquaculture earnings, in addition to customary and recreational harvesting;
- that are associated with high impacts, such as *Perna perna, Perna viridis, Mytilopsis sallei*, are yet to establish in New Zealand;

it is concluded that non-indigenous bivalves pose a non-negligible risk to New Zealand's coastal waters.

4.4.4. Risk management

The treatment of hulls to manage the risks posed by bivalves is covered by the generic options (Section 3). Specific considerations for hull fouling bivalves should include appropriate local containment. While it may be possible to remove large specimens or clumps by hand, caution should be exercised due to the likelihood of release of associated non-indigenous organisms into the receiving environment. Moreover, physical damage may cause the accidental release of gametes which could fertilise and subsequently found a population.

4.4.5. References

Alberini, A; Zanatta, V; Rosato, P (2007) Combining actual and contingent behaviour to estimate the value of sports fishing in the Lagoon of Venice. *Ecological Economics*, 61: 530-541.

Apte, S; Holland, B S; Godwin, L S; Gardner, J P A (2000) Jumping ship: a stepping stone event mediating transfer of a non-indigenous species via a potentially unsuitable environment. *Biological Invasions* 2: 75-79.

Bayne, B L (1976) *Marine mussels: their ecology and physiology*. Cambridge University Press; Cambridge, United Kingdom.

Bownes, S J; McQuaid, C D (2006) Will the invasive mussel *Mytilus galloprovincialis* Lamarck replace the indigenous *Perna perna* L. on the south coast of South Africa? *Journal of Experimental Marine Biology and Ecology* 338: 140-151.

Chapman, M G; People, J; Blockley, D (2005) Intertidal assemblages associated with natural *corallina* turf and invasive mussel beds. *Biodiversity and Conservation* 14: 1761-1776.

Creese, R; Hooker, S; De Luca, S; Wharton, Y (1997) Ecology and environmental impact of *Musculista senhousia* (Mollusca: Bivalvia: Mytilidae) in Tamaki Estuary, Auckland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 31: 225-236.

Crow, S P (1979) Marine fouling on the Maui A platform. Unpublished thesis, The University of Auckland, Auckland.

Dafforn, K A; Glasby, T M; Johnston, E L (2008) Differential effects of tributyltin and copper antifoulants of recruitment of non-indigenous species. *Biofouling* 24: 23-33.

Dinamani, P (1971) Occurrence of the Japanese oyster, *Crassostrea gigas* (Thunberg), in northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 5: 352-357.

Dinamani, P (1991) Introduced Pacific oysters in New Zealand. In: Leffler, M; Greer, J R (Eds) *The ecology of* Crassostrea gigas *in Australia, New Zealand, France and Washington*. Maryland Sea Grant, College Park, Maryland. VM-SG-TS-92-07.

Farrapeira, C M R; de Melo, A V de O M; Barbosa, D F; da Silva, K M E (2007) Ship hull fouling in the port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55: 207-221.

Gardner, J P A (2004) A historical perspective of the genus *Mytilus* (Bivalvia: Mollusca) in New Zealand: multivariate morphometric analyses of fossil, midden and contemporary blue mussels. *Biological Journal of the Linnean Society* 82: 329-344.

Galil, B S; Bogi, C (2008) *Mytilopsis sallei* (Mollusca: Bivalvia: Dreissenidae) established on the coast of Israel. *JMBA 2 Biodiversity Records* Published Online

Glasby, T M (1999) Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney, Australia. *Estuarine, Coastal and Shelf Science* 48: 281-290.

Gollasch, S (2002) The importance of ship hull fouling as a vector of species introductions into the North Sea. *Biofouling* 18: 105-121.

Hanekom, N (2008) Invasion of an indigenous *Perna perna* mussel bed on the south coast of South Africa by an alien mussel *Mytilus galloprovincialis* and its effect on the associated fauna. *Biological Invasions* 10: 233-244.

Hayward, B W; Morley, M S; Hayward, J J; Stephenson, A B; Blom, W M; Hayward, K A; Grenfell, H R (1999a) Monitoring studies of the benthic ecology of Waitemata Harbour, New Zealand. *Records of the Auckland Museum* 36: 95-117.

Hayward, B W; Morley, M S; Stephenson, A B; Blom, W M; Grenfell, H R; Prasad, R; Rogan, D; Thompson, F; Cheetham, J; Webb, M (1999b) *Intertidal and subtidal biota and habitats of the central Waitemata Harbour*. Technical Publication 127, Auckland Regional Council; Auckland, New Zealand.

Hicks, D W; Tunnell, J W (1995) Ecological notes and patterns of dispersal in the recently introduced mussel, *Perna perna* (Linne, 1758), in the Gulf of Mexico. *American Malacological Bulletin* 11: 203-206

His, E; Robert, R (1987) Comparative effects of two anti-fouling paints on the oyster *Crassostrea gigas*. *Marine Biology* 95: 83-86.

Hoare, K; Beaumont, A R; Davenport, J (1995a) Variation among populations in the resistance of *Mytilus edulis* embryos to copper: adaptation to pollution? *Marine Ecology Progress Series* 120: 155-161.

Hoare, K; Davenport, J; Beaumont, A R (1995b) Effects of exposure and previous exposure to copper on growth of veliger larvae and survivorship of *Mytilus edulis* juveniles. *Marine Ecology Progress Series* 120: 163-168.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

James, P; Hayden, B (2000) *The potential for the introduction of exotic species by vessel hull fouling: a preliminary study*. NIWA client report WLG00/51.

Kimmerer, W J (2006) Response of anchovies dampens effects of the invasive bivalve *Corbula amurensis* on the San Francisco Estuary foodweb. *Marine Ecology Progress Series*, 324: 207-218.

Kochmann, J; Buschbaum, C; Volkenborn, N; Reise, K (2008) Shift from native mussels to alien oysters: Differential effects of ecosystem engineers. *Journal of Experimental Marine Biology and Ecology* 364: 1-10.

Lee, J E; Chown, S L (2007) *Mytilus* on the move: transport of an invasive bivalve to the Antarctic. *Marine Ecology Progress Series* 339: 307-310.

Lewis, J A (2004) Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research. *Ballast Water Research Series* Report No. 14 Department of Agriculture, Fisheries and Forestry; Canberra, Australia.

Lewis, J A; Coutts, A D M (2009) Chapter 24 Biofouling Invasions. In: Dürr, S; Thomason, J (Eds) *Biofouling* Wiley-Blackwell.

Lohrer, D; Townsend, M; Morrison, M; Hewitt, J (2008) *Change in the benthic assemblages of the Waitemata Harbour: Invasion risk as a function of community structure.* Biosecurity New Zealand Technical Paper No: 2008/17 MAF Biosecurity New Zealand; Wellington, New Zealand.

MAFBNZ (2009) *Perna perna* http://www.biosecurity.govt.nz/pests/perna-perna Accessed 20 April 2009.
Mitchem, E L: Fajans, J S; Baker, S M (2007). Contrasting response of two native crustaceans to non-indigenous prey, the green mussel, *Perna viridis*. *Florida Scientist* 70(2), 180-188.
Nicastro, K R; Zardi, G I; McQuaid, C D (2008) Movement behaviour and mortality in invasive and indigenous mussels: resilience and resistance strategies at different spatial scales. *Marine Ecology Progress Series* 372: 119-126.
Newell, R I E (2004) Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. *Journal of Shellfish Research* 23: 51-61.
Nichols FH, Thompson JK and Schemel LE (1990) Remarkable invasion of San Francisco Bay (California, USA) by the Asian clam *Potamocorbula amurensis*. II. Displacement of a former community. Marine Ecology Progress Series 66: 95–101
Norling, P; Kautsky, N (2007) Structural and functional effects of *Mytilus edulis* on diversity of associated species and ecosystem functioning. *Marine Ecology Progress Series* 351: 163-175.
Pauley, G B; Van Der Raay, B; Troutt, D (1988) *Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest)--Pacific oyster*. U.S. Fisheries and Wildlife Service Biological Report 82(11.85). U.S. Army Corps of Engineers, TR EL-82.4.
Pimentel, D; Zuniga, R; Morrison, D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* 52: 273-288.
Pollard, D A; Hutchings, P A (1990) A review of exotic marine organisms introduced to the Australian region. II. Invertebrates and algae. *Asian Fisheries Science* 3:223-250.
Porter, E T; Cornwell, J C; Sanford, L P (2004) Effect of oysters *Crassostrea virginica* and bottom shear velocity on benthic-pelagic coupling and estuarine water quality. *Marine Ecology Progress Series* 271: 61-75.
Rajagopal, S; Sasikumar, N; Azariah, J; Nair, K V K (1991) Some observations on biofouling in the cooling water conduits of a coastal power plant. *Biofouling* 3: 311-324.
Rajagopal, S; Venugopalan, V P; van der Velde, G; Jenner, H A (2006) Greening of the coasts: a review of the *Perna viridis* success story. *Aquatic Ecology* 40: 273-297.
Robinson, T B; Griffiths, C L; McQuaid, C D; Rius, M (2005) Marine alien species of South Africa - status and impacts. *African Journal of Marine Science* 27: 297-306.
Ruesink, J L; Feist, B E; Harvey, C J; Hong, J S; Trimble, A C; Wisehart, L M (2006) Changes in productivity associated with four introduced species: ecosystem transformation of a ‘pristine’ estuary. *Marine Ecology Progress Series* 311: 203-215.
Skerman, T M (1960) Ship fouling in New Zealand waters: A survey of marine fouling organisms from vessels of the coastal and overseas trade. *New Zealand Journal of Science* 3: 620-648.
Stafford, H; Willan, R C; Neil, K M (2007) The invasive Asian Green Mussel, *Perna viridis* (Linnaeus, 1758) (Bivalvia: Mytilidae), breeds in Trinity Inlet, tropical northern Australia. *Molluscan Research* 27: 105-109.

Willan, R C; Russel, B C; Murfet, N B; Moore, K L; McEnnulty, F R; Horner, S K; Hewitt, C L; Dally, G M; Campbell, M L; Bourke, S T (2000) Outbreak of Mytilopsis sallei (Récluz, 1849) (Bivalvia: Dreissenidae) in Australia. Molluscan Research 20: 25–30.

4.5. BRISTLEWORMS (CLASS: POLYCHAETA)

4.5.1. Hazard Identification

Bristleworms (polychaetes) are the largest class of worms (annelids) with more than 13,000 species represented in marine, brackish and freshwater environments with pelagic, epibenthic and infaunal habitats (Read 2007). Although bristleworm lengths range from 1 mm to 3 m, the majority are less than 10 cm (Ruppert et al. 2004). Bristleworms are differentiated by the presence of paired appendages (parapodia) on most segments that contain bundles of bristles (chaetae).

Bristleworms have varied life histories and reproductive strategies (Gherardi et al. 2007, Read 2007). Although the majority of species reproduce sexually, asexual reproduction does occur (Ruppert et al. 2004, Bybee et al. 2007, Gherardi et al. 2007). Fertilised gametes may develop in the plankton, be brooded, or develop within the adult body (resulting in live birth) (Giangrande et al. 2000, Bybee et al. 2007). Most bristleworms have subannual life-spans, although some species, such as the known invasive *Sabella spallanzanii*, may live for more than one year (Bybee et al. 2007, Read 2007, Ramey 2008).

Ecologically, bristleworms are important as food for other animals, bioturbators and habitat providers (Hutchings 1998, Volkenborn & Reise 2007). Bristleworms can be divided into sedentary and mobile forms. Sedentary forms are suspension feeders, living either infaunally or epifaunally in tubes, burrows or calcareous tubes (Ruppert et al. 2004). Suspension feeding forms commonly have elaborate appendages for feeding and respiration and their gregarious nature and tubes provide habitat for other organisms (Dubois et al. 2002). Mobile forms include active swimmers, crawlers and burrowers, and may be carnivores, herbivores or scavengers (Ruppert et al. 2004). Bristleworms can be extremely abundant and may reach densities of thousands of individuals per m^2 (Hutchings 1998, Dubois et al. 2002, Essink & Dekker 2002, Labrune et al. 2007).

There are many well known occurrences of non-indigenous bristleworms impacting upon core values. For example, *Polydora* spp. *Hydroides ezoensis* and *Terabrasabella heterounicinata* are recognised aquaculture pests, responsible for the loss of significant quantities of stock, such as oysters in Japan, abalone in the USA and scallops in Norway (Culver & Kuris 2000, Nishi 2002, Radashevsky & Olivares 2005, Simon et al. 2005). Reef-forming bristleworms, such as *Ficopomatus enigmaticus*, can dramatically alter community and sediment characteristics (Wilson et al. 1996 *In* Hutchings 1998, Schwindt et al. 2001, Bruschetti et al. 2008). Some introduced species remain restricted to disturbed environments such as ports where their habitat forming capabilities may increase biodiversity (Ranasinghe et al. 2005, Cinar et al. 2006, Marrs undated).

Given that bristleworms:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that bristleworm species represent a potential hazard via vessel biofouling.

4.5.2. Risk Assessment

4.5.2.1. Entry assessment

Bristleworms are present in the majority of marine environments and are most prominent in soft-bottom sediments (Hutchings 1998, Read 2007). Fouling and errant bristleworms are components of shallow marine fouling assemblages worldwide and are likely to be found within ports and other vessel hubs (OECD 1967). Latitudinal trends in general bristleworm diversity are contradictory and uncertain (Hutchings 1998, Giangrande & Licciano 2004, Metcalfe & Glasby 2008). For example, fouling serpulid tubeworms are reported to have heavier settlement in tropical waters (OECD 1967) but this may be confounded by the dramatically higher settlement and growth in areas of high nutrients and sheltered conditions, such as the Mediterranean, the Indian Ocean, and ports in general (OECD 1967, Simon et al. 2005, Marrs undated).

Bristleworms are common foulers of vessel hulls (Lewis 2004). For example, in the MAFBNZ commissioned research on biofouling 22% of vessels sampled had bristleworm fouling (Inglis et al. in prep.). Tubeworms, in particular, can be dominant fouling organisms aided by their ability to grow and reproduce rapidly (Lewis 2004), and their tolerance of previously established algal growth (OECD 1967). Moreover, a number of tubeworms show tolerance towards heavy metals, including copper and tributyltin, which aids in both their fouling of vessels and survival in port areas (OECD 1967, Lewis 2004, Lewis et al. 2006, Burlinson & Lawrence 2007, Qui et al. 2007). Errant bristleworms have also been recorded as hull biofouling (Coutts & Dodgshun 2007), however, unlike tubeworms errant bristleworms require existing biofouling to provide shelter and food.

Bristleworms are known to be translocated via hull fouling and remain viable. Lewis et al. (2006) attribute hull fouling to the introduction of *H. sanctaecrucis* to Australia from the Pacific side of the Americas or possibly Asia. The introduction of *H. ezoensis* to and throughout Europe from the north-west Pacific has also been attributed to hull fouling (Zibrowius 1992). Thirty three species of non-indigenous bristleworms have been recorded in New Zealand (Kospartov et al. in press).

Due to their tubes remaining after death, the actual numbers of viable tubeworms entering receiving environments is uncertain. However, although the original organism is gone, the tube still represents a potential shelter or substrate for other non-indigenous organisms.

Given that bristleworms:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous bristleworm species into New Zealand via vessel biofouling is non-negligible.

4.5.2.2. Establishment assessment

Bristleworms entering as biofouling are likely to find suitable habitat in New Zealand, particularly in ports of first arrival, where there are abundant settlement substrates and high nutrient levels. Moreover, many bristleworm species can tolerate both polluted and pristine habitats (OECD 1967, Simon et al. 2005, Bruschetti et al. 2008). Errant bristleworms are not likely to be restricted by feeding specialisation, therefore it is likely that if an errant bristleworm can survive translocation it will be able to survive in New Zealand port environments.

Bristleworms are able to transfer from vessels to New Zealand substrates through release of gametes/larvae. Some mobile species may be able to swim but most successful releases will be in association with other fouling being removed. Release, settlement and growth rates of larval worms may depend on specific temperatures or conditions (OECD 1967, Sion et al. 2005, Bybee et al. 2007). For example, OECD (1967) notes that "*Hydroides norvegica* in the mouth of the Clyde took three months to attain a length which specimens in Madras Harbour reached in three weeks." In temperate waters, reproduction and settlement generally occurs in summer (OECD 1967, Currie et al. 2000, Giangrande et al. 2000, Simon et al. 2005). Ports offer sheltered, nutrient rich environments increasing the likelihood of establishment if propagules are released (Simon et al. 2005, Marrs undated). As sexual reproduction is probably contingent on conspecifics being close enough to receive spawning cues (Ruppert et al. 2004), mature conspecifics must reach critical density to trigger gamete release.

The physical movement of adult non-indigenous bristleworms from vessels onto New Zealand substrates may also result in population establishment. Many sedentary forms of bristleworm can regenerate body parts (Ruppert et al. 2004, Gherardi et al. 2007), therefore, abrasion of individuals from the hull surface may result in viable adults entering the New Zealand environment. Alternatively, entire tubes or aggregations may be dislodged from the hull surface. This is particularly relevant if poor hull cleaning practices are employed. Errant bristleworms can swim off the hull and onto receiving substrates where they may establish. Spawning requirements will become a factor after the establishment of adult populations/individuals.

There are known recent recordings of non-indigenous bristleworms establishing in New Zealand. Of the thirty three species of non-indigenous bristleworms have been recorded in New Zealand with eighteen confirmed as established (Kospartov et al. in press). In the last five years three species (*H. ezoensis*, *S. spallanzanii*, and *Paralepidonotus ampulliferus*) have become established (Kospartov et al. in press).

Given that particular bristleworm species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous bristleworm species in New Zealand via vessel biofouling is non-negligible.

4.5.2.3. Consequence assessment

The establishment of non-indigenous bristleworms in New Zealand is likely to result in impacts to benthic ecosystems. For example, tube dwelling species may produce reefs which modify physical habitat or change plankton communities and light attenuation through filter-feeding (Holloway 1999, Bruschetti et al. 2008). Bristleworms may facilitate biodiversity loss or gain (Holloway 1999) with impacts being most apparent when individuals are densely aggregated (Ross et al. 2007). For example, the mud tubes built by *P. ciliata* can agglomerate to form layers of mud up to 20 cm thick. These layers can eliminate the native flora and fauna or alter ecosystem dynamics (Daro and Polk 1973 *In* NIMPIS 2010). By contrast, crab populations in the SW Atlantic increased following the introduction of *F. enigmaticus* reefs, which then impacted other organisms through predation (Schwindt et al. 2001).

Some introduced bristleworms have been reported to increase diversity in disturbed/polluted environments where they provide habitat and food (Ranasinghe et al. 2005, Cinar et al. 2006, Marrs undated). However, bristleworm reefs formed by *F. enigmaticus,* have been shown to have had positive effects on the densities of other non-indigenous species when compared to habitats created by native species (Heiman et al. 2008). Therefore, the establishment of non-indigenous bristleworm reefs may aid the establishment of other non-indigenous species. It is noted, however, that bristleworms can be quite ephemeral and colonies can wax and wane with anthropogenic and natural disturbances (Zulkhe, 2001, Dubois et al. 2002, Chollett & Bone 2007).

The proliferation of calcareous tubes on vessels can compromise antifouling systems, block intake pipes, increase cleaning costs and decrease fuel efficiency. In addition, bristleworms may impact the economic value of aquaculture and fisheries. For example, *Polydora* spp. are well known shell-borers that are able to thrive under aquaculture conditions, i.e. nutrient rich environments with dense organism aggregations. Examples of negative impacts of *Polydora* spp. include: greater than 50% mortality in abalone/paua (Lleonart et al. 2003a), reduced meat and shell weight in oysters (*C. gigas*) (Royer et al. 2006), and the loss of one third of Norways 1997 scallop spat (Mortensen et al. 2000). In addition, tubeworms, of both boring and non-boring species, compete with cultured bivalves for food and oxygen (Ropert & Goulletquer 2000).

In New Zealand, *P. websteri* and *P. hoplura* are non-indigenous tubeworm species that have had negative effects on oyster aquaculture (Handley 1995, Handley & Bergquist 1997). Other boring species such as *Boccardia knoxi* also have similar effects (Llenart et al 2003b). The recent incursion response to *S. spallanzanii* has cost New Zealand $1 million, with a further $1.5 million earmarked for further control.

4.5.3. Risk Estimation

Given that particular bristleworm species:

- are commonly associated with vessel biofouling;
- are capable of forming establishing populations in New Zealand;
- have recently established populations within New Zealand;
- have well known invasive behaviours;
- can be ecosystem engineers;
- can facilitate the entry and establishment of a variety of other non-indigenous organisms;
- are known to impact marine ecosystems;
- are known to impact aquaculture and other economic values;

it is concluded that the import of non-indigenous bristleworm species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.5.4. Risk Management

A generic, broad spectrum management option for biofouling should mitigate bristleworm associated risks (Section 3). However, the mobility of bristleworms presents further management difficulties, and as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape. In addition, the ability of bristleworms to regenerate body parts and the potential to spawn should be considered during the application of management options.

4.5.5. References

Bruschetti, M; Luppi, T; Fanjul, E; Rosenthal, A; Iribarne, O (2008) Grazing effect of the invasive reef-forming polychaete *Ficopomatus enigmaticus* (Fauvel) on phytoplankton biomass in a SW Atlantic coastal lagoon. *Journal of Experimental Marine Biology* 345: 212-219.

Burlinson, F C; Lawrence, A J (2007) A comparison of acute and chronic toxicity tests used to examine the temporal stability of a gradient in copper tolerance of *Hediste diversicolor* from the Fal estuary, Cornwall, UK. *Marine Pollution Bulletin* 54: 66-71.

Bybee, D R; Bailey-Brock, J H; Tamaru, C S (2007) Gametogenesis and spawning periodicity in the fan worm *Sabellastarte spectablis* (Polychaeta: Sabellidae). *Marine Biology* 151: 639-648.

Chollett, I; Bone, D (2007) Effects of heavy rainfall on polychates: Differential spatial patterns generated by a large scale disturbance. *Journal of Experimental Biology and Ecology* 340: 113-125.

Cinar, M E; Katagn, T; Ozturk, B; Egemen, O; Ergen, Z; Kocatas, A; Onen, M; Kirkim, F; Bakir, K; Kurt, G; Dagli, E; Kaymakci, A; Acik, S; Dogan, A; Ozcan, T (2006) Temporal changes of soft- bottom zoobenthic communities in and around Alsancak Harbour (Izmir Bay, Aegean Sea), with special attention to the autecology of exotic species. *Marine Ecology* 27: 229-246.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.

Culver, C S; Kuris, A M (2000) The apparent eradication of a locally established introduced marine pest. *Biological Invasions* 2: 245-253.

Dubois, S; Retiere, C; Olivier, F (2002) Biodiversity associated with *Sabellaria alveolata* (Polychaeta: Sabellariidae) reefs: effects of human disturbances. *Journal of the Marine Biological Association of the United Kingdom* 82: 817-826.

Essink, K; Dekker, R (2002) General patterns in invasion ecology tested in the Dutch Wadden Sea: the case of a brackish-marine polychaetous worm. *Biological Invasions* 4: 359-368.

Gherardi, M; Sciscioli, M; Lepore, E; Todisco, G; Giangrande, A (2007) The role of schiogenesis in population dynamics of *Timarete filigera* (Polychaeta: Cirratulidae) 2-years observations in the Port of Bari (South Adriatic Sea). *Marine Ecology* 28: 1-9.

Giangrande, A; Licciano, M (2004) Factors influencing latitudinal pattern of biodiversity: an example using Sabellidae (Annelida, Polychaeta). *Biodiversity and Conservation* 13: 1633-1646.

Giangrande, A; Licciano, M; Pagliara, P; Gambi, M C (2000) Gametogenesis and larval development in *Sabella spallanzanii* (Polychaeta: Sabellidae) from the Mediterranean Sea. *Marine Biology* 136: 847-861.

Handley, S J (1995) Spionid polychaetes in Pacific Oysters, *Crassostrea gigas* (Thunberg) from Admiralty Bay, Marlborough Sounds, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 29: 305-309.

Handley, S J; Bergquist, P R (1997) Spionid polychaete infestations of intertidal Pacific oysters *Crassostrea gigas* (Thunberg), Maharangi Harbour, northern New Zealand. *Aquaculture* 153: 191-205.

Heiman, K W; Vidargas, N; Micehi, F (2008). Non-native habitat as home for non-native species: comparison of communities associated with invasive tubeworm and native oyster reefs. *Aquatic Biology* 2: 47-56.

Holloway, M G (1999) Ecological effects of the Introduced Fanworm, *Sabella spallanzanii*, on hard substrate epifaunal assemblages. PhD Thesis; University of Melbourne, Australia.

Hutchings, P (1998) Biodiversity and functioning of polychaetes in benthic sediments. *Biodiversity and Conservation* 7: 1133-1145.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Labrune, C; Gremare, A; Amouroux, J-M; Sarda, R; Taboada, S (2007) Assessment of soft-bottom polychaete assemblages in the Gulf of Lions (NW Mediterranean) based on a mesoscale survey. *Estuarine, Coastal and Shelf Science* 71: 133-147.

Lewis, J (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research*. Ballast Water Research Series Report No. 14. Dept of Agriculture, Fisheries and Forestry; Canberra, Australia; 129 pp.

Lewis, J A; Watson, C; ten Hove, H A (2006) Establishment of the Caribbean serpulid tubeworm *Hydroides sanctaecrucis* Kroyer [in] Morch, 1863, in Northern Australia. Biological Invasions 8: 665-671.

Marrs, R (undated) *Invasive non-native species in the UK. Hydroides ezoensis*. http://138.253.199.114/IAAP%20Web/IAAPwebsite/Marinesppintro2.asp?ID=30. Accessed 20 February 2009.

Metcalfe, K N; Glasby, C J (2008) Diversity of Polychaeta (Annelida) and other worm taxa in mangrove habitats of Darwin Harbour, northern Australia. *Journal of Sea Research* 59: 70-82.

NIMPIS (2010). *Polydora ciliata* impact details. *National Introduced Marine Pest Information System* (Eds: Hewitt C.L., Martin R.B., Sliwa C., McEnnulty, F.R., Murphy, N.E., Jones T. & Cooper, S.). Web publication <http://crimp.marine.csiro.au/nimpis>, Accessed: 2nd February 2010.

Nishi, E (2002) *Hydroides ezoensis. In:* The Ecological Society of Japan (Eds) *Handbook of Alien Species in Japan.* Chijin-Shokan, Tokyo; p180. (in Japanese)

Qiu, J-W; Tang, X; Zheng, C; Li, Y; Huang, Y (2007) Copper complexation by fulvic acid affects copper toxicity to the larvae of the polychaete *Hydroides elegans*. *Marine Environment Research* 64: 563-573.

Radashevsky, V I; Olivares, C (2005) *Polydora uncinata* (Polychaeta: Spionidae) in Chile: an accidental transportation across the Pacific. *Biological Invasions* 7: 489-496.

Railkin, A I (2004) Marine Biofouling: Colonization Processes and Defences. CRC Press; 303pp.

Ramey, P (2008) Life history of a dominant polychaete, *Polygordius jouinae*, in inner continental shelf sands of the Mid-Atlantic Bight, USA. *Marine Biology* 154: 443-452.

Ranasinghe, J A; Mikel, T K; Velarde, R G; Weisberg, S B; Montagne, D E; Cadien, D B; Dalkey, A (2005) The prevalence of non-indigenous species in southern Californian embayments and their effects on benthic macroinvertebrate communities. *Biological Invasions* 7: 679-686.

Read, G (2007) Bristleworms (Class Polychaeta of Phylum Annelida). In MacDiarmid A (Ed) *The Treasures of the Sea: Nga Toanga a Tangaroa. A Summary of the Biodiversity in the New Zealand Marine Ecoregion*. WWF-New Zealand; Wellington, New Zealand.

Ross, D J; Keough, M J; Longmore, A R; Knott, N A (2007) Impacts of two introduced suspension feeders in Port Phillip Bay, Australia. *Marine Ecology Progress Series* 340: 41-53.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Schwindt, E; Bortolus, A; Iribarne, O O (2001) Invasion of reef-builder polychaete: direct and indirect impacts on the native community structure. *Biological Invasions* 3: 137-149.

Simon, C A; Kaiser, H; Britz, P J (2005) The life history responses of the abalone pest, *Terebrasabella heterouncinata*, under natural and aquaculture conditions. *Marine Biology* 147: 135-144.

Volkenborne, N; Reise, K (2007) Effects of *Arenicola marina* on polychaete functional diversity revealed by large-scale experimental lugworm exclusion. *Journal of Sea Research* 57: 78-88.

Zibrowius H (1992) Ongoing modification of the Mediterranean marine and flora by the establishment of exotic species. Mesogee 51: 83-107.

Zuhlke, R (2001) Polychaete tubes create ephemeral community patterns: Lanice conchilega (Pallas, 1766) associations studied over six years. Journal of Sea Research 46 (3-4): 261-272.

4.6. BRYOZOANS (PHYLUM: BRYOZOA)

4.6.1. Hazard Identification

Bryozoans are typically sessile colonial marine organisms of which there are over 4,000 known species. Bryozoans are common components of shallow marine and continental shelf biota with a few species found in deep sea and freshwater environments (Gordon & Mawatari 1992, Hughes 2001, Stone 2006). Although most bryozoans are sessile colonial benthic filter feeders, one genus is solitary and motile and one polar species forms pelagic colonies (Peck et al. 1995, Ruppert et al. 2004). Representatives can be found in environments ranging from tropical to polar and sheltered to energetic (Barnes & Whittington 1999, Creary 2002, Ruppert et al. 2004, Mallela 2007).

The typical bryozoan life history involves a short pelagic larval phase and a sessile colonial adult phase. Larvae are brooded before being expelled into the water column for minutes to hours, before settling on nearby substrate. A single settled larva transforms into the adult form, called a zooid, which is composed of the animal encased in a tiny calcareous tube (Ruppert et al. 2004). This first zooid is called the ancestrula. A bryozoan colony is formed following consecutive budding of zooids from the ancestrula and then from the other zooids. After colony formation many zooids exhibit polymorphisms that assist with the division of labour, for example, the zooids on the edge of the colony continue to bud while others may be solely dedicated to cleaning the colony surface or reproduction (Ryland 1965). Although all colonies have ovaries and testes, the development and release of eggs and sperm is usually staggered to allow enough cross-fertilisation between colonies to ensure outbreeding (Ruppert et al. 2004).

Bryozoans are ecologically important as reef builders and as a result create habitat for a variety of organisms (Martindale 1992, Stone 2006, Mallela 2007, Gordon et al. 2009). Bryozoans are able to dominate benthic substrates in energetic environments, with encrusting forms growing quickly to occupy available space and smothering other organisms. In more sheltered environments erect forms out-compete other organisms by vertical growth (Ward & Thorpe 1989, Barnes & Whittington 1999, Gordon et al. 2009).

Bryozoa are known components of marine biofouling assemblages and have a history of introductions (Ryland 1965, Gordan & Mawatari 1992). For example, the introduction of *Tricellaria inopinata* to the Venice Lagoon, Italy, resulted in almost total dominance of the lagoon's benthic habitat between 1989 and 1993 and reduced the native bryozoan fauna to small marginal habitats (Occhipinti Ambrogi 2000). Non-indigenous bryozoa may modify habitat and provide hard substrates for other organisms including other non-indigenous species (Zabin et al. 2007).

Given that bryozoans:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that bryozoan species represent a potential hazard via vessel biofouling.

4.6.2. Risk Assessment

4.6.2.1. Entry assessment

Bryozoans are found worldwide although diversity is richest at mid to high latitudes, with an emphasis in the southern hemisphere (Barnes & Griffith 2008). Australasia is considered the centre of bryozoan biodiversity, with New Zealand currently considered the richest in diversity with the highest percentage of endemic species (60%) (Barnes & Griffith 2008). As common members of fouling communities in ports it is likely that propagules will encounter vessels in port, especially within Australasia (Ryland 1965).

While bryozoans have multiple possible entry pathways, including floating debris, other organisms and ballast water (Mackie et al. 2006), hull fouling is probably the most common introduction pathway (Key et al. 1999, Barnes & Fraser 2003, Barnes et al. 2004, Barnes & Milner 2005). Generally bryozoans are one of the more common hull fouling organisms, and are often amongst the first and most rapid colonisers of hull surfaces (Ryland 1965, Railkin 2004). Many bryozoans are tolerant of the heavy metal concentrations found in ports and anti-fouling paints and are attracted to the bare metal of the rudder shaft and propeller (Wisely 1963, Piola & Johnston 2006a). They are also known to grow profusely in pipes (Railkin 2004).

Typically, hull fouling bryozoans are those that inhabit estuaries and ports (Ryland 1965, Gordan & Mawatari 1992). As a result, these species are likely to have broad tolerances to changing temperature, salinity and turbidity. For example, Allen (1953) noted that *Watersiporia cucullata*, found on many Australian Naval vessels, survived voyages of up to three months at speeds reaching 30 knots. Moreover, bryozoans, particularly the seasonal species, can regress or become “dormant” during unfavourable conditions and rejuvenate once conditions become favourable, a behaviour which may aid in translocation (Keough 1986, Piola & Johnstone 2006b). In terms of morphology, both encrusting (e.g. *W. subtorquata*) and branching species (e.g. *Bugula neritina*) are known to survive translocation as hull fouling (Ryland 1965).

All non-indigenous bryozoans introduced to New Zealand are likely to have been introduced via hull fouling (Kospartov et al. in press). In the MAFBNZ commissioned research on biofouling, bryozoans were found on 26% of vessels entering New Zealand (Inglis et al. in prep.). Inglis et al. (in prep.) noted that 17 of the 30 non-indigenous bryozoan species translocated to New Zealand were not yet established.

Given that particular bryozoan species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous bryozoan species into New Zealand via vessel biofouling is non-negligible.

4.6.2.2. Establishment assessment

Bryozoans are likely to find suitable habitat and environments in New Zealand. The majority of hull fouling bryozoans are likely to be limited only by the temperature or environment encountered in New Zealand’s ports of first arrival. Low temperatures are known to affect bryozoan growth, reproduction, and mortality (Amui-Vedal et al. 2007, Borque et al. 2007). As a result, at-risk locations to temperate bryozoa establishment are ports and marinas during spring and summer months. Due to low temperatures, tropical bryozoa are unlikely to be able to establish in New Zealand.

Bryozoans are able to transfer from vessels through the release of gametes, larvae or colony fragments. Bryozoan growth and reproduction are generally greatest in the summer months, which coincide with the period of high water productivity in New Zealand, making this time the most likely for bryozoan establishment (Ryland 1965, Amui-Vedal et al. 2007, Borque et al. 2007). Opportunities for propagule release which will result in successful establishment are probably restricted to port and marina environments. Moreover, given the short time bryozoan larvae spend in the water column and the short distance viable fragments are likely to disperse, these environments are the most likely point of initial establishment.

There are recent recordings of non-indigenous bryozoa establishing in New Zealand. Of the 23 established non-indigenous species of bryozoa in New Zealand five have been reported since 2000 (Kospartov et al. in press). Moreover, 42% of bryozoans were identified as non-indigenous in a survey of New Zealand ports (Gordon & Mawatari 1992).

Given that particular bryozoan species:
- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous bryozoan species in New Zealand via vessel biofouling is non-negligible.

4.6.2.3. Consequence assessment

The consequences of non-indigenous bryozoan establishment to New Zealand may range from nuisance fouling to modified habitat and ecosystem function. Introduced bryozoa in New Zealand have already been found to significantly impact environmental and economic values. For example, *W. arcuata* overgrew all subtidal benthic habitat in the Auckland region when first introduced in 1957 (Skerman 1960, Morton and Miller 1968). More recently, in the same region and habitat, *Zoobotryon verticillatum* has formed gelatinous masses up to a metre long (Gordon et al. 2009). Similarly, *Membraniporopsis tubigera* has negatively impacted fisheries as large calcareous formations have fouled fishing equipment (Gordon et al. 2006).

Bryozoa introductions have also impacted values elsewhere. For example, the large calcareous formations formed by *Schizoporella variabilis* in the soft sediments of San Francisco Bay provide habitat for a range of organisms (Zabin et al. 2007). This has had both positive and negative consequences as while the overall biodiversity has increased the bryozoans provide habitat for a number of non-indigenous species, for example, they are a 'nursery' for the commercially destructive Atlantic oyster drill, *Urosalpinx cinera* (Zabin et al. 2007). In addition, in the Gulf of Maine, USA, colonies of the non-indigenous bryozoan *Membranipora membranacea* have formed on kelp fronds leaving them brittle and susceptible to breakage. As the kelp forest has receded, the habitat has been taken over by the non-indigenous algae *Codium fragile* ssp. *tomentosoides* (Harris & Tyrrell 2001, Pratt & Grason 2007).

Although it is unlikely that hull maintenance will have to be undertaken on account of bryozoan colonies alone, it must be considered that dead bryozoan colonies facilitate the attachment and growth of other fouling organisms including calcareous worms, hydroids and those that do not readily settle on anti-fouling paint (Allen 1953, Floerl et al. 2004, Floerl et al. 2005).

Consequences to biodiversity and habitats may be restricted to disturbed and port environments. Hull fouling and port dwelling bryozoan's tolerance to pollution and thermal variation may not give them an advantage outside of these environments (Gordon & Mawatari 1992, Piola & Johnston 2006).

4.6.3. Risk Estimation

Given that particular bryozoan species:

- have recent records of entry into New Zealand via hull fouling;
- have recent records of establishment in New Zealand;
- have well known invasive behaviours;
- may facilitate the entry and establishment of a variety of non-indigenous organisms;
- may have significant impacts on core values, such as, those attributed to aquaculture and biodiversity;

it is concluded that the import of non-indigenous bryozoan species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.6.4. Risk Management

The generic broad spectrum approach to biofouling mitigation is considered appropriate for mitigation of the risks associated with hull fouling bryozoans (Section 3). Specific considerations for hull fouling bryozoans should include appropriate local containment due to the likelihood of release of associated non-indigenous organisms into the receiving environment.

4.6.5. References

Allen, F E (1953) Distribution of Marine Invertebrates by Ships. *Australian Journal of Marine and Freshwater Research* 4: 307-316.

Amui-Vedel, A-M; Hayward, P J; Porter, J S (2007) Zooid size and growth rate of the bryozoan *Cryptosula pallasiana* Moll in relation to temperature, in culture and its natural environment. *Journal of Experimental Marine Biology and Ecology* 353: 1-12.

Barnes, D K A; Griffiths, H J (2008) Biodiversity and biogeography of southern temperate and polar bryozoans. *Global Ecology and Biogeography* 17: 84-99.

Barnes, D K A; Fraser, K P P (2003) Rafting by five phyla on man-made flotsam in the Southern Ocean. *Marine Ecology Progress Series* 262: 289-291.

Barnes, D K A; Milner, P (2005) Drifting plastic and its consequences for sessile organism dispersal in the Atlantic Ocean. *Marine Biology* 146: 815-825.

Barnes, D K A; Whittington, M (1999) Biomechanics and mass mortality of erect bryozoans on a coral reef. *Journal of the Marine Biological Association of the United Kingdom* 79: 745-747.

Barnes D K A; Warren N L; Webb K; Phalan B; Reid K (2004) Polar pedunculate barnacles piggy-back on pycnogona, penguins, pinniped seals and plastics. *Marine Ecology Progress Series* 284: 305-310.

Borque, D; Davidson, J; MacNair, N G; Arsenault, G; LeBlanc, A R; Landry, T; Miron, G (2007) Reproduction and early life history of an invasive ascidian *Styela clava* Herdman in Prince Edward Island, Canada. *Journal of Experimental Marine Biology and Ecology* 342: 78-84.

Cocito, S; Sgorbini, S; Bianchi, C N (1998) Aspects of the biology of the bryozoan *Pentapora fascialis* in the northwestern Mediterranean. *Marine Biology* 131: 73-82.
Creary, M M (2002) Spatial distribution of epibenthic byozoans found on the roots of *Rhizophora mangle*, Kingston Harbour, Jamaica, W.I. *Bulletin of Marine Science* 73(2): 477-490.
Floerl, O; Inglis, G J (2003) Boat harbour design can exacerbate hull fouling. *Austral Ecology* 28: 116-127.
Floerl, O; Pool, T K; Inglis, G J (2004) Positive interactions between non-indigenous species facilitate transport by human vectors. *Ecological Applications* 14: 1724-1736.
Floerl, O; Inglis, G J; Marsh, H M (2005) Selectivity in vector management: an investigation of the effectiveness of measures used to prevent transport of non-indigenous species. *Biological Invasions* 7: 459-475.
Gordon, D P; Mawatari, S F (1992) Atlas of marine-fouling bryozoa of New Zealand ports and harbours. Miscellaneous publication 107. New Zealand Oceanographic Institute; New Zealand. 52 pp.
Gordon, D; Smith, F (2007) Lace Corals (Phylum Bryozoa). In MacDiarmid, A (Ed) *The Treasures of the Sea: Nga Taonga a Tangaroa. A Summary of the Biodiversity in the New Zealand Marine Ecoregion.* WWF-New Zealand; Wellington, New Zealand.
Gordon, D P; Ramalho, L V; Taylor, P D (2006) An unreported invasive bryozoan that can affect livelihoods – *Membraniporopsis tubiger* in New Zealand and Brasil. *Bulletin of Marine Science* 78(2): 331-342.
Gordon, D P; Taylor, P D; Bigey, F P (2009) Phylum Bryozoa. In Gordon, D P (ed) *New Zealand Inventory of Biodiversity. Volume 1*: Kingdom Animalia; Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press; Christchurch, New Zealand; pp 271-297.
Harris, L G; Tyrrell, M C (2001) Changing community states in the Gulf of Maine: synergism between invaders, overfishing and climate change. *Biological Invasions* 3: 9-21.
Hayward, B; Morley, M (2009) Contrasts in the recent introductions of two tunicates to northern New Zealand. In O'Shea, S; Costello, M J; O'Connell, K (eds) *Programme and Abstracts for Annual Conference in Auckland 2-4 September 2009*. Accessed at http://nzmss.rsnz.org, p 92.
Hughes, D J (2001) Quantitative analysis of a deep-water bryozoan collection from the Hebridean continental slope. *Journal of the Marine Biological Association of the United Kingdom*. 81: 987-993.
Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.
Keough, M J (1986) The distribution of the bryozoan Bugula neritina on seagrass blades: settlementgrowth and mortality. Ecology 67: 846-857.
Key, M M Jr; Winston, J E; Volpe, J W; Jeffries, W B; Voris, H K (1999) Bryozoan fouling of the Blue Crab *Callinectes sapidus* at Beaufort, North Carolina. *Bulletin of Marine Science* 64(3): 513-533.
Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.
Lewis, J (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research*. Ballast Water Research Series Report No. 14. Dept of Agriculture, Fisheries and Forestry; Canberra, Australia; 129 pp.
Lewis, P N; Hewitt, C L; Riddle, M; McMinn, A (2003) Marine introductions in the Southern Ocean: an unrecognised hazard to biosecurity. *Marine Pollution Bulletin* 46: 213-223.

Lewis, P N; Riddle, M J; Hewitt, C L (2004) Management of exogenous threats to Antarctica and the sub-Antarctic Islands: balancing risks from TBT and non-indigenous organisms. *Marine Pollution Bulletin* 49: 999-1005.

Mackie, J A; Keough, M J; Christidis, L (2006) Invasion patterns inferred from cytochrome oxidase I sequences in three bryozoans, *Bugula neritina*, *Watersiporia subtorquata*, and *Watersiporia arcuata*. *Marine Biology* 149: 285-295.

Mallela, J (2007) Coral reef encruster communities and carbonate production in cryptic and exposed coral reef habitats along a gradient of terrestrial disturbance. *Coral Reefs* 26: 775-785.

Martindale, W (1992) Calcified epibionts as palaeological tools: examples from Recent and Pleistocene reefs of Barbados. *Coral Reefs* 11: 167-177.

McCann, L D; Hitchcock, N G; Winston, J E; Ruiz, G M (2007) Non-native bryozoans in coastal embayments of the Southern United States: New records for the western Atlantic. *Bulletin of Marine Science* 80(2): 319-342.

Morton, J; Miller, M (1968) *The New Zealand Seashore*. Collins; Auckland, New Zealand; 638 pp.

Occhpinti Ambrogi, A (2000) Biotic Invasions in a Mediterranean Lagoon. *Biological Invasions* 2: 165-176.

O'Dea, A (2003) Seasonality and zooid variation in Panamanian encrusting bryozoans. *Journal of the Marine Biological Assosiation of the United Kingdom* 83: 1107-1108.

Peck, L S; Hayward, P J; Spencer-Jones, M E (1995) A pelagic bryozoan from Antarctica. *Marine Biology* 123:757-762.

Piola, R F; Johnston, E L (2006a) Differential tolerance to metals among populations of the introduced bryozoan Bugula neritina. *Marine Biology* 148: 997-1010.

Piola, R F; Johnstone, E L (2006b) Differential resistance to extended copper exposure in four introduced bryozoans. *Marine Ecology and Progress Series* 311: 103-114.

Pratt, M (undated) *Membranipora membranacea*: an invasive bryozoan in the Gulf of Maine. http://www.bowdoin.edu/~mpratt/invasives.html - accessed 8 July 2008.

Pratt, M C; Grason, E W (2007) Invasive species as a new food source: does a nudibranch predator prefer eating an invasive bryozoan? *Biological Invasions* 9: 645-655.

Railkin, A I (2004) Marine Biofouling: Colonization Processes and Defences. CRC Press; 303pp.

Ryland, J S (1965) Catalogue of Main Fouling Organisms (found on ships coming into European water): Polyzoa. Organisation for Economic Co-operation and Development; Paris, France.

Rubin, J A (1985) Mortality and avoidance of competitive overgrowth in encrusting Bryozoa. Marine Ecology Progress Series 23: 291-299.

Ruppert, E E; Fox, R S; Barnes, R D (2004) Invertebrate Zoology: A functional evolutionary approach (7th edition). Thompson Learning; Victoria, Australia.

Saunders, M; Mexetas, A (2008) High recruitment of the introduced bryozoan Membranipora membranacea is associated with kelp bed defoliation in Nova Scotia, Canada. Marine Ecology Progress Series 369: 139-151.

Skerman, T M (1960) The recent establishment of the Polyzoan *Watersipora cucullata* (Busk) in Auckland Harbour, New Zealand. *New Zealand Journal of Science* 3: 615-619.

Stone, R P (2006) Coral habitat in the Aleutian Islands of Alaska: depth distribution, fine scale species associations, and fisheries interactions. *Coral Reefs* 25: 229-238.

Ward, M A: Thorpe, J P (1989) Assessment of space utilisation in a subtidal temperate bryozoan community. *Marine Biology* 103: 215-224.

Winston, J E (1977) Distribution and Ecology of Estuarine Ectoprocts: A Critical Review, *Chesapeake Science* 18(1): 34-57.

Wisely, B (1963) Effects of anti-fouling paints on settling larvae of the bryozoan *Bugula neritina* L. *Australian Journal of Marine and Freshwater Research* 14(1): 44-59.

Wood, R A (1999) *Reef Evolution*. Oxford University Press.
Wyatt, A S J; Hewitt, C L; Walker, D I; Ward, T J (2005) Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment, *Diversity and Distributions* 11(1): 33-44.
Zabin, C J; Obermolte, R; Gentry, J (2007) Great balls of bryos: novel substrate on the mud flats in San Francisco Bay. *Fifth International Conference on Marine Bioinvasions*. Cambridge, Massachusetts.

4.7. CRABS (INFRAORDER: BRACHYURA)

4.7.1. Hazard Identification

The Brachyura, or true crabs, are successful and highly specialised decapods. This group contains over 4,500 species, the majority of which are found in benthic marine habitats, however, freshwater and terrestrial species exist (Ruppert et al. 2004). Crabs range in size from only a few millimetres wide to the Japanese spider crab whose chelipeds may span three metres (Ruppert et al. 2004). Although the majority of crabs cannot swim, notable exceptions are members of the Portunidae family (Ruppert et al. 2004).

Ecologically, crabs are important components of benthic communities due to their feeding and foraging activities. Many crabs are active food gatherers and feed upon a range of organisms, for example, the highly invasive *Carcinus maenus* has been observed to prey upon organisms from more than 100 families and 158 genera in 5 plant/protist and 14 animal groups (Cohen et al. 1995). Predation by crabs can regulate the abundance and species composition of local fauna, such as, marine and estuarine infaunal soft-bottom communities (Quammen 1984), epibenthic microbiota (Foreman 1985) and meiofauna (Hoffman et al. 1984). Burrowing and foraging activity can also impact the environment, especially in estuarine intertidal areas (Montague 1982, DePatra & Levin 1989). For example, crabs can retain nitrogen in the system, increase oxygenation and drainage of the soil (Montague 1982), and increase meiofauna (DePatra & Levin 1989). In addition, crabs provide food for many apex predators such as sharks and octopuses and are targeted by humans for consumption both commercially and recreationally.

The life-cycle of crabs involves a pelagic larval stage and a mobile benthic adult stage. Crabs have separate sexes and usually copulate, exchanging spermatophores, with the eggs brooded by the females (Ruppert et al. 2004). The majority of crabs spawn in seawater irrespective of whether they inhabit freshwater or terrestrial environments (Ruppert et al. 2004). Brackish-water forms, such as blue crabs (*Callinectes* spp.) and river immigrants such as *Eriocheir sinensis,* return to more saline waters for breeding and larval release (Rudnick et al. 2005).

Species of crab are known to have established outside their native range and some have had severe ecological, economic and social impacts. *C. maenas*, *Charybdis japonica* and *E. sinensis* are well known as invasive marine organisms. These crabs, like many high impact invasive marine organisms, are highly fecund, broadly tolerant to abiotic variation, active food gatherers and can impact core values. For example *C. maenus*, *E. sinensis*, *Hemigrapsus sanguineus* and *Rithropanopeus harrisii* have all been recorded feeding upon fish trapped in nets, bait fish and fish and mussels subject to aquaculture (Cohen et al. 1995, Brousseau et al. 2001, Rudnick et al. 2005, Roche & Torchin 2007). The monetary impact of *E. sinensis* in Germany since 1912 is estimated at approximately 80 million Euro. This estimation included impacts to catchment gear installation and maintenance, bank erosion, feeding, and on commercial fisheries and aquaculture (Gollasch 2006).

Given that crabs:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that crab species represent a potential hazard via vessel biofouling.

4.7.2. Risk Assessment

4.7.2.1. Entry assessment

Crabs have a global distribution with diversity decreasing with increasing latitude (Bruce & Patuawa 2007). The southwest Pacific is specifically noted for high diversity in many groups (Bruce & Patuawa 2007). In addition to shelter, ports and harbours provide a productive habitat for crabs as water temperatures are often higher than the open coast (Cohen et al. 1995). Both larval and adult life-stages are, therefore, likely to encounter vessels within ports and marinas.

Most crab introductions are attributed to ballast water, dry ballast, and aquaculture (Cohen et al. 1995, Carlton & Cohen 2003, Herborget al. 2005, Miller et al 2006, de Lafontaine et al. 2008) however, crabs have also been found in association with hull fouling (Farrapiera et al. 2007) and within seachests (Coutts et al. 2003, Coutts & Dodgshun 2007). For example, Minchin and Gollasch (2003) observed hundreds crabs, *Porcellana longicornis,* within the shells of dead barnacles on a commercial vessel hull. In addition, the Global Invasive Species Database (http://www.issg.org/database/welcome/) fact sheet on *E. sinensis* notes that "some specimens in empty shells of cirripeds have been reported on a ships hull." Pre-moult and moulting crabs also have a tendency to seek shelter which may result in their transport within vessel crevices (Dodgshun & Coutts 2003).

Despite the instances recorded above, crabs are rarely found in hull fouling surveys and are usually low in abundance. This may be due to adult and larger juvenile crabs requiring a high level of fouling to provide adequate food and shelter, however, heavily fouled vessels are generally rare. The available sheltered space, such as dead barnacle shells and gratings, may also limit the number and size of crabs.

Crabs are known to survive translocation with the capacity to tolerate high variation in abiotic conditions being a common attribute among introduced crabs. For example, adult *C. maenus* can withstand over wintering temperatures as low as 0^{o}C and temperatures as high as 31-36^{o}C for brief periods (Hidalgo et al. 2005). *C. maenus* are also capable of resisting salinities as low as 4 ppt (Hidalgo et al. 2005).

Cuculescu et al. (1998) observed that eurythermal species of crab are able to survive larger changes in temperatures than stenothermal species. In addition, crabs sampled in summer survived exposure to higher temperatures than those sampled in winter. This indicates that individuals translocated in summer would be able to survive higher temperature changes in transit. Larvae and juvenile crabs are more sensitive to temperature and salinity differences than adult crabs (Baylon and Suzuki 2007).

There are recent known introductions of non-indigenous crabs to New Zealand. In the MAFBNZ commissioned research on biofouling, Inglis et al. (in prep.) found 7 species of non-indigenous crabs that are not established in New Zealand.

Given that particular crab species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling (although usually restricted to heavily fouled vessels);
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous crab species into New Zealand via vessel biofouling is non-negligible.

4.7.2.2. Establishment assessment

There are known suitable habitats for introduced crabs in New Zealand. The similarity of various receiving environments (i.e. climate and physical habitat) has been suggested as a major factor contributing to the success of invasive crab species (Lohrer et al. 2000). Although many invasive crabs have high tolerance to various abiotic conditions, they are most likely to establish in similar non-indigenous habitats to their native environments (Lohrer et al. 2000, Thresher et al. 2003, Rudnick et al. 2005, Gust & Inglis 2006, Roche & Torchin 2007). Resources play a significant part in the success of non-indigenous species, whether through superior competition or utilisation of a free resource (Lohrer et al. 2000, Kado 2003). Therefore, the depauperate nature and absence of large coastal crabs in New Zealand (Dell 1968, Morton 2004, Bruce & Patuawa 2007) may present open niches that will aid the establishment of non-indigenous crabs. As crab diversity generally decreases with increasing latitude (Bruce and Patuawa 2007), northern New Zealand is more likely to host suitable environments for the establishment of non-indigenous crabs.

Crabs can transfer from a vessel through release of eggs, larvae or individual migration into the environment where the vessel is moored/berthed. However, reproduction is required to establish a self sustaining population, for example, there are many cases of individual crabs being recorded beyond their natural ranges that have not resulted in established populations (Carlton & Cohen 2003, Thresher et al. 2003). Crabs tend to have localised recruitment (Thresher et al. 2003, Roman & Palumbi 2004) and spread (Thresher et al. 2003, Herborg et al. 2005), therefore, reproductive success is likely if propagule numbers are high enough. Depending on larval settlement, a self-sustaining population with low genetic variation may also form.

Temperature and salinity are likely to play a crucial role in whether larvae released into the water column develop, settle, and contribute towards establishment. In general, higher temperatures[11] result in faster larval development (Epifanio et al. 1998, Rudnick et al. 2005, Baylon & Suzuki 2007). However, the megalope stage of development requires a particularly narrow temperature and salinity range (Epifanio et al. 1998, Carlton & Cohen 2003, Bravo et al. 2007). The tolerance of adult crabs to temperature and salinity variation increases dramatically after larval settlement, however, temperature is important for the regulation of adult growth, development and reproduction. For example, in southern China *E. sinensis* becomes sexually mature within 1-2 years (Jin et al. 2001, Zhang et al. 2001), by comparison the same species in the San Francisco estuary and in northern Europe can take up to 3 and 5 years to mature, respectively (Gollasch 1999, Rudnick et al. 2005). Temperature can be protective of non-indigenous establishment, for example, *C. maenas* has been found in numerous tropical locations such as Sri Lanka and Brazil, but has not been able to establish (Carlton & Cohen 2003). Further, if temperatures are particularly low, crab growth and reproduction may be slowed or even halted until the next season (Rudnick et al. 2005). As a result of the influence of temperature on the crab life-cycle, the most conducive location for the establishment of non-indigenous crabs is likely to be northern New Zealand.

There are recent known examples of non-indigenous crabs establishing in New Zealand. Kospartov et al. (in press) list four species that have established since 2000, however, it is uncertain whether these species were translocated via hull fouling.

The likelihood of establishment appears to be dependent on many factors. Though an organism may be introduced to a suitable environment, establishment depends on a raft of conditions being favourable, such as climate and oceanography. This is exemplified by the

[11] As a general rule, the summer temperatures experienced within the crabs native range.

fact that despite being introduced to England on several occasions and being already established on the European continent, *E. sinensis* did not establish populations in England until recently (Herborg et al. 2005). Human mediated transport has increased the distribution of non-indigenous crabs, for example, the wide distribution of *E. sinesis* and the long distance transport and establishment of *C. maenas* and *C. japonica* populations (Anon 2001 *In* Gust and Inglis 2006, Thresher et al. 2003, Carlton & Cohen 2003). While crabs are likely have a low frequency on vessel pathways, as international maritime traffic intensifies (Hewitt et al. 2004), the increased vessel dockings will result in increases to propagule exposure and thus the likelihood of establishment (Roman & Palumbi 2004, Petersen 2006, Roman 2006, Herborg et al. 2007).

Population numbers of invasive crabs can be highly variable. Under normal conditions crab recruitment is localised, however, given the right conditions populations can occasionally explode. Environmental conditions are sometimes conducive to the retention of the large larval output in the immediate and surrounding areas, for example, Gimenez & Dick (2007) found that the numbers of *C. maenas* settled in the North Sea are related to the direction, strength and duration of the wind. In the Thames (UK) and San Francisco (USA) estuaries high recruitment of invasive crabs was observed during dry winters, when freshwater input and flow-through was lower (Carlton & Cohen 1995, Herborg et al. 2005). In addition, *C. maenas* and *R. harrisii* both spread further up the western North American coast when the El Nino current advected onto the coast and swept northwards (Petersen 2006). However, after these climatically driven events, populations can dwindle and be restricted to small pockets. In contrast to population explosions, populations can also experience rapid declines due to unfavourable conditions, for example, the cooler Auckland summer of 2002-03 resulted in a 66% drop in catch rates of *C. japonica* (Gust & Inglis 2006).

Given that particular crab species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous crab species in New Zealand via vessel biofouling is non-negligible.

4.7.2.3. Consequence assessment

Crabs can be active predators impacting upon commercial and recreational fishing. Native New Zealand crabs are not particularly large, aggressive or wide-spread in marine near-shore environments, however, the majority of invasive crab species have the aforementioned characteristics (Gust & Inglis 2006, MacDonald et al. 2007). As many invasive crabs are known to actively and aggressively seek out food sources, commercial and recreational fisheries may be impacted, for example, invasive crabs may monopolise baited traps used to commercially catch New Zealand paddle crabs (*Ovalipes catharus*). They may also become stuck in fishing gear and feed on bait or catch reducing fishing efficiency.

The introduction of the European green crab (*C. maenus*) to North America resulted in severe impacts on shore communities, affecting the survival and recruitment of gastropods and bivalves (Vermeij 1982b; Williams 1984 *In* NIMPIS 2010). The introduction of this crab was one of the factors responsible for the decline of the soft shelled clam (*Mya arenaria*) fishery (Whitlow et al. 2003 *In* NIMPIS 2010), which affected the livelihoods of thousands of workers in eastern North America. At the time the crab extended its range into the fisheries areas, catches fell from 14.5 million pounds in 1938 to 2.3 million pounds in 1959 (IUCN 2009). *C. maenus* has subsequently established in the west coast of North America, South

Africa, south eastern Australia, Brazil, Panama, the Red Sea, Pakistan, Sri Lanka, Myanmar, Japan, Patagonia and Hawai'i. In Australia (Tasmania) *C. maenas* has been reported to be a significant predator of the clam species, *Katelysia scalarina* (Walton et al. 2002 *In* NIMPIS 2010). In addition, since the arrival of *C. maenus* to California in 1993, clam and native shore crab populations have dropped significantly (NIMPIS 2010).

The introduction of non-indigenous species may not create immediate impacts to core values, as founding populations may require a shift in their new range to encounter a more suitable habitat. For example, the giant red king crab (*Paralithodes camtschaticus*), introduced by Russia to the Barents Sea in 1960 to form a fishery did not establish numbers nor have recorded impacts for approximately twenty years. In the 1980's this crab had migrated to Norway and by the 1990's had increased to numbers significant enough to raise concerns amongst local fishermen and environmentalists (IUCN 2009). It has since been shown that *P. camtschaticus* has impacted on local scallop populations (Jorgensen 2005). In Norway, these crabs are capable of foraging on 150 to 335 g of scallop per m^2 over 48 hr compared to the natural biomass of the commercially harvested Iceland scallop (*Chlamys islandica*) of 400 to 1200 g per m^2 (Jorgensen 2005).

Social and cultural values may also be impacted by established non-indigenous crab populations. In addition to affecting employment related to fisheries, the aggressive nature of the crabs may make them a nuisance for recreational users of coastal waters, such as beach users or fishermen trying to clean fishing gear. Invasive crabs may feed on local bivalve species which make up an important commercial and food component of Māori life and for recreational fishers.

Established non-indigenous crabs may modify biotic communities and physical environments. The burrowing behaviour of crabs may impact marine shorelines and riverine banks by altering erosion rates and sediment characteristics of nearby and down-current areas. Possible knock-on effects may include the smothering of benthic organisms and reduced water quality.

4.7.3. Risk Estimation

Given that particular crab species:

- are associated with vessel biofouling (when extensive fouling communities are present);
- are capable of forming establishing populations in New Zealand;
- have recently established populations within New Zealand (although, the vector remains unknown);
- have well known invasive behaviours;
- are known to impact core values;

it is concluded that the import of non-indigenous crab species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.7.4. Risk Management

The cryptic nature of crabs makes their presence on vessel hulls difficult to predict or detect. Although a hull may appear clean, due to habitat preferences, crabs may be located behind gratings that have macro-fouling or reside within the tests of large or small barnacles. Therefore, given their association with biofouling, generic measures are likely to be appropriate (Section 3). However, the mobility of crabs presents further management difficulties, and as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape.

4.7.5. References

Baylon, J; Suzuki, H (2007) Effects of changes in salinity and temperature on the survival and development of larvae and juveniles of the crucifix crab *Charybdis feriatus* (Crustacea: Decapoda: Portunidae). *Aquaculture* 269: 390-401.

Bravo, M A; Cameron, B; Metaxas, A (2007) Salinity tolerance in the early larval stages of *Carcinus maenas* (Decapoda, Brachyura), a recent invader of the Bras d'Or Lakes, Nova Scotia, Canada. Crustaceana 80: 475-490.

Brousseau, D J; Filipowicz, A; Baglivo, J A (2001) Laboratory investigations on the effects of predator sex and size on prey selection by the Asian crab, *Hemigrapsus sanguineus*. *Journal of Experimental Marine Biology and Ecology* 262(2): 199-210.

Bruce, N; Patuawa, P (2007) Crustaceans (Subphylum Crustacea). In MacDiarmid, A (Ed) *The Treasures of the Sea: Nga Taonga a Tangaroa. A Summary of the Biodiversity in the New Zealand Marine Ecoregion*. WWF-New Zealand; Wellington, New Zealand.

Carlton, J T, Cohen, A N (2003) Episodic global dispersal in shallow water marine organisms: the case history of the European shore crabs *Carcinus maenas* and *C. aestuarii*. *Journal of Biogeography* 30: 1809-1820.

Clark, P F; Rainbow, P S; Robbins, R S; Smith, B; Yeomans, W E; Thomas, M; Dobson, G (1998) The alien Chinese mitten crab, *Eriocheir sinensis* (Crustacea: Decapoda: Brachyura), in the Thames catchment. *Journal of the Marine Biological Association of the United Kingdom* 78: 1215-1221.

Cohen, A N; Carlton, J T; Fountain, M C (1995) Introduction, dispersal and potential impacts of the green crab *Carcinus maenas* in San Francisco Bay, California. *Marine Biology* 122: 225-237.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in seachests: a protected mechanism for marine bioinvasion. *Marine Pollution Buletin* 54: 875-886.

Coutts, A D M; Moore, K M; Hewitt, C L (2003) Ships' seachests: an overlooked transfer mechanism for non-indigenous species? *Marine Pollution Bulletin* 46: 1504-1515.

Cuculescu, M; Hyde, D; Bowler, K (1998) Thermal tolerance of two species of marine crab, *Cancer pagurus* and *Carcinus maenas*. *Journal of Thermal Biology* 23: 107-110.

Dell, R K (1968) Composition and distribution of the New Zealand Brachyuran fauna. *Transactions of the Royal Society of New Zealand* 10: 225-240.

DePatra, K D; Levin, L A (1989) Evidence of the passive deposition of meiofauna into fiddler crab burrows. *Journal of Experimental Marine Biology and Ecology* 125(3): 173-192.

Epifianio, C E; Dittel, A I; Park, S; Schwalm, S; Fouts, A (1998) Early life history of *Hemigrapsus sanguineus*, a non-indigenous crab in the Middle Atlantic Blight (USA). *Marine Ecology Progress Series* 170: 231-238.

Foreman, K (1985) Do predators, resources or physical factors control the seasonal cycle of meiofauna of marshes? *Estuaries* 8: 48.

Gimenez, L; Dick, S (2007) Settlement of shore crab *Carcinus maenas* on a mesotidal open habitat as a function of transport mechanisms. *Marine Ecology Progress Series* 338: 159-168.

Gollasch, S (2006) NOBANIS – Invasive Alien Species Fact Sheet – *Eriochier sinensis*. www.nobanis.org. Accessed 9 Jan 2009.

Gust, N; Inglis, G J (2006) Adaptive multi-scale sampling to determine an invasive crab's habitat usage and range in New Zealand. *Biological Invasions* 8: 339-353.

Hanfling, B; Carvalho, G R; Brandl R (2002) mt-DNA sequences and possible invasion pathways of the Chinese mitten crab. *Marine Ecology Progress Series* 238: 307-310.

Herborg, L-F; Weetman, D; Oosterhout, O V; Hanfling, B (2007) Genetic population structure and contemporary dispersal patterns of a recent European invader, the Chinese mitten crab, *Eriocheir sinensis*. *Molecular Ecology* 16: 231-242.

Herborg, L M; Rushton, S P; Clare, A S; Bentley, M G (2005) The invasion of the Chinese mitten crab (*Eriocheir sinensis*) in the United Kingdom and its comparison to continental Europe. *Biological Invasions* 7: 959-968.

Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B; Boyd, S; Cohen, B F; Currie, D R; Gomon, M F; Keough, M J; Lewis, J A; Lockett, M M; Mays, N; MacArthur, M A; O'Hara, T D; Poore, G C B; Ross, D J; Storey, M J; Watson, J E; Wilson, R S (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology* 144: 183-202.

Hidalgo, F J; Baron, P J; Orensanz, J M (2005) A prediction comes true: the green crab invades the Patagonian coast. *Biological Invasions* 7: 547-552.

Hoffman, J A; Katz, J; Bertness, M D (1984) Fiddler crab deposit-feeding and meifaunal abundance in salt marsh habitats. *Journal of Experimental Marine Biology and Ecology* 82: 161-174.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

IUCN (2009) Marine menace: alien invasive species in the marine environment. http://www.iucn.org/about/work/programmes/marine/marine_resources/marine_publications/?3255/marinemenace. Accessed 17 November 2009.

Jin, G; Li, Z; Xie, P (2001) The growth patterns of juvenile and precocious Chinese mitten crabs, *Eriocheir sinensis* (Decapoda, Grapsida), stocked in freshwater lakes of China. *Crustaceana* 74: 261-273.

Jorgensen, L L (2005) Impact scenario for an introduced decapod on Arctic epibenthic communities. *Biological Invasions* 7: 949-957.

Kado, R (2003) Invasion of Japanese shores by the NE Pacific barnacle *Balanus glandula* and its ecological and biogeographical impact. *Marine Ecology Progress Series*. 249: 199-206.

Ko, H S (2005) First zoea of *Eriphea sebana* (Crustacea: Decapoda: Xanthoidea: Menippidae) hatched in the laboratory. *Integrative Biosciences* 9: 135-138.

de Lafontaine, Y; Sevigny, J-M; Calve, R; Verreault, G; Despatie, S-P; Veilleux, E (2008) Chinese mitten crabs (*Eriocheir sinensis*) in the St. Lawrence River and Estuary, Canada: new records and risk of invasion. *Aquatic Invasions* 3(2): 153-163.

Lohrer, A M; Whitlatch, R B; Wada, K; Fukui, Y (2000) Home and Away: comparisons of resource utilization by a marine species in native and invaded habitats. *Biological Invasions* 2: 41-57.

MacDonald, J A; Roudez, R; Glover, T; Weis, J S (2007) The invasive green crab and Japanese shore crab: behavioural interactions with a native crab species, the blue crab. *Biological Invasions* 9: 837-848.

Maeda, M; Itami, T; Furumoto, A; Hennig, O; Imamura, T; Kondo, M; Hirono, I; Aoki, T; Takahashi, Y (1998) Detection of penaeid red-shaped DNA virus (PRDV) in wild-caught shrimp and other crustaceans. *Fish Pathology* 33: 381-387.

McDermott, J J (1998) The western Pacific brachyuran (*Hemigrapsus sanguineus*: Grapsidae), in its new habitat along the Atlantic coast of the United States: geographic distribution and ecology. *ICES Journal of Marine Science* 55: 289-298.

Minchin, D; Gollasch, S (2003) Fouling and ships' hulls: how changing circumstances and spawning events may result in the spread of exotic species. *Biofouling* 19 (supplement): 111-122.

Montague, C L (1982) The influence of fiddler crab burrows and burrowing on metabolic processes in saltmarsh sediment. In Kennedy, V S (Ed) *Estuarine Comparisons*. Academic Press; New York, USA; pp 238-301.

Morton, J (2004) Seashore Ecology of New Zealand and the Pacific. David Bateman Ltd; Auckland, New Zealand; 504 pp.

NIMPIS (2010). *Carcinus maenas* impact details. *National Introduced Marine Pest Information System* (Eds: Hewitt C.L., Martin R.B., Sliwa C., McEnnulty, F.R., Murphy, N.E., Jones T. & Cooper, S.). Web publication <http://crimp.marine.csiro.au/nimpis>, Accessed: 2nd February 2010.

Pardo, M L; Palma, A T; Prieto, C; Sepulveda, P; Valdivia, I; Ojeda, P (2007) Processes regulating early post-settlement habitat use in a subtidal assemblage of brachyuran decapods. *Journal of Experimental Marine Biology and Ecology* 344: 10-22.

Payen, G G; Bonami, J R (1979) Mise en évidence de particules d'allure virale associées aux noyaux des cellules mésodermiques de la zone germinative testiculaire du crabe *Rhithropanopeus harrisii* (Gould) (Brachyura, Xanthidae). *Revue des Travaux de l'Institut des Pêches Maritimes* 43: 361-365.

Petersen, C (2006) Range expansion in the northeast Pacific by an estuary mud crab – a molecular study. *Biological Invasions* 8: 565-576.

Quammen, M L (1984) Predation by shorebirds, fish, and crabs on invertebrates in intertidal mudflats: an experimental test. *Ecology* 65(2): 529-537.

Reyns, N B; Eggleston, D B (2004) Environmentally-controlled, density-dependant secondary dispersal in a local estuarine crab population. *Oecologia* 140: 280-288.

Roche, D G; Torchin, M E (2007) Established population of the North American Harris mud crab, *Rhithropanopeus harrisii* (Gould 1841) (Crustacea: Brachyura: Xanthidae) inn the Panama Canal. *Aquatic Invasions* 3: 155-161.

Rodriguez, G; Suarez, H (2001) Anthropogenic Dispersal of Decapod Crustaceans in Aquatic Environments. *Interciencia* 26: 282-288.

Roman, J (2006) Diluting the founder effect: cryptic invasions expand a marine invader's range. *Proceedings of the Royal Society Series B* 273: 2453-2459.

Roman, J; Palumbi, S R (2004) A global invader at home: population structure of the green crab, *Carcinus maenas*, in Europe. *Molecular Ecology* 13: 2891-2898.

Rudnick, D; Veldhuizen, T; Tullis, R; Culver, C; Heib, K; Tsukimura, B (2005) A life history model for the San Francisco Estuary population of the Chinese mitten crab, *Eriocheir sinensis* (Decapoda: Grapsoidea). *Biological Invasions* 7: 333-350.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Thresher, R; Proctor, C; Ruiz, G; Gurney, R; MacKinnon, C; Walton, W; Rodriguez, L; Bax, N (2003) Invasion dynamics of the European shore crab, *Carcinus maenas*, in Australia. *Marine Biology* 142: 867-876.

Zhang, T; Li, Z; Cui, Y (2001) Survival, growth, sex ratio, and maturity of the Chinese mitten crab (*Eriocheir sinensis*) reared in a Chinese pond. *Journal of Freshwater Ecology* 16: 633-640.

4.8. ECHINODERMS (PHYLUM: ECHINODERMATA)

4.8.1. Hazard Identification

Approximately 6,000 species of echinoderms cover organisms, such as, sea stars (Asteroids), brittle stars (Ophiuroids), sea urchins, sand dollars (Echinoids), sea cucumbers (Holothurians), and sea lilies and feather stars (Crinoids) (Ruppert et al. 2004). New Zealand has 618 described echinoderm species covering all five classes, of which 37% are endemic to the EEZ (Gordon et al. 2007).

Echinoderms are found in environments ranging from intertidal to abyssal depths and perform ecological functions ranging from predator to detritivore. Echinoderms can be conspicuous components of the benthic fauna such as the sea urchin (*Evechinus chloroticus*) grazing fronts of New Zealand's rocky reefs (Andrew 1988) or the feeding swarms of the sea star, *Asterias rubens,* in the Irish Sea (Dare 1982). The habitats of sea urchins alternate between species rich kelp forests and sea urchin-dominated "barrens", these states can alternate depending on factors such as storms, disease and recruitment (Pearse 2006). Some sea stars are keystone predators important for the structure and function of marine benthic communities. Their capacity for population outbreaks and significant impacts to areas of fishery and aquaculture importance is well documented (Sloan 1980, Menge 1982).

The life-cycle of echinoderms is comparatively similar throughout all the classes. The majority of species have separate sexes and are broadcast spawners (Ruppert et al. 2004). With regard to asexual reproduction, echinoderms have varying abilities to regenerate lost body parts and in some cases two individuals can be produced. Certain sea stars can regenerate from as little as one arm and a fifth of the central disc (Ruppert et al. 2004). Asexual reproduction in the sea star *Coscinasterias acutispina* appears to be governed by a variety of factors including nutrition, temperature and parasitism (Haramoto et al. 2007).

There are few reports on the occurrence of non-indigenous echinoderms and even fewer reports of impacts from translocated echinoderms. However, this may be due to a deficiency in the sampling of seachests and impact studies. The northern Pacific sea star (*Asterias amurensis*), the most high profile invasive echinoderm, has been linked to the decline of large bivalves within its introduced range of Tasmania (Ross et al. 2002, 2003). Other non-indigenous echinoderms have been recorded within Port Phillip Bay, Australia (Hewitt et al. 2004), and along the coast of Israel (Galil 2007).

Given that echinoderms:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that echinoderm species represent a potential hazard via vessel biofouling.

4.8.2. Risk Assessment

4.8.2.1. Entry Assessment

Echinoderms are generally thought to be translocated beyond their natural range via ballast water (Rainer 1995), however, there is increasing evidence that vessel internal seawater systems (including seachests) can also be a significant translocation mechanism (Thresher 2000 *In* Ross et al. 2003). For example, Coutts & Dodgshun (2007) found a sea cucumber (unidentified), sea urchin (*E. chloroticus*), and the eleven armed sea star (*Coscinasterias muricata*), all indigenous, within the seachests of commercial vessels sampled in New Zealand. In addition, a sea star (*Pateriella regularis*) was found amongst substantial algal fouling on a barge situated in the Derwent estuary, Australia (Lewis et al. 2006). Despite the presence of *A. amurensis* being detected in samples of biofouling from vessel hulls using molecular probes (Hayes et al. 2004), there is no evidence that echinoderms would be able to remain on a clean vessel hull when underway, therefore, their translocation is probably restricted to sheltered niche areas (Thresher 2000 *In* Ross et al. 2003, Inglis et al. in prep.).

Minimal information exists regarding the environmental tolerances of echinoderms, as such there is considerable uncertainty regarding their ability to survive translocation across oceanic boundaries. However, it is known that *A. amurensis* survived translocation from the temperate regions of the north Pacific to Tasmania and that the non-indigenous echinoderms found in Israel were translocated from the Red Sea and surrounding areas (Galil 2007). In addition, the eight species (five brittle stars, one sea cucumber, two sea urchins) of non-indigenous echinoderms most recently introduced to New Zealand were all native to the Indo-Pacific region (Kospartov et al. 2008). As a wide variety of echinoderms inhabit high stress environments such as intertidal rockpools, it can be assumed that some species will have a range of thermal and physical tolerances.

In addition to the uncertainty regarding environmental tolerances, there is no knowledge on the effects of anti-fouling paints on the various groups of echinoderms. However, sea stars (*A. rubens*) have previously been observed to inhabit environments of high heavy metal contamination (Temara et al. 1998).

Given that particular echinoderm species:
- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous echinoderm species into New Zealand via vessel biofouling is non-negligible.

4.8.2.2. Establishment Assessment

It is assumed that New Zealand would provide a suitable environment/habitat for a number of non-indigenous echinoderm species. Difficulty in establishment within a new environment may depend on the organisms' feeding strategy, for example, detritivores and omnivores may experience less difficulty meeting dietary requirements than predators and herbivores. This may be due to uncertainty introduced by the differences between native and local fauna, for example, herbivorous echinoderms may experience difficulty recognising unfamiliar algae (Prince & LeBlanc 1992) or receive less nutritional value (Lyons & Schiebling 2007).

Alternatively, introduced echinoderms may find readily available and nutritious food sources, experience predator/competition release, and go on to become a major structuring force in the ecosystem, as has been the case with *A. amurensis* in Tasmania (Ross et al. 2002, 2003). Translocated from Tasmania to Port Phillip Bay, Victoria (Australia) in 1995, the *A. amurensis* population had reached an estimated 75 million individuals by 2000 (Parry & Cohen 2001).

For establishment in New Zealand waters, echinoderms of both sexes must be present within spawning distance. It is assumed that it is possible for two individuals to be present within the one niche area (i.e. seachest) and for spawning to occur within this area. Alternatively, individuals may leave the vessel and encounter individuals of the opposite sex in the introduced environment.

At present, there are no known establishments of non-indigenous echinoderms in New Zealand (Kospartov et al. in press).

Given that particular echinoderm species:
- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous echinoderm species in New Zealand via vessel biofouling is non-negligible.

4.8.2.3. Consequence Assessment

The potential consequences associated with large predatory sea stars and sea urchins are profound as many are noted keystone species. Sea urchins have the ability to clear macroalgal stands to produce ‘urchin barrens’ (Pearse 2006), therefore, the establishment of non-indigenous sea urchins may exacerbate this phenomenon in the New Zealand environment. In addition, non-indigenous sea urchins would compete with indigenous grazing species, some of which have important commercial, recreational and cultural fisheries, such as abalone and kina (Pearse 2006).

As predatory sea stars are important to the structure of their native ecosystems, their introduction and establishment may have significant effects on bivalve populations. For example, the introduction of *A. amurensis* to the Derwent Estuary in 1986 is likely responsible for the decline and subsequent rarity of, surface or shallow sediment dwelling, bivalve species (Ross et al. 2003). If other areas of the Tasmanian coast were to attain the sea star densities within the Derwent Estuary, the authors predict large direct effects on native assemblages, particularly on populations of large surface dwelling bivalves, including several commercial species. In addition, *A. amurensis* is potentially preventing the re-establishment and recovery of adult populations in the Derwent Estuary, as the diet of the sea star is found to have a high prevalence of juvenile molluscs (Ross et al. 2003). For example, predation losses of commercial scallop spat over a settlement season are reported to be up to 50% (Crawford pers. comm. *In* Hutson et al. 2005). It has since been shown that the Australian scallops (*Pecten fumatus* and *Chlamys asperrima*) have a low escape response frequency to *A. amurensis* compared to the indigenous *C. muricata* suggesting that the risk of predation on scallops will be greater for the introduced sea star. Similar effects can be expected for New Zealand bivalve species not previously exposed to predatory sea stars.

In Port Phillip Bay, Victoria (Australia) *A. amurensis* been observed predating on spat and on grow-out ropes (J. Mercer, G. Parry, pers. comm. *In* Dommisse & Hough 2004). However, despite these occurrences, sea star predation is not considered a big threat to stock losses of commercially grown mussels (Dommisse & Hough 2004). This is perhaps due to the fact that in Port Phillip Bay, spat are "resocked" and immersed in freshwater before on-growing for the removal of predators (Gunthorpe et al. 2001 *In* Dommisse & Hough 2004). Although such practice is not followed in Tasmania, those farmers interviewed did not consider stock losses to the sea star as significant (Dommisse & Hough 2004). The introduction of a non-indigenous predatory sea star to New Zealand may therefore have indirect effects on New Zealand aquaculture due to the alteration of standard practices, rather than direct effects on aquaculture species. There is little doubt that benthic assemblages have been altered where populations of *A. amurensis* have established. It is likely that the introduction of a predatory sea star to New Zealand would impact on benthic assemblages and specific commercial bivalve species.

Apart from those consequences associated with predatory sea stars, another consequence of non-indigenous echinoderm introductions is likely to be competition for resources. For instance, the sea star *Pateriella regularis* shows intraspecific and interspecific competition with the chiton *Chiton glaucus* on the intertidal cobble fields of Wellington, New Zealand (Palmer & Phillips 2009). These effects on marine benthic assemblages may have negative impacts on commercial and foods source for Māori culture and recreational gatherers.

4.8.3. Risk Estimation

Given that particular echinoderm species:

- are associated with vessel biofouling, particularly in niches areas;
- are capable of forming establishing populations in New Zealand;
- have well known invasive behaviours;
- can be keystone species that structure marine communities through predation or herbivory;
- associated with high impacts, such as *Asterias amurensis*, are present in southeastern Australia;
- have had high impacts on core values following establishment in other countries;

it is concluded that the import of non-indigenous echinoderm species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.8.4. Risk Management

To manage the risks associated with echinoderms a generic approach to removal of fouling is appropriate (Section 3). As seachests present a likely habitat of translocated echinoderms, this area should be given due attention by inspectors on the dock and during haul out and cleaning activities. The ability of some echinoderm species to regenerate must also be considered during haul out and cleaning procedures whereby, the entire organism, including jettisoned body parts, should be contained. The mobility of some echinoderm species presents further management difficulties, and as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape.

4.8.5. References

Andrew, N L (1988) Ecological aspects of the common sea urchin, *Evechinus chloroticus*, in northern New Zealand: a review. *New Zealand Journal of Marine and Freshwater Research* 22**:** 415-426.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in vessel seachests: A protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875–886.

Dare, P J (1982) Notes on the swarming behaviour and population density of *Asterias rubens* L. (Echinodermata: Asteroidea) feeding on the mussel, *Mytilus edulis* L *ICES Journal of Marine Science* 40: 112-118.

Dommisse, M; Hough, D (2004) Controlling the northern Pacific seastar *Asterias amurensis* in Australia. Final report for the Australian Government Department of the Environment and Heritage.

Gordon, D; Smith, F; Patuawa, T (2007) Sea stars, sea urchins and sea cucumbers (Phylum Echinodermata). In MacDiarmid, A (Ed) *The Treasures of the sea, ngā taonga a Tangaroa. A summary of biodiversity in the New Zealand marine ecoregion.* WWF – New Zealand; Wellington, New Zealand.

Haramoto, S; Komatsu, M; Yamazaki, Y (2007) Patterns of asexual reproduction in the fissiparous seastar *Coscinasterias acutispina* (Asteroidea: Echinodermata) in Japan. *Zoological Science* 24(11): 1075-1081.

Hayes K; Sutton, C; Gunasekera, R; Sliwa, C; Migus, S; McEnnulty, F; Dunstan, P; Green, M; Patil, J (2004) Empirical validation: Small vessel translocation of key threatening species. Stage I – *Asterias amurensis*. Department of the Environment and Heritage; Canberra, Australia.

Hutson, K S; Ross, D J; Day, R W; Ahern, J J (2005) Australian scallops do not recognise the introduced predatory seastar *Asterias amurensis. Marine Ecology Progress Series* 298: 305–309

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Lewis, P N; Bergstrom, D M; Whinam, J (2006) Barging in: a temperate marine community travels to the subantarctic. *Biological Invasions* 8: 787-795.

Menge, B A (1982) Effects of feeding on the environment: Asteroidea. *In*: Janguoux M and Lawrence J M (eds) *Echinoderm Nutrition*, pp 521–554. A.A. Balkema, Rotterdam, The Netherlands

Palmer, S; Phillips, N (2009) The ecological role of an omnivorous seastar (*Patiriella regularis*) in intertidal cobble fields. In O'Shea, S; Costello, M J; O'Connell, K (eds) *Programme and Abstracts for Annual Conference in Auckland 2-4 September 2009*. New Zealand Marine Sciences Society; Accessed at http://nzmss.rsnz.org; 1-126 pp.

Parry, G D; Cohen, B F (2001) The distribution, abundance andpopulation dynamics of the exotic seastar *Asterias amurensis* during the first three years of its invasion of Port Phillip Bay. Report No. 33, Marine and Freshwater Research Institute, Queenscliff.

Pearse, J S (2006) Ecological role of purple sea urchins. *Science*, 314: 940-941.

Rainer, S F (1995) Potential for the introduction and translocation of exotic species by hull fouling: A preliminary assessment. Centre for Research on Introduced Marine Pests, Technical Report 1. CSIRO; Australia.

Ross, D J; Johnson, C R; Hewitt, C L (2003) Variability in the impact of an introduced predator (*Asterias amurensis*: Asteroidea) on soft-sediment assemblages. *Journal of Experimental Marine Biology and Ecology* 288: 257–278.

Ross, D J; Johnson, C R; Hewitt, C L (2002) Impact of introduced seastars *Asterias amurensis* on survivorship of juvenile commercial bivalves *Fulvia tenuicostata. Marine Ecology Progress Series* 241: 99-112.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Sloan NA (1980) Aspects of the feeding biology of asteroids. *Oceanography and Marine Biology: an Annual Review* 18: 57–124

Talman, S G; Keough, M J (2001) Impact of an exotic clam, *Corbula gibba*, on the commercial scallop *Pecten fumatus* in Port Phillip Bay, south-east Australia: evidence of resource restricted growth in a subtidal environment. *Marine Ecology Progress Series* 221: 135-143.

Temara, A; Skei, J M; Gillan, D; Warnau, M; Jangoux, M; Dubois, P (1998) Validation of the asteroid *Asterias rubens* (Echinodermata) as a bioindicator of spatial and temporal trends of Pb, Cd, and Zn contamination in the field. *Marine Environmental Research* 45(4-5): 341-356.

4.9. FLATWORMS (PHYLUM: PLATYHELMINTHES)

4.9.1. Hazard Identification

The phylum Platyhelminthes includes free-living flatworms (Turbellaria), parasitic flukes (Trematoda) and tapeworms (Cestoda) (Species 2000). The classes Trematoda and Cestoda are entirely parasitic and spend the vast proportion of their life-histories endosymbiotically, and as such, are exempt from this risk assessment. Within Turbellaria, the order Polycladida (polyclads) contains greater than 1,000 described marine flatworm species. More than half of these species occur between latitudes 30^{o} N and 30^{o} S, with approximately two thirds of these species inhabiting the Indo-Pacific Region (Rawlinson 2008). Marine flatworms are generally sensitive to temperature, thus many species are restricted within their latitudinal range (Rawlinson 2008). The order Polycladida is divided into two suborders, Cotylea and Acotylea, based on the presence or absence of a sucker. The Cotylea (sucker present) are more species rich in lower latitudes than the Acotylea (Rawlinson 2008).

All flatworms are hermaphrodites, reproducing either through copulation or internal fertilisation (Ruppert et al. 2004, Lee 2006). In addition, flatworms have a well developed ability to regenerate. Flatworm numbers appear to peak in the summer and early autumn (Skerman 1960, Rivest et al. 1999) which coincides with their breeding season in which more than one batch of eggs can be produced (Lee 2006). Predatory flatworms often deposit eggs inside the empty shells of their prey, whereas other species lay gelatinous masses, chains, or plates of eggs on hard substrata (Skerman 1960, Lee 2006). Flatworms exhibit brooding behaviour by lying on their eggs which develop directly and hatch live young (Ruppert et al. 2004). As adult flatworms carry toxins, they are rarely preyed upon (Lee 2006).

Flatworms are considered to be important mobile predators of sessile hard-substrate marine communities (Newman et al. 2000 *In* Rawlinson 2008). However, some species and juveniles of predaceous species are herbivorous (Ruppert et al. 2004). As an order, flatworms feed on a variety of organisms from protozoa, rotifers, small crustaceans and annelids to ascidians, bivalves, crustaceans and gastropods (Newman et al. 1993, Rivest et al. 1999, Ruppert et al. 2004, Lee 2006). However, individual species of flatworms are usually specialist feeders that show strong feeding preferences in both prey species and size (Skerman 1960, Galleni et al. 1980, O'Connor & Newman 2003). Flatworms detect their prey through chemoreception or by water movement (Skerman 1960, O'Connor & Newman 2003, Lee et al. 2006). Feeding methods vary from simply ingesting or extruding the stomach directly onto the prey to the use of mucous and toxins for prey subdual (Skerman 1960, Galleni et al. 1980, Ruppert et al. 2004, Lee 2006).

Introductions and impacts of non-indigenous flatworms have been recorded globally. Flatworms are known to impact sessile marine populations, for example, up to 90% of the youngest oysters were lost in some areas of the Mediterranean due to flatworm predation (Galleni et al. 1980). Flatworm predation of cultivated shellfish has also caused major impacts in North America and Japan, whilst France has significant control measures in place to stop the introduction of flatworms associated with the importation of Japanese oysters (Galleni et al. 1980). However, the impacts associated with the introduction and establishment of non-indigenous flatworm species have not always been recorded. For example, after initial high densities (19 individuals per cm^2), numbers of *Convoluta convoluta* in North America have fallen (to 3 individuals per cm^2) with no impacts recorded (Rivest et al. 1999, Byrnes & Whitman 2003). Likewise, concerns were raised over impacts to aquaculture and wild oyster stocks when the Asian flatworm was found in the Netherlands (Sluys et al. 2005), however, no impacts have since been recorded. It should be noted, that flatworms are not well studied and, as a result, there is uncertainty with regard to the risks associated with their introduction.

Given that flatworms:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that flatworm species represent a potential hazard via vessel biofouling.

4.9.2. Risk Assessment

4.9.2.1. Entry assessment

Flatworms are known to inhabit the vessel hulls and they have previously been recorded on these structures in New Zealand (Coutts & Dodgshun 2007, Inglis et al. in prep.), Germany (Gollash 2002) and Brazil (Farrapeira et al. 2007). Some flatworms may be able to stay attached to the surface of a relatively clean hull as they possess suckers (Minchin & Gollasch 2003). However, many flatworms are known to associate with barnacles, oysters and ascidians which they use for shelter, food and sites for their egg masses (Skerman 1960, Johns et al. 2009). As a result, flatworms are more likely to be found amongst a biofouling assemblage rather than on a clean open hull.

Flatworms are known to remain viable following translocation. Survival following a voyage to New Zealand from Australia or the Pacific Islands is thought to be feasible as flatworms are able to undergo extended periods without feeding (Galleni et al. 1980, O'Connor & Newman 2003). However, given that flatworms are sensitive to differences in water temperature (O'Connor & Newman 2003), the survival of adults from or through the Pacific Islands is unlikely, therefore, temperate Australia remains the likely source of viable non-indigenous flatworms.

Uncertainty exists with regard the translocation of non-indigenous species due to taxonomic resolution, and as a result, the majority flatworms present in hull fouling are only identified at the phylum or order level. Therefore, while it is almost certain that flatworms have the ability to be transported via hull fouling, the number of positive identifications of non-indigenous species is low and as yet none have been recorded in New Zealand.

The translocation of the eggs of non-indigenous species should also be considered as flatworms use the empty shells of their prey (e.g. barnacles) as sites to lay their eggs. At present, there are no records of egg masses found on vessels, though this is likely to be due to a deficiency in targeted investigations. Assuming that it is possible for egg masses to be transported via hull biofouling, there are questions regarding their arrival in a viable state. Although brooding is common among flatworms, experiments conducted on two flatworm species showed that brooding had little effect on hatching success (Johnstone & Lee 2008). As the flatworms studied are relatively common Australian species, the possibility of unattended eggs being transported to New Zealand as viable propagules is highlighted. However, Johnstone & Lee (2008) note that brooding behaviour may have advantages under changing salinities and temperatures, reinforcing the uncertainty associated with the transit of unattended eggs.

There are few recorded occurrences of flatworms on vessel hulls. This may be due to their naturally low numbers, although this may also be exacerbated by sampling bias. As flatworms are small, motile and cryptic, often found within barnacles and bivalves, there are a variety of reasons that may contribute to an underestimation of their numbers. In addition, flatworms are likely to be associated with high fouling in low flow areas, such as the environment provided by seachests, which are perhaps less well studied than the outer hull surface. Seven international vessels were found carrying (unidentified) flatworms during the MAFBNZ commissioned research on biofouling (Inglis et al. in prep.). Prior to this research, no introductions of marine flatworms to New Zealand via hull fouling had been recorded.

The effect of anti-fouling paint on flatworms is unknown as it is uncertain whether their slime would form a protective layer from the biocides. However, their known habitats, such as barnacle shells, are likely to provide such protection.

Given that particular flatworm species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous flatworm species into New Zealand via vessel biofouling is non-negligible.

4.9.2.2. Establishment assessment

There are likely to be suitable habitats for the establishment of non-indigenous flatworms in New Zealand as they are commonly found under boulders, in crevices of oyster clumps and other cryptic locations. Despite prey preference being restricted to specific organism groups, flatworms may feed on a variety of species within the target group (Skerman 1960, Galleni et al. 1980, O'Connor & Newman 2003). As such, flatworms are likely to find suitable prey in a new location. Furthermore, many common biofouling species are present in New Zealand ports.

As flatworms are capable of swimming short distances, they would be able to transfer directly from vessel hulls to local substrata. Non-indigenous flatworms can also establish a population in New Zealand through the entry of egg masses, sexual reproduction and asexual reproduction (fragmentation and regeneration of an individual or multiple individuals).

There are no records of non-indigenous marine flatworms establishing in New Zealand. However, it has been noted that "at the moment we are hard-pressed to give definitive names to any of the several species of ‘oyster leeches’ that are being encountered by aquaculturists” (Diggles et al. 2002 *In* Johns et al. 2009). Therefore, in New Zealand, further taxonomic research is required on flatworms not only for the protection of important economic values, such as bivalve aquaculture, but also environmental, social and cultural values. At present, it is unlikely that a non-indigenous flatworm population established in New Zealand would be able to be correctly identified.

Given that particular flatworm species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- are unlikely to be correctly identified following establishment in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous flatworm species in New Zealand via vessel biofouling is non-negligible.

4.9.3. Consequence assessment

Flatworms are well known pest organisms impacting the global aquaculture of oysters and mussels (Galleni et al. 1980). If flatworms that prey on commercial species of bivalve were to establish in New Zealand, the likely impacts would include reductions in spat and adult growth. Given that the prey of flatworms can include both oysters and mussels, and the known impacts to commercial stocks in other jurisdictions, the potential consequences to New Zealand’s aquaculture industry could be wide-ranging. In 2008, mussels (*Perna canaliculus*) and oysters (*C. gigas*) comprised of 77% (NZ$204.3 million) and 6% of the value (NZ$16.9 million) of New Zealand’s aquaculture exports, respectively (Aquaculture New Zealand 2009). Given the value of bivalve aquaculture to New Zealand, the possible establishment of novel flatworm species via hull biofouling is a non-negligible concern. Additional costs would be involved if protective measures were introduced to reduce the number of flatworms on farms, the transfer of flatworm between farms, or their occurrence in stock for sale. Further, the presence of predatory flatworms may lead to the formation of market barriers from countries where populations are not established.

The impact of non-indigenous flatworms on wild bivalve populations in New Zealand is unknown. It is likely that environmental variation would restrict their numbers in all but the most sheltered environments, as has been demonstrated for the invasive *C. convoluta* in North America (O’Conner & Newman 2003). While flatworms are selective consumers of juvenile sessile organisms, the large numbers of settling larvae and juveniles tend to swamp these predatory effects (Skerman 1960, Byrnes & Witman 2003). Therefore, it is likely that non-indigenous flatworms would have a localised chronic (low intensity but sustained) effect on wild bivalve species, the consequences of which are unknown.

4.9.4. Risk Estimation

Given that particular flatworm species:

- are known components of hull fouling assemblages;

- are known to have been introduced to new locations;
- are unlikely to be correctly identified upon entry into New Zealand;
- are capable of forming establishing populations in New Zealand;
- are likely to have adverse effects on core values, such as, bivalve aquaculture and coastal benthic communities;

it is concluded that the import of non-indigenous flatworm species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.9.5. Risk Management

To manage the risks presented by flatworms via hull fouling, a generic approach to removal of biofouling is appropriate (Section 3). In addition, the risk management strategy should include the removal of barnacle shells, and other macro-fouling, due to their role as shelter for flatworm adults and eggs. In addition, the mobility of flatworms presents further management difficulties, as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape. As flatworms are sensitive to temperature and salinity changes, these factors may be used in the mitigation of viable transfers (O' Connor & Newman 2001).

4.9.6. References

Aquaculture New Zealand (2009) New Zealand Aquaculture farm facts (2nd edition). http://www.aquaculture.org.nz/aquaculture-in-nz/industry-overview/overview/. Accessed 20 January 2010.

Byrnes, J; Witman, J D (2003) Impact assessment of an invasive flatworm, *Convoluta convolute*, in the Southern Gulf of Maine. *Journal of Experimental Marine Biology and Ecology* 293: 173-191.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.

Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship Hull Fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55(3): 207- 221.

Galleni, L; Tongiorgi, P; Ferrero, E; Salghetti, U (1980) *Stylochus mediterraneus* (Turbellaria: Polycladida), predator on the mussel *Mytilus galloprovincialis*. *Marine Biology* 55: 317-326.

Gollasch, S (2002) The importance of ship hull fouling as a vector of species introductions into the North Sea. *Biofouling* 18(2): 105-121.

Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B (1999) Marine Biological Invasions of Port Phillip Bay, Victoria. Centre for Research on Introduced Pests. Technical Report No. 20. CSIRO Marine Research; Hobart, Australia; 344 pp.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Johns, P; Newman, L J; Holleman, J J; Dawson, E W; Sterrer, W; Allison, F R; Diggles, B K; Andrews, J R H; Hine, P M; McKenna, P B; Poulin, R (2009)

Phylum Platyhelminthes. In Gordon, D P (ed) *New Zealand Inventory of Biodiversity. Volume 1*: Kingdom Animalia; Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press; Christchurch, New Zealand; pp 102-128.

Lee, K-M (2006) *Taxonomy and ecology of predatory marine flatworms (Platyhelminthes: Polycladida) in Botany Bay, New South Wales, Australia*. MSc thesis; University of New South Wales, Australia.

Lee, K-M; Beal, M A; Johnston, E L (2006) a new predatory flatworm (Platyhelminthes, Polycladida) from Botany Bay, New South Wales, Australia. *Journal of Natural History* 39(47): 3987-3995.

Minchin, D; Gollasch, S (2003) Fouling and Ships' Hulls: how Changing Circumstances and Spawning Events may Result in the Spread of Exotic Species. *Biofouling* 19: 111-122.

Newman, L J; Cannon, L R G; Govan, H (1993) *Stylochus* (*Imogene*) *matatasi* n. sp. (Platyhelminthes, Polycladida): pest of cultured giant clams and pearl oysters from the Solomon Islands. *Hydrobiologia* 257: 185-189.

O'Connor, W A; Newman, L J (2001) Halotolerance of the oyster pradator, *Imogine mcgrathi*, a stylochid flatworm from Port Stephens, New South Wales, Australia. *Hydrobiologia* 459: 157-163.

O'Connor, W A; Newman, L J (2003) Predation of cultured mussels, *Mytilus galloprovincialis*, by stylochid flatworms, *Imogine mcgrathi*, from Twofold Bay, New South Wales, Australia. *Asian Fisheries Science* 16: 133-140.

Rawlinson, K A (2008) Biodiversity of coastal polyclad flatworm assemblages in the Wider Caribbean. *Marine Biology* 153: 769-778.

Rivest, B R; Coyer, J; Tyler, S (1999) The first known invasion of a free-living marine flatworm. *Biological Invasions* 1: 393-394.

Ruppert E E; Fox R S; Barnes R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Skerman TM (1960) Note on *Stylochus zanzibaricus* Laidlaw, (Turbellaria, Polycladida), a suspected predator of barnacles in the Port of Auckland, New Zealand. *New Zealand Journal of Science* 3: 610-614.

Sluys, R; Faubel, A; Rajagopal, S; Velde, G van der (2005) A new and alien species of "oyster leech" (Platyhelminthes, Polycladida, Stylochidae) from the brackish North Sea Canal, The Netherlands. *Helgoland Marine Research* 59: 310-314.

Species 2000. *Catalogue of Life: Dynamic Checklist*. http://www.catalogueoflife.org/dynamic-checklist/browse_taxa.php. Accessed 3 March 2009.

4.10. GASTROPODS AND CHITONS (CLASS: GASTROPODA AND POLYPLACOPHORA)

4.10.1. Hazard Identification

Gastropoda (gastropods) and Polyplacophora (chitons) are two classes of non-bivalve molluscs. The Gastropoda are a diverse class of mobile snail- or slug-like organisms primarily adapted to benthic life. They are divided into the subclasses of Prosobranchia, Opistobranchia and Pulmonata. The prosobranchs are marine, freshwater and terrestrial snails that usually have a coiled shell and operculum[12]. By contrast, opistobranchs have a shell that is uncoiled, reduced or entirely absent, e.g. sea slugs, sea hares and sea butterflies. Subclass Pulmonata includes a few marine intertidal species, such as limpets, as well as land snails and many freshwater forms.

The chitons are small, armoured, herbivorous molluscs that live attached to hard substrata by a broad flat foot. They are dorsoventrally flattened and oval-shaped, with eight overlapping calcareous plates (Ruppert et al. 2004). When immersed, chitons graze algae from hard substrata. Many chiton species have a restricted range, returning to the same niche in the substrata before being emmersed at low tide. The chiton makes a watertight seal protecting it against desiccation and dislodgement (Ng & Williams 2006).

Reproduction varies among the gastropod subclasses. Prosobranchs are gonochoric with external fertilisation while the hermaphroditic opisthobranchs and pulmonates copulate and fertilise internally (Ruppert et al. 2004). The majority of gastropods lay gelatinous egg masses, some hatching as planktonic veliger larvae while others hatch juveniles. Chitons have separate sexes and reproduce via broadcast spawning.

Ecologically, the grazing of hard surfaces by chiton and herbivourous gastropods influences the structure and diversity of fouling communities (Duffy & Hay 2001, Aguilera & Navarrete 2007, Nydam & Stachowicz 2007). In addition, predatory gastropods may contribute to the shaping of sessile fouling communities (Nydam & Stachowicz 2007), while detritivorous gastropods disperse the soft sediments in which they feed, incorporating the oxygen that is essential for infaunal organisms.

Globally, there have been many recorded introductions of gastropods, particularly, opistobranchs (Boyd 1999, Albayrak & Caglar 2006, Ozturk & Can 2006, Daskos & Zenetos 2007). The impacts of these invasions include economic losses and changes to the benthic community, for example, loss of diversity and alteration of trophic webs. Large predatory whelk species (*Rapana venosa, Ocinebrallus inornatus* and *Urosalpinx cinerea*) threaten wild and commercial bivalve populations in a variety of locations (Mann & Harding 2000, Martel et al. 2004, Savini & Occhipinti-Ambrogi 2006, Faasse & Ligthart 2007). The introduction of the unusual filter-feeding limpet, *Crepidula fornicata,* has resulted in a variety of environmental impacts (Richard et al. 2006).

[12] A round 'lid' used to block the opening once the organism has retracted into the shell.

There are two recorded cases of non-indigenous chitons forming established populations. Although the "Japanese chiton" *Acanthichitona achates* was inadvertently introduced to Washington State, USA in 1948 through seed oysters, no impacts have been recorded (Ray 2005, Elder 2009). However, the chiton, *Chiton glaucus,* introduced to southeastern Tasmania prior to 1910, has become "one of the most conspicuous and common chiton species and hence must be having some impact on the native species" (Hayes et al. 2005).

Given that gastropods and chitons:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that gastropod and chiton species represent a potential hazard via vessel biofouling.

4.10.2. Risk Assessment

4.10.2.1. Entry assessment

Gastropods and chitons are known components of vessel biofouling assemblages. For example, six gastropods and one vermitid species were recorded in a hull fouling survey at the Port of Recife, Brazil (Farrapeira et al. 2007). Two species of gastropod have established in New Zealand after being introduced as fouling on oil rigs (Cranfield 1999). In the MAFBNZ commissioned research on biofouling, approximately 1% of vessels that entered New Zealand had either gastropods and/or chitons recorded (Inglis et al. in prep.).

The gastropods and chitons usually found in association with vessel biofouling are small intertidal/upper subtidal species with high attachment strength and/or close association with other hull fouling species (Cranfield et al. 1999, Mann and Harding 2000). For example, many nudibranchs introduced to New Zealand and Port Phillip Bay (Australia) have been associated with the hull fouling bryozoans upon which they feed (Miller and Willan 1986 *In* Boyd 1999, Cranfield 1999). Nudibranchs are capable of swimming directly onto the hull of a vessel and it has been suggested that egg masses attached to a hull may result in introductions (Kerckhof et al. 2006, Faasse & Ligthart 2007). In general, adults of the larger predatory molluscs and *C. fornicata* occupy infaunal habitats, however, juveniles spend the first part of their life in fouling communities on hard surfaces. Within the hull environment, seachests may provide the most suitable habitat for gastropods and chitons. Depending on their life-history, they could either settle independently or be drawn in by the action of the seachest (Mann & Harding 2000).

Gastropods and chitons have a known ability to survive translocation. For example, many of the introduced gastropods recorded in New Zealand are native to either Japan or Europe (Cranfield 1999). Gastropods with shells are able to retreat and plug the aperture with mucous or their operculum, protecting themselves from adverse physical conditions. In addition, many of the molluscs found in hull fouling are from coastal habitats and are thus adapted to changes in environmental conditions. For example, *Rapana venosa* adults can tolerate salinities in the range of 18-28 ppt whilst juveniles show no effects when exposed to salinities as low as 10 ppt (Mann & Harding 2000).

There have been no known introductions of non-indigenous gastropods or chitons in New Zealand since the early 1980's (Kospartov et al. in press).

The effect of anti-fouling paint on gastropods and chitons is uncertain. The biocide tributyltin was responsible for the near extinction of certain whelk species (Faasse & Ligthart 2007), however, this biocide has been banned internationally since 2008. As modern anti-fouling paints require regular current movement to be effective (Floerl et al. 2005), it is possible that gastropods and chitons may settle in the niche areas of the hull where the water current results in suboptimal hull protection. In addition to the physical barrier provided by mucous production, close association with other biofouling is also expected to diminish contact between these organisms and anti-fouling biocides.

Given that particular gastropod and chiton species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous gastropod and chiton species into New Zealand via vessel biofouling is non-negligible.

4.10.2.2. Establishment assessment

Suitable habitat for non-indigenous gastropods and chitons is known to exist in New Zealand. The prevalence of hard substrate in ports and marinas within New Zealand may provide suitable substrate for establishment of gastropods and chitons. In event of the introduction of predatory whelks or *C. fornicata*, the harbours close to ports and marinas commonly encompass estuarine and marine infaunal (muddy/sandy) habitats.

Gastropods and chitons can enter and establish through the movement of adults and/or juveniles directly off the vessel, for example nudibranchs, or through spawning and settlement. The dislodgement of closely associated biofouling or individuals may also lead to the introduction of non-indigenous gastropods and chitons to New Zealand. As most of the gastropods found on vessel biofouling are likely to be broadcast spawners (Mann and Harding 2000), for fertilisation to take place there must be at least two individuals in close proximity. Pelagic larval phases allow for long distance dispersal and establishment in other locations (Gilman 2006, Viard et al. 2006) by increasing the chances of encountering suitable habitat. Some snails and many nudibranchs associated with ephemeral biofouling have short life-cycles and may mature in a matter of days or weeks (Willan 2003). The shorter life-cycle may increase the chances of establishment in New Zealand waters as species are more likely to be reproductively active following entry into New Zealand. The sex and reproductive output of some gastropod species is dependent on the chemicals released by other conspecifics, for example, the establishment of *C. fornicata* is reliant on the presence of other juveniles or adults in the near vicinity (ISSG 2005).

The rates of non-indigenous species establishment can often be correlated with the abundance of the organisms at the source location. For example, Miller et al. (2007) investigated the successful and failed molluscan invaders in San Francisco Bay in association with oyster transfers from the east coast of the USA. They found that the relative abundance of molluscs in the source region was the strongest predictor of their invasion success, thus emphasising the importance of propagule pressure to invasion success (Miller et al. 2007).

The most recent known establishment of a gastropod or chiton in New Zealand was the nudibranch *Janolus hyalinus* prior to 1981 (Kospartov et al. in press). There have been no recorded establishments of chitons in New Zealand (Kospartov et al. in press).

Given that particular gastropod and chiton species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been previously recorded as established in New Zealand (gastropods only);

it is concluded that the likelihood of establishment of non-indigenous gastropod and chiton species in New Zealand via vessel biofouling is non-negligible.

4.10.2.3. Consequence assessment

The consequences of herbivorous gastropods or chitons in New Zealand are likely to be interspecific competition with other grazers. A number of common grazing gastropods and chitons of various sizes are present in New Zealand coastal waters, including *C. glaucus* (Morton 2004). This chiton is known to exhibit intraspecific competition and interspecific competition with the omnivorous seastar *Patiriella regularis* in intertidal cobble fields (Palmer & Phillips 2009). It is uncertain whether non-indigenous introductions will compete, be excluded, co-exist, or find a new niche.

Ecological consequences of predatory gastropods in New Zealand are likely to be interspecific competition with other predators, and possible negative effects on indigenous fauna. The establishment of predatory gastropods may negatively impact on bivalve aquaculture.

Oyster drills are gastropods of the family Muricidae that prey on oysters, other bivalves, including mussels (Hanks 1957), and barnacles (Faasse & Ligthart 2007). Among pest species, oyster drills are widely considered to have the highest impacts to world oyster production. These organisms can prey on oysters of any size and impacts can range from nil to over 90% predation depending on local conditions (Butler 1985). Specifically, the American oyster drill (*Urosalpinx cinerea*) was recorded as consuming more than half the oyster spat in certain European estuaries (Cole 1942 *In* Hulme et al. 2008). In 2003, the global value of oyster (*Crassostrea gigas*) production was estimated at US$3.69 billion (FAO 2006). Despite enormous research efforts, there appears that little can be done to mitigate the effects of oyster drills once established, the only mitigation measures are manually scrubbing snail eggs off the oysters, leaving areas fallow for periods of time, or growing in less saline areas (National Sea Grant College Program 2003).

Given that the prey of oyster drills can include both oysters and mussels, and the known impacts to commercial stocks in other jurisdictions, the potential consequences to New Zealand's aquaculture industry could be wide-ranging. In 2008, mussels (*Perna canaliculus*) and oysters (*C. gigas*) comprised of 77% (NZ$204.3 million) and 6% of the value (NZ$16.9 million) of New Zealand's aquaculture exports, respectively (Aquaculture New Zealand 2009). Given the value of bivalve aquaculture and commercial fishing to New Zealand, the possible establishment of specific gastropod species via hull biofouling is a non-negligible concern.

The slipper limpet *C. fornicata* is a filter feeding gastropod that forms large aggregations and may occur at densities exceeding 1700 m^{-2} (DAISIE 2011). These structures have been reported to change local sedimentation dynamics, impacting the benthic community. In addition, their filter feeding behaviour has been implicated in recruitment failures for commercial oyster species and declines of juvenile sole (commercially harvested *Solea solea*) in nursery habitats (Le Pape et al. 2004, Richard et al 2006, Beninger et al 2007, Decottignies et al 2007). Further, slipper limpets require removal from oyster and scallops before marketing and as such increase commercial costs (Minchin 2008). Introductions of this species have occurred through shellfish stock movement and vessel biofouling (DAISIE 2011).

4.10.3. Risk Estimation

Given that particular gastropod species:
- are known components of hull fouling assemblages;
- are known to have been introduced to new locations;
- are capable of forming establishing populations in New Zealand;
- have well known invasive behaviours;
- have had significant impacts on core values;
- are likely to have adverse effects on core values, such as, bivalve aquaculture;

it is concluded that the import of non-indigenous gastropod species via biofouling poses a non-negligible risk to New Zealand's marine core values.

Given that particular chiton species:
- are not common components of vessel hull biofouling assemblages;
- have not been recently recorded as introduced to New Zealand;
- are not recorded as being responsible for impacts to core values;

it is concluded that the import of non-indigenous chiton species via biofouling poses a negligible risk to New Zealand's marine core values.

4.10.4. Risk Management

A generic approach is suitable to managing the risk associated with gastropods (Section 3). In addition, the mobility of gastropods and the dislodgement of their egg masses present further management difficulties, as a result, systems that contain all organisms in-water prior to treatment may be required to reduce the risk of pre-treatment escape.

4.10.5. References

Aguilera, M A; Navarrete, S A (2007) Effects of *Chiton granosus* (Frembly, 1827) and other molluscan grazers on algal succession in wave exposed mid-intertidal rocky shores of central Chile. *Journal of Experimental Marine Biology and Ecology* 349: 84-98.

Albayrak, S; Caglar, S (2006) On the presence of *Siphonia belcheri* Hnaley, 1858 [Gastropoda: Siphonariidae] and *Septifer bilocularis* (Linnaeus, 1758) [Bivalvia: Mytilidae] in the Iskenderun Bay (SE Turkey). *Aquatic Invasions* 1(4): 292-294.

Aquaculture New Zealand (2009) New Zealand Aquaculture farm facts (2nd edition). http://www.aquaculture.org.nz/aquaculture-in-nz/industry-overview/overview/. Accessed 20 January 2010.

Beninger, P G; Decottignies, P; Guiheneuf, F; Barille, L; Rince, Y (2007) Comparison of particle processing by two introduced suspension feeders: selection in *Crepidula fornicata* and *Crassostrea gigas*. *Marine Ecology Progress Series* 334:165-177.

Boyd, S (1999) The Introduced Mollusca of Port Phillip Bay. In: Hewitt C L; Campbell M L; Thresher R E; Martin R B (Eds.) *Marine Biological Invasions of Port Phillip Bay , Victoria*. Centre for Research on Introduced Marine Pests. Technical Report No. 20. CSIRO Marine Research; Hobart, Australia; pp.129-149.

Butler, P A (1985) Synoptic review of the literature on the Southern Oyster Drill *Thais haemastoma floridana*. NOAA Technical Report NMFS 35. 9pp.

Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) Adventive marine species in New Zealand. NIWA Technical Report 34. 48 pp.

DAISIE European Invasive Alien Species Gateway, 2011. *Crepidula fornicata*. Available from: http://www.europealiens.org/speciesFactsheet.do?speciesId=50159# Accessed 5 January 2011.

Daskos, A; Zenetos, A (2007) Additions to the knowledge of alien Opistobranchia of Greece. *Aquatic Invasions* 2(3): 258-260.

Decottignies, P; Beninger, P G; Rince, Y; Robins, R J; Riera, P (2007) Exploitation of natural food sources by two sympatric, invasive suspension feeder: *Crassostrea gigas* and *Crepidula fornicata*. *Marine Ecology Progress Series* 334: 179-192.

Duffy, J E; Hay, M E (2001) The ecology and evolution of marine consumer-prey interactions. In Bertness, M D; Hay, M E; Gaines, S D (eds) Marine Community Ecology. Sinaur Associates; Sunderland, United States of America; pp 131-157.

Elder, N (2009) *Acanthochitona achates*. USGS Non-indigenous Aquatic Species Database. http://nas.er.usgs.gov/queries/FactSheet.asp?speciesID=1190. Accessed 24 September 2009

Faasse, M; Ligthart, M (2007) The American oyster drill, *Urosalpinx cinerea* (Say, 1822) introduced to the Netherlands – increased risks after ban on TBT? *Aquatic Invasions* 2: 402-406.

FAO (2006) *Crassostrea gigas* fact sheet: Cultured Aquatic Species Information Program. http://www.fao.org/fishery/culturedspecies/Crassostrea_gigas/en. Accessed 3 February 2010.

Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship hull fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55: 207-221.

Gilman, S E (2006) The northern geographic range limit of the intertidal limpet *Collisella scabra*: a test of performance, recruitment, and temperature hypotheses. *Ecogeography* 29: 709-720.

Hanks, J E (1957) The feeding rates of the common oyster drill, *Urosalpinx cinerea* (Say), at controlled water temperatures. *Biological Bulletin* 112:330-335.

Hulme, P E; Roy, D B; Cunha, T; Larsson, T-B (2008) A pan-European inventory of alien species: rationale, implementation and implications for managing biological invasions. In DAISIE (eds.) *The Handbook of European Alien Species*. Springer, Dordrecht.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Le Pape, O: Guerault, D; Desaunay, Y (2004) Effect of an invasive mollusc, American slipper limpet *Crepidula fornicata*, on habitat suitability for juvenile common sole Solea solea in the Bay of Biscay. *Marine Ecology Progress Series* 277: 107–115.

Mann, R; Harding, J M (2000) Invasion of the North American Atlantic coast by a large predatory Asian mollusc. *Biological Invasions* 2: 7-22.

Martel, C; Guarini, J M; Blanchard, G; Sauriau, P G; Trichet, C; Robert, S; Garcia-Meunier (2004) Invasion by the marine gastropod *Ocinebrellus inornatus* in France. III. Comparison of biological traits with the resident species *Ocenebra erinacea*. *Marine Biology* 146: 93-102.

Miller, A W; Ruiz, G M; Minton, M S; Ambrose, R F (2007) Differentiating successful and failed molluscan invaders in estuarine ecosystems. *Marine Ecology Progress Series* 332: 41-51.

Morton, J (2004) *Seashore ecology of New Zealand and the Pacific*. David Bateman, Auckland, New Zealand.

National Sea Grant College Program (2003) The Gulf Oyster Industry: seizing a better future. Communictions office, Louisiana Sea grant College Program; Louisiana State University, Louisiana; USA. 17pp.

Ng, J S S; Williams, G A (2006) Intraspecific variation in foraging behaviour: influence of shore height on temporal organization of activity in the chiton *Acanthopleura japonica*. *Marine Ecology Progress Series* 321:183-192.

Nydam, M; Stachowicz, J J (2007) Predator effects on fouling community development. *Marine Ecology Progress Series* 337: 93-101.

Ozturk, B; Can, A (2006) Indo-Pacific gastropod species in the Levantine and Aegean Seas. *Aquatic Invasions* 1(3): 124-129.

Palmer, S; Phillips, N (2009) The ecological role of an omnivorous seastar (*Patiriella regularis*) in intertidal cobble fields. In O'Shea, S; Costello, M J; O'Connell, K (eds) *Programme and Abstracts for Annual Conference in Auckland 2-4 September 2009*. New Zealand Marine Sciences Society; Accessed at http://nzmss.rsnz.org; 1-126 pp.

Ray, G L (2005) Invasive marine and estuarine animals of the Pacific northwest and Alaska. U.S. Army Engineer Research and Development Center; Vicksburg, U.S.A.; 19 pp.

Richard, J; Huet, M; Thouzeau, G; Paulet, Y-M (2006) Reproduction of the invasive slipper limpet, Crepidula fornicata, in the Bay of Brest, France. *Marine Biology* 149: 789-801.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Savini, D; Occhipinti-Ambrogi, A (2006) Consumption rates and prey preference of the invasive gastropod *Rapana venosa* in the Northern Adriatic Sea. *Helgoland Marine Research* 60: 153-159.

Viard, F; Ellien, C; Dupont, L (2006) Dispersal ability and invasion success of Crepidula fornicata in a single gulf: insights from genetic markers and larval-dispersal model. *Helgoland Marine Research* 60: 144-152.

Willan, R (2003) *Nudibranchs*. In Andrew, N; Francis, M (eds) The Living Reef – the ecology of New Zealand's rocky reefs. Craig Potton Publishing; Nelson, New Zealand.

4.11. HYDROIDS (CLASS: HYDROZOA)

4.11.1. Hazard Identification

Hydrozoa are a diverse class of predatory animals belonging to the phylum Cnidaria. This class consists of approximately 3,000 species, the majority of which occupy marine habitats (Ruppert et al. 2004). Hydrozoans have two alternative body plans: the polyp and the medusa. A typical polyp is a sedentary organism that has the mouth uppermost which is surrounded by tentacles laden with stinging cells. The polyp body plan is represented by the sea anemone. A typical medusa (upside-down polyp) has the mouth on the underside of a bell that is surrounded by a trail of tentacles. These species swim by pulsing of the bell. The medusa body plan is represented by the jellyfish. Polyps often reproduce by budding off other polyps, however, in some species these buds remain attached to form colonies. Colonial species include seaweed-like hydroids, pelagic Portuguese man-of-war, and reef building fire and rose corals. The term hydroid refers to sessile benthic colonies, while colonial pelagic species are called siphonophores (Ruppert et al. 2004, Cairns et al. 2009).

The life-cycle of hydrozoa involves a benthic polyp phase and a pelagic medusa phase. Hydroid colonies form from an individual polyp by budding asexually. A mature colony consists of both feeding polyps, with tentacles and stinging cells, and reproductive polyps without tentacles, which regularly bud off medusae (Ruppert et al. 2004). The medusae are the dispersal and sexual stage of the life-cycle. Cells in mature medusae undergo meiosis and give rise to either sperm or eggs, which are released into the surrounding water to be fertilised. After development, the larva settles onto a suitable substrate to give rise to a new colony (Ruppert et al. 2004).

Ecologically, hydrozoans are successful colonists and can have high abundances under certain environmental conditions (Ma & Purcell 2005). Hydroids constitute an important component of the fouling community, growing rapidly and profusely on both natural and anthropogenic hard substrata. As micro-carnivores it has been suggested that hydroids may play an important role in the energy transfer between benthos and plankton (Ma & Purcell 2005). The hydromedusae produced by the polyps have also been shown to impact zooplankton and ichthyoplankton populations (Larson 1987, Purcell & Grover 1990). Due to their effects on larval settlement and community structure, the presence of hydroids on vessel hulls is a concern for both aquaculture (Mortensen et al. 2000) and wildlife heritage areas (Wyatt et al. 2005, Lewis et al. 2006). Well known introduced hydrozoan species include *Obelia dichotoma, Moerisia lyonsi, Maeotias marginata, Blackfordia virginica* (Cohen & Carlton 1995, Vainola & Oulasvirta 2001, Ma & Purcell 2005, Bardi & Marques 2009).

Given that hydroids:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that hydroid species represent a potential hazard via vessel biofouling.

4.11.2. Risk Assessment

4.11.2.1. Entry assessment

Hydroids are common fouling components in ports, on wharf piles, and other anthropogenic surfaces (Lewis 2004, Cairns et al. 2009). Therefore, hydroids are likely to encounter the hulls of international vessels destined for New Zealand.

Hydroids are common components of hull fouling. For example, Woods Hole (1952) listed more species of hydroids than any other macrofaunal hull fouling group. Hydroids can be found on hulls with or without other biofouling and are tolerant to high concentrations of copper ions, the active ingredient of the majority of anti-fouling paints (Pyefinch & Downing 1949 *In* Lewis 2004, Railkin 2004). Hydroids have traditionally been found in greater numbers on slower moving vessels and on hull locations that experience slower flow velocities (Skerman 1960, Railkin 2004). For example, in the MAFBNZ commissioned research hydrozoan fouling was present on 8% of sampled vessels, 78% of which were recreational vessels, a slow moving vessel class.

Hydroids are known to survive translocation. The "contamination" of Pacific coast harbours by ship-introduced hydroids was recorded as early as 1915 (Cohen & Carlton 1995). To be transported by vessels, species must occur in estuarine and coastal habitats and therefore be tolerant of a range of salinities and high water turbidity (Watson 1999). Cryptic and small hydroids are more likely to survive voyages as they are likely to be sheltered by larger fouling organisms or be present in niche areas (Watson 1999). The larval stages of some species can also be cryptic and it has been suggested that larger species could be translocated via this lifestage (Watson 1999). Until recently, coincident with the increased interest in marine biosecurity, there were few data regarding the transport of hydroids via shipping (Watson 1999).

Taxonomic identification of hydrozoans is problematic. Difficulties in identification may be due to species similarities, high colony variability or poor specimen quality, for example, distinguishing features may be missing, particularly towards the end of the reproductive season and as predators and currents take their effect (Boyd et al. 2002). As a result there has been a history of misidentifications, for instance in San Francisco Bay, New England, and Europe (Cohen & Carlton 1995, Boyd et al. 2002). In the MAFBNZ commissioned research on biofouling, 22 of 58 hydrozoan samples were classified as indeterminate (Inglis et al. in prep.).

Given that particular hydroid species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous hydroid species into New Zealand via vessel biofouling is non-negligible.

4.11.2.2. Establishment assessment

Hydroids are general opportunist foulers of hard substrates and, as such, are likely to find suitable and available habitats in New Zealand. Watson (1999) noted that the majority of fouling species would succeed in the high flow environments provided by the jetty pilings of port and harbours. The temperature range of the receiving environment is likely to be an important influence of the establishment of hydroid populations, particularly for seasonal species that are adapted to narrow bands of summer or winter temperatures (Watson 1999). Given their wide tolerances, salinity and turbidity are unlikely to be major influences.

Hydroids can establish in the New Zealand environment through fragmentation of colonies (asexual reproduction) or the release of medusae (sexual reproduction). In ports or harbours, fertile colonies or fragments may be released into the water column by abrasion during berth or during hull cleaning operations. The release of medusae may occur if fertile colonies are components of the biofouling assemblage. As smaller hydroid species grow and reach maturity faster than larger species, they are more likely to pass through a life-cycle and release propagules in New Zealand waters (Watson 1999).

Twenty six species of non-indigenous hydroid are recorded as established in New Zealand (Kospartov et al. in press). Five of these species have established since 2000. Non-indigenous hydroids have been recorded throughout New Zealand from the Auckland Islands to the Kermadec Islands (Kospartov et al. in press).

Given that particular hydroid species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous hydroid species in New Zealand via vessel biofouling is non-negligible.

4.11.2.3. Consequence assessment

The consequences of the establishment of non-indigenous hydroids to New Zealand is likely to range from nuisance fouling to the structuring of benthic fouling communities. The fouling of vessel hulls and general water intake pipes may lead to increased maintenance, however, this should be resolved by a regular schedule and should not result in significant economic impacts.

Ecologically, hydroids can dominate parts of the sea floor; as micro-carnivores they may have unforeseen effects on New Zealand marine assemblages. Hydrozoans feed on a range of eggs, larvae, and small fauna present in the water column, transferring energy from the pelagic environment to the benthos (Gili et al. 1996). Moreover, their predation of eggs and larvae in the water column can structure the benthic fouling community. For example, Standing (1976) found that the predation of the settling barnacle cyprids (*Balanus crenatus*) by *Obelia dichotoma* in Bodega Bay, California, USA, allowed the settlement of the otherwise excluded solitary ascidian (*Ascidia ceratodes*). In addition, Wyatt et al. (2005) expressed concern over introduced hydroids in the Shark Bay World Heritage Site, Australia, because of their effects on larval settlement.

Consequences of non-indigenous species of hydroids are likely to include fouling of anthropogenic structures in coastal marine settings. Hydroids are a substantial part of aquaculture biofouling assemblages. The impact of fouling of aquaculture operations may include the inhibition of bivalve growth, competition for resources and the consumption of settling larvae (Sandifer et al. 1974, Claereboudt et al. 1994). The costs of biofouling to the aquaculture industry can be significant. For example, biofouling can reduce bivalve growth by over 40% and the cleaning of oyster cultures is estimated account for 20% of their market value. The annual cost of fouling on cultured mussels in Scotland is estimated to be up to 750,000 Euro (Campbell & Kelly 2002). Hydroid colonies can recover within a week of cleaning, with some colonies having increased thickness (Guenther et al. in press).

As hydroids provide habitat and chemical cues for a wide range of other organisms such as amphipods and polychaetes, they may facilitate the settlement of other non-indigenous species in port/marina and aquaculture environments which may result in further consequences to core values.

4.11.3. Risk Estimation

Given that particular hydrozoan species:

- have been recorded entering New Zealand via hull fouling;
- are unlikely to be correctly identified upon entry into New Zealand;
- may facilitate the entry of a variety of other non-indigenous organisms;
- are capable of forming establishing populations in New Zealand;
- have known impacts to economic values, such as, aquaculture production;
- have the potential to impact benthic community structures with far reaching indirect effects;

it is concluded that the import of non-indigenous hydrozoan species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.11.4. Risk Management

Due to their small and cryptic nature and the high likelihood of non-identification, a generic approach to the removal of biofouling would be the appropriate risk management option of hydrozoans (Section 3). Specific considerations for hull fouling hydrozoans should consider appropriate local containment due to the likelihood of release of associated non-indigenous organisms into the receiving environment.

4.11.5. References

Bardi, J; Marques, A C (2009) The invasive hydromedusae *Blackfordia virginica* Mayer, 1910 (Cnidaria: Blackfordiidae) in southern Bazil, with comments on taxonomy and distribution of the genus *Blackfordia*. *Zootaxa* 2198: 41-50.

Boyd, M J; Mulligan, T J; Shaughnessy, F J (2002) Non-Indigenous of Humboldt Bay, California. A Report to the California Department of Fish and Game. February 28, 2002.

Cairns, S D; Gershwin, L-A; Brook, F J; Pugh, P; Dawson, E W; Ocana, O V; Vervoort, W; Williams, G; Watson, J E; Opresko, D M; Schuchert, P; Hine, M;

Gordon, D P; Campbell, H J; Wright, A J; Sanchez, J A; Fautin, D G (2009) Phylum Cnidaria. In Gordon, D P (ed) *New Zealand Inventory of Biodiversity. Volume 1*: Kingdom Animalia; Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press; Christchurch, New Zealand; pp 59-101.

Campbell, D A; Kelly, M S (2002) Settlement of *Pomatoceros triqueter* (L.) in two Scottish lochs, and factors determing its abundance on mussels grown in suspended culture. *Journal of Shelfish Research* 21: 519-527.

Claereboudt, M R; Bureau, D; Cote, J; Himmelman, J H (1994) Fouling development and its effect on the growth of juvenile giant scallops (*Placopecten magellanicus*) in suspended culture. *Aquaculture* 121(4): 327-342.

Cohen, A N, Carlton, J T (1995) Non-indigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta. A Report for the United States Fish and Wildlife Service, Washington D.C. and The National Sea Grant Program, Connecticut Sea Grant (NOAA Grant Number NA36RG0467).

Gili, J-M; Hughes, R G; Alva, V (1996) A quantitative study of the feeding of the hydroid *Tubularia larynx* Ellis and Solander, 1786. *Scientia Marina* 60(1): 43-54.

Guenther, J; Misimi, E; Sunde, L M (in press) The development of biofouling, particularly the hydroid *Ectopluera larynx*, on commercial salmon cage nets in Mid-Norway. *Aquaculture*.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Larson, R J (1987) Trophic ecology of planktonic gelatinous predators in Saanich Inlet, British Columbia: diet and prey selection. *Journal of Plankton Research* 9: 811-820.

Lewis, J (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research*. Ballast Water Research Series Report No. 14. Dept of Agriculture, Fisheries and Forestry; Canberra, Australia; 129 pp.

Lewis, P N; Bergstrom, D M; Whinam J (2006) Barging in: a temperate marine community travels to the sub-Antarctic. *Biological Invasions* 8: 787-795.

Ma, X; Purcell, J E (2005) Effects of temperature, salinity, and predators on mortality of and colonization by the invasive hydrozoan *Moerisia lyonsi*. *Marine Biology* 147: 215-224.

Mortensen, S; Van Der Meeren, T; Fosshagen, A; Hernar, I; Harkestad, I; Torkildsen, L; Bergh, O (2000) Mortality of scallop spat in cultivation, infested with tube dwelling bristle worms, *Polydora* sp. *Aquaculture International* 8: 267-271.

Purcell, J E; Grover, J J (1990) Predation and food limitation as causes of mortality in larval herring at a spawning ground in British Columbia. *Marine Ecology Progress Series* 59: 55-61.

Railkin, A I (2004) Marine Biofouling: Colonization Processes and Defences. CRC Press; 303 pp.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Sandifer, P A; Smith, P I J; Calder, D R (1974) Hydrozoans as pests in closed-system culture of larval decapod crustaceans. *Aquaculture* 4: 55-59.

Skerman, T M (1960) Ship-fouling in New Zealand waters: a survey of marine fouling organisms from vessels of the coastal and overseas trade routes. *New Zealand Journal of Science* 3: 620-648.

Standing, J D (1976) Fouling community structure: effects of the hydroid, *Obelia dichotoma*, on larval recruitment. In Mackie, G O (ed) *Coelenterate ecology and behaviour. The Third International Symposium on Coelenterate Biology held at University of Victoria, Victoria, British Columbia May 10-13*; Plenum Press, New York, USA; pp 155-164.

Vainola, R; Oulasvirta, P (2001) The first record of *Maeotias marginata* (Cnidaria, Hydrozoa) from the Baltic Sea: a Pontocaspian invader. *Sarsia* 86: 401-404.

Watson, J E (1999) *Review of Hydroids Introduced to Victorian Waters*. pp. 88-106. In Hewitt, C L; Campbell, M L; Thresher, R E; Martin, R B (Eds) Marine Biological Invasions of Port Phillip Bay, Victoria. Centre for Research on Introduced Marine Pests. Technical Report No. 20. CSIRO Marine Research; Hobart, Australia. 344 pp.

Woods Hole Oceanographic Institution (1952) *Marine Fouling and its Prevention*. United States Naval Institute; Annapolis, United States of America.

Wyatt, A S J; Hewitt, C L; Walker, D I; Ward, T J (2005) Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment, *Diversity and Distributions* 11(1): 33-44.

4.12. MACROALGAE

4.12.1. Hazard Identification

Macroalgae are attached macroscopic algae ranging in size from filaments a few millimetres long (e.g. *Ulva* spp. Heesch et al. 2007) to large plants greater than 45 metres long (e.g. *Macrocystis pyrifera* Schiel 2003). While some species are of cryptic habit many are significant habitat formers, particularly along rocky coastlines (Schiel 1990, Schiel & Hickford 2001). Macroalgae can be generally classified as red, brown or green which correspond to the phylogenetic groupings of Rhodophyta, Ochrophyta and Chlorophyta, respectively (Adams 1994, Algaebase 2008).

Brown kelps, such as *Ecklonia radiata* and *Durvillaea antarctica*, are recognised as ecosystem engineers that provide "vertical structure, habitat, nurseries and food" (Schiel 2003), regulate community composition and are often conspicuous as low intertidal and/or shallow subtidal forests (Taylor & Schiel 2005). Although the green and red macroalgae are generally smaller in size and less conspicuous than brown algae they remain important components of both intertidal and subtidal ecosystems with some forming specific biogenic habitats, such as rhodolith beds. Green and red macroalgae are morphologically diverse ranging from calcified, encrusting coralline species, through filamentous, foliose bushes to larger bladed species. Of the species known to be introduced into New Zealand, most are small red or green species (Nelson & Parsons 2003) although significant exceptions include *Codium fragile* ssp. *fragile*, *Undaria pinnatifida* and *Grateloupia turuturu*.

A global review of macroalgal species introductions provides a list of 277 species with representatives from Rhodophyta, Ochrophyta and Chlorophyta (Williams & Smith 2007). Arguably the most prominent of these species are the Japanese kelp *Undaria pinnatifida* and the aquarium weed *Caulerpa taxifolia* (Lowe et al. 2004). However, there are other species reported to have significant invasive impacts, such as, *Caulerpa racemosa* varieties, *Sargassum muticum*, *Grateloupia turuturu* and *Codium fragile* ssp. *tomentosoides* (Scheibling & Gagnon 2006, Strong et al. 2006, Mathieson et al. 2008, Piazzi & Balata 2009).

Of the non-indigenous macroalgal species established in New Zealand *U. pinnatifida* is arguably the most conspicuous and has had the greatest recorded impact, substantially modifying subtidal habitat (Battershill et al. 1998, Russell et al. 2008). Ribera Siguan (2003) found most macroalgal translocations could be attributed to vessels and aquaculture. Similarly, Williams and Smith (2007) determined that hull fouling was the dominant vector of macroalgal introductions. These factors combined account for between 79% (Ribera Siguan 2003) and 85% (Williams & Smith 2007) of all macroalgal translocations. The morphologies of macroalgal species associated with hull fouling are characterised as being small with flexible, filamentous vegetative structures and high growth rate (Ribera Siguan 2003, Williams & Smith 2007). Callow (1986) noted that the dominant macroalgal foulers of artificial substrates are of the species Enteromorpha[13] and Ectocarpus, with both taxa having forms resistant to anti-fouling biocides. However, larger macroalgae, such as *U. pinnatifida*, have also been associated with hull fouling (Hay 1990). These larger species may be more likely to survive on slower moving vessels, such as yachts and barges, or are translocated as other life stages which can survive on hulls, such as has been demonstrated for the microscopic stage of *U. pinnatifida* (Coutts 1999).

[13] Enteromorpha is now considered a synonym of Ulva (Hayden et al. 2003).

In the MAFBNZ commissioned research on biofouling, the majority of the identified algal species on recreational vessels were small and filamentous with only one species not known to be established in New Zealand (Inglis et al. in prep.). However, many of the samples were unable to be identified to species level and as a result it was not possible to determine if the larger macroalgae were non-indigenous to New Zealand. The identification of algal samples is difficult due to complex taxonomy and the requirement of ephemeral features (e.g. reproductive structures) to make conclusive identifications.

Given that macroalgae:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that macroalgal species represent a potential hazard via vessel biofouling.

4.12.2. Risk Assessment

4.12.2.1. Entry assessment

Macroalgae are well known hull fouling organisms (Stubbings 1947, Callow 1986a, Mineur et al. 2007). Ribera Siguan (2003) and Williams & Smith (2007) have identified hull fouling as the most significant vector for the translocation of macroalgal species. Macroalgae establishment on a vessel hull would require a relatively static hull surface, a compromised, ineffective or absent fouling control system and a supply of propagules.

Macroalgal spores can settle and attach to the biofilm on a hull. The initial attachment strength is low whereby spores can easily be dislodged, however, once differentiation has taken place rhizoid cells provide a considerably stronger attachment (Railkin 2004). The spores of some common fouling species can form a strong surface attachment. For example, *Ulva* sp. have been found to have propagules that can resist flow forces equivalent to 40 knots within 4 hours of settlement (Findlay et al. 2002).

Anti-fouling paints are generally effective in preventing macroalgal settlement, however, some species of macroalgae have become tolerant of anti-fouling coatings (Callow 1986b, Mineur et al. 2008). Vessels will have areas of the hull where the anti-fouling system has been compromised (e.g. along the boot top and where the hull rubs against pontoons when moored) or absent (e.g. sacrificial anodes and the propeller). The majority of macroalgal species require an appropriately lit environment which restricts their potential zone of recruitment to conspicuous areas of the hull, such as just below the waterline. As a consequence, vessel maintenance for aesthetic and/or performance enhancement will mitigate algal accumulation for most vessels. However, for some vessel owners/operators the benefit of in-service maintenance is perceived as marginal relative to the costs, particularly for recreational vessels (Hilliard et al. 2006). In addition, where macroalgal growth has been allowed other species, such as mobile invertebrates, are likely to utilise the habitat the macroalgae has created. As a result, macroalgae may facilitate the entry of other non-indigenous species.

Some macroalgal species may survive on vessel hulls in life stages that are difficult to remove and/or are not visible, for example *U. pinnatifida* (Coutts 1999). In addition, macroalgae can become entangled in hull appendages which, if occurring on departure, may not be removed until the vessel reaches its destination. *C. taxifolia*, for example, has invaded estuarine areas of New South Wales where recreational vessels are known to moor. As *C. taxifolia* can regenerate from fragments, the clearing of entangled fragments within the arrival port may lead to establishment. Other species, such as *C. fragile* ssp. *fragile,* may also be translocated in this manner.

Many algal species have been known to have wide tolerances to environmental stressors. For example, the invasive *Gracilaria salicornia* can withstand both high and low temperatures and salinities, and *U. pinnatifida* is known to be able to withstand exposure environments ranging from sheltered harbours to open coasts (Russell et al. 2008). To survive translocation macroalgal species need to be able to tolerate a range of conditions which may include biocides from anti-fouling paint, shear forces, temperature variation and nutrient variation. Many macroalgal species have been found to tolerate most, or all, of these factors.

Given that particular macroalgal species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous macroalgal species into New Zealand via vessel biofouling is non-negligible.

4.12.2.2. Establishment assessment

The macroalgal species arriving as hull fouling are likely to be tolerant to a wide variety of environmental stressors (e.g. *Ulva* spp. Heesch et al. 2007, 2009). As a result, it is assumed that most species surviving translocation to New Zealand have the potential to settle and establish, either as fragments or gametes. However, the long-term viability of the population will depend upon the species ability to adapt to the local conditions and compete against the local flora.

The ability of a non-indigenous species to compete for sheltered space in a disturbed environment is key to establishment success (Mineur et al. 2008). Moreover, it has been noted that invasive species are competitively superior on artificial substrates such as marina pontoons (Bulleri et al. 2006, Farrell & Fletcher 2006). As a consequence, ports may provide ‘stepping stones’ for macroalgal species to establish in New Zealand (Floerl et al. 2009). For example, the major ports of first arrival for recreational vessels (Opua and Auckland) have substantial areas occupied by marina and mooring structures as well as many vessels which may be susceptible to fouling. From such stepping stones non-indigenous species can take advantage of disturbance generated gaps in the natural assemblages to colonise natural substrates. *C. fragile* ssp. *fragile* has been reported to colonise such gaps with dense stands which prevent the natural assemblage re-establishing (Scheibling & Gagnon 2006).

The attributes of invasive macroalgae have been comprehensively reviewed (Williams & Smith 2007). While the three macroalgal classes all have representatives known to be invasive, some genera have characteristics which predispose them to being invasive. For example, successful green macroalgae are from the genera *Codium*, *Caulerpa* and *Bryopsis* which are characterised by their siphonous construction and tolerance to a broad range of habitats. Of the families Williams & Smith (2007) identified as having significantly more invasive species than expected only the Ceramiaceae are substantially represented in the results of the MAFBNZ commissioned research on biofouling (Inglis et al. in prep.). However, the majority of species found conformed to the functional grouping of ‘filamentous’ which contains species expected to be most successful where disturbance potential is high, such as port and marina environments (Williams & Smith 2007).

In New Zealand ten species of Chlorophyta, 11 species of Ochrophyta and 14 species of Rhodophyta have been identified as either non-indigenous or cryptogenic[14] and established in New Zealand (Kospartov et al. in press).

Given that particular macroalgal species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous macroalgal species in New Zealand via vessel biofouling is non-negligible.

4.12.2.3. Consequence assessment

The impacts of introduced macroalgae are generally poorly understood, with Williams & Smith (2007) reporting impact studies for just 17 of the 277 macroalgal species known to be introduced. From these impact studies, changes in community composition, structure and function have been reported (Battershill et al. 1998, Gribben & Wright 2006, Strong et al. 2006, Antoniadou & Chintinaglou 2007, Wright et al. 2007, Piazzi & Balata 2009).

The case of the invasive macroalgae species *C. taxifolia* has been well documented. In 1984, a cold tolerant and fast-growing strain of *C. taxifolia* was accidentally released from the Oceanographic Museum of Monaco into the Mediterranean Sea. By 2000, it had spread to Mallorca, Elba, Sicily and Tunisia (Meinesz et al. 2001 *In* Galil et al. 2008). Although not initially introduced via hull fouling, the hulls of fishing boats and recreational craft were important vectors for the secondary spread of this species (Knoepffler-Peguy et al. 1985, Meinesz, 1992 *In* Galil et al. 2008).

Areas supporting stands of native Mediterranean seagrasses have been smothered by *C. taxifolia* (Zibrowius 1991 *In* NIMPIS 2010) which has had negative effects on shoot density and size of native seagrass (Ceccherelli & Cinelli, 1997 *In* NIMPIS 2010). In addition, the toxic secondary metabolites produced by *C. taxifolia* prevents it being grazed by macro-herbivores and inhibits the growth of native macroalgae (Galil 2000 *In* NIMPIS 2010). As a result of this protective mechanism, the remaining native species may suffer from overgrazing (Boudouresque 1997 *In* NIMPIS 2010). In addition, *C. taxifolia* limits the light levels to benthic species, and traps sediments and organic matter (Boudouresque 1997 *In* NIMPIS 2010).

As a result of the aforementioned effects, the invasive *C. taxifolia* has lead to the disappearance of encrusting algae and reduced diversity of associated benthic assemblages with impacts observed on populations of fish, amphipods, sea urchins and polychaetes (Boudouresque et al. 1995 *In* NIMPIS 2010).

Reduction of available fish habitat has reduced catches. In addition, fish that can consume *C. taxifolia*, such as the Mediterranean bream (*Sarpa salpa*), are rendered unsuitable for human consumption by their accumulation of toxins (Meinesz & Hesse 1991 *In* GISD 2006). Decreases in fishing efficiency have been caused by entanglements in boat anchors, fishing nets and trawling gear (NIMPIS 2010).

[14] Cryptogenic species are defined as those species for which the native range is ambiguous. This is usually due to a lack of information about the species or they are new to science or they have been present in New Zealand for a long time (Cranfield et al. 1998, Kospartov et al. 2008).

In addition, the introduction of macroalgae can alter social values, for example, the introduction of *C. taxifolia* has decreased the aesthetic value of underwater habitats and reducing tourism and scuba diving activities (Meinesz 1999). In the State of Hawai'i, the island of Maui contends with a US$20 million yearly loss of tourism revenue due to the fouling of beaches by non-indigenous algae (Beck et al. 2008).

In New Zealand, the seagrass habitats of the Kaipara harbour are essential for the snapper fishery of the west coast of the North Island as they provides habitat for approximately 90% of juvenile fish (Morrison 2008). Given that sea grass habitats have been overgrown by the invasive strain of *C. taxifolia* (Williams & Smith 2007), such New Zealand habitats are likely to be at risk along with their associated ecosystem services including fisheries.

Economic impacts resulting from the cost of *Caulerpa* spp. eradication included approximately $US6 million spent in Southern California up to 2004 (Anderson, 2004 *In* GISD 2006) and $AUS6-8 million in South Australia (GISD 2006).

Many native macroalgae provide the structure, habitat and food which drive many coastal ecosystems. As a result, any modification to the species composition and abundance is likely to have flow-on effects to coastal marine values.

4.12.3. Risk estimation

Given that particular macroalgal species:

- are known to be associated with vessel biofouling;
- may facilitate the entry of a variety of other non-indigenous organisms;
- are capable of forming establishing populations in New Zealand;
- have well known invasive behaviours;
- where introduced, have been associated with high impacts;
- have the potential impact to disturb the native macroalgal structure which may have far reaching indirect impacts to core values;

it is concluded that the import of non-indigenous macroalgal species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.12.4. Risk management

A generic approach to the removal of biofouling would be the appropriate risk management option of macroalgal hull fouling (Section 3). Key considerations for macroalgal risk management during mitigation actions are the containment (and appropriate neutralisation) of macroalgal fragments, including reproductive structures (such as spores, gametophytes), and any associated mobile organisms.

4.12.5. References

Adams, N M (1994) *Seaweeds of New Zealand: an illustrated guide*. Canterbury University Press; Christchurch, New Zealand.

Antoniadou, C; Chintinoglou, C (2007) Zoobenthos associated with the invasive red alga *Womersleyella setacea* (Rhodomelacea) in the northern Aegean sea. *Journal of the Marine Biological Association of the United Kingdom* 87: 629-641.

Battershill, C; Miller, K; Cole, R (1998) The understorey of marine invasions. *Seafood New Zealand* 6: 31-33.

Beck, KG; Zimmerman, K; Schardt, JD; Stone, J; Lukens, R R; Reichard, S; Randall J; Cangelosi, A A; Cooper, D; Thompson, J P (2008) Invasive species defined in a policy context: Recommendations from the Federal Invasive Species Advisory Committee. *Invasive Plant Science and Management* 1: 414–421.
Bulleri, F; Abbiati, M; Airoldi, L (2006) The colonisation of human-made structures by the invasive alga *Codium fragile* ssp. *tomentosoides* in the north Adriatic Sea (NE Mediterranean). *Hydrobiologia* 555: 263-269.
Callow, M E (1986a) Fouling algae from in-service ships. *Botanica Marina* 29:351-357.
Callow, M E (1986b) A worldwide survey of slime formation on anti-fouling paints. In Evans, L V; Hoagland, K D (Eds) Algal Biofouling. *Studies in Environmental Science* 28; Elsevier; Amsterdam.
Coutts, A D M (1999) Hull fouling as a modern vector for marine biological invasions: investigation of merchant vessels visiting northern Tasmania. MSc Thesis, Australian Maritime College, Launceston.
Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) Adventive marine species in New Zealand. *NIWA Technical Report* 34.
Finlay, J A; Callow, M E; Schultz, M P; Swain, G W; Callow, J A (2002) Adhesion strength of settled spores of the green alga *Enteromorpha*. *Biofouling* 18: 251-256.
Floerl, O; Inglis, G J; Day, K; Smith, A (2009) The importance of transport hubs in stepping-stone invasions. *Journal of Applied Ecology* 46: 37-45.
Galil, B S; Occhihipinti-Ambrogi, A; Gollasch, S (2008). In: Abdulla, A; Linden, O; (eds). *Maritime traffic effects on biodiversity in the Mediterranean Sea: Review of impacts, priority areas and mitigation measures*. Malaga, Spain: IUCN Centre for Mediterranean Cooperation. 184 pp.
Global Invasive Species Database, 2006. *Caulerpa taxifolia*. Available from: http://www.issg.org/database/species/ecology.asp?fr=1&si=115 [Accessed 4th February 2010].
Gribben, P E; Wright, J T (2006) Sublethal effects on reproduction in native fauna: are females more vulnerable to biological invasion. *Oecologia* 149: 352-361.
Hay, C H (1990) The dispersal of sporophytes of *Undaria pinnatifida* by coastal shipping in New Zealand, and implications for further dispersal of *Undaria* in France. *British Phycological Journal* 25: 301-313.
Hayden, H S; Blomster, J; Maggs, C A, Silva, P C; Stanhope, M J; Walland, R J (2003) Linnaeus was right all along: *Ulva* and *Enteromorpha* are not distinct genera. *Journal of Phycology* 38: 277-294.
Heesch, S; Broom, J; Neill, K; Farr, T; Dalen, J; Nelson, W (2007) *Genetic diversity and possible origins of New Zealand populations of* Ulva. Biosecurity New Zealand Technical Paper 2007/01, MAF Biosecurity New Zealand; Wellington, New Zealand
Heesch, S; Broom, J; Neill, K; Farr, T; Dalen, J; Nelson, W (2009) *Ulva, Umbraulva* and *Gemina*: genetic survey of New Zealand taxa reveals diversity and introduced species. *European Journal of Phycology* 44: 143-154.
Hilliard, R; Polglaze, J; LeProvost, I (2006) *Review and evaluation of the biofouling protocol for vessels less than 25 m in length.* Australian Quarantine and Inspection Service; Canberra, Australia.
Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.
Lowe S; Browne M; Boudjelas S; De Poorter M. (2004) *100 of the World's Worst Invasive Alien Species A selection from the Global Invasive Species Database*. The Invasive

Species Specialist Group, Species Survival Commission, World Conservation Union (IUCN), 12pp.

Mathieson, A C; Dawes, C J; Pederson, J; Gladych, R A; Carlton, J T (2008) The Asian red seaweed *Grateloupia turuturu* (Rhodophyta) invades the Gulf of Maine. *Biological Invasions* 10: 985-988.

Meinesz, A (1999) *Killer algae: the true tale of a biological invasion.* The University of Chicago Press: Chicago, USA. ISBN 0-226-51922-8. XVI, 360 pp.

Mineur, F; Johnson, M P; Maggs, C A; Stegenga, H (2007) Hull fouling on commercial ships as a vector of macroalgal introduction. *Marine Biology* 151:1299-1307.

Mineur, F; Johnson, M P; Maggs, C A (2008 Non-indigenous marine macroalgae in native communities: a case study in the British Isles. *Journal of the Marine Biological Association of the United Kingdom* 88:693-698.

Morrison, M (2008) Tracking snapper origins. *Water and Atmosphere* 16:4.

Murphy, G (2006) *Ecological effects of* Ulva lactuca *L. in Avon-Heathcote estuary*. MSc Thesis, University of Canterbury; Christchurch, New Zealand.

NIMPIS (2010) *Caulerpa taxifolia* impact details. In Hewitt, C L; Martin, R B; Sliwa, C; McEnnulty, F R; Murphy, N E; Jones, T; Cooper, S (Eds) National Introduced Marine Pest Information System. http://crimp.marine.csiro.au/nimpis - accessed 04 February 2010.

Piazzi, L; Balata, D (2009) Invasion of alien macroalgae in different Mediterranean habitats. *Biological Invasions* 11: 193-204.

Railkin, A I (2004) Marine biofouling: colonization processes and defences. CRC Press, Boca Raton.

Ribera Siguan, M A (2003) *Pathways of biological invasions of marine plants.* In Ruiz, G M; Carlton, J T (Eds) Invasive species: Vectors and management strategies. Island Press Washington.

Russell, L K; Hepburn, C D; Hurd, C L; Stuart, M D (2008) The expanding range of *Undaria pinnatifida* in southern New Zealand: distribution, dispersal mechanisms and the invasion of wave-exposed environments. *Biological Invasions* 10: 103-115.

Schaffelke, B; Hewitt, C L (2007) Impacts of introduced seaweeds. *Botanica Marina* 50: 397-417.

Scheibling, R E; Gagnon, P (2006) Competitive interactions between the invasive green alga *Codium fragile* ssp. *tomentosoides* and native canopy-forming seaweeds in Nova Scotia (Canada). *Marine Ecology Progress Series* 325: 1-14.

Schiel, D R (2003) *Kelp beds.* In Andrew, N; Francis, M (Eds) *The living reef: The ecology of New Zealand's rocky reefs*. Craig Potton Publishing; Nelson, New Zealand; pp38-47

Schiel, D R (1990) Macroalgal assemblages in New Zealand: structure, interactions and demography, *Hydrobiologia* 192: 59-76.

Schiel, D R; Hickford, M J (2001) Biological structure of near shore rocky subtidal habitats in southern New Zealand. *Science for Conservation* 182 New Zealand Department of Conservation.

Strong, J A; Dring, M J; Maggs, C A (2006) Colonisation and modification of soft substratum habitats by the invasive macroalgae *Sargassum muticum*. *Marine Ecology Progress Series* 321: 87-97.

Stubbings, H G (1947) Biology of the underwater fouling of ships' hulls. *Journal of Naval Engineering* 1:41-50.

Taylor, DI; Schiel, D R (2005) Self-replacement and community modification by the southern bull kelp *Durvillaea antarctica. Marine Ecology Progress Series* 288: 87–102.

Williams, S L; Smith, J E (2007) A global review of the distribution, taxonomy, and impacts of introduced seaweeds. *Annual Review of Ecology, Evolution, and Systematics* 38: 327-359.

Wright, J T; McKenzie, L A; Gribben, P E (2007) A decline in the abundance and condition of a native bivalve associated with *Caulerpa taxifolia* invasion. *Marine and Freshwater Research* 58: 263-272.

4.13. MICROALGAE

4.13.1. Hazard identification

Microalgae are a taxonomically diverse group of unicellular algae which can be found in the plankton or on the benthos, singularly or in aggregations, as free-living, commensal or symbiotic organisms. As primary producers, microalgae are important components of marine ecosystems providing up to 40% of global primary production (Falkowski 1994). In addition, microalgae link dissolved nutrients to macro-organisms in the zooplankton and on, or in, the benthos. Some microalgae can form blooms with harmful or negative consequences, such as impacts on human health and other values related to the marine environment. In New Zealand *Alexandrium* spp. and *Gymnodinium* spp. have caused outbreaks of shellfish poisoning which have had significant impacts on aquaculture operations, recreational and cultural harvests and human health (MacKenzie et al. 1997, Mackenzie & Beauchamp 2001). However, while toxic bloom events have been relatively recent occurrences in New Zealand it is difficult to determine if the causative species are native or non-indigenous.

Globally it has been difficult to determine if microalgal species are introduced due to taxonomic difficulties and/or a lack of baseline information (Bolch & Reynolds 2002, Lundholm & Moestrup 2006). Those microalgae that have been identified as non-indigenous species in the literature tend to be species detected due to negative bloom events, for example, *Gymnodinium catenatum, Thecadinium yashimaense*, and *Karenia mikimotoi* (Hallegraeff & Bolch 1992, McMinn et al. 1997, Hoppenrath et al. 2007). However, many microalgal species are known to be carried in ballast water (Hallegraeff & Bolch 1992) and a link between shipping and invasion of Tasmanian waters by *G. catenatum* has been shown (McMinn et al. 1997). Whilst all documented non-indigenous microalgae are thought to have been translocated by ballast water or with aquaculture species (Hallegraeff & Gollasch 2006), microalgae are also well known components of biofilms.

Microalgae associated with marine biofilms are typically diatoms of *Navicula* spp. *Achnanthes* spp. *Amphora* spp. and *Nitzschia* spp. (Callow 1986, Anil et al. 2006) which are generally considered innocuous in terms of potential impacts. While species of these genera normally dominate biofilms they are not necessarily the only microalgae fouling the surface. Maso et al. (2003), for example, have identified harmful algal bloom species (dinoflagellates of *Ostreopsis*, *Coolia* and *Alexandrium* spp.) adhering to the surface of plastic debris in the Mediterranean. This finding flags the possibility that hull biofouling has been an overlooked vector for the translocation of microalgal species. Similarly, Maso et al. (2003) found both vegetative and encysted forms of the dinoflagellates of concern adhering to debris. In addition, *G. catenatum* cysts were found to be fouling oyster sticks during the 2000/2001 bloom in the northwest of the North Island of New Zealand (Mackenzie & Beauchamp 2001).

The cyst stages of many harmful algal species are known to be resistant to a variety of stressors, thus cysts adhered to vessel hulls may not be vulnerable to treatments other than physical removal. Cysts do not require light and therefore are not restricted, as are vegetative cells, to areas of the hull near the waterline where shear stress and abrasion are likely to lead to dislodgement. Therefore, cysts could be lodged in niche areas and consequently be protected.

Microalgae have been identified as components of hull biofilms (Callow 1986, Cooksey & Wigglesworth-Cooksey 1995, Callow & Callow 2002, Cassé & Swain 2006). However, the identified species are usually restricted to dominant, commonly identified and reportedly innocuous species. Biofilm species were not targeted in the MAFBNZ commissioned research on biofouling, however the few diatom samples collected were only able to be identified as Bacillariaceae and *Melosirai* (Inglis et al. in prep.). However, given the ubiquitous presence of biofilms on submerged surfaces, it is likely that microalgae are present on the hulls of most vessels arriving in New Zealand.

Given that microalgae:
- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that microalgal species represent a potential hazard via vessel biofouling.

As current anti-fouling system technologies only have the capacity to control biofouling to the biofilm (slime layer) level, the prevention of the entry of non-indigenous microalgae via vessel hull fouling cannot be practically achieved. As a consequence, further research is required to confirm the presence and translocation potential of potentially hazardous microalgae before the development and implementation of biofilm control measures can be justified specifically from the biosecurity perspective.

4.13.2. References

Anil, A C; Patil, J S; Mitbaukar, S; D'Costa, P M; D'Silva, S; Hegde, S; Naik, R (2006) Role of diatoms in marine biofouling. In Tewari, A (Ed) *Recent advances on applied aspects of Indian marine algae with reference to global scenario*. Volume 1. Central Salt and Marine Chemicals Institute. pp. 351-365.

Bolch, C J S; Reynolds, M J (2002) Species resolution and global distribution of microreticulate dinoflagellate cysts. *Journal of Plankton Research* 24: 565-578.

Callow, M E (1986) A worldwide survey of slime formation on anti-fouling paints. In Evans, L V; Hoagland, K D (Eds) Algal Biofouling. *Studies in Environmental Science* 28. Elsevier; Amsterdam, The Netherlands.

Callow, M E; Callow, J A (2002) Marine biofouling: a sticky problem. *Biologist* 49: 1-5.

Cassé, F; Swain, G W (2006) The development of micro-fouling on four commercial anti-fouling coatings under static and dynamic immersion. *International Biodeterioration and Biodegradation* 57: 179-185.

Cooksey, K E; Wigglesworth-Cooksey, B (1995) Adhesion of bacteria and diatoms to surfaces in the sea: a review. *Aquatic Microbial Ecology* 9: 87-96.

Falkowski, P G (1994) The role of phytoplankton photosynthesis in global biogeochemical cycles. *Photosynthesis Research* 39: 235-258.

Hallegraeff, G M; Bolch, C J (1992) Transport of diatom and dinoflagellate resting spores in ships' ballast water: implications for plankton biogeography and aquaculture. *Journal of Plankton Research* 14: 1067-1084.

Hallegraeff, G; Gollasch, S (2006) Anthropogenic introductions of microalgae. In Granéli, E; Turner, T (Eds) *Ecology of Harmful Algae*. Ecological Studies 189. Springer Verlag; Berlin, Germany; pp. 379-390.

Hoppenrath, M; Elbrächter, M; Halliger, H; Koeman, R P T; Krakhmalnyy, A; Surek, B; Erler, K; Luckas, B (2007) First records of the benthic, bloom-forming, non-toxic dinoflagellate *Thecadinium yashimaense* (Dinophyceae) in Europe: with special emphasis on the invasion in the North Sea. *Helgoland Marine Research* 61: 157–165.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international*

vessels arriving in New Zealand. Summary of the patterns and predictors of fouling. Biosecurity New Zealand Technical Paper.
Lundholm, N; Moestrup, Ø (2006) The biogeography of harmful algae. In Granéli, E; Turner, J T (Eds) *Ecology of Harmful Algae*. Ecological Studies Vol 189. Springer-Verlag; Berlin, Germany; pp 23-35.
Mackenzie L; Beauchamp T (2001) *Gymnodinium catenatum* in New Zealand: a new problem for public health and the shellfish industry. Website publication, The Cawthron Institute www.cawthron.org.nz.
MacKenzie, L; Adamson, J; Todd, K (1997) *Alexandrium* spp. and associated PSP-toxin contamination of shellfish on the North Island east coast during 1997. In *Proceedings of the marine biotoxin science workshop No. 8*, MAF Regulatory Authority; Wellington, New Zealand; pp 58-65.
McMinn, A; Hallegraeff, G F; Thomson, P; Jenkinson, A V; Heijnis, H (1997) Cyst and radionucleotide evidence for the recent introduction of the toxic dinoflagellate *Gymnodinium catenatum* into Tasmanian waters. *Marine Ecology Progress Series* 161: 165-172.
Maso, M; Garcés, E; Pagés, F; Camp, J (2003) Drifting plastic debris as a potential vector for dispersing Harmful Algal Bloom (HAB) species. *Scientia Marina* 67: 107-111.

4.14. OTHER ARTHROPODS

4.14.1. Hazard Identification

Phylum Arthropoda consists of two major extant taxa - Chelicerata (e.g. horseshoe crabs, sea spiders, and arachnids), and Mandibulata. Mandibulata contains the classes Crustacea (e.g. crabs, barnacles, water fleas) and the Tracheata (e.g. insects, centipedes, millipedes). These taxa share several characteristics, including; a segmented body, a chitinous exoskeleton with ecdysis (moulting), jointed and paired segmental appendages, and the absence of locomotary cilia (Ruppert et al. 2004). For the purposes of this hazard identification the following groups have been excluded from consideration under the term "other arthropods": the Tracheata as they are primarily terrestrial species and within Crustacea, amphipods and isopods, barnacles and true crabs have been examined separately. The classes within the scope of "other arthropods" include Merostomata, Malacostraca, Ostracoda and Pycnogonida.

The 'other arthropods' have a variety of life-histories. Their feeding and reproductive methods are highly varied and most are relatively small to medium in size.

The majority of invasive species included within 'other arthropods' have freshwater affinities. For example, non-indigenous species of crayfish (Niwa & Ohtaka 2006), small crustaceans such as the tanaid C*ercopagis pengoi,* and the cladoceran *Daphnia lumholtzi*, are from planktonic freshwater/brackish habitats. Due to their habitat restrictions it can be assumed that it is unlikely that these organisms will survive translocation through marine environments via vessel biofouling. Although resting egg stages are known, the risk of their transport can be discounted as they develop in soft sediment and are sensitive to small temperature variations (Sopanen 2008).

There are few records of 'other arthropods' in hull fouling surveys. Groups that have been recorded during hull fouling surveys include the Mysidacea, Tanaidacea, Calanoidea and the Chelicerate groups Pycnogonida and Xiphosura (Cranfield et al. 1998, Cohen et al. 2001, Gollasch 2002, Cardigos et al. 2006, Lewis et al. 2006, Coutts & Dodgshun 2007, Farrapeira et al. 2007). The MAFBNZ commissioned research on biofouling found an indeterminate Tanaidacea on one vessel and the non-indigenous unestablished pycnogonid *Endeis flaccida* on 7 vessels (Inglis et al. in prep.).

Given that "other arthropods":
- are rarely known as components of vessel biofouling assemblages;
- have few known introductions to new marine locations;
- have no known impacts on core values;

it is concluded that "other arthropod species" do not represent a potential hazard via vessel biofouling.

4.14.2. References

Cardigos, F; Tempera, T; Avila, S; Goncalves, J; Colaco, A; Santos, R S (2006) Non-indigenous marine species of the Azores. *Helgoland Marine Research* 60: 160-169.
Cohen, A N; Berry, H; Mills, C; Milne, D; Britton-Simmons, K; Wonham, M; Secord, D; Barkas, J A; Bingham, B; Bookheim, B; Byers, J; Chapman, J W; Cordell, J; Dumbauld, B; Fukuyama, A; Harris, L H; Kohn, A; Li, K; Mumford, T; Radashevsky,

V; Sewell, A; Welch, K (2001) Report of the Washington State Exotics Expedition 2000. Nearshore Habitat Program, Washington State Department of Natural Resources.
Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.
Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) Adventive marine species in New Zealand. NIWA Technical Report 34. 48 pp.
Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship Hull Fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55(3): 207- 221.
Gollasch, S (2002) The importance of ship hull fouling as a vector of species introductions into the North Sea. *Biolfouling* 18(2): 105-121.
Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.
Lewis, P N; Bergstrom, D M; Whinam, J (2006) Barging in: a temperate marine community travels to the subantarctic. *Biological Invasions* 8: 787-795.
Niwa N, Ohtaka A (2006) *Accidental introduction of symbionts with freshwater shrimps*. In Koike, F; Clout, M N; Kawamichi, M; De Poorter, M; Iwatsuki, K (Eds) Assessment and Control of Biological Invasion Risks. Shoukadoh Book Sellers; Kyoto, Japan; and, IUCN; Gland, Switzerland; pp 182-186.
Ruppert, E E; Fox, R S; Barnes, R D (2004) Invertebrate Zoology: a functional evolutionary approach (7^{th} Ed). Brooks/Cole-Thomson Learning, Belmont, USA.
Sopanen, S (2008) The effect of temperature on the development and hatching of resting eggs of non-indigenous predatory cladoceran *Cercopagis pengoi* in the Gulf of Finland, Baltic Sea. *Marine Biology* 154: 99-108.

4.15. PEANUT WORMS (PHYLUM: SIPUNCULA)

4.15.1. Hazard Identification

Peanut worms (sipunculans) are unsegmented benthic worm-like organisms, related to annelids, molluscs and echiurans (Ruppert et al. 2004). There are approximately 370 known sipunculan species that range from the shallow subtidal habitats to the abyssal plains (Murina 1984, Flach and Heip 1996, Ramirez-Llodra et al. 2008). Although the majority of species inhabit protective crevices (the substratum, dead barnacles or echinoderm tests), others bore tubes into limestone, or are sediment dwellers (Ruppert et al. 2004). Peanut worms are normally rare, however, they can in some circumstances be highly abundant, for example, averaging 8,000 individuals per m^2 on coral reefs (Chapman 1955 *In* Murina 1984).

Ecologically, peanut worms feed non-selectively on particulate organic matter from the water column or on the benthos (Ruppert et al. 2004). They have muscular bodies that lack any major calcareous formations, shell, or tube (Murina 1984). In addition to being a food source for Asian populations, due to their unprotected nature peanut worms are prey items for molluscs, actinians, crabs, sea stars, and fish (Kohn 1975 *In* Murina 1984, Nelson 1981 *In* Murina 1984). Peanut worms are proposed to be important bioeroders on coral reefs (Rice and McIntyre 1972 *In* Murina 1984, Rice 1976 *In* Murina 1984, Wood 1999).

The peanut worm life-cycle involves a pelagic larval phase and a sedentary adult phase. Almost all species have separate sexes, although sexual dimorphism is not obvious (Cutler 2009). Gametes are broadcast spawned into the water column where they fertilise, disperse and develop (Cutler 2009).

Penaut worms are not common members of hull fouling communities and there have been no recorded introductions of these organisms. However, two individual sipuncula (identified to phylum level) were recorded in seachests by Coutts and Dodgshun (2007), while in the MAFBNZ commissioned research on biofouling, Inglis et al. (in prep.) identified one sipunculan (identified to phylum level). This lone specimen was sampled from the keel of a recreational vessel and associated with heavy fouling consisting of predominantly barnacles.

The uncertainties and assumptions associated with the introduction and establishment of peanut worms via vessel biofouling include the general poor understanding of sipunculan taxonomy, functional role, life-cycle, and associations in the coastal marine environment (Cutler 2009).

Given that peanut worms:

- are rarely associated with marine biofouling assemblages;
- are not known to have be introduced to new locations;
- have had no known effects on core values;

it is concluded that peanut worms are not considered a potential hazard.

References

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54:875-886.

Cutler, E B (2009) Phylum Sipuncula. In Gordon, D P (ed) *New Zealand Inventory of Biodiversity. Volume 1*: Kingdom Animalia; Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press; Christchurch, New Zealand; pp 302-307.

Flach, E; Hiep, C (1996) Vertical distribution of macrozoobenthos within the sediment on the continental slope of the Goban Spur area (NE Atlantic). *Marine Ecology Progress Series* 141:55-66.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Murina, G V V (1984) Ecology of Sipuncula. *Marine Ecology Progress Series* 17: 1-7.

Ramirez-Llodra, E; Ballesteros, M; Company, J B; Dantart, L; Sarda, F (2008) Spatio-temporal variations in biomass and abundance in bathyal non-crustacean megafauna in the Catalan Sea (North-west Atlantic). *Marine Biology* 153: 297-309.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Smith, L D; Wonham, M J; McCann, L D; Ruiz, G M; Hines, A H; Carlton, J T (1999) Invasion pressure to a ballast-flooded estuary and an assessment of inoculant survival. *Biological Invasions* 1:67-87.

Wood, R A (1999) *Reef Evolution*. Oxford University Press.

4.16. ROUND WORMS (PHYLUM: NEMATODA)

4.16.1. Hazard Identification

Round worms (nematodes), are among the most widespread and abundant animals on earth. They inhabit moist interstitial environments in all habitats, including the bodies of plants and animals. In the marine benthos, free-living round worms are the most abundant animals present (Warwick 1981), for example, Jensen (1987) found 2.9 million individuals per m^2 down to a 20 cm depth.

Ecologically, despite being ubiquitous and abundant benthic meiofauna the role of round worms and their importance to food chains remains unestablished (Heip et al. 1985 *In* Vranken et al. 1988, Herman & Vranken 1988). Free-living round worms feed on a range of organisms and particulate organic matter, although, many are specialised feeders. Marine round worms may be carnivores, herbivores or detritivores, feeding on, for example, other round worms, diatoms, algae, fungi, bacteria, and particulate organic matter (Ruppert et al. 2004).

Reproduction in marine round worms varies with temperature, salinity and quality and quantity of food. Under optimal conditions short-lived species are able to produce tens of generations per year (Tietjen & Lee 1972, Warwick 1981, Vranken et al. 1988). Round worms usually have separate sexes and reproduce by internal fertilisation (Ruppert et al. 2004).

The only identified invasive/introduced round worm recorded is *Anguillicola crassus* which was introduced to Germany and Ireland. However, as an eel parasite, this round worm is not associated with organisms of the fouling community or thought to be vectored by hull fouling (Gollasch & Nehring 2006, Minchin 2007). While parasitic and terrestrial round worms have been recorded as having impacts on organisms and plant commodities, no impacts have been recorded for free-living marine round worms.

Records of round worms on hull fouling are scarce. Records include *Oncholaimus oxyuris* found on container vessels in the North Sea (Gollasch 2002), an unidentified species found in a seachest in New Zealand (Coutts & Dodgshun 2007), and unidentified individuals on fishing, research, tug, pilot and cargo vessels in the Port of Recife (Brazil) (Farrapeira et al. 2007). In the MAFBNZ commissioned research on biofouling, no round worms were recorded on the hulls of vessels sampled entering New Zealand (Inglis et al. in prep.).

Given that round worms:

- are rarely known as components of vessel biofouling assemblages;
- have few known introductions to new locations;
- have no known impacts on core values;

it is concluded that round worm species do not represent a potential hazard via vessel biofouling.

4.16.2. References

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.

Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship Hull Fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55(3): 207- 221.

Gollasch, S (2002) The Importance of Ship Hull Fouling as a Vector of Species Introductions into the North Sea. *Biofouling* 18(2): 105-121.

Gollasch, S; Nehring, S (2006) National checklist for aquatic alien species in Germany. *Aquatic Invasions* 1(4): 245-269.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Herman, P M J; Vranken, G (1988) Studies of the life-history and energetics of marine and brackish-water nematodes. II. Production, respiration and food uptake by *Monhystera disjuncta*. *Oecologia* 77: 457-463.

Jensen, P (1987) Differences in microhabitat, abundance, biomass and body size between oxybiotic and thiobiotic free-living marine nematodes. *Oecologia* 71: 564-567.

Minchin, D (2007) A checklist of alien and cryptogenic aquatic species in Ireland. *Aquatic Invasions* 2(4): 341-366.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Tietjen, J H; Lee, J J (1972) Life-cycles of Marine Nematode – Influence of Temperature and Salinity on the Development of *Monhystera denticulata* Timm. *Oecologia* 10: 167-176.

Vranken, G; Herman, P M J; Heip, C (1988) Studies of the life-history and energetics of marine and brackish-water nematodes. I. Demography of *Monhystera disjuncta* at different temperature and feeding conditions. *Oecologia* 77: 296-301.

Warwick, R M (1981) The Influence of Temperature and Salinity on Energy Partitioning in the Marine Nematode *Diplolaimelloides bruciei*. *Oecologia* 51: 318-325.

4.17. RIBBON WORMS (PHYLUM: NEMERTEA)

4.17.1. Hazard Identification

Ribbon worms (nemerteans) are bilateral unsegmented worm-like organisms. The phylum comprises of approximately 1,500 species, the majority of which occupy marine habitats, however, freshwater and terrestrial species exist (Gibson 1995, Thollesson and Norenburg 2003, Ruppert et al. 2004). Marine ribbon worms are commonly found in sediments and in the crevices among shells, stones, holdfasts of algae and sessile animals (Ruppert et al. 2004). Ribbon worms have few predators as they often contain toxins or noxious chemicals (Carroll et al. 2003).

Ecologically, ribbon worms are ubiquitous, however, their role and importance is undefined. Ribbon worms are carnivores that feed on the soft parts of arthropods, annelids, molluscs, fish and are parasites of crab eggs (Wickham et al. 1984; McDermott & Roe 1985, Torchin et al. 2001, Ruppert et al. 2004). A unique feature of ribbon worms is their long, eversible proboscis that is used in feeding and burrowing (McDermott & Roe 1985, Turbeville & Ruppert 1985, Ruppert et al. 2004). In armed ribbon worms, the proboscis is everted to expose a stylet at the tip from which a toxin is injected to subdue prey (McDermott & Roe 1985, Ruppert et al. 2004). The discharged proboscis of unarmed ribbon worms is used to immobilise prey by the release of sticky toxic secretions as it is coiled around them (Ruppert et al. 2004).

Reproduction in ribbon worms can be either sexual or asexual. Most ribbon worm species have separate sexes, aggregate before spawning and fertilise externally (Thiel & Junoy 2006). Fertilisation may involve free spawning or mucous bundles, however, some species undergo internal fertilisation (Thiel & Junoy 2006). Many ribbon worms, especially large-bodied species, fragment when disturbed and subsequently regenerate (Ruppert et al. 2004).

Ribbon worms are not common components of hull fouling communities and there have been no reported invasive/introduced ribbon worm or recorded impacts on core values. Ribbon worms have been recorded on the hulls of various vessels in Brazil (Farrapeira et al. 2007) and in seachests of vessels in New Zealand (Coutts & Dodgshun 2007). Ribbon worm larvae were also found in the ballast and cargo tanks of container ships in North America (Smith et al. 1999). However, no ribbon worms were recorded in the MAFBNZ commissioned research on biofouling (Inglis et al. in prep.).

Given that ribbon worms:

- are rarely known as components of vessel biofouling assemblages;
- have few known introductions to new locations;
- have no known impacts on core values;

it is concluded that ribbon worm species do not represent a potential hazard via vessel biofouling.

4.17.2. References

Carroll, S; McEvoy, E G; Gibson, R (2003) The production of tetrodotoxin-like substances by nemertean worms in conjunction with bacteria. *Journal of Experimental Marine Biology and Ecology* 288: 51-63.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.
Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship Hull Fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55(3):207- 221.
Gibson, R (1995) Nemertean genera and species of the world: an annotated checklist of original names and description citations, synonyms, current taxonomic status, habitats and recorded zoogeographic distribution. *Journal of Natural History* 29(2): 271-561.
Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.
McDermott, J J; Roe, P (1985) Food, Feeding Behavior and Feeding Ecology of Nemerteans. *American Zoologist* 25: 113-125.
Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.
Smith, L D; Wonham, M J; McCann, L D; Ruiz, G M; Hines, A H; Carlton, J T (1999) Invasion pressure to a ballast-flooded estuary and an assessment of inoculant survival. *Biological Invasions* 1: 67-87.
Thiel, M; Junoy, J (2006) Mating behavior of nemerteans: present knowledge and future directions. *Journal of Natural History* 40(15-16): 1021-1034.
Thollesson, M; Norenburg, J L (2003) Ribbon worm relationships: a phylogeny of the phylum Nemertea. *Proceedings of the Royal Society of London Series B* 270: 407-415.
Torchin, M E; Lafferty, K D; Kuris, A M (2001) Release from parasites as natural enemies: increased performance of a globally introduced crab. *Biological Invasions* 3: 333-345.
Turbeville, J M; Ruppert, E E (1985) Comparative ultrastructure and the evolution of Nemertines. *American Zoologist* 25: 53-71.
Wickham, D E; Roe, P; Kuris, A M (1984) Transfer of nemertean egg predators during host molting and copulation. *Biological Bulletin* 167: 331-338.

4.18. SEA ANEMONES AND CORALS (CLASS: ANTHOZOA)

4.18.1. Hazard Identification

Anthozoa is the largest class within phylum Cnidaria, containing over 6,000 species of exclusively marine anemones and corals (Ruppert et al. 2004). This class is split into several orders, including sea anemones (Actiniaria), hard corals (Scleractinia), and black corals (Antipatharia) (Species 2000). Colonial corals, solitary corals and sea anemones are ubiquitous in the marine environment ranging from shallow tropical seas to deep polar sea-beds (Cairns 1995, Freiwald & Roberts 2005).

Unlike the other cnidarians, the anthozoan life-cycle does not have a medusa lifestage, therefore, the polyp alone undergoes sexual and asexual reproduction (Ruppert al. 2004). Most corals and some sea anemones form genetically identical clones through budding of new polyps (asexual reproduction). Individual polyps may remain attached to each other to form a colonial organism, or separate to form individual organisms (Karlson 1986, Ting & Geller 2000). As a result, colonies are essentially a single genetic organism that can live for a hundred years or more (Dahan & Benayahu 1997). Species that broadcast spawn are often observed in stable, benign environments, while in more variable environments species that brood are more common (Szmant 1986 *In* Vermeij et al. 2007).

Ecologically, corals and sea anemones play important roles as benthic suspension feeders, space occupiers, and habitat formers. Tentacles on the polyp extend into the water column or along the substratum and capture food using stinging cells. Depending on the species, food can range from plankton and particulate organic matter to crabs and fish (Anthony 1999, Smith & Gordon 2003, Lira et al. 2008). Some shallow water anthozoans harbour photosynthetic symbionts, typically zooxanthellae, which exude carbohydrates that are utilised by the host (Falkowski et al. 1984, Hoegh-Guldberg 1999). Most shallow water anthozoans participate in both phototrophic and heterotrophic feeding to varying degrees (Anthony & Fabricius 2000). Corals and their calcium carbonate skeletons provide habitat for many organisms and support commercial, subsistence and recreational fisheries in both shallow and deep waters (Bellwood et al. 2004, Roberts et al. 2005, Stone 2006).

Known invasive anthozoa include *Carijoa riisei*, *Tubastrea coccinea*, *Haliplanella lineata*, and *Oculina patagonica*. Typical invasive life history characteristics exhibited include high tolerance towards abiotic factors, high growth rate, early maturity, localised recruitment of larvae, and high asexual reproductive output (Gollasch & Reimann-Zurneck 1996, Fine et al. 2001, Glynn et al. 2008, Kahng et al. 2008, Lira et al. 2008, Rodolfo-Metalpa et al. 2008b). In addition, nuisance anthozoan species have, to some degree, similar aspects of their life history. For example, budding/monoclonal stands can be seen in sea anemones such as *Nematostella vectensis* in the northern hemisphere (Reitzel et al. 2008) and *Actinia tenebrosa* in the southern hemisphere (Ayre 1984).

Given that corals and sea anemones:

- are known components of marine biofouling assemblages;
- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that coral and sea anemone species represent a potential hazard via vessel biofouling.

4.18.2. Risk Assessment

4.18.2.1. Entry assessment

Corals and sea anemones are components of benthic fouling assemblages globally. Some anthozoan species, such as *H. lineata* and *C. riisei*, are found amongst port and harbour fouling in close proximity to commercial and recreational vessels (Elredge & Smith 2001, Grigg 2003).

Corals and sea anemones are known to be associated with vessel biofouling. In general, these organisms constitute a minor component of the hull fouling assemblages surveyed (Zibrowius 1992, Lewis 2004), however, they have been found on all vessel types (Gollasch & Reimann-Zurneck 1996, Fenner 2001, Gollasch 2002, Godwin 2003, de Paula & Creed 2004, Coutts & Dodgshun 2007, Creed & de Paula 2007, Farrapeira et al. 2007). It has been suggested that anthozoa recruit to substrata that have been submerged from between four months to a year (Gollasch & Reimann-Zurneck 1996, Creed & de Paula 2007, Farrapeira et al. 2007). While this may be applicable to corals, the wandering nature of some sea anemones, their pedal disc and heavy mucous may enable attachment to clean hulls. For example, in the MAFBNZ commissioned research on biofouling, Inglis et al. (in prep.) found two unidentifiable and two local species of sea anemone on the relatively clean hulls of recreational vessels.

Corals appear to be associated with vessels that have extended periods of inactivity. The soft tissue of corals is susceptible to the abrasion caused by macro-fouling and sediment in areas of high water movement. As a result oil rigs and barges are the likely vessels to present suitable substrates for coral colonisation. This appears to be the most likely scenario for the majority of *Tubastrea coccinea* introductions and in the case of the introduction of *Balanophyllia elegans* to Hawai'i. Recreational and fishing vessels have been implicated in the introduction of *Oculina patagonica* to Algeria (Sartotetto et al. 2008), however, there is a high likelihood that these vessels have been subject to poor maintenance regimes and have spent extended periods idle. These factors increase the likelihood of coral settlement and maturation on a vessel hull. However, temperate corals are unlikely to be associated with vessel hull fouling as they most frequently inhabit continental slopes, plateux, rises and seamounts. They are rare in ports and on coastlines due to factors related to these environments, such as abrasion, and their life-histories.

Uncertainty exists with regard to how anti-fouling systems affect corals and sea anemones. It is assumed that coral and sea anemone larvae are not likely to survive settlement on bare anti-fouling paint. Therefore the most probable locations for successful attachment are on niche areas that are already fouled. The pedal disc and mucous created by adult sea anemones may afford some protection from anti-fouling biocides. Therefore, if an individual were to settle on an exposed hull area, it may be able to migrate to a niche area.

Coral and sea anemones are known to survive translocation. Some translocations have been recorded across relatively short distances, for example, the translocation of *Sagartiogeton undatus* from the United Kingdom and Ireland to Germany (Gollasch 2002). However, corals and sea anemones have been translocated over large distances, for example, the sea anemone *Aiptasia pallida* was translocated from its native range in the Carribean and eastern USA to the Port of Recife on the eastern tip of Brazil (Farrapeira 2007). Other examples of long distance translocations include the coral *B. elegans* from California to Hawai'i (Hellberg 1995, Godwin 2003), and the sea anemone *H. luciae* being found on a vessel hull in Germany (Gollasch and Reimann-Zurneck 1996).

The MAFBNZ commissioned research on biofouling found five sea anemones on vessels entering New Zealand (Inglis et al. in prep.). This research included the first record of the unestablished tropical sea anemone *Anemonia* cf *manjano* (Kospartov et al. in press). The four other native species identified were *Anthopleura aureoradiata*, *Anthothoe albocincta*, *Diadume neozelanica*, *Oulactis mucosa*.

Given that particular coral and sea anemone species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous coral and sea anemone species into New Zealand via vessel biofouling is non-negligible.

4.18.2.2. Establisment assessment

There are known suitable environments and habitats in New Zealand for the establishment of non-indigenous species of corals and sea anemones as these organisms are found throughout New Zealand from coastal waters to the abyssal depths (Cairns 1995). Given that some non-indigenous hull fouling corals and sea anemones are associated with port and harbour environments, it is likely that they will find similar habitats upon entry into New Zealand.

Corals and sea anemones are able to establish through fragmentation or the release of adults or larvae. Corals may become dislodged from hull fouling. However most corals, especially azooxanthellate species, grow at rates of millimetres per year, therefore, for a sizable fragment to be produced the organism would likely be several years old. As a result, dislodgement of fragments large enough to survive and establish are unlikely to occur from a well maintained vessel.

The sea anemone, *H. lineata,* has been translocated from its native Japan to numerous locations in both hemispheres (Molina et al. 2009). *H. lineata* has many characteristics that have contributed to its success as an invasive species, including, a propensity to bud, a high tolerance to physical conditions and the ability to detach from substrata and float on the water surface (Ayre 1984, Gollasch & Reimann-Zurneck 1996, Ting & Geller 2000, Reitzel et al. 2008). Detachment of sea anemones is likely a response to unfavourable conditions, such as, disturbances or changes to physical conditions or spatial congestion. *H. lineata* is the only non-indigenous sea anemone to known to have established in New Zealand.

There are no recent records of coral establishment in New Zealand. The two non-indigenous coral species, *Hoplangia durotrix* and *Tethocyathus cylindraceus,* established in New Zealand were introduced in the 1940's, are restricted to the north/north eastern North Island and have had no recorded impacts (Cranfield et al. 1998). The native distributions of these corals are north-east Atlantic and western Atlantic, respectively (Kospartov et al. in press). Tropical and subtropical coral species are restricted to the northern parts of New Zealand due to the presence of warmer currents (Cairns 1995, Laing & Chiswell 2003). Seasonal variations become more pronounced with increasing latitude, in addition, there is reduced irradiance and a greater supply of inorganic nutrients, particulate food (Muller-Parker & Davy 2001) and increased disturbance. These factors lead to decreased metabolic rates in tropical species and increased competition from macroalgae and other fouling organisms (Veron 1995, Fine et al. 2001, Rodolfo-Metalpa et al. 2008a). Therefore, corals adapted to temperate regions are slow-growing azooxanthellate, and are often restricted to sheltered depths and locations.

Given that particular coral and sea anemone species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous coral and sea anemone species in New Zealand via vessel biofouling is non-negligible.

4.18.2.3. Consequence assessment

The invasive *C. riisei* was first recorded in Hawai'i in 1972 (Evans et al. 1974 In Kahng & Grigg 2005), at this time it was considered to be an environmentally and economically benign species. However, *C. riisei* have smothered native black coral colonies *Antipathes* spp.. This has had negative impacts upon the US$30 million per annum black coral jewellery trade (Grigg 2003, Grigg 2004). With regard to the protection of New Zealand native black coral, especially those that grow in the Kermadec Islands and the shallow depths of Fjordland, temperature appears to be a significant protective factor as *C. riisei* does not thrive below 23°C (Kahng & Grigg 2005).

T. coccinea, native to the Indo-Pacific (Glynn et al. 2008), is commonly regarded as one of the most invasive corals and has spread through the tropical western Atlantic (Fenner 2001, de Paula & Creed 2004, Creed & de Paula 2007). In New Zealand this coral is only found in the Kermadec Islands (corals are common but do not form coral reefs) suggesting that, as elucidated in the establishment assessment, corals are not suited to temperate coastal environments. The Kermadec Islands are isolated and highly exposed which influences organism and larval recruitment. The corals on these islands, are few in species, do not form reefs, and cover only 20 to 40% of substrate (Francis & Nelson 2003). It is likely that the sub-optimal conditions for *T. coccinea* growth and competition from other species will buffer the impacts of any non-indigenous coral introductions.

As a result of the information reviewed, the likelihood of tropical or sub-tropical corals establishing and exhibiting significant consequences to New Zealand's core values is considered low.

Despite occasional high abundances of invasive sea anemones, there is a deficiency of ecological studies regarding their impacts (Elridge & Smith 2001, Molina et al. 2009, Inglis et al. in press). The sea anemone, *H. lineata,* was introduced to New Zealand prior to 1983 and has since established populations in the Waitemata Harbour, Motueka and Lyttleton Harbour (Cranfield et al. 1998, Inglis et al. in press). At present, there have been no reported impacts attributed to this invasion (Inglis et al. in press). For example, no significant impacts on the indigenous anemone *A. aureoradiata* has been recorded since the introduction of *H. lineata,* as these species have been observed to occur in the same habitats and attached to similar substrata (Molina et al. 2009, Phillips *pers. obs.*). Due to the prevalence of native sea anemone species and the lack of recorded impacts of invasive sea anemones, it is deemed unlikely that non-indigenous anemones will have significant consequences on native populations.

Contact with stinging sea anemones can have significant consequences to human health. For example, the toxins of the blue striped anemone can cause severe reactions such as urticaria, haemorrhage, ulceration or a chronic rash (Rucks & Heasman 2006). A likely mode of contact between humans and sea anemones would be during the clearance of fouling organisms from mussel buoys. However, the consequences of contact with sea anemones in this manner can be managed by the wearing of personal protective equipment (Rucks & Heasman 2006). It is assumed that the introduction of non-indigenous stinging sea anemones is not common given the lack of literature regarding impacts to human health. Therefore, it is unlikely that the establishment of common non-indigenous sea anemone species will lead to increased human health concerns.

4.18.3. Risk Estimation

Given that particular coral species:

- are not common components of vessel hull biofouling assemblages;
- are only associated with fouling on inactive, unmaintained vessels;
- are unlikely to establish in the majority of New Zealand's marine environments;
- are few in the number of introductions and invasive species recorded;
- have not been recently recorded as introduced to New Zealand;
- are not recorded as being responsible for impacts to core values relevant to New Zealand;

it is concluded that the import of non-indigenous coral species via biofouling poses a negligible risk to New Zealand's marine core values.

Given that particular sea anemone species:

- are not common components of vessel hull biofouling assemblages;
- have few establishments recorded in New Zealand;
- are not widely recorded as being responsible for impacts to core values;

it is concluded that the import of non-indigenous sea anemone species via biofouling poses a negligible risk to New Zealand's marine core values.

4.18.4. References

Anthony, K R N (1999) Coral suspension feeding on fine particulate matter. *Journal of Experimental Marine Biology and Ecology* 232: 85-106.

Anthony, K R N; Fabricius, K E (2000) Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. *Journal of Experimental Marine Biology and Ecology* 252: 221-253.

Ayre, D J (1984) The effects of sexual and asexual reproduction on geographic variation in the sea anemone *Actinia tenebrosa*. *Oecologia*. 62: 222-229.

Barnes, D J; Chalker, B E (1990) Calcification and photosynthesis in reef-building corals and algae. In Dubinsky Z (Ed) *Coral Reefs Ecosystems of the World*. Elsevier; New York, USA; pp 75-87.

Batson, P (2003) *Deep New Zealand: Blue water, black abyss*. Canterbury University Press. Christchurch, New Zealand.

Bellwood, D R; Hughes, T P; Folke, C; Nystrom, M (2004) Confronting the coral reef crisis. *Nature* 429: 827-833.

Cairns, S D (1995) The Marine Fauna of New Zealand: Scleractinia (Cnidaria: Anthozoa). *New Zealand Oceanographic Institute Memoir* 103: 139 pp; 44 pls.

Cairns, S D (2000) A revision of the shallow-water azooxanthellate Scleractinia of the western Atlantic. *Studies in Natural History of the Caribbean Region.* 75: 1-240.

Coles, S; Elridge, L (2002) Non-indigenous species introductions on coral reefs: a need for information. *Pacific Science* 56: 191-209.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in sea-vessels: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886.

Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) Adventive marine species in New Zealand. NIWA Technical Report 34. 48 p.

Creed, J C (2006) Two invasive alien azooxanthellate corals, *Tubastrea coccinea* and *Tubastrea tagusensis*, dominate the native zooxanthellate *Mussismilia hispida* in Brazil. *Coral Reefs*. 25: 350.

Creed, J C; de Paula, A F (2007) Substratum preference during recruitment of two invasive alien corals onto shallow-subtidal tropical rocky shores. *Marine Ecology Progress Series* 330: 101-111.

Dahan, M; Benayahu, Y (1997) Clonal propagation by the azooxanthellate octocoral *Dendronephthya hemprichi*. *Coral Reefs* 16: 5-12.

de Paula, A F; Creed, J C (2004) Two species of the coral *Tubastrea* (Cnidaria, Scleractinia) in Brazil: a case of accidental introduction. *Bulletin of Marine Science*. 74(1): 175-183.

Eldredge, L G; Smith, C M (Eds)(2001) *A Guidebook of Introduced Marine Species in Hawai'i*. Bishop Museum Technical Report 21.

Falkowski, P G; Dubinsky, Z; Muscatine, L; Porter, J W (1984) Light and the bioenergetics of a symbiotic coral. *Bioscience* 34: 705-709.

Farrapeira, C M R; Melo, A V de O M de; Barbosa, D F; Silva, K M E da (2007) Ship Hull Fouling in the Port of Recife, Pernambuco. *Brazilian Journal of Oceanography* 55(3):207- 221.
Fenner, D (2001) Biogeography of three Caribbean corals (Scleractinia) and the invasion of *Tubastrea coccinea* into the Gulf of Mexico. *Bulletin of Marine Science*. 69(3): 1175-1189.
Fine, M; Loya, Y (1995) the coral *Oculina patagonica* a new immigrant to the Mediterranean coast of Israel. *Israeli Journal of Zoology* 41: 84.
Fine, M; Zibrowius, H; Loya, Y (2001) *Oculina patagonica*: a non-lessepian scleractinian coral invading the Mediterranean Sea. *Marine Biology*. 138: 1195-1203.
Francis, M; Nelson, W (2003) In Andrew, N; Francis, M (Eds) *The living reef: The ecology of New Zealand's rocky reefs*. Craig Potton Publishing; Nelson, New Zealand; pp192-201.
Freiwald, A; Roberts, J M (Eds) (2005) *Cold-water Corals and Ecosystems*. Springer-Verlag; Berlin Heidelberg, Germany; pp 467-479.
Fukui, Y (1991) Embryonic and larval development of the sea anemone *Haliplanella lineate* from Japan. *Hydrobiologia*. 216/217: 137-142.
Glynn, P W; Colley, S B; Mate, J L; Cortes, J; Guzman, H M; Bailey, R L; Feingold, J S; Enochs, I C (2008) Reproductive ecology of the azooxanthellate coral *Tubastrea coccinea* in the Equatorial Eastern Pacific: Part V. Dendrophylliidae. *Marine Biology*. 153: 529-544.
Godwin, L S (2003) Hull Fouling of Maritime Vessels as a Pathway for Marine Species Invasions to the Hawai'ian Islands. *Biofouling* 19: 123-131.
Goffredo, S; Caroselli, E; Mattioli, G; Pignotti, E; Zaccanti, F (2008) Relationships between growth, population structure and sea surface temperature in the temperate solitary coral *Balanophyllia europaea* (Scleractinia, Dendrophlliidae). *Coral Reefs* 27: 623-632.
Gollasch, S (2002) The Importance of Ship Hull Fouling as a Vector of Species Introductions into the North Sea. *Biofouling* 18(2): 105-121
Gollasch, S; Reimann-Zurneck, K (1996) Transoceanic dispersal of benthic macrofauna: *Haliplanella luciae* (Verrill, 1898) (Anthozoa, Actinaria) found on a ship's hull in a shipyard dock in Hamburg Harbour, Germany. *Helgolander Meeresunters*. 50: 253-258.
Gordon, D; Smith, F (2007) Corals, Medusae, and Hydroids (Phylum Cnidaria). In MacDiarmid A (Ed) *The Treasures of the Sea: Nga Taonga a Tangaroa. A summary of the Biodiversity in the New Zealand Marine Ecoregion*. WWF; Wellington, New Zealand; pp 105-112.
Grigg, R W (2003) Invasion of a deep black coral bed by an alien species, *Carijoa riisei,* off Maui, Hawai'i. *Coral Reefs* 23: 121-122.
Grigg, R W (2004) Harvesting Impacts and Invasion by an Alien Species Decrease estimates of Black Coral Yield off Maui, Hawai'i. *Pacific Science* 58(1): 1-6.
Hamoutene, D; Puestow, T; Miller-Banoub, J; Wareham, V (2008) Main lipid classes in some species of deep sea corals in the Newfoundland and Labrador region (Northwest Atlantic Ocean). *Coral Reefs* 27: 237-246.
Hellberg, M E (1995) Stepping-stone gene flow in the solitary coral *Balanophyllia elegans*: equilibrium and nonequilibrium at different spatial scales. *Marine Biology* 123: 573-581.
Hoegh-Guldberg, O (1999) Climate change, coral bleaching and the future of the world's coral reefs. *Marine and Freshwater Resear*ch 50: 839-866.
Huston, M (1985) Variation in coral growth rates with depth at Discovery Bay, Jamaica. *Coral Reefs* 4: 19-25.
Inglis, G; Gust, N; Fitridge, I; Floerl, O; Woods, C; Hayden, B; Fenwick, G (in press) Baseline survey for non-indigenous marine species: Port of Lyttelton. Biosecurity New Zealand Technical Paper No. 2005/01.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kahng, S E; Benyahu, Y; Wagner, D; Rothe, N (2008) Sexual reproduction in the invasive octocoral *Carijoa riisei* in Hawai'i. *Bulletin of Marine Science*. 82(1): 1-17.

Kahng, S E; Grigg, R W (2005) Impact of an alien octocoral, *Carijoa riisei*, on black corals in Hawai'i. *Coral Reefs*. 24: 556-562.

Karlson, R H (1986) Disturbance, colonial fragmentation, and size-dependant life history variation in two coral reef cnidarians. *Marine Ecology Progress Series*. 28: 245-249.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Laing, A; Chiswell, S (2003) The Ocean Medium. In Andrew N; Francis M (2003) *The Living Reef – the ecology of New Zealand's rocky reefs*. Craig Potton Publishing; Nelson, New Zealand; pp 24-31.

Lewis, J (2004) *Hull fouling as a vector for the translocation of marine organisms: Report 1: Hull fouling research*. Ballast Water Research Series Report No. 14. Dept of Agriculture, Fisheries and Forestry; Canberra, Australia; 129 pp.

Linares, C; Coma, R; Zabala, M (2008) Effects of a mass mortality event on gorgonian reproduction. *Coral Reefs*. 27: 27-34.

Lira, A K F; Naud, J-P; Gomes, P B, Santos, A M, Perez, C D (2009) Trophic ecology of the octocoral Carijoa riisei from littoral of Pernambuco, Brazil. I. Composition and spatio-temporal variation of the diet. *Journal of the Marine Biological Association of the United Kingdom* 89(1): 89-99.

Marshall, A T; Clode, P (2004) Calcification rate and the effect of temperature in a zooxanthellate and azooxanthellate scleractinian reef coral. *Coral Reefs* 23: 218-224.

Molina, L M; Valinas, M S; Pratolongo, P D; Elias, R; Perillo, G M E (2009) First record of the sea anemone *Diadumene lineata* (Verrill 1871) associated to *Spartina alterniflora* roots and stems, in marshes at the Bahia Blanca estuary, Argentina. *Biological Invasions* 11: 409-416.

Muller-Parker, G; Davy, S K (2001) Temperate and tropical algal-sea anemone symbioses. *Invertebrate Biology* 120: 104-123.

Pearse, V B; Muscatine, L (1971) Role of symbiotic algae (zooxanthellae) in coral calcification. *Biological Bulletin* 141: 350-363.

Reitzel, A M; Darling, J A; Sullican, J C; Finnerty, J R (2008) Global population genetic structure of the starlet anemone Nematostella vectensis: multiple introductions and implications for conservation policy. *Biological Invasions*. 10: 1197-1213.

Roberts, J M; Brown, C J; Long, D; Bates, C R (2005) Acoustic mapping using a multibeam echosounder reveals cold-water coral reefs and surrounding habitats. *Coral Reefs* 24: 654-669.

Rodolfo-Metalpa, R; Peirano, A; Houlbreque, F; Abbate, M; Ferrier-Pages, C (2008a) Effects of temperature, light and heterotrophy on the growth rate and budding of the temperate coral *Cladocora caespitose*. *Coral Reefs* 27: 17-25.

Rodolfo-Metalpa, R; Reynaud, S; Allemand, D; Ferrier-Pages, C (2008b) Temporal and depth responses of two temperate corals, *Cladocora caespitosa* and *Oculina patagonica*, from the North Mediterranean Sea. *Marine Ecology Progress Series*. 369:103-114.

Rucks, B; Heasman, K (2006) Blue Striped Sea Anemone: a safety issue for marine farms. New Zealand Aquaculture Magazine March/April 2006. http://www.nzaquaculture.co.nz/. Accessed 16 January 2010.

Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.

Sartoretto, S; Harmelin, J-G; Bachet, F; Bejaoui, N; Lebrun, O; Zibrowius, H (2008) The alien coral *Oculina patagonica* De Angelis, 1908 (Cnidaria, Scleractinia) in Algeria and Tunisia. *Aquatic Invasions* 3(2): 173-180.

Smith, F; Gordon, D (2003) Sessile Invertebrates. In Andrew N; Francis M (2003) *The Living Reef – the ecology of New Zealand's rocky reefs*. Craig Potton Publishing; Nelson, New Zealand; pp 80-91.

Species 2000. *Catalogue of Life: Dynamic Checklist*. http://www.catalogueoflife.org/dynamic-checklist/browse_taxa.php. Accessed 3 March 2009.

Stone, R P (2006) Coral habitat in the Aleutian Islands of Alaska: depth distribution, fine scale species associations, and fisheries interactions. *Coral Reefs* 25: 229-238.

Ting, J H; Geller, J B (2000) Clonal diversity in introduced populations of an Asian sea anemone in North America. *Biological Invasions*. 2: 23-32.

Vermeij, M J A; Frade, P R; Jacinto, R I R; Debrot, A O; Bak, R P M (2007) Effects of reproductive mode on habitat-related differences in population structure of eight Caribbean coral species. *Marine Ecology Progress Series* 351: 91-102.

Veron, J E N (1986) *Corals of Australia and the Indo-Pacific*. Angus and Robertson; North Ryde, Australia.

Veron, J E N (1995) *Corals in space and time: the biogeography and evolution of the scleractinia*. UNSW Press; Sydney, Australia.

Ward, S (1995) Two patterns of energy allocation for growth, reproduction and lipid storage in the scleractinian coral *Pocilopora damicornis*. *Coral Reefs* 14: 87-90.

Zibrowius, H (1992) Ongoing modification of the Mediterranean fauna and flora by the establishment of exotic species. *Mesogee.* 51: 83-107.

4.19. SEA SQUIRTS (CLASS: ASCIDIACEA)

4.19.1. Hazard Identification

Sea squirts (Ascidians) are sessile, filter-feeding organisms that have a global marine distribution (Millar 1969). Sea squirts are classified within phylum Chordata as their tadpole-like larvae possess a notochord (Ruppert et al. 2004). The generalised sea squirt is a filter feeder. Water is drawn in the inhalant siphon and over a "branchial basket" where food is trapped within a mucous net. The filtered water is then expelled via the exhalant siphon (Petersen 2007). The majority of sea squirts occur in shallow, sheltered waters attached to hard natural and artificial substrata, however, soft sediment dwellers and pelagic forms (salps) also exist (Lambert and Lambert 2003, Ruppert et al. 2004, Davis and Davis 2005).

The general sea squirt life-cycle involves a short-lived planktonic larval phase and a sessile adult phase. Adult sea squirts may be solitary or colonial. While solitary forms each have their own test (protective covering), colonial forms may share the same test (Millar 1969, Ruppert et al. 2004). In addition, some colonies share a common exhalant opening (Millar 1969, Ruppert et al. 2004). Solitary sea squirts are usually spherical or cylindrical, while colonial sea squirts often form mats or hanging aggregations. Colonial sea squirts can often survive fragmentation and form new colonies (Bullard et al. 2007a).

Sea squirts are hermaphroditic and in general, solitary species are broadcast spawners while colonial species fertilise eggs internally (Lambert 2005, Kott and Page 2007). Sea squirt larvae are non-feeding, tadpole-like and free-swimming. The larvae can swim for a few hours before settlement on an appropriate substrate where they metamorphose into the sessile adult form (Lambert 2005, Kott and Page 2007). Sea squirt dispersal is very limited under natural conditions due to the short time the larvae spend in the water column.

Ecologically, sea squirts are important suspension feeders, capable of filtering vast quantities of water for the removal of plankton and organic material (Kott et al. 2009). Sea squirts are also major space occupiers on the benthos and in the plankton (Kott et al. 2009). Their ability to bud enables them to rapidly cover available space on the ground or in the water column (Kott et al. 2009). Sea squirts have been recorded from the tropics to the sub-Antarctic (Primo and Vazquez 2008).

There are many instances of sea squirt introductions, particularly in North America and Europe. Impacts include decreased local biodiversity (Blum et al. 2007) and negative effects on fisheries and aquaculture (Thompson and MacNair 2004, Davis and Davis 2006, Howes et al. 2007, Locke et al. 2007, Valentine et al. 2007b). Many non-indigenous sea squirts experience competitor and predator release in introduced areas, allowing significant aggregations to develop (Locke et al. 2007, Osman and Whitlatch 2007). They often become gregarious during summer, as temperatures and nutrients increase, and can smother sessile benthic structures and organisms (Bullard et al. 2007b, Howes et al. 2007). Well known invasive species include *Botrylloides violaceus*, *Diplosoma listerianum*, *Stylea clava*, *Ascidiella aspersa*, *Didemnum* sp, *Botryllus schlosseri*, *Ciona intesinalis*. In New Zealand the introduced *S. clava*, *D. vexillium*, *A. aspersa*, and *Eudistoma elongatum* have become nuisance or pest species. Non-indigenous sea squirts also facilitate the introduction of their epibionts, such as amphipods (Howey 1998, Lambert & Lambert 1998, Wyatt et al. 2005).

Given that sea squirts:

- are known components of marine biofouling assemblages;

- are known to have been introduced to new locations;
- have known impacts on core values;

it is concluded that sea squirt species represent a potential hazard via vessel biofouling.

4.19.2. Risk Assessment

4.19.2.1. Entry assessment

Sea squirts are known to occur on entry pathways into New Zealand. The conditions found in port and marina environments provide highly suitable habitat for sea squirts including the provision of hard substrates in shallow water and sheltered and disturbed environments (Locke et al. 2007, Tyrrell and Byers 2007). Sea squirts have very limited natural dispersal ability, however, the increase of artificial structures such as aquaculture equipment and ocean going vessels has resulted in the dispersal of sea squirts far beyond their natural boundaries (Lambert 2005, Minchin and Sides 2006).

Sea squirts are known to have been translocated beyond their natural boundaries via hull fouling (Lambert 2005, Wyatt et al. 2005, Fuller 2007). Recreational vessels have been reported to have spread sea squirts, for example *Didemnum* spp. to Ireland (Minchin and Sides 2006), *S. clava* to Ireland and the German Bight (Minchin et al. 2006, Krone et al. 2007), and an Atlantic clade of *Clavelina lepadiformis* to the Mediterranean (Turon et al. 2003). In the MAFBNZ commissioned research on biofouling, sea squirts were present on 5% of vessels sampled (Inglis et al. in prep.). Four non-indigenous species were recorded by Inglis et al. (in prep.), two of which are not established in New Zealand.

Sea squirts appear to be sensitive to anti-fouling systems. Hull fouling sea squirts often appear to be secondary foulers requiring the presence of other organisms to act either as a buffer, between them and the biocides, or as a stimulant for larval settlement (Edmondson 1944, Wood & Allen 1958). Typical of fouling species, sea squirts take advantage of failures anti-fouling systems, and as a result are often found in niche areas (Edmondson 1944, Wood & Allen 1958). Lambert et al. (2005) states that the adult stages of a number of species are tolerant to heavy metals.

The ability of sea squirts to survive translocation is perhaps facilitated by their protective tests and mucous formation. In general, sea squirts are not found on the hulls of fast moving vessels, however, they may survive in sheltered locations or on low speed voyages (Davis and Davis 2004, Coutts et al. in press). Sea squirts may have relatively high attachment strengths, for example, Coutts et al. (in press) found *Asterocarpa humilis* remained attached at speeds of up to 18 knots. The ability of small colonial fragments to rapidly regrow means that, for certain species, the whole colony need not survive translocation.

Given that particular sea squirt species:

- are known to occur along vessel pathways;
- are known to be associated with vessel biofouling;
- are known to survive translocation;

it is concluded that the likelihood of entry of non-indigenous sea squirt species into New Zealand via vessel biofouling is non-negligible.

4.19.2.2. Establishment assessment

There are known suitable environments and habitats in New Zealand for the establishment of sea squirts. Most hull fouling sea squirts are adapted to port/marina conditions and have a relatively high tolerance to environmental stressors (Lambert 2005). For example, *S. plicata* can tolerate brackish waters ranging from 22–34 ppt salinity, and also survive in polluted environments (Thiyagarajan & Qian 2003). *S. clava* has also been noted to tolerate a wide range of salinities (Kluza et al. 2006). Although cold temperatures may slow the growth or kill sea squirts, it should not be assumed that the warmer waters of northern New Zealand offer the best sites for establishment as species are adapted to inhabit a variety of climates. For example, in northeastern North America *Didemnum* sp. are more successful in colder seasons and in deeper water than *Botrylloides violaceus* (Dijkstra et al. 2007, McCarthy et al. 2007). Non-indigenous sea squirts may also succeed in conditions not thought to be suitable for establishment, for example, *E. elongatum* from tropical/subtropical eastern Australia has established in Northland, New Zealand, despite some conditions, such as flooding events, considered as being outside of its tolerance.

Hard substrata are essential for the majority of sea squirts. Artificial structures, such as port and marina structures provide good settlement substrates that are shaded (most larvae are negatively phototactic at settlement), offers little competition, and are relatively predator free (Lambert 2005). The availability of artificial substrata was a critical factor in the successful invasion of *S. clava* in the Gulf of St. Lawrence (Canada) where natural hard substrates are scarce (Locke et al. 2007). Establishment on artificial substrates provides local sources of propagules to enable colonisation of other areas, including those with natural substrates (Minchin and Sides 2006, Coutts and Forrest 2007, Tyrell and Byers 2007).

Sea squirts can establish if colony fragments are dislodged or if fertilised larvae are released, settle, and reproduce. The gregarious nature of sea squirts may result in multiple individuals occupying the same vessel hull. However, sea squirt establishment does not necessarily depend on the presence of more than one individual. For example, solitary forms, such as *Corella inflata*, can self-fertilise (Lambert 1968) and colonial forms can asexually grow and disperse. In addition, non-identical clones of colonial sea squirts, such as *Diplosoma listerianum*, can effectively fuse/interbreed (Sommerfeldt and Bishop 1999).

Three non-indigenous sea squirt species have established in New Zealand since 2004 (Kospartov et al. in press).

Given that particular sea squirt species:

- are known to inhabit similar climates/environments to those that exist in New Zealand;
- are capable of forming establishing populations in New Zealand;
- have been recently recorded as established in New Zealand;

it is concluded that the likelihood of establishment of non-indigenous sea squirt species in New Zealand via vessel biofouling is non-negligible.

4.19.2.3. Consequence assessment

In general, non-indigenous sea squirts can have ecological impacts by acting as larval predators and competing for resources leading to the reduction of growth and survival of co-occurring species. For example, the presence of *S. clava* was shown to reduce the settlement of other species, the exceptions being barnacles and spirobiid polychaetes (Osman and Whitlach 1999 *In* NIMPIS 2010). By using experimentally manipulated settlement panels, Blum et al. (2007) examined the effects of non-indigenous *C. intestinalis* on the sessile invertebrate community in San Francisco Bay, USA. They concluded that the presence of *C. intestinalis* both depresses local species diversity and alters community assembly processes to change the community composition (Blum et al. 2007).

D. vexillum has significant impacts on the species composition of benthic communities. For example, the abundance of two polychaete species, *Nereis zonata* and *Harmothoe extenuata*, increased significantly in infested areas compared with uninfested areas (Lengyel et al. 2009). By contrast, Mercer et al. (2009) found that *D. vexillum* mats had minimal influence on benthic macro-invertebrate species abundance and richness on pebble-cobble habitats in eastern Long Island Sound, New England, USA.

The establishment of non-indigenous sea squirts has resulted in severe economic consequences to North American shellfish aquaculture. In Nova Scotia, Canada, the sea squirt *C. intestinalis* has caused up to 50% mortality to blue mussels (*Mytilus edulis*) and incurs a loss of up to C$1.86 for every 1 kg of sea squirt per m of mussel rope (Daigle & Herbinger 2009). The clubbed tunicate (*S. clava*) has also been recorded to cause major economic impacts due to resource competition and smothering resulting in reduced mussel production in Prince Edward Island, Canada (Davis & Davis 2009). In addition to decreasing stock marketability and yields, sea squirt fouling has increased labour and processing costs. For example, for mussel aquaculture in Prince Edward Island, Canada, these additional costs were conservatively estimated at 19% per pound of the market price (Thompson & McNair 2004). In north east North America, *D. vexillum* has threatened the US$400 million sea scallop industry by having smothered up to 75% of surface area in the Georges Bank fishery (Morris et al. 2009). In addition, Atlantic cod, Atlantic herring, and haddock fisheries (US$55 million in 2004) have been negatively affected (Daley & Scavia 2008). Further, dense fouling of sea squirts on fishing equipment, moorings and ropes can be time consuming to remove and can result in tangling of fishing gear.

Human health consequences within the mariculture industry have been related to the introduction of non-indigenous sea squirts. For example, asthma-like conditions have developed in workers cleaning *S. clava* fouled shellfish in poorly ventilated conditions (Kluza et al. 2006). In addition, soft tissue injuries have resulted as workers pulling in shellfish lines have to contend with the increased weight due to sea squirt fouling (Kluza et al. 2006). In some cases, workers are now exposed to caustic or harmful products used to manage sea squirt fouling. In addition, worker morale can also be negatively affected due to the extra work involved in cleaning (Thompson & MacNair 2004).

In New Zealand *D. vexillum* was originally recorded in Whangamata Harbour, northern North Island, but was introduced shortly afterward to the Marlborough District, top of the South Island (Coutts & Forrest 2007). Given the characteristics of the organism, the shift in habitat to cooler sheltered waters may have aided its establishment in New Zealand (Dijkstra et al. 2007, McCarthy et al. 2007, Valentine et al. 2007a). *D. vexillum* fouls commercial mussel lines in the Marlborough Sounds, costing growers an estimated NZ$2.2 million per year in production losses (NZIER 2008). The Marlborough and Coromandel regions would likely be the most vulnerable aquaculture areas to sea squirt establishment as they produce 68 and 21% of New Zealand's green lipped mussel exports, respectively (Aquaculture New Zealand 2009). In 2008, the aquaculture industry was valued at NZ$265 million, of which green lipped mussels made up 77% of exports by value.

The eradication, or mitigation, of sea squirts is resource intensive. Solitary sea squirts can be prolific, for example, *S. clava* can reach densities of 500 to 1,500 individuals per m^2 (NIMPIS 2010). Thus far, the New Zealand incursion response to *S. clava* has cost $2.2 million. Colonial sea squirts are particularly difficult to manage, for example, as *D. vexillum* is highly fragmentary, 100% of the colony has to removed/contained or it can re-proliferate. In addition, *D. vexillum* has highly variable growth forms allowing for the colonisation of a variety of substrata which can also create difficulties for eradication (Coutts & Forrest 2007, Pannell & Coutts 2007). In 2004, a preliminary eradication program at Shakespeare Bay, Marlborough, cost approximately NZ$0.65 million.

It is noted that in some cases *S. clava* is being harvested and sold to Asian food markets at US$8 a pound. Although this is likely to be one of the only species able to be sold as it is large, relatively fleshy and solitary compared to colonial sea squirts.

4.19.3. Risk Estimation

Given that particular sea squirts species:

- are known to be associated with vessel biofouling;
- may facilitate the entry of a variety of other non-indigenous organisms;
- are capable of forming establishing populations in New Zealand;
- have well known invasive behaviours;
- have had high impacts on core values following establishment;
- are likely to have adverse effects on core values, such as, bivalve aquaculture and on coastal benthic communities;

it is concluded that the import of non-indigenous sea squirts species via biofouling poses a non-negligible risk to New Zealand's marine core values.

4.19.4. Risk Management

The presence of sea squirts on hull fouling is best treated with broad spectrum options (Section 3). However, consideration should be given to the potential for the expulsion of propagules or fragmentation of colonies during vessel haul out. Where identified, solitary forms may be removed by hand where it can be certain all specimens are visible, it can be ensured that any stimulated spawning can be contained, and the removed organisms are contained for shore based disposal. Considerations should also include appropriate local containment due to the likelihood of release of associated non-indigenous organisms into the receiving environment.

4.19.5. References

Aquaculture New Zealand (2009) New Zealand Aquaculture Farm Facts. 2nd Edition, June 2009.

Blum, J C; Chang, A L; Liljesthröm, M; Schenk, M; Steinberg, M K; Ruiz, G M (2007) The non-native solitary ascidian *Ciona intestinalis* (L.) depresses species richness. *Journal of Experimental Marine Biology and Ecology* 342: 5-14

Borque, D; Davidson, J; MacNair, N G; Arsenault, G; LeBlanc, A R; Landry, T; Miron, G (2007) Reproduction and early life history of an invasive ascidian *Styela clava* Herdman in Prince Edward Island, Canada. *Journal of Experimental Marine Biology and Ecology* 342: 78-84.

Bullard, S G; Sedlack, B; Reinhardt, J F; Litty, C; Gareau, K; Whitlatch, R B (2007a) Fragmentation of colonial ascidians: differences in reattachment capability among species. *Journal of Experimental Marine Biology and Ecology* 342:166-168.

Bullard, S G; Lambert, G; Carman, M R; Byrnes, J; Whitlatch, R B; Ruiz, G; Miller, R J, Harris, L; Valentine, P C; Collie, J S; Pederson, J; McNaught, D C; Cohen, A N; Asch, R G; Dijkstra, J; Heinonen, K (2007b) The colonial ascidian *Didemnum* sp. A: Current distribution, basic biology and potential threat to marine communities of the northwest and west coasts of North America. *Journal of Experimental Marine Biology and Ecology* 342: 99-108.

Coutts, A D M; Forrest, B M (2007) Development and application of tools for incursion response: Lessons learned from the management of the fouling pest *Didemnum vexillum*. *Journal of Experimental Marine Biology and Ecology* 342:154–162.

Coutts, A D M; Sinner, J (2004) An updated cost-benefit analysis of management options for *Didemnum vexillum* in Queen Charlotte Sound. Cawthron Report No. 925 prepared for the Marlborough District Council, New Zealand. 29pp.

Daigle, R M; Herbinger, C M (2009) Ecological interactions between the vase tunicate (*Ciona intestinalis*) and the farmed blue mussel (*Mytilus edulis*) in Nova Scotia, Canada. *Aquatic Invasions* 4(1): 177-187.

Daley, B A; Scavia, D (2008) An integrated assessment of the continued spread and potential impacts of the colonial ascidian, *Didemnum* sp. A, in U.S. waters. NOAA Technical Memorandum NOS NCCOS 78, 61pp.

Davis, M H; Davis, M E (2004) The role of man-aided dispersal in the spread of the immigrant *Styela clava* Herdman, 1882. *Journal of Marine Science and Environment* 1:18-24.

Davis, M H; Davis, M E (2005) *Styela clava* (Tunicata: Ascidiacea) - a new addition to the fauna of the Portuguese coast. *Journal of the Marine Biological Association of the United Kingdom* 85:403-404.

Davis, M H; Davis, M E (2006) *Styela clava* (Tunicata: Ascideacea) a new edition to the fauna of New Zealand. *Porcupine Marine Natural History Society Newsletter* 20: 23-28.

Davis, M H; Davis, M E (2009) *Styela clava* – a new threat to the Mediterranean shellfish industry? *Journal of Aquatic Invasions* 4(1): 283-289.

Dijkstra, J; Harris, L G; Westerman, E (2007) Distribution and long-term temporal patterns of four invasive colonial ascidians in the Gulf of Maine. *Journal of Experimental Marine Biology and Ecology* 342(1): 61-68.

Edmondson, C H (1944) Incidence of fouling in Pearl Harbour. *Occasional Papers of the Bishop Museum* 18: 1-34

Fuller, P (2007) *Styela plicata*. USGS Non-indigenous Aquatic Species Database, Gainesville, FL. http://nas.er.usgs.gov/queries/FactSheet.asp?speciesID=1293 - accessed 20 June 2008

Howes, S; Herbinger, C M; Darnell, P; Vercaemer, B (2007) Spatial and temporal patterns of recruitment of the tunicate *Ciona intestinalis* on a mussel farm in Nova Scotia, Canada. *Journal of Experimental Marine Biology and Ecology* 342: 85-92.

Howey, R (1998) Tunicates with salad on the side. *Micscape Magazine*. http://www.microscopy-uk.org.uk/mag/indexmag.html?http://www.microscopy-uk.org.uk/mag/artnov98/rhtunic.html - accessed 20 June 2008.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kott, P; Page, M (2007) Ascidians (Phylum Tunicata, Class Ascidiacea). In MacDiarmid, A (Ed) *The Treasures of the Sea: Nga Taonga a Tangaroa. A Summary of the Biodiversity in the New Zealand Marine Ecoregion*. WWF-New Zealand; Wellington, New Zealand.

Kott, P; Bradford-Grieve, J; Esnal, G; Murdoch, R (2009) Phylum Tunicata. In Gordon, D P (ed) *New Zealand Inventory of Biodiversity. Volume 1*: Kingdom Animalia; Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press; Christchurch, New Zealand; pp 409-430.

Kluza, D; Ridgway, I; Kleeman, S; Gould, B (2006) *Styela clava (clubbed tunicate): Organism Impact Report*. MAF Biosecurity New Zealand. http://www.biosecurity.govt.nz/pest-and-disease-response/pests-and-diseases-watchlist/sea-squirt - accessed 20 June 2008.

Krone, R; Wanke, C; Schroder, A (2007) A new record of Styela clava Herdman, 1882 (Urochordata, Ascidiacea) from the central German Bight. Aquatic Invasions 2(4): 442-444.

Lambert, G (1968) The general ecology and growth fo a solitary ascidian, *Corella willmeriana*. *Biological Bulletin* 135: 296-307. [The species is actually *Corella inflata*.]

Lambert, G (2005) Ecology and natural history of the protochordates. *Canadian Journal of Zoology* 83: 34-50.

Lambert, C C; Lambert, G (1998) Non-indigenous ascidians in southern California harbours and Marinas. *Marine Biology* 130: 675-688.

Lambert, C C; Lambert, G (2003) Persistence and differential distribution of non-indigenous ascidians in harbours of the Southern California Blight. *Marine Ecology Progress Series* 259:145-161.

Lambert, G; Lambert, C C (1995) Non-indigenous sea squirts in California Harbors. *Aquatic Nuisance Species Digest* 1:17-20.

Lengyel, N L; Collie, J S; Valentine, P C (2009) The invasive colonial ascidian *Didemnum vexillum* on Georges Bank — Ecological effects and genetic identification. *Aquatic Invasions* 4(1): 143-152.

Locke, A; Hanson, J M; Ellis, K M; Thompson, J; Rochette, R (2007) Invasion of the southern Gulf of St. Lawrence by the clubbed tunicate (*Styela clava* Herdman): Potential mechanisms for invasions of Prince Edward Island estuaries. *Journal of Experimental Marine Biology and Ecology* 342: 69-77.

McCarthy, A; Osman, R W; Whitlatch, R B (2007) Effects of temperature on growth rates of colonial ascidians: A comparison of Didemnum sp. to Botryllus schlosseri and Botrylloides violaceus. Journal of Experimental Marine Biology and Ecology 342(1): 172-174.

Mercer, J M; Whitlatch, R B; Osman, R B (2009). Potential effects of the invasive colonial ascidian (*Didemnum vexillum* Kott, 2002) on pebble-cobble bottom habitats in Long Island Sound, USA. *Aquatic Invasions* 4(1): 133-142

Millar, R H (1969) *Catalogue of Main Fouling Organisms (found on ships coming into European water): Ascidians*. Organisation for Economic Co-operation and Development; Paris, France

Minchin, D; Sides, E (2006) Appearance of a cryptogenic tunicate, a *Didemnum* sp. fouling marina pontoons and leisure craft in Ireland. *Aquatic Invasions* 1(3): 143-147.

Minchin, D; Davis, M H; Davis, M E (2006) Spread of the Asian tunicate *Styela clava* Herdman, 1882 to the east and south-west coasts of Ireland. *Aquatic Invasions* 1(2): 91-96.

Morris, J A; Carman, M R; Hoagland, K E; Green-Beach, E R M; Karney, R C (2009) Impact of the invasive colonial tunicate *Didemnum vexillum* on the recruitment of the bay scallop (*Argopecten irradians irradians*) and the implications for recruitment of the recruitment of the sea scallop (*Placopecten magellanicus*) on Georges Bank. *Journal of Aquatic Invasions* 4(1): 201-211.

NIMPIS (2010). *Styela clava* impact details. *National Introduced Marine Pest Information System* (Eds: Hewitt C.L., Martin R.B., Sliwa C., McEnnulty, F.R., Murphy, N.E., Jones T. & Cooper, S.). Web publication <http://crimp.marine.csiro.au/nimpis>, Accessed: 4th February 2010.

NZIER (New Zealand Institute of Economic Research), 2008. The Economic impact of *Didemnum vexillum*. Report to Marine Farming Association. 48 pp.

Osman, R W; Whitlatch, R B (2007) Variation in the ability of *Didemnum* sp. to invade established communities. *Journal of Experimental Marine Biology and Ecology* 342: 40-53.

Pannell, A; Coutts, A D M (2007) Treatment methods used to manage *Didemnum vexillum* in New Zealand. A report prepared for Biosecurity New Zealand. Biosecurity New Zealand, New Zealand; 43pp.
Peterson, J K (2007) Ascidian suspension feeding. *Journal of Experimental Marine Biology and Ecology* 342: 127-137.
Primo, C; Vasquez, E (2008) Zoogeography of the southern New Zealand, Tasman and southern African ascidian fauna. *New Zealand Journal of Marine and Freshwater Research* 42: 233-256.
Ruppert, E E; Fox, R S; Barnes, R D (2004) *Invertebrate Zoology: A functional evolutionary approach* (7th edition). Thompson Learning; Victoria, Australia.
Sommerfeldt, A D; Bishop J D D (1999) Random amplified polymorphic DNA (RAPD) analysis reveals extensive chimerism in a marine protochordate. *Molecular Ecology* 8: 885-890.
Sutherland, P (1978) Functional roles of *Schizoporella* and *Styela* in the fouling community at Beaufort, North Carolina. *Ecology* 59:257-264
Thiyagarajan, V; Qian, P (2003) Effect of temperature, salinity and delayed attachment on development of the solitary ascidian *Styela plicata* (Lesueur). *Journal of Experimental Marine Biology and Ecology* 290:133–146
Thompson, R; MacNair, N (2004) *An overview of the clubbed tunicate (*Styela clava*) in Prince Edward Island*. PEI Department of Agriculture, Fisheries, Aquaculture and Forestry Technical Report 234
Turon, X; Tarjuelo, I; Duran, S; Pascual, M (2003) Characterising invasion processes with genetic data: an Atlantic clade of *Clavelina lepadiformis* (Ascidiacea) introduction into Mediterranean harbours. *Hydrobiologia* 503:29-35
Tyrell, M C; Byers, J E (2007) Do artificial substrates favour non-indigenous fouling species over native species? *Journal of Experimental Marine Biology and Ecology* 342: 54-60.
Valentine, P C; Carman, M R; Blackwood, D S; Heffron, E J (2007a) Ecological observations on the colonial ascidian Didemnum sp. in a New England tide pool habitat. *Journal of Experimental Marine Biology and Ecology* 342: 109-121.
Valentine, P C; Collie, J S; Ried, R N; Asch, R G; Guida, V G; Blackwood, D S (2007b) The occurrence of the colonial ascidian *Didemnum* sp. on Georges Bank gravel habitat – Ecological observations and potential effects on groundfish and scallop fisheries. *Journal of Experimental Marine Biology and Ecology* 342: 179-181.
Wood, E J F, Allen, F E (1958) Common marine fouling organisms of Australian waters. Department of the Navy, Navy Office; Melbourne, Australia; 23 pp.
Wyatt, A S J; Hewitt, C L; Walker, D I; Ward, T J (2005) Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment, *Diversity and Distributions* 11: 33-44.

4.20. SPONGES (PHYLUM: PORIFERA)

4.20.1. Hazard Identification

Sponges (phylum Porifera) are suspension feeders commonly found in subtidal benthic habitats from tropical to polar environments. Sponge habitat can range from the intertidal down to abyssal depths. Sponges can range in size and morphology from thin encrusting, or erect branching, to massive forms. Generally they become more dominant members of the benthic fauna as depth increases, hard substrates become available and the water becomes relatively free from terreginous sediment and large salinity fluctuations. Sponges are also abundant in high energy, subtidal environments where fast flowing currents can assist feeding efficiency, for example, seamounts (Rowden et al. 2002).

Sponges are important components of marine ecosystems. Sponges provide and facilitate the maintenance of biogenic habitats, couple the benthic and pelagic environments, recycle nutrients, provide protection for other organisms within their matrix and are a source of primary production via their symbiotic and commensal relationships (Bell 2008).

Records of sponge introductions are few which is likely due to difficulties associated with their taxonomy. For example, Kospartov et al. (in press) listed eight sponge species found in New Zealand waters as non-indigenous with a further 105 species listed as cryptogenic. Similarly, Streftaris et al. (2005) listed eight species of Porifera as introduced to the Mediterranean, while Longo et al. (2007) described *Paraleucilla magna* as "the first record of an invasion due to Porifera for the Mediterranean Sea".

There are few reported impacts from non-indigenous sponges. Significant smothering impacts have been recorded as a result of the introduction of *Mycale grandis* to Hawai'i (Coles and Bolick 2007) and *Terpios hoshinata* to Guam (Bryan, 1973, Rützler & Muzik 1993). In addition, boring sponges (in particular *Cliona* spp.) have been shown to impact shellfish (Fromont et al. 2005). Bioerosion, including shell boring, is a common attribute of sponge fauna and has been suggested as necessary to maintain coral reef health (Hein & Risk 1975). Shell boring can have impacts on both the productivity, and later marketability, of shellfish which can impact social, cultural and economic values. For example, Fromont et al. (2005) reported that pearl oysters (*Pinctada maxima*) lost value, growth and resistance to predation when infected with boring species (*C. orientalis*, *C. dissimilis* and *Pione velans*). In New Zealand, the non-indigenous boring sponge *C. celata* has been collected from habitats ranging from the low intertidal to over 200 m in depth with records dating back to the mid-1960's (Probert et al. 1979). While *C. celata* is known to impact upon commercially important species such as paua and oysters it is generally considered a minor problem compared to boring Spionid bristleworms such as *Polydora* spp..

Sponges are known members of the fouling community and may be found on artificial substrates. Sponges are not typically primary foulers and tend to recruit later in the succession of organisms by using secondary metabolites to clear and maintain space for growth. While introductions have been attributed to hull fouling by some authors (Godwin 2003) there have been few specimens recorded by others. For example, in the MAFBNZ commissioned research on biofouling, the solitary sponge sample recorded was the New Zealand native species *Haliclonia* cf *maxima* (Inglis et al. in prep.).

Given that sponges:

- have a low likelihood of fouling vessels;
- are likely to be associated with substantial fouling by other species;

- are not typically invasive;
- that are known to cause impacts, such as boring sponges, are already present in New Zealand;

it is concluded that sponge species do not represent a potential hazard via vessel biofouling.

4.20.2. References

Bell, J J (2008) The functional role of marine sponges. *Estuarine, Coastal and Shelf Science* 79: 341-353.

Bryan PG (1973) Growth rate, toxicity, and distribution of the encrusting sponge *Terpios* sp. (Hadromerida: Suberitidae) in Guam, Mariana Islands. *Micronesica* 9(2):237–242.

Coles, S L; Bolick, H (2007) Invasive introduced sponge *Mycale grandis* overgrows reef corals in Kāne'ohe Bay, O'ahu, Hawai'i. *Coral Reefs* 26:911

Fromont, J; Craig, R; Rawlinson, L; Alder, J; (2005) Excavating sponges that are destructive to farmed pearl oysters in Western and Northern Australia. *Aquaculture Research* 36: 150-162.

Godwin, L S; (2003) Hull fouling of maritime vessels as a pathway for marine species invasions to the Hawai'ian Islands. *Biofouling* 19(1) Supplement 1: 123-131.

Hein, F J; Risk, M J; (1975) Bioerosion of coral heads: inner patch reefs, Florida reef tract. *Bulletin of Marine Science* 25: 133-138.

Inglis, G J; Floerl, O; Unwin, M; Ponder-Sutton, A; Seaward, K; Kospartov, M; Bell, A; Kluza, D (in prep.). *The biosecurity risks associated with biofouling on international vessels arriving in New Zealand. Summary of the patterns and predictors of fouling.* Biosecurity New Zealand Technical Paper.

Kospartov K; Inglis G; Seaward K; Brink A van den; D'Archino R; Ahyong S (in press) Non-*indigenous and cryptogenic marine species in New Zealand*. Current state of knowledge – Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand. Wellington, New Zealand.

Longo, C; Mastrototaro, F; Corriero, G (2007) Occurrence of *Paraleucilla magna* (Porifera: Calcarea) in the Mediterranean Sea. *Journal of the Marine Biological Association of the United Kingdom* 87: 1749-1755.

Probert, P K; Batham, E J (1979) Epibenthic macrofauna off south-eastern New Zealand and mid-shelf bryozoan dominance. *New Zealand Journal of Marine and Freshwater Research* 13: 379-392.

Rowden, A A; O'Shea, S; Clark, M R (2002) *Benthic biodiversity of seamounts on the northwest Chatham Rise*. Marine Biodiversity Biosecurity Report 2, Ministry of Fisheries; Wellington, New Zealand.

Rützler, K; Muzik, K (1993) *Terpios hoshinaota*, a new cyanobacteriosponge threatening Pacific reefs. *Scientia Marina* 57(4): 395-403.

Streftaris, N; Zenetos, A; Papathanassiou, E (2005) Globalisation in marine ecosystems: the story of non-indigenous marine species across European seas. *Oceanography and Marine Biology: An Annual Review* 43: 419-453.