

Assessment and Management of Risks from Biofouling

Assessment and Management of Risks from Biofouling

by

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PREFACE

In the last decade, the extent to which non-indigenous marine species have been transported to New Zealand with human activities has been revealed by government-funded baseline ecological inventories in the country's main ports and harbours. It is now recognised that c. 200 non-indigenous species (NIS) have become established in New Zealand. Among these are a number of benthic organisms that have become problematic fouling species, adversely affecting both commercial sectors (e.g., aquaculture) and natural ecosystems. Transport to New Zealand via biofouling on the hulls of international vessels has been identified or implicated as the pathway of initial introduction for these species and there have been several high-profile incursions. For example, a domestic barge transported several tonnes of the colonial ascidian *Didemnum vexillum* into the Marlborough Sounds, where it has since become a pest to the aquaculture industry. The large number of NIS thought to have been introduced to New Zealand through biofouling highlights the deficiencies in existing border management in this country and a lack of fundamental ecological knowledge in relation to the invasion risk, management feasibility, and the collateral effects of different management decisions.

This thesis comprises six technical chapters that collectively aim to provide underpinning knowledge for the improvement of pre- and post-border vessel biofouling management in New Zealand. The publication of research undertaken in this thesis is timely, as several countries, including New Zealand and Australia, are in the process of developing border standards for biofouling. At a global scale, the International Maritime Organisation has been requested to include vessel biofouling in the agenda of their work plan. Restrictions on in-water vessel defouling in Australia and New Zealand are also currently under review.

At the time of submitting this thesis, Chapter 3 has been published, Chapters 2 and 4 have been accepted for publication, and Chapters 5 to 7 have been submitted to international journals. As each chapter presents discrete pieces of work, the introductions can be at times repetitive. For this reason, I have used the preface for each chapter to highlight the linkages and continuity among chapters. The prefaces also describe where the work has been submitted or published and, for multi-authored publications, the contribution made by co-authors.

ABSTRACT

Vessel biofouling is a well recognised modern-day pathway for the transfer of non-indigenous species (NIS). However despite awareness of these risks, marine incursions as a result of vessel biofouling continue to occur at a growing rate. The objective of this thesis is to provide underpinning knowledge to improve pre- and post-border management strategies for vessel biofouling. Chapter 2 provides a baseline assessment of the biofouling extent and assemblage composition on slow-moving vessels arriving at New Zealand's border. Slow-movers were targeted because their operational profile is widely considered to favour the accumulation of extensive biofouling communities (i.e., potentially high risk vectors of NIS). Interestingly, this research revealed low fouling levels and a low incidence of NIS. Highest levels of fouling were observed in areas where antifouling paint condition was poor or absent (e.g., dry-docking support strips and niche areas), which is consistent with recent studies of biofouling on other vessel types. Despite these findings, there have been several documented examples where heavily fouled slow movers have had high risk NIS on them. As such, risk profiling of slow-moving vessels is recommended. This should be based on operational characteristics such as maintenance history, exposure to regions where pest species are known to be present and intended vessel movements in the recipient region, and should ideally be undertaken on a case-by-case basis prior to arrival from international or distant source-regions.

There are limited biofouling risk mitigation options available upon the discovery of NIS at the border, particularly for large vessels (e.g., barges) or towed structures (e.g., oil rigs) where removal to land is often not feasible and in-water defouling may be the only option available. Chapter 3 provides a conceptual framework that identifies biosecurity benefits and risks posed by in-water defouling. Among the latter are the survivorship of defouled material, the release of viable propagules via spawning, and enhanced colonisation of recently defouled surfaces by high risk NIS. Chapter 4 then assesses the operational performance of two diver-operated defouling tools (rotating brush devices) that were designed to retain defouled material during operation (i.e., mitigating one of the main risks associated with in-water defouling identified in Chapter 3). These devices proved effective in removing low-to-moderate levels of fouling from flat and curved experimental surfaces. However, performance was generally poorer at removing more advanced levels of fouling. Furthermore, neither system was capable of retaining

all material defouled; c. 4% was lost to the environment, of which around 20% was viable. A significant component of material lost comprised fragmented colonial organisms (e.g., the ascidian *Diplosoma* sp.), which are theoretically capable of forming new colonies from fragments. The study also concluded that the defouling brush devices were not suitable for treating niche areas of vessel hulls such as gratings and water cooling intakes, areas where earlier work in Chapter 2 identified fouling levels to be the greatest.

Observations of fully intact and seemingly viable fragments being lost to the environment during in-water defouling trials led to a series of laboratory- and field-based experiments designed to elucidate factors influencing the survivorship of defouled material on the seabed (Chapter 5). This work showed that for some colonial organisms (e.g., ascidians), the size of fragments generated during removal affected reattachment success. Thus the defouling method is an important consideration for vessels fouled by colonial NIS. Manipulative field experiments demonstrated that exposure to sediments and benthic predation can play a major role in post-defouling survivorship. Sediment-induced mortality and susceptibility to predation was also taxon-specific. For example, soft-bodied organisms (e.g., sponges, colonial ascidians) were more affected by sedimentation and predation than calcareous taxa (e.g., tubeworms).

Chapter 6 provides a “real world” example of in-water defouling. In December 2007, the defouling of an oil rig over soft-sediments in Tasman Bay, and the subsequent discovery of NIS amongst the defouled material on the seabed, led to a dredge-based incursion response whose goal was eradication of the NIS, in particular the brown mussel *Perna perna*. During the response, c. 35 tonnes of defouled material was removed from the seabed, and target pests were reduced to densities considered too low for successful reproduction (and therefore establishment in the region) to occur. This chapter evaluates the efficacy of the response method and demonstrates that where complete elimination of a pest (i.e., removal of all organisms) is not feasible, alternative eradication success criteria based on density thresholds can be developed to mitigate biosecurity risks posed by an incursion.

The preceding technical chapters highlight the risks posed by biofouling and identify that there are presently limited post-border risk mitigation tools available. This reinforces the widely held belief that more effort should be put into pre-border management. In Chapter 7, I use two case studies of oil rig biofouling to highlight the

many challenges associated with pre-border management, and identify the urgent need for the development of treatment tools and strategies to mitigate biosecurity risks posed by vessels and structures where removal to land (e.g., dry-docking) is not feasible.

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Chapter 1

General Introduction

1.1 INTRODUCTION

1.1.1 Background

It has long been recognised that human activities have been a major pathway for the inadvertent spread of marine organisms well beyond their natural distributions (e.g., Chilton 1910; Elton 1958; Skerman 1960; Carlton 1985). The literature over the last two decades suggests that the rate of non-indigenous (NIS) introductions is steadily increasing (Carlton and Geller 1993; Cohen and Carlton 1998; Ruiz et al. 1999; Harris and Tyrrell 2001; Grosholz 2005), reflecting the greater frequency of vessel movements, changing patterns of shipping trade (Kolar and Lodge 2001; Perrings et al. 2005), changing environmental conditions that facilitate invasion success (Dukes and Mooney 1999; Harris and Tyrrell 2001; Diederich et al. 2005; Grosholz 2005; Nehls et al. 2006) and greater awareness (e.g., baseline monitoring etc) of the problem. Some introductions have had significant ecological, economic and social consequences (Carlton 1996; 2001; Hewitt 2003; Hewitt et al. 2004; Pimentel et al. 2005). In New Zealand, a number of historical and recent introductions threaten some of New Zealand's highly valued resources such as aquaculture (e.g., the ascidians *Ciona intestinalis*, *Styela clava*, and *Didemnum vexillum*), whereas others contribute substantially to the country's economy (e.g., the Pacific oyster *Crassostrea gigas*).

Shipping has been identified as a particularly important pathway for the spread of NIS (Hewitt et al. 1999; Gollasch 2002; Fofonoff et al. 2003; Ruiz and Carlton 2003; Piola and Johnston 2008; Yamaguchi et al. 2009), with the main vessel-related transport mechanisms being ballast water (Carlton 1985; Olenin et al. 2000; Taylor et al. 2007), hull fouling (Lewis et al. 2003; Coutts and Taylor 2004; Davidson et al. 2009), and fouling of niche areas such as sea chests, intake pipes and gratings (Carlton et al. 1995; Coutts et al. 2003; Minchin and Gollasch 2003; Coutts and Dodgshun 2007). More than 200 marine species have been accidentally introduced into New Zealand waters,

primarily through ballast water and hull fouling (Cranfield et al. 1998; Hayden et al. 2009). In a similar vein, Hewitt and Campbell (2008) estimate that 55–69% of the ~1780 NIS detected in ports and harbours around the world are likely to have been introduced by vessel biofouling, based on their life-history characteristics.

1.1.2 Vessel biofouling

The deleterious effects of biofouling have plagued ocean-going vessels for centuries. In addition to marine biosecurity risks from NIS transfers, biofouling reduces vessel speed due to a reduction in hydrodynamics and manoeuvrability, causing increased fuel and maintenance costs (Townsin 2003; Tribou and Swain 2010). In recent decades, biofouling has been managed by the use of biocidal (toxic) paints, of which tributyltin self-polishing copolymer paints (TBT-SPC paints) have been the most successful (Yebra et al. 2004; Finnie and Williams 2010). Biofouling organisms settling on toxic paints may survive short periods, but soon die or become detached, provided the coating is functioning (Swain and Schultz 1996). However, due to growing awareness of the adverse impacts of TBT to the marine environment (e.g., Stewart et al. 1992; Svavarsson 2000), the application of TBT-based paints has been regulated internationally since 1990. Copper-based alternatives have been developed but are not 100% effective (Piola et al. 2009a; Finnie and Williams 2010), hence the antifouling paint companies and ship operators are presently faced with the challenge of replacing toxic biocidal paints with coatings that provide the same economic benefits and cause less harmful effects to the environment. This has led to the development of alternative non-toxic antifouling strategies (see Yebra et al. 2004 for a review).

Most vessel types routinely undergo maintenance to remove hull fouling organisms in order to maintain/optimize fuel efficiency rather than to manage the biosecurity risks posed by the vessel. This can occur *in situ* (i.e., in-water defouling) or on land (e.g., dry-docking, careening¹). Arguably the most ‘biosecure’ method to treat biofouling is the removal of the vessel to land where defouling can occur and the defouled material can be contained (Woods et al. 2007). However, for many vessels (e.g., large merchant vessels), removal to land can be difficult and has to be planned for in advance, and is dependent on the availability of suitable facilities in the port/country in question. Thus, in some situations, in-water defouling may be the only available option to remove

¹ A common method used for recreational vessels where a craft is beached on its side and cleaned.

biofouling from a hull. Furthermore, the recent global recession (Floerl and Coutts 2009), rising fuel costs and copper-resistant organisms (Russell and Morris 1970; Ng and Keough 2003; Dafforn et al. 2008) are expected to result in a worldwide increase in the amount of in-water defouling undertaken. In most cases, biofouling material is not retained by in-water defouling devices and may therefore settle on natural seabed habitats or artificial structures adjacent to the vessel, or be more widely dispersed by currents (Floerl et al. 2004). In part because of the perceived ecological risk from the release of this material, a number of countries have placed restrictions on this approach or are considering doing so.

The application of antifouling paints and routine vessel hull maintenance are reasonably effective in reducing biofouling transfers resulting from vessel movements (Callow and Callow 2002; Coutts and Taylor 2004). However, despite the widespread use of antifouling paint coatings, fouled vessels continue to ply the world's oceans. There are several main reasons for this: (i) not all vessels undergo routine maintenance and have antifouling paints re-applied within recommended timeframes, including obsolete vessels (Davidson et al. 2008) and oil rigs (Hopkins and Forrest 2009); (ii) sub-standard paint application or inappropriate selection of paint for vessel type/use; (iii) some vessels remain idle for extended periods (e.g., barges, tugs, oil rigs), and as such, the efficacy of self-polishing antifouling paints is compromised (Ferriera et al. 2006); (iv) biofouling can occur on non-hull areas of the vessel where antifouling paint condition is often poor; such as sea chests², gratings and water intake pipes (Coutts and Dodgshun 2007); and (v) some taxa are able to colonise recently antifouled surfaces (e.g., copper-resistant taxa).

1.1.3 Management of biofouling risks

Marine biosecurity is the protection of the marine environment from impacts of non-indigenous species (NIS), and typically involves pre-border and post-border management of vectors and risk species (Hewitt et al. 2004). Preventing an incursion from occurring in the first instance is favoured over attempting to manage or eradicate a pest once it arrives or becomes established (Ruiz and Carlton 2003; Hewitt et al. 2004; Finnoff et al. 2007). This is because there are presently limited options to eradicate

² Sea chests are recesses built into a ship's hull that house intake piping via which water is pumped aboard for ballast, engine cooling and fire fighting.

pests from the marine environment (Piola et al. 2009b). In New Zealand, this has been highlighted by failed attempts to eradicate the Asian kelp *Undaria pinnatifida* (Hunt et al. 2009) and the colonial ascidian *Didemnum vexillum* (Coutts and Forrest 2007).

With the exception of ballast water discharge, marine biosecurity risks posed by vessels arriving in New Zealand are largely unmanaged, partly because there are at present few effective or practical management tools. For ballast water, New Zealand has an import health standard (IHS) requiring that ballast water from all vessels be exchanged in mid-ocean *en route* to New Zealand, and only the exchanged water can be discharged in New Zealand waters (Hewitt et al. 2004). Ballast water exchange is recognised as the only practical way of reducing the spread of marine pests via ballast water discharge, although it is not completely effective (Taylor et al. 2007). Exemptions to the IHS are made in certain situations (e.g., poor weather conditions precluding safe ballast water exchange), unless the vessel contains ballast water from Port Phillip Bay and Tasmania for which un-exchanged ballast cannot be discharged in New Zealand waters. For vessel biofouling, there have been efforts to identify high risk vectors (e.g., Coutts and Taylor 2004; Inglis et al. 2010; Piola and Conwell 2010). However, routine and ongoing pre-border management (e.g., inspection regimes) of hull fouling does not exist in any nation, including New Zealand (Hewitt et al. 2004). Thus, the lack of effective pre-border vector management means that NIS will continue to arrive in New Zealand, and some will become problematic pests.

At the time of writing, New Zealand and Australia are in the process of developing border standards for biofouling, and at a global scale, the International Maritime Organisation has been requested to include vessel biofouling in the agenda of their work plan. However, despite the growing awareness of the need for pre- and post-border management of biofouling risks, there remain many technical challenges to overcome before efforts are effective in reducing NIS transfers, including: (i) the inability to reliably predict the presence of NIS on the thousands of potential vectors arriving at the border, (ii) the lack of mitigation tools presently available to eliminate risks posed by NIS while attached to a vector or when released to the environment (e.g., via spawning or intentional removal), and (iii) our present lack of understanding regarding biosecurity risks associated with various biofouling treatment options (e.g., in-water defouling). Hence, the goal of this thesis is to provide underpinning research to address the current

lack of knowledge in these key areas, and to contribute to the improved management of biofouling risks in New Zealand.

1.2 SCOPE AND CONTENT OF THESIS

This thesis contains laboratory- and field-based manipulative experiments, and “real world” case studies that collectively highlight the challenges of pre- and post-border management of biofouling risks. Specifically, Chapter 2 aimed to determine the nature and extent of fouling on international slow-moving vessels and towed structures visiting New Zealand. Slow-movers were targeted because their operational profile is widely considered to favour the accumulation of extensive biofouling communities (i.e., potentially high risk vectors of NIS). Chapter 3 presents options for managing biosecurity risks posed by vessel biofouling, and provides a conceptual framework of risks posed by in-water defouling.

The aim of Chapter 4 was to evaluate the efficacy of two diver-operated defouling tools that were developed to retain all biofouling material removed. Both tools were tested across a range of biofouling levels that had accumulated on experimental surfaces and on a fouled fishing vessel. Observations of viable biofouling organisms being released to the environment during in-water defouling trials using these devices led to a series of field-based experiments where the seabed survivorship of intact organisms and fragmented colonial species was assessed in contrasting port environments (Chapter 5). The aim of this research was to determine whether fragment size affected the survivorship of colonial organisms and to elucidate environmental factors that influenced the survivorship of defouled material in port environments.

In December 2007, a semi-submersible drilling rig was defouled in Tasman Bay prior to departure for Australia. During the defouling operation, several NIS were discovered amongst samples collected during an earlier ecological survey of the rig, prompting a full-scale eradication effort initiated by MAF Biosecurity New Zealand. The defouling of an oil rig in Tasman Bay and the subsequent incursion response provided a rare opportunity to evaluate the efficacy of dredging as a method for removing target NIS from the seabed (Chapter 6), and to assess the survivorship of defouled material (Chapter 7). Chapter 7 also uses two oil rig case studies to highlight the many challenges associated with the pre-border management of biofouling risks. The General Discussion (Chapter 8) is used to expand on some of the main findings of this research.

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Chapter 2

Biofouling and Non-Indigenous Marine Species Associated with Slow-moving Vessels

PREFACE

This chapter provides an insight to the level of risk posed by unmanaged slow moving vessels arriving at New Zealand's border. The operational profile of slow movers means they are often stationary for long periods, which will decrease the efficacy of their antifouling coatings and lead to an accumulation of biofouling organisms. Furthermore, the slow speed at which such vessels travel (cruising speed of 5 - 10 knots) is generally considered to favour the survival of their associated fouling assemblages. As such, vessels in this category are considered to be high risk, and this appears deserved in light of recent documented NIS introductions attributable to the movements of slow movers (e.g., the invasive ascidian *Didemnum vexillum*). My thesis supervisors Barrie Forrest and Jonathan Gardner provided useful comments on this chapter, which was accepted for publication in the international journal Biofouling in May 2010.

ABSTRACT

Vessel traffic is the primary pathway for non-indigenous marine species introductions to New Zealand, with hull fouling recognised as being an important mechanism. This chapter describes hull fouling on seven slow-moving commercial vessels sampled over a one year period. Sampling involved the collection of images and fouling specimens from different hull locations using a standardised protocol developed to assess vessel biofouling in New Zealand. A total of 29 taxa was identified by expert taxonomists, of which 24% were indigenous to New Zealand and 17% non-indigenous. No first records to New Zealand were reported, however 59% of species were classified as “unknown” due to insufficient taxonomic resolution. The extent of fouling was low compared to that described for other slow-movers. Fouling cover, biomass and richness were on average 17.1% (SE = 1.8%), 5.2 g (SE = 1.1 g) and 0.8 (SE = 0.07) per photoquadrat (200 x 200 mm), respectively. Fouling extent was lowest on the main hull areas where antifouling paint was in good condition. In contrast, highest levels of fouling were associated with dry-docking support strips and other niche areas of the hull where paint condition was poor. Future studies should target vessels from a broader range of bioregions, including vessels that remain idle for extended periods (i.e., months) between voyages, to increase our understanding of biosecurity risks posed by international commercial slow-movers.

2.1 INTRODUCTION

Hull fouling is a feature of all maritime vessels, and has been implicated in the global spread of non-indigenous species (NIS) since international vessel traffic began (Bishop 1951; Carlton and Hodder 1995 and references therein). Hull fouling is also recognised as an important modern-day pathway for the human-mediated spread of NIS (Dafforn et al. 2008; Piola and Johnston 2008; Yamaguchi et al. 2009; Coutts et al. 2010), especially with a global ban on the use of highly effective organotin-based antifouling coatings (Nehring 2001; Champ 2003; Yebra et al. 2004; Sonak et al. 2009). Several studies have characterised biofouling on merchant ships, international fishing vessels and international yachts (e.g., Coutts and Taylor 2004; Coutts and Dodgshun 2007; Davidson et al. 2009; Piola and Conwell 2010) but much remains to be understood about predictors of biofouling and NIS transfer.

Increasingly, commercial vessels such as barges and their tugs have also been recognised as high risk pathways for the spread of non-indigenous species (Lewis et al. 2006; Coutts et al. 2010), and in some cases identified or implicated in the international and domestic spread of some high-profile marine pests (Coutts and Forrest 2007). Like merchant vessels and yachts, the condition of antifouling coatings is a factor that is expected to affect the susceptibility of barges and tugs to fouling. However, the operational profile of craft such as barges means they are often stationary for long periods, which will decrease the efficacy of their antifouling coatings due to lower biocidal release rates and lead to an accumulation of biofouling organisms (Ferreira et al. 2006). Moreover, the slow speed at which such vessels travel (cruising speed of 5 - 10 knots) is likely to favour the survival of their associated fouling assemblages (Davidson et al. 2006; Coutts et al. 2010).

The international movement of barges and tugs provides an interesting addendum to our understanding of biosecurity risk from vessel hull fouling. While it is apparent that merchant vessels are numerically the dominant international vessel type, comprising c. 75% of international traffic to New Zealand (MAF Biosecurity New Zealand, unpublished data), the incidence of fouling on such vessels is typically low (Coutts and Taylor 2004). This reflects the fact that merchant vessels are largely in constant use and well-maintained to improve hydrodynamic efficiencies (Coutts and Taylor 2004; Schultz 2007). In contrast, barge and tug movements internationally are considerably less in number, but have the potential to be heavily fouled. In New Zealand, for example, a poorly maintained and heavily fouled barge was identified as the vector by which the invasive colonial ascidian *Didemnum vexillum* was transferred into a nationally important aquaculture region (Coutts and Forrest 2007).

The above and other examples of extensive fouling on international (e.g., Davidson et al. 2008a,b) and domestic (e.g., Coutts 2002) slow-moving vessels clearly highlight the potential biosecurity risk that arises with the movement of such craft. However, from the literature it is unclear whether such examples are representative of biosecurity risk generally, as there appear to have been no systematic surveys of fouling on operational international slow-moving commercial vessels. Hence, the objective of this chapter was to provide a preliminary characterisation of fouling extent and the occurrence of non-indigenous marine species on commercial barges and tugs arriving in New Zealand over a one year period.

2.2 METHODS

2.2.1 Description of vessels sampled

Five barges and two tugs were sampled at four New Zealand ports between May 2006 and May 2007. All vessels surveyed had arrived from Australia and had been operating on New Zealand - Australia routes. Although tugs can travel at > 10 knots, they were included in the sampling because they travel at slow speeds (c. 5 knots) when towing, and often remain idle with their barge while not underway. Five of the vessels were sampled on one occasion only. However, repeat visits by the tug *Katea* (May and August 2006, May 2007) and the barge *Sea-Tow 60* (September 2006 and May 2007) enabled a preliminary evaluation of changes in fouling over time. Therefore, there were 10 sampling occasions in total, representing c. 75% of commercial barges arriving in New Zealand over the one year period (Sea-Tow Ltd., unpublished data).

Barges sampled ranged in length from 47 - 97 m (beam = 8.9 and 24.0 m, respectively), while tugs ranged from 29 - 34 m (beam = 9.0 and 10.8 m, respectively). Vessel speeds (while towing or being towed) ranged from 5 - 7.5 knots (Table 2.1). All Sea-Tow vessels had ablative antifouling coatings (Sea-Barrier 3000TM). The barges *Soundcem I* and *Soundcem II* also had an ablative antifouling coating, but the owners could not specify which paint brand. None of the vessels sampled during this study had been cleaned in-water since their last dry-docking. Residency periods ranged from 1 - 37 days (based on the previous 20 ports visited before sampling), with a slightly higher average residency period for barges (mean = 5.0 days, SE = 0.8 days) compared with tugs (mean = 3.4 days, SE = 0.6).

2.2.2 Vessel sampling procedures and determination of fouling extent

Hull fouling on each vessel was assessed using a standard sampling protocol developed for international yacht arrivals to New Zealand (Floerl et al. 2005) and later applied by Ministry of Agriculture and Forestry (MAF) Biosecurity New Zealand for assessing fouling across a range of international vessel types. First, a level of fouling (LoF) rank (0 - 5) was assigned based on surface (i.e., out of water) observations along the length of the vessel, as follows: 0 = no visible fouling, 1 = partial biofilm, 2 = 1 - 5% of patchy macrofouling or filamentous algal cover, 3 = 6 - 15% patchy cover, 4 = 16 - 40% cover, and 5 = $> 40\%$ fouling cover. For comparison with surface observations, divers

recorded in-water LoF at the same vessel regions without prior knowledge of the surface scores assigned.

Divers then took photoquadrats (200 x 200 mm) at four vessel regions (Figure 2.1), i.e., the bow, amidships, stern, and opportunistically sampled niche areas (e.g., gratings, propeller shaft), using an 8 megapixel Canon EOS digital camera (18-55 mm lens kit, IkeliteTM underwater housing, 2 x IkeliteTM DS50 strobes). Sampling within the bow, amidships and stern regions was conducted in zones, with replicate ($n = 3$) samples haphazardly taken near-surface (0.5 m), inside dry-docking support strips (DDSS) (where feasible), and on sub-surface areas of the hull where antifouling paint was present. For each photoquadrat, divers assessed paint condition as good (no imperfections), average (minor chipping and visible paint wear to base layers) or poor (substantial areas of no paint, and/or bare hull). Organisms within the quadrat were scraped into labelled sample bags. At the surface, samples were sieved to 1 mm, blotted, weighed, sorted into broad taxonomic groups and preserved. Samples were identified to the highest level of taxonomic resolution feasible by specialist taxonomists and classified as being either native (indigenous), non-indigenous, cryptogenic (uncertain origins), or unknown (due to insufficient taxonomic resolution).

At the completion of sampling, photoquadrats were rectified in ArcMap 9.2 (ESRI, Redlands, CA, USA). Fouling biota present within each image were individually traced to create a map from which percent cover of fouling could be calculated by dividing the total area of fouling taxa by the quadrat area and multiplying by 100. Fouling richness was determined as the number of different taxa within each photoquadrat image. Fouling biomass was expressed as wet weight of the blotted samples.

Table 2.1 Summary information, maintenance history and residency periods for each of the vessels surveyed. None of the vessels had been in-water cleaned since dry-docking.

Vessel name	Date sampled	Location sampled (Latitude)	Average speed (knots)	Length/beam/draft (m)	Time since last dry dock	Average residency period - days (SE)
Tugs						
<i>Katea</i> (1)	25/05/2006	Auckland (36°S)	6.0	29.0/9.0/3.5	11 months	2.8 (1.0)
<i>Koranui</i>	7/06/2006	Tauranga (37°S)	7.5	34.4/10.8/5.6	1 year 3 months	5.3 (1.8)
<i>Katea</i> (2)	29/08/2006	Westport (41°S)	5.5	29.0/9.0/3.5	1 year 2 months	2.1 (0.2)
<i>Katea</i> (3)	10/05/2007	Nelson (41°S)	6.5	29.0/9.0/3.5	1 year 11 months	3.4 (1.2)
Barges						
<i>Soundcem II</i>	25/05/2006	Auckland (36°S)	6.0	47.0/8.9/2.8	1 month	30
<i>Soundcem I</i>	25/05/2006	Auckland (36°S)	6.0	47.0/8.9/2.8	1 month	30
<i>Sea-Tow 80</i>	7/06/2006	Tauranga (37°S)	7.5	97.0/24.0/4.8	6 years, 1 month	5.3 (1.8)
<i>Sea-Tow 61</i>	29/08/2006	Westport (41°S)	5.5	85.3/24.4/5.5	1 year 9 months	2.1 (0.2)
<i>Sea-Tow 60</i> (1)	28/09/2006	Nelson (41°S)	6.0	85.3/24.4/5.5	1 year 10 months	7 (1.8)
<i>Sea-Tow 60</i> (2)	10/05/2007	Nelson (41°S)	6.5	85.3/24.4/5.5	2 years 6 months	3.4 (1.2)

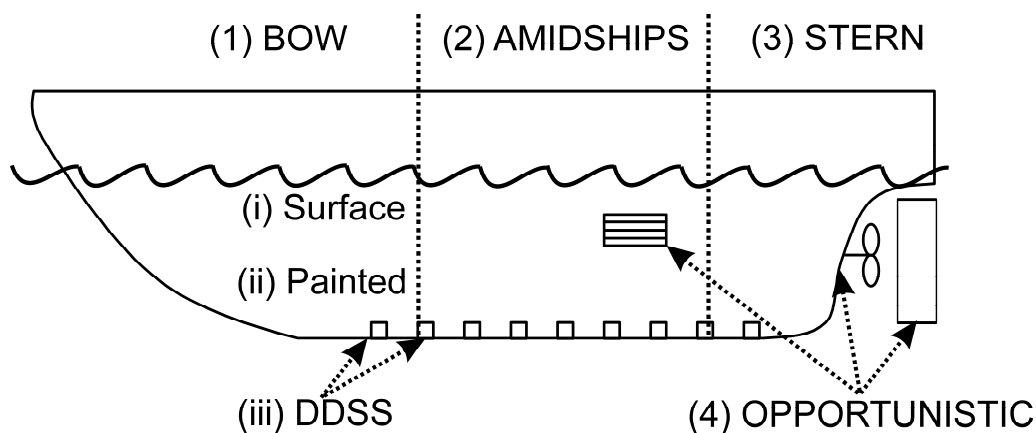


Figure 2.1 Diagram of a vessel hull identifying vessel regions (1 - 4) and vertical sampling zones (i - iii). Opportunistically sampled areas of a vessel hull typically included gratings, propeller shafts, seawater intake pipes and behind anodes; however these were not always present on the vessels sampled.

2.2.3 Statistical analyses

Given the low vessel sample size, data analyses were restricted to descriptive statistics for the quantitative measures of fouling extent and qualitative assessments of paint condition, and categorical levels of fouling. Relationships between fouling extent (% cover, biomass and richness) and categorical levels of fouling (LoF) were tested using one-way ANOVA. Data were explored for homogeneity and normality using Statistica Version 7 (StatSoft Inc., Tulsa, OK, USA), and dependent variables were $\log(x+1)$ transformed where necessary to meet the assumptions of Generalised Linear Modelling. Differences between surface and diver observations of fouling were tested using the non-parametric Wilcoxon match pairs test. For these analyses, data from repeat sampling events of the same vessel were excluded.

2.3 RESULTS

2.3.1 Antifouling paint condition

Paint condition on the seven vessels was consistently rated as poor in opportunistically sampled niche areas (e.g., sea chests, gratings) and on the dry-docking support strips (DDSS) where paint had not been applied during the previous dry-docking event. Paint condition on the main hull surface of the tugs was rated as good for 100% of the 36

observations, but barges were assigned a higher proportion of poor (11%) and average (17%) scores. In particular, the barge *Sea-Tow 80* (> 6 years since last dry-dock) had poor paint condition present on all sub-surfaces inspected. There was also a higher proportion of average and poor paint scores assigned to surface sampling zones (approx. 1 m below the waterline) for both barges (8.3 and 8.3%, respectively) and tugs (6 and 9%, respectively).

2.3.2 Fouling identity, cover, biomass and richness

Twenty-nine taxa were identified from 125 samples collected during the one year survey (Table 2.2). Of these, 41% were identified to species-level, 31% to genus-level and the remaining 28% to phylum. A relatively diverse range of taxa was encountered, representing four animal and four algal phyla (Figure 2.2). Samples were numerically dominated by arthropods (mainly crustaceans), molluscs and macroalgae. Approximately 24% of taxa were indigenous to New Zealand and 17% non-indigenous, and a high proportion of taxa (59%) had “unknown” status due to insufficient taxonomic resolution (i.e., as a result of partial/damaged specimens or lack of distinguishing features in juveniles). Non-indigenous taxa were found on both barges and tugs; however no first records for New Zealand were present in the samples taken (Table 2.2).

Table 2.2 Presence of taxa on vertical sampling zones and opportunistically sampled niche areas of slow-movers and their current biosecurity status in New Zealand. X = present.

Taxon	Description	Biosecurity Status	Tug				Barge			
			Surface	Painted	DDSS	Niche	Surface	Painted	DDSS	Niche
<i>Acryptolaria</i> sp.	Hydroid	Status unknown							X	
<i>Amphibalanus amphitrite</i>	Acorn barnacle	Non-indigenous		X			X			X
<i>Amphibalanus variegatus</i>	Acorn barnacle	Indigenous		X	X	X	X	X	X	X
Anthozoa	Anemone	Status unknown						X		
<i>Austrominius modestus</i>	Acorn barnacle	Indigenous	X		X	X	X	X	X	X
<i>Bangia</i> sp.	Red algae	Status unknown	X							X
Bivalvia	Unid. Bivalve	Status unknown			X					
<i>Cladophora</i> sp.	Green alga	Status unknown	X							X
<i>Conchoderma auritum</i>	Goose barnacle	Indigenous			X	X				
<i>Coryne pusilla</i>	Hydroid	Non-indigenous							X	X
<i>Crassostrea gigas</i>	Oyster	Non-indigenous				X		X		
<i>Crassostrea</i> sp.	Oyster	Status unknown			X					
Cyanobacteria	Cyanobacteria	Status unknown								X
Ectocarpales	Brown algae	Status unknown	X				X			
<i>Ectocarpus fasciculatus</i>	Brown alga	Indigenous	X			X		X		X
<i>Eudendrium</i> sp.	Hydroid	Status unknown				X			X	
Hydrozoa	Unidentified hydroid	Status unknown			X			X	X	X
<i>Lepas anatifera</i>	Goose barnacle	Indigenous	X	X					X	
Maxillopoda	Unid. maxillopod	Status unknown			X		X		X	X
<i>Mytilus galloprovincialis</i>	Blue mussel	Indigenous			X				X	X
<i>Obelia dichotoma</i>	Hydroid	Non-indigenous								X
<i>Obelia</i> sp.	Hydroid	Status unknown							X	X
Ostreidae	Oyster	Status unknown						X	X	
<i>Paracerceis sculpta</i>	Isopod	Non-indigenous	X							
<i>Polysiphonia</i> sp.	Red algae	Status unknown	X							
<i>Rhizoclonium</i> sp.	Green alga	Status unknown	X							
Serpulidae	Calcareous tubeworm	Status unknown								X
<i>Stylonema alsidii</i>	Red algae	Indigenous								X
<i>Ulva</i> sp.	Green algae	Status unknown	X			X	X			X

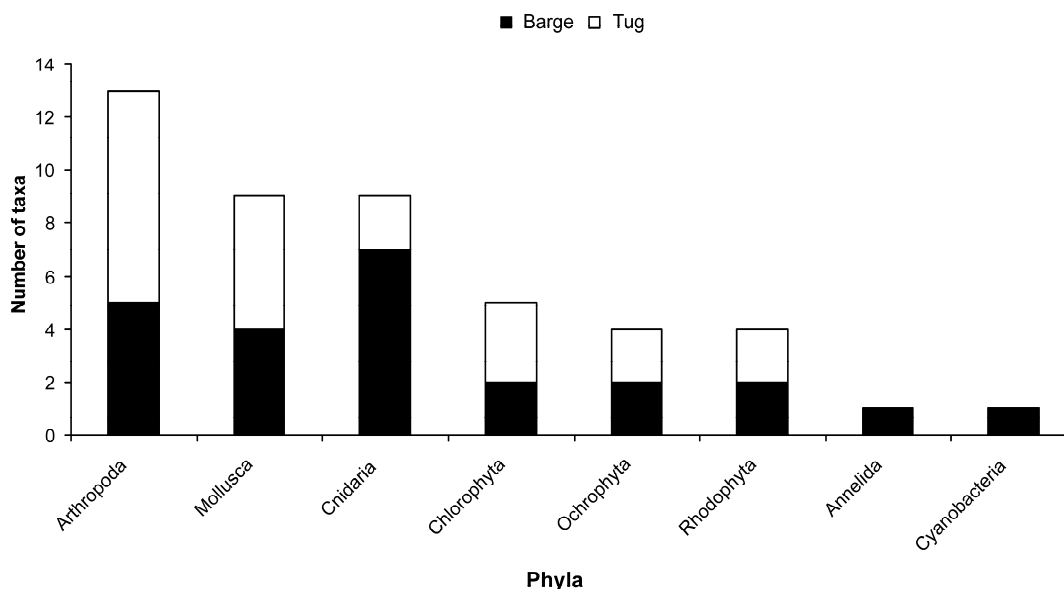


Figure 2.2. Number of taxa (assigned to phyla) on vessels surveyed.

In general, fouling assemblages encountered on the vessels were two-dimensional in structure rather than well-developed, three-dimensional late successional stages. Fouling cover ranged from 0 - 100% (overall mean = 17%), with higher levels observed on tugs compared with barges (Table 2.3). Fouling cover did not vary greatly along the vessel regions (bow, amidships, stern and opportunistic areas) for either vessel type. However, fouling cover on vertical sampling zones (i.e., surface, painted and DDSS) was more variable, with higher levels on the DDSS (where paint condition was poor) compared with painted areas of the hull. Taxon richness per photoquadrat on barges and tugs was very low (mean = 0.89 and 0.8 taxa, respectively). Overall vessel taxon richness ranged between 3 - 10 taxa for barges (mean = 8.5, SE = 1.3), and 6 - 12 for tugs (mean = 7.0, SE = 1.2). Fouling biomass ranged between 0 - 4.4 kg.m⁻² (overall mean = 0.13 kg.m⁻², SE = 0.03 kg.m⁻²), with highest levels observed on DDSS (mean = 0.3 kg.m⁻², SE = 0.1 kg.m⁻²) and on niche areas (mean = 0.3 kg.m⁻², SE = 0.05 kg.m⁻²) of the vessels.

Samples collected opportunistically from niche areas of the vessels where paint condition was typically poor, were often characterised by higher taxon richness (Tables 2.2 and 2.3); including fouling taxa typically associated with later stages of fouling (e.g., bivalves). In contrast, painted areas of the vessel hulls had low richness with

mainly barnacles and hydroids present. Surface zones were characterised by a high incidence of macroalgae. Dry docking support strips (i.e., where antifouling paint was absent) had a diverse range of taxa present (e.g., barnacles, bivalves and hydroids), with macroalgae noticeably absent within this zone (Table 2.2).

Fouling characteristics on vessels sampled more than once during the one year sampling period changed over time. For *Katea*, changes in fouling are evident as marked differences in taxa richness and biomass between the first and last sampling events (Figure 2.3).

Table 2.3 Mean fouling cover (%) and richness per photoquadrat (0.04 m²) taken within vertical sampling zones (Surface, Painted, DDSS) across the vessel sampling regions (refer Figure 2.1). Associated standard error (bracketed value) is shown.

Vessels	Bow			Amidships			Stern			Opportunistic
	Surface	Painted	DDSS	Surface	Painted	DDSS	Surface	Painted	DDSS	Niche
Cover (%)										
Tugs										
<i>Katea</i> (1)	100 (0)	0	7.9 (3.3)	100 (0)	0	0.2 (0.2)	0	0	0	0.2 (0.1)
<i>Koranui</i>	0	0	64.9 (20.4)	0	0	60.1 (14.1)	0	0	4 (0.18)	31.6 (11.8)
<i>Katea</i> (2)	100 (0)	0	39.0 (13.5)	100 (0)	0	0.2 (0.1)	100 (0)	0	0	44.7 (20.1)
<i>Katea</i> (3)	38.1 (7.6)	0	100 (0)	74.5 (9.6)	0	3.1 (2.2)	89.0 (4.7)	0	14.9 (6.7)	29.8 (15.5)
Barges										
<i>Soundcem II</i>	0	0	0	0	0	2.2 (1.5)	0	0	0.2 (0.1)	8.8 (5.1)
<i>Soundcem I</i>	0	0	0	0	0	2.8 (1.4)	0	0	0	81.5 (0)
<i>Sea-Tow 80</i>	1.5 (0.4)	0.4 (0.1)	1.0 (0.2)	3.3 (2.9)	24.9 (7.1)	8.7 (3.6)	0.02 (0.02)	5.0 (0.7)	0	11.5 (3.3)
<i>Sea-Tow 61</i>	0	0	2.6 (0.7)	0	0	2.8 (0.6)	0	0	0.1 (0.1)	6.1 (0.1)
<i>Sea-Tow 60</i> (1)	0	0	65.1 (16.9)	0	0	82.9 (17.1)	52.5 (20.1)	0	63.8 (7.8)	36.6 (12.3)
<i>Sea-Tow 60</i> (2)	0	0	79.7 (6.1)	0	0	84.7 (15.4)	0	0	0	24.7(0.1)
Richness										
Tugs										
<i>Katea</i> (1)	0	0	1 (0)	2.3 (1.2)	0	0.7 (0.7)	0	0	0	0.5 (0.4)
<i>Koranui</i>	0	0	1.3 (0.3)	0	0	1.0 (0)	0	0	1.0 (0)	2.0 (1)
<i>Katea</i> (2)	3.0 (0.6)	0	2.0 (0.6)	4.3 (0.3)	0	0	3.0 (0)	0	0	1.0 (0.4)
<i>Katea</i> (3)	2.0 (0)	0	3.5 (0.5)	2.0 (0)	0	0.7 (0.3)	2.0 (0)	0	1.0 (0)	2.5 (0.5)
Barges										
<i>Soundcem II</i>	0	0	0	0	0	1.0 (0.6)	0	0	0.7 (0.3)	1.7 (0.3)
<i>Soundcem I</i>	0	0	0	0	0	1.7 (0.3)	0	0	0	1.0 (0)
<i>Sea-Tow 80</i>	1.0 (0)	1 (0)	1.0 (0)	2.7 (0.9)	2.7 (0.3)	2.7 (0.3)	0.3 (0.3)	1.3 (0.3)	0	1.0 (0.2)
<i>Sea-Tow 61</i>	0	0	3.0 (0)	0	0	1.0 (0)	0	0	0.3 (0.3)	1.0 (0)
<i>Sea-Tow 60</i> (1)	0	0	2.0 (0)	0	0	2.0 (0)	0	0	2.0 (0)	3.8 (1.2)
<i>Sea-Tow 60</i> (2)	0	0	2.3 (0.3)	0	0	2.6 (0.3)	0	0	0	3.0 (0)

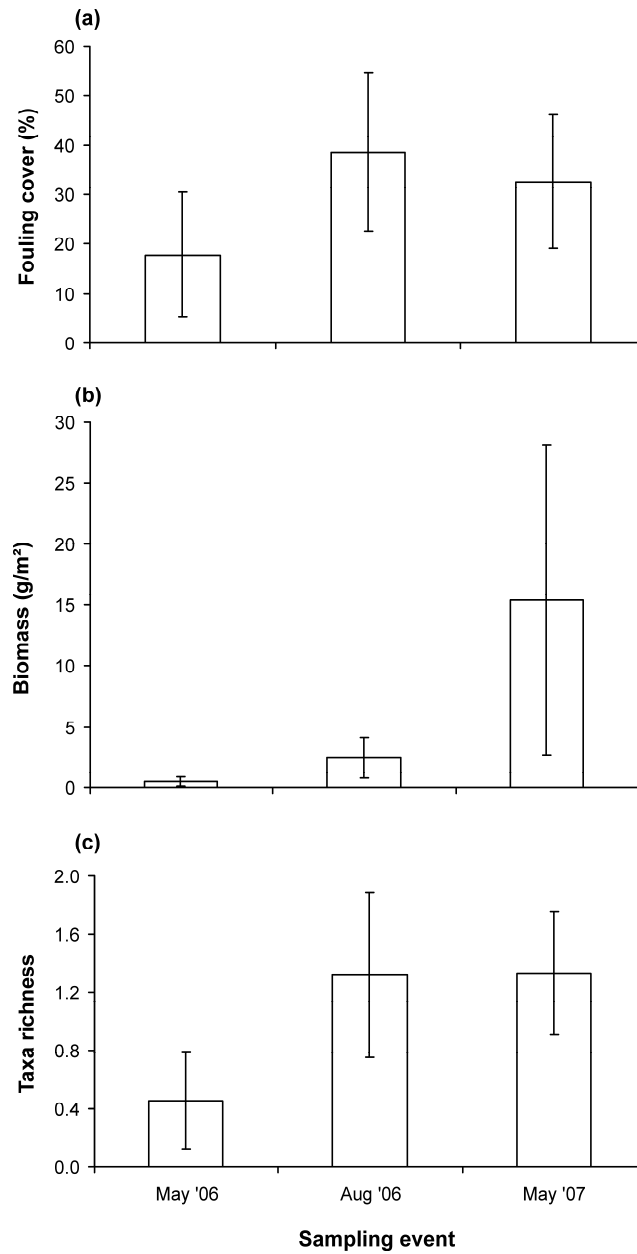


Figure 2.3. Changes in mean (a) fouling cover, (b) fouling biomass, and (c) taxa richness per photoquadrat on the tug *Katea* over the three separate sampling events (time since last dry docking = 11, 13 and 21 months) undertaken over the one year study period. Error bars denote 95% confidence intervals.

2.3.3 Utility of LoF as a measure of fouling

Strong positive linear relationships were evident between categorical level of fouling (LoF) scores assigned by divers and quantitative measures of fouling cover ($F_{4,297} = 54.74$, $P < 0.001$) and taxon richness ($F_{4,297} = 119.25$, $P < 0.001$). Fouling biomass also increased with increasing LoF ($F_{2,297} = 37.83$, $P < 0.001$); however this relationship was non-linear (exponential), with a marked increase in biomass at LoF scores ≥ 3 (Figure 2.4).

Given the relative ease in which surface scores for LoF can be assigned, it was of interest to determine whether they corresponded to the LoF below the surface of the vessel. As expected, there was no significant difference between LoF values assigned at the surface by non-divers and by divers (Wilcoxon Match Pairs Test, $Z = < 0.01$, $df = 63$, $P = 1.00$). However, surface observations of fouling were unable to reliably predict fouling levels on painted areas of the vessel below the waterline ($Z = 4.29$, $df = 63$, $P < 0.001$), on DDSS ($Z = 5.18$, $df = 56$, $P < 0.001$) or on opportunistically sampled regions of the hull ($Z = 4.22$, $df = 34$, $P < 0.001$).

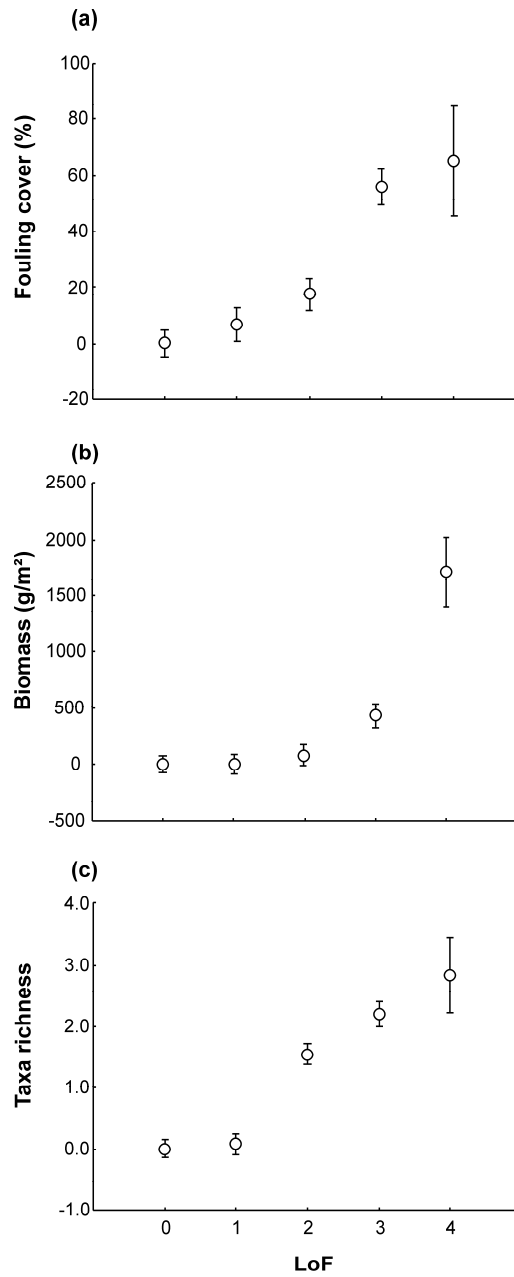


Figure 2.4. Categorical levels of fouling and corresponding (a) fouling cover, (b) biomass, and (c) taxa richness. Error bars denote 95% confidence intervals.

2.4 DISCUSSION

2.4.1 Fouling composition and patterns

Biofouling on commercial slow-movers arriving in New Zealand over the one year study period was neither taxon rich nor extensive, and in fact comparable to that described for merchant vessels visiting New Zealand (richness range = 1 - 11 taxa per vessel, average = 4.8 taxa; Coutts and Taylor 2004). A low incidence of fouling was also described on cargo barges arriving in Hawaii (Godwin 2003; Godwin et al. 2004), yet some fouling studies on slow-movers have described extensive and/or diverse biofouling communities (e.g., Lewis et al. 2006; Coutts and Forrest 2007).

Observations of low fouling extent in the present study was most likely due to the fact that most vessels were sampled within two years of their most recent antifouling coating (average of c. 2 years for barges and c. 16 months for tugs). Furthermore, vessels typically spent short periods of time idle between voyages (i.e., 85% of port visits were < 5 days), thus the window of opportunity for colonisation and growth by local taxa was considerably less than that of vessels with long residency periods (e.g., obsolete vessels, Davidson et al. 2008b).

It appears for slow-moving vessels that differences in location on the hull are likely to be less pronounced than observed on faster moving vessels such as merchant ships (speeds > 20 knots), for which fouling is greater in hydrodynamically protected areas (Coutts et al. 2003; Coutts and Taylor 2004). Conceivably, the forces on a vessel moving at slow speeds (c. 5 knots) are not sufficient to adversely affect (e.g., dislodge, damage) fouling assemblages, such that patterns of fouling across the hull are independent of location. However, our observation of higher fouling on DDSS and niche areas is consistent with most other vessel hull fouling studies (e.g., Godwin and Eldredge 2001; Coutts and Taylor 2004; Coutts and Dodgshun 2007; Davidson et al. 2009), and is intuitive given that antifouling paint was generally in poor condition and was unlikely to contain sufficient active biocides to prevent colonisation by the planktonic propagules of fouling biota (Coutts and Taylor 2004; Yebra et al. 2004; Almeida et al. 2007). Thus, these vessel regions prone to accumulate fouling probably pose the greatest biosecurity risk because they tend to have the greatest number of taxa present (Coutts and Taylor 2004; Davidson et al. 2009).

Surface observations of fouling on the vessels did not appear to be a useful predictor of sub-surface fouling, given that they did not reliably predict fouling levels on painted and unpainted (i.e., dry-docking support strips) surfaces of the hull, nor niche areas (e.g., gratings, intake pipes, anode straps, etc). Essentially, high levels of fouling can occur in niche areas, despite low fouling visible from surface inspection. As niche areas may be of most significance from a biosecurity perspective, a reliable assessment of risk will generally need to be based on in-water inspection rather than surface observation alone. On the other hand we note that high surface fouling will generally reflect a vessel that is heavily fouled overall (e.g., Coutts and Forrest 2007; G. Hopkins pers. obs.), hence surface-based observation can justifiably be used to identify “rogue” vessels that often present a high biosecurity risk (e.g., Piola and Forrest 2009).

2.4.2 Biosecurity risks from NIS on slow-moving commercial vessels

The low number of NIS (five taxa) encountered on the seven vessels sampled in the present study were globally ubiquitous and none were first time records in New Zealand. By comparison, there are some 2000 international merchant vessel visits per year (Dodgshun et al. 2007), for which recent studies indicate a relatively high occurrence of NIS and cryptogenic species (c. 4 per vessel) (Inglis et al. 2010). Hence, merchant vessels conceivably represented a far greater biosecurity risk to New Zealand over the study period.

Nonetheless, the actual risk from the slow-movers sampled in the present study cannot be dismissed as trivial, as a large proportion (c. 59%) of taxa sampled were unknown. Past experience shows that the translocation of relatively unknown species with no history of invasiveness can lead to significant problems. In New Zealand, this was highlighted in 2001 when an unknown didemnid ascidian (later identified as *Didemnum vexillum*; Kott 2002) was discovered on a barge that had been moored for several months and was heavily-fouled (Coutts and Forrest 2007). This species subsequently spread from the barge and is now a fouling pest to aquaculture (Pannell and Coutts 2007).

Although the data are preliminary, repeat sampling in the present study also showed how fouling can change (e.g., biomass increases) over time, conceivably reflecting decreased invasion resistance with age of antifouling paint. While greater fouling may be associated with greater biosecurity risk, risk is also related to voyage history and the

interaction between vessels and source populations of NIS (Floerl and Inglis 2005). A recent example highlighting this point was the discovery of a large number (c. 700) of Mediterranean fanworms (*Sabella spallanzanii*) on the barge *Sea-Tow 80* during diver surveys in Auckland Harbour (November 2009). *Sea-Tow 80* was sampled in the present study (July 2006) > 6 years after being dry-docked, but had relatively low levels of hull fouling comprising solely indigenous taxa. The barge was then dry-docked in August 2006 (i.e., one month after our sampling) and worked in Australia and New Zealand over the following three years. This voyage history included Port Phillip Bay, where there are established populations of *S. spallanzanii* (Hewitt et al. 2004). Hence, there is clearly a range of complex factors that must be considered in order to understand vessel risk.

A number of other examples similarly highlight the potential for significant biosecurity risk to arise from the international movement of fouled tugs and barges (Lewis et al. 2006), oil rigs (Foster and Willan 1979; Hopkins and Forrest 2009) and other towed structures (DeFelice 1999; Apte et al. 2000). This situation parallels the recognised biosecurity risk posed by slow-moving recreational vessels such as yachts (Floerl and Inglis 2005; Piola and Forrest 2009; Inglis et al. 2010).

2.5 CONCLUSIONS

There will always be stochastic processes that determine vessel fouling and related risk from NIS. However, further sampling of commercial slow-movers will improve our ability to predict fouling status and NIS risk profiles. Vessels from a broad range of bioregions and service industries should be targeted, for this purpose, in particular vessels that remain idle for extended periods between voyages (e.g., months rather than days/weeks as in the present study). Although there appear to be very few documented cases in which NIS transported by slow-movers have led to adverse effects in a recipient region, the potential nonetheless exists for these low likelihood events to have high consequences. On the basis that commercial slow-mover movements are only a fraction of global vessel movements, several management options are conceivable. In the New Zealand case, the very low number of slow-moving vessel arrivals each year makes it feasible to assess vessel risk on a case-by-case basis prior to their entry into the country, and to implement appropriate mitigation strategies (e.g., inspections for NIS and treatment where necessary) pre-border.

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Chapter 3

Management Options for Vessel Hull Fouling: An Overview of Risks Posed by In-water Defouling

PREFACE

Several countries have placed restrictions on in-water defouling in recognition of the environmental risks from the release of contaminants associated with antifouling paints, and the potential biosecurity risks associated with such activities. In New Zealand and Australia, guidelines governing in-water defouling activities (ANZECC 1997) are under review, due in part to the lack of suitable land-based facilities available to treat fouled vessels, and recognition of advances in in-water treatment technologies and changes in paint strategies (e.g., non-toxic coatings). A better understanding of biosecurity risks associated with in-water defouling is likely to make a major contribution to the re-drafting of these guidelines, as well as improve management of biosecurity risks posed by biofouling at the border. This chapter contains results of preliminary in-water defouling trials using rotating brushes that led to the study presented in Chapter 4.

This work has been published in a refereed journal and is presented below in identical form. My co-author and thesis supervisor Barrie Forrest reviewed the draft manuscript. The citation for the original publication is:

Hopkins GA, Forrest BM. 2008. Management options for vessel hull fouling: an overview of risks posed by in-water cleaning. *ICES Journal of Marine Science* 65: 811–815

ABSTRACT

Hull fouling has been identified as an important pathway for the spread of non-indigenous marine species. However, the management of associated biosecurity risks has proved challenging. Left unmanaged, a fouled vessel can pose a biosecurity risk through the detachment and dispersal of viable material, and through spawning by adult taxa upon arrival in a recipient port or region. These risks can be effectively managed through the removal of the vessel to land for defouling (e.g., dry-docking). However, alternative methods are needed for small (e.g., recreational) vessels, as well as for large vessels fouled outside their dry-docking schedule. Among the various treatment options, in-water defouling is relatively common, although some countries have placed restrictions on this method because of perceived biosecurity risks. A conceptual framework is presented here that identifies risks posed by in-water defouling compared with alternatives, including no management. Decisions on the appropriate management option will be influenced by many factors, including the species present, the level of fouling, and the time a vessel spends in a recipient region. It is important that any regulatory changes regarding in-water defouling are supported by relevant research that quantifies the risks associated with the various management options.

3.1 BACKGROUND

Marine biosecurity is the protection of the marine environment from impacts of non-indigenous species (NIS), and typically involves pre- and post-border management of vectors and high risk organisms (Hewitt et al. 2004). Given the constraints in controlling NIS once they have established adventive populations, it is clearly preferable to manage pathways of spread to reduce the risk of initial incursion (Bax et al. 2001; Ruiz and Carlton 2003; Hewitt et al. 2004; Finnoff et al. 2007). Of the many pathways for NIS in the marine environment, vessel traffic has been identified as being particularly important (Hewitt et al. 1999; Gollasch 2002), with the main vessel-related mechanisms being ballast water (Carlton 1985; Olenin et al. 2000; Taylor et al. 2007) hull fouling (Lewis et al. 2003; Coutts and Taylor 2004), and fouling of niche areas such as sea chests, intake pipes and gratings (Carlton et al. 1995; Coutts et al. 2003; Minchin and Gollasch 2003; Coutts and Dodgshun 2007).

There has been considerable research globally into treatment solutions for ballast water (e.g., Mountfort et al. 1999; Oemcke et al. 2004). Nonetheless, practical tools to reduce

ballast-water-related introductions are still unavailable, are not completely effective, or have the potential to enhance the survival of some groups of organisms (Taylor et al. 2007). Despite the evidence of biosecurity risks associated with fouling-related mechanisms, the range of opportunities for treatments other than application of antifoulant coatings are poorly understood. The use of tributyltin (TBT) self-polishing copolymer paints has to date been the most successful means of combating fouling on vessels (Yebra et al. 2004). However, owing to the negative impacts of TBT on the marine environment (Stewart et al. 1992), the application of TBT-based paints will be phased out by January 2008. Although other biocidal (e.g., copper) paints are reasonably effective (Floerl and Inglis 2005), their efficacy is limited on vessels that are frequently idle. Similarly, non-toxic hull coatings are likely to become more widespread in their use, but require reasonable vessel speeds (> 20 knots) to remove fouling organisms (Yebra et al. 2004). Along with factors such as poor vessel maintenance, the limitations of present antifouling coatings highlight a need for alternative strategies to manage fouling on vessels, and to reduce the spread of NIS (Brady 2001).

There are several management options when a high-risk vessel (i.e., a vessel fouled with NIS or pest organisms) arrives at a recipient port. Refusing entry (risk avoidance) is arguably the most desirable approach, at least from the perspective of the recipient port or country. In New Zealand this option is supported by relevant biosecurity legislation, but in reality is rarely enforced. Instead, vessel fouling is typically managed through in-water defouling or removal of the vessel to land for defouling (e.g., dry-docking). In-water defouling is often used for small vessels and may be the only alternative for large vessels outside their dry-docking schedule. However, a number of countries have placed restrictions on this approach or are considering doing so. This move is due in part to concerns over the release of fouling organisms to the environment, based on the perception that biosecurity risks will be exacerbated by in-water defouling (ANZECC 1997; Woods et al. 2007). However, there appears to be little information to support this stance, and conceivably there may be situations where biosecurity risks as a result of in-water defouling are less than that from unmanaged vessel fouling. Here, I discuss this issue and present preliminary findings from an experimental evaluation of in-water hull defouling systems.

3.2 RISKS FROM IN-WATER DEFOULING

Understanding risks from in-water hull defouling requires an understanding of baseline (i.e., unmanaged) hazards and their magnitude, and how these change with management. The key sources of hull-fouling risk with and without management are represented conceptually in Figure 3.1.

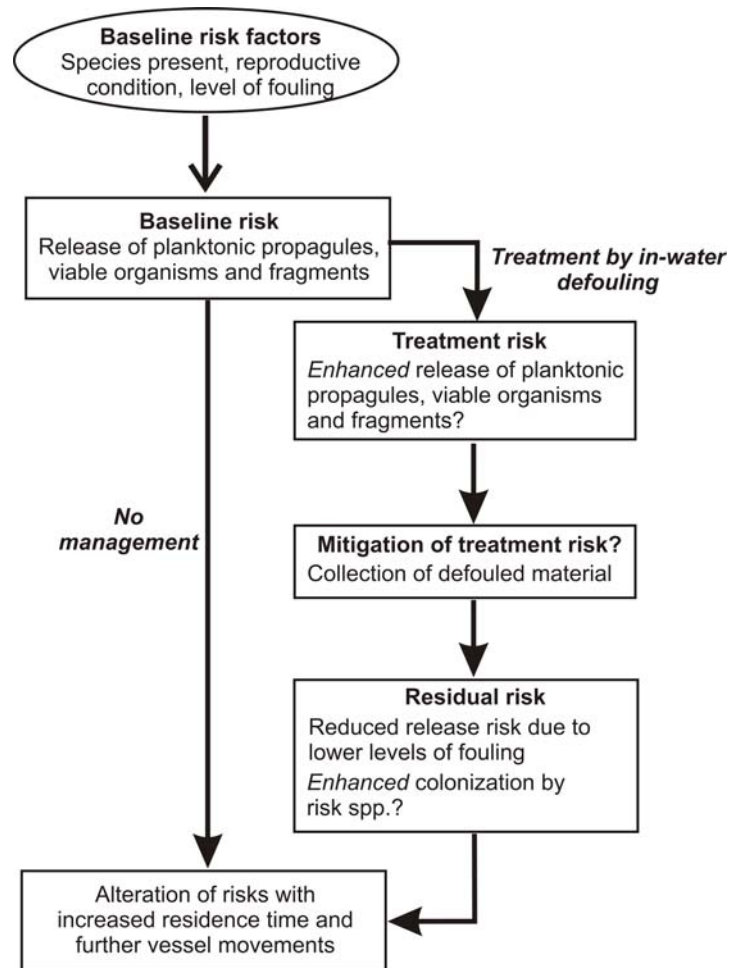


Figure 3.1 Conceptual diagram of the risks posed by a fouled vessel if left unmanaged compared with the risks from in-water hull defouling.

3.2.1 Unmanaged risk

Given a suitable habitat, biosecurity risks from hull fouling primarily arise when competent pest organisms are released into a recipient region in the form of adult life stages or planktonic propagules (Figure 3.1). For some species (e.g., certain macroalgae, colonial bryozoans, compound ascidians, sponges), dispersal of fragments may also be a pathway for establishment in a new habitat (Valentine et al. 2007). For planktonic propagules, it is well recognized that factors such as increased density or frequency of release are related to invasion success (Ruiz et al. 2000), and the same concept of “propagule pressure” applies to adult organisms or fragments (Lockwood et al. 2005). Clearly, therefore, biosecurity risk will depend on various characteristics of the fouling assemblage (e.g., species composition, dispersal modes, reproductive status) and the level of fouling. Such factors are in turn determined by numerous other processes that occur between source and recipient regions, with the fouling assemblage that arrives in a new region reflecting factors such as voyage and maintenance history of the vessel, species assemblages present in source regions, and the survivorship of the fouling community during vessel passage (Hayes 1998).

Several other factors contribute to risk from the point of a vessel’s arrival. Fouling organisms may release viable propagules in response to new environmental cues (e.g., altered salinity or temperature) in a recipient region, and inoculate surrounding habitats including artificial structures (Apte et al. 2000; Minchin and Gollasch 2003). Hull-fouling organisms or viable fragments may also detach from contact with wharf piles and other structures, or through other mechanisms (e.g., predation, water currents). Assuming suitable environmental conditions, risk is likely to increase with the residence time of a vessel in a recipient region for all release modes (Floerl and Inglis 2005), e.g., by providing attached organisms sufficient time to become reproductively viable. However, it is worth noting that some vessels may also visit a port or region where suboptimal environmental conditions prevail (e.g., low salinity, high turbidity), and in such cases, release risks may be mitigated through die-off of the fouling organisms.

3.2.2 Risks posed by in-water hull defouling

Relative to no management, the risk of in-water defouling reflects the combined risk from the defouling operation itself, and from the residual risk posed by the reduced level of hull fouling (Figure 3.1). In-water defouling operations typically involve

mechanical removal of fouling and can be undertaken using a range of devices, depending on the vessel size, build composition (i.e., wood, steel, fibreglass), and the type of paint coating used. For example, a small recreational yacht is likely to be defouled using plastic or metal hand-held scrapers (which may take several hours), whereas a large merchant ship is more likely to be defouled over 1–2 d using diver-operated devices such as rotating brushes. Most in-water hull defouling devices are not designed to defoul the entire hull of the vessel, and niche areas (e.g., sea chests and intake pipes) often remain untreated between dry-docking periods. In most cases, defouled material is not retained by the defouling devices and may settle on natural seabed habitat or artificial structures adjacent to the vessel, or be more widely dispersed by currents.

Release of viable organisms

The physical disturbance of fouling communities by in-water defouling methods may trigger the release of viable gametes or propagules (ANZECC 1997). If this occurs, in-water defouling may increase the biosecurity risk relative to natural spawning cues. However, the likelihood of this is currently unknown and is the subject of my ongoing research. Fouling organisms or viable fragments dislodged during manual defouling may survive and establish (ANZECC 1997). This is potentially a significant issue for in-water methods that do not retain defouled material. However, recent experimental work reveals that intact organisms can also be dislodged from the hull and lost to the environment when a vessel is treated by devices that aim to retain all defouled material (Chapter 4).

Two diver-operated rotating brush systems were tested on a fouled vessel and on two settlement plate shapes (flat plates, and curved plates to mimic a vessel's hull) with varying levels of biofouling. Each system consisted of a single brush rotating at 700 rpm, coupled with a purpose-built suction and collection capability. In pilot trials, up to 12% (mean = 5.6%, S.E. = 2.3%) of defouled material was not retained, with retention less on curved surfaces (see Chapter 4 for the complete study). Most biofouling not captured by the systems was crushed and fragmented, but viable organisms such as barnacles and hydroids were almost always present. Moreover, as fouling levels became more advanced, larger calcareous organisms, for instance serpulid polychaetes, were resistant to the rotating brushes and remained relatively intact on the experimental surfaces (Figure 3.2). Other sources of risk identified during these trials

included the unintentional detachment of fouling organisms by divers operating the devices (e.g., by divers' fins) and by equipment associated with the rotating brush devices, e.g., hoses and ropes. Results of a more comprehensive experimental study, including an evaluation of seasonal changes in efficacy of defouling and retention, are currently being analysed. Factors affecting the survival of this defouled material are also being investigated, including the extent of damage sustained during the defouling process, the environmental conditions of the receiving environment (e.g., substratum type, water temperature, salinity, light), and the biological processes in recipient ports, for instance predation and competition.

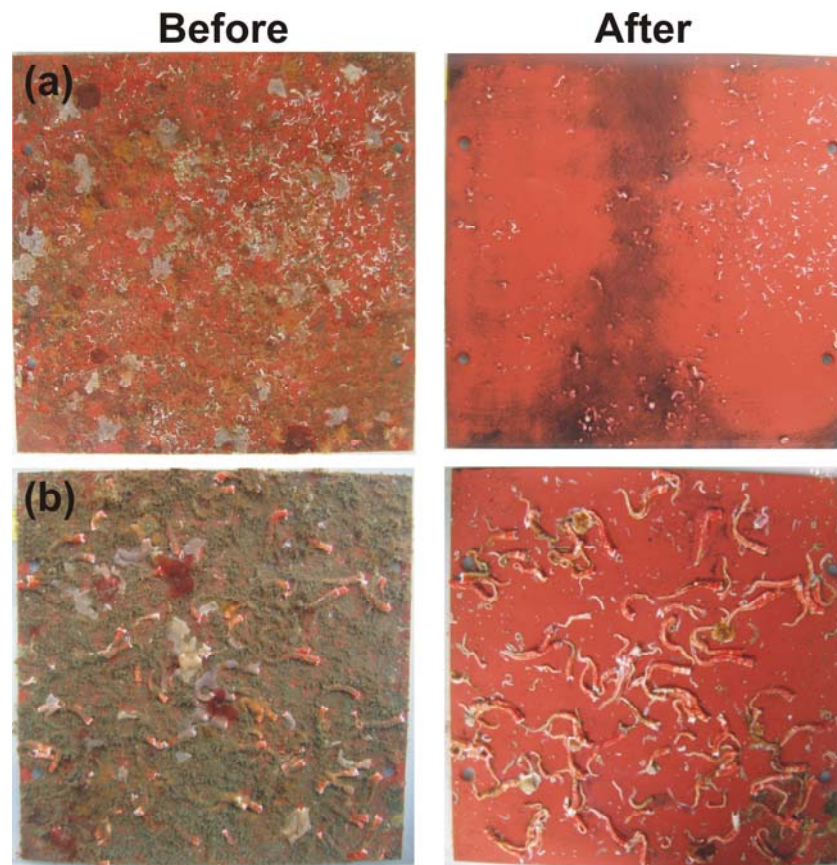


Figure 3.2 Photographs of settlement plates (350 × 350 mm) before and after treatment by rotating brushes after (a) 3 months and (b) 12 months of fouling.

Enhanced recolonisation

The removal of fouling from a vessel without the reapplication of antifouling paint may increase the susceptibility of the surface to new fouling, so exacerbating future biosecurity risk. For example, Floerl et al. (2005) found that defouled boat surfaces in a tropical region of Australia had up to six times more recruitment than surfaces that had been either chemically sterilized or contained intact fouling assemblages. Several theories were advanced to explain this finding, including: (i) the liberation of chemical or physical cues for settlement during defouling; (ii) the predation of settling larvae by the existing fouling communities; (iii) larval avoidance behaviour; and (iv) that the presence of a fouling community may provide resistance against recruitment by taking up space.

3.3 ALTERNATIVES TO IN-WATER DEFOULING

Regular defouling of a vessel hull is an effective biosecurity management practice that minimizes the transfer of fouling organisms, in particular when followed by the reapplication of antifouling coatings within recommended time-frames (Floerl and Inglis 2005). This typically takes place during dry-docking, following the physical removal of hull-fouling organisms, using devices such as water blasters and scrapers. Biosecurity risks posed by the use of dry-docking or haul-out (e.g., travel lift) facilities are likely to be less significant than in-water defouling methods, and can be managed through the installation of barriers such as filters and containment tanks, to prevent defouled material re-entering the marine environment (Woods et al. 2007).

In-water encapsulation techniques have recently been developed to reduce the risks posed by hull fouling. For example, fouled vessels in New Zealand have been wrapped in plastic (encapsulated) *in situ* in an attempt to eliminate (by creation of anoxic conditions) the solitary ascidian *Styela clava* and the colonial ascidian *Didemnum vexillum*, based on approaches described by Coutts and Forrest (2005, 2007). Although these techniques appear promising, the development of anoxic conditions may be too slow to result in mortality where the time-frames of visiting vessels are short (< 48 h). Mortality could be accelerated through the addition of chemical agents to the encapsulated seawater (Coutts and Forrest 2005), although collateral damage to the wider ecosystem would need to be considered. An advantage of encapsulation methods

is that risk organisms are contained once the wrap is in place, although fouling material may be detached during the wrapping process (Denny 2007).

3.4 CONCLUSIONS AND FUTURE DIRECTIONS

Regulatory moves to ban or restrict in-water defouling need to account for the possibility that risks posed by this method may, under certain circumstances, be less than those from no management intervention. For example, in the case of domestic vessel risk and the management of internal borders, a restriction on in-water defouling could act as a disincentive to vessel operators to defoul, especially when faced with potentially expensive alternatives (e.g., dry-docking or slipping). Clearly in such instances, unmanaged biosecurity risks may be exacerbated and exceed the risks posed by in-water defouling, especially where biosecurity practices are adopted.

Although understanding the risks from in-water defouling and the development of other defouling methods will contribute to biosecurity programmes, the effective management of hull-fouling risks will ultimately require a broader suite of measures. These include the development of specific management programmes for vessels visiting high-value areas (Lewis et al. 2006), educating and promoting awareness among vessel operators, research to understand better the factors contributing to vessel risk, and targeted surveillance programmes for vessels or vessel types identified as high risk. Decisions on subsequent management options for high-risk vessels will need to consider many factors, including the fouling species present, the level of fouling, the residence time of a vessel in a recipient region, and the risks from treatment.

In relation to in-water defouling, it is important that any regulatory changes be supported by research that quantifies the relative risks associated with the various methods. To achieve this, future research should focus on gaining a better understanding of environmental factors affecting the survivorship of defouled material, the effects of defouling disturbance on propagule release, and the colonization of recently defouled surfaces by high-risk species. The relative efficacy, costs and benefits of other in-water techniques, such as hull wrapping, also need to be quantified.

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Chapter 4

The Effectiveness of Rotating Brush Devices for Management of Vessel Hull Fouling

PREFACE

To address one of the issues identified in the conceptual framework presented in Chapter 3, I evaluated the efficacy of two diver-operated rotating brush devices that had been designed to retain defouled material following removal. I aimed to establish whether either of the devices was likely to be capable of removing and retaining 100% of biofouling from a vessel hull. At the time of this study, these devices were arguably the most biosecure in-water hull cleaning devices in the world.

Ashley Coutts provided significant input into experimental design, and Barrie Forrest and Jonathan Gardner provided useful comments on this chapter, which was accepted for publication in the international journal *Biofouling* in May 2010.

ABSTRACT

The present study tested two diver-operated rotating brush systems, coupled with suction and collection capabilities, to determine their efficacy in the management of vessel biofouling. Both rotating brush systems proved effective ($> 80\%$) in removing low-to-moderate levels of fouling from flat and curved experimental surfaces (Perspex plates); however, performance was generally poorer at removing more advanced levels of fouling. In particular, mature calcareous organisms were relatively resistant to the rotating brushes, with a high proportion (up to 50%) remaining on plates following treatment. On average, $> 95\%$ of defouled material was collected and retained by both systems. The amount of lost material generally increased when treating curved plates with increasing biomass, whereas the material lost from flat plates was typically less and remained relatively constant throughout the trials. The majority ($> 80\%$) of fouling not captured by the systems was crushed by the brushes (i.e., non-viable); however, a diverse range of viable organisms (e.g., barnacles, hydroids, etc) was lost to the environment during the defouling trials. When defouling a vessel, unintentional detachment of fouling organisms is likely to be high through physical disturbance by divers operating the devices and by associated equipment (e.g., hoses). Furthermore, residual biosecurity risks are also likely to remain due to diver error, persistent fouling remaining on treated surfaces and the inaccessibility of niche areas to the brush systems. To address these limitations, further research into alternative treatment methods is required.

4.1 INTRODUCTION

The accumulation of biofouling on vessel hulls has important implications for maritime activities. From an operational perspective these include a reduction in vessel speed and manoeuvrability, causing increased fuel and maintenance costs (Townsin 2003; Tribou and Swain 2010). The role of hull fouling in the inadvertent spread of non-indigenous species (NIS) has also been recognised for over a century (Chilton 1910; Bishop 1951). However, it is only in more recent times that scientific attention has been given to the transport of NIS (both historical and contemporary), accompanied by a ground-swell of interest in the role of hull fouling on modern-day vessels as a pathway for their spread (Hewitt et al. 1999; Gollasch 2002; Lewis et al. 2003; Ruiz and Carlton 2003; Coutts and Taylor 2004; Dafforn et al. 2008; Piola and Johnston 2008; Yamaguchi et al. 2009).

A number of studies have described the amount and composition of fouling on a range of vessel types, including merchant vessels (Coutts and Taylor 2004; Davidson et al. 2009), yachts (Floerl et al. 2005a) and barges (Lewis et al. 2006; Chapter 2). In addition to fouling on the general hull area, relatively high levels of fouling can occur in “niche” areas such as sea chests, intake pipes and gratings. Hence such areas are recognised as being of particular importance in the transfer of NIS (Carlton et al. 1995; Coutts et al. 2003; Minchin and Gollasch 2003; Coutts and Dodgshun 2007). A range of other vessel-related mechanisms are recognised as being of importance in the transfer of NIS; including ballast water (Carlton 1985; Carlton and Geller 1993; Ruiz et al. 2000), contamination of bilge water (Darbyson et al. 2009), anchors (e.g., because of entrained sediments) and associated equipment such as fishing gear (Acosta and Forrest 2009). However, various studies highlight biofouling as the most important mechanism for the spread of NIS. For example, Hewitt and Campbell (2008) estimate that, based on their life-history characteristics, 55–69% of the c. 1780 NIS detected in ports and harbours around the world are likely to have been introduced by vessel hull fouling.

Recognition of the ongoing importance of modern-day hull biofouling is likely to have significant implications for vessel operations, as regulatory agencies and the International Maritime Organization (IMO) seek to develop and implement management requirements (MAFBNZ 2009). Accordingly, there is considerable interest in the parallel development of management tools that are complementary to the use of antifouling coatings, or can be used in situations where there is an immediate need to reduce fouling levels. The latter could include a desire to meet operational performance requirements (e.g., reducing drag to increase fuel efficiency) or a mandatory requirement to address a biosecurity threat. Refusing entry to vessels presenting an unacceptable biosecurity risk is arguably the most effective approach from a technical perspective, but in reality has rarely been enforced. In Chapter 3, several additional management options for vessels considered high risk from a biosecurity perspective were identified, which apply equally to vessels having levels of fouling that affect operational performance. For smaller vessels, slipping and treatment on land may be an option, but for larger vessels that cannot be removed to land or readily dry-docked, in-water cleaning may be the only available option.

Traditionally, in-water cleaning methods have involved the manual removal of fouling by divers; for example using plastic or metal handheld scrapers or brushes. In most

cases, defouled material is not collected and has the potential to exacerbate biosecurity risk (Chapter 2). Primarily as a result of the potential risk from the release of this material, as well as from paint fines containing toxic substances, a number of countries have placed restrictions on this type of in-water cleaning or are considering doing so. The development of more sophisticated in-water cleaning technologies that have a low ecological risk is in its infancy. In New Zealand, a vacuum system designed to remove an estimated 26 tonnes of fouling (including the invasive ascidian *Didemnum vexillum*) from the hull of a barge was only 80% effective in reducing biomass (Coutts 2002). Although the majority of defouled material was captured by the suction device, the amount of loss and its ecological risk was not quantified. An alternative strategy involves the *in situ* plastic wrapping of vessel, a system that has been developed and widely applied in New Zealand (Coutts and Forrest 2007; Piola et al. 2009) and Australia (Aqueal 2009), and has the advantage of containing the biofouling. This method has been applied successfully to vessels up to 60 m in length but not to larger merchant ships (Denny 2007).

Two New Zealand companies have also independently developed diver-operated rotating brush systems that are designed to clean vessel hulls to improve fuel efficiency, and collect the defouled material to minimise risks to the environment. Both systems have been in commercial operation for more than five years now, yet there still exists only a few devices worldwide that have been designed to collect fouling material during removal (Bohlander 2009). Gaining an understanding of the efficacy of these systems in removal of biofouling, and their success in retaining defouled material, is an important step towards the development of effective and environmentally friendly methods to manage vessel biofouling.

This chapter describes experimental studies that collectively aimed to: (i) assess the performance of the brush systems in relation to the level of fouling, morphology of the fouling assemblage, and the curvature of the defouled surface; and (ii) characterise the nature, extent and viability of organisms in the defouled material lost to the environment from the brush capture systems. The biosecurity risks that potentially arise as a result of the defouling process are also discussed.

4.2 METHODS

4.2.1 Description of systems tested

The project initially tested the two brush systems at an experimental site on fouled settlement plates, with a subsequent trial conducted on a vessel. Brush System A consisted of a PhosmarineTM brush unit and hydraulic pump, while System B consisted of a CharlynTM hydraulic motor running a commercial road-sweeping brush head (Table 4.1). Both brush units were fitted with a purpose-built shroud that was designed to enhance the retention of defouled material (Figure 4.1), which was pumped to a collection facility for land disposal.

Table 4.1 Specifications of the rotating brush devices and pumping/filtration units tested.

Specifications	System A	System B
Rotating brush unit		
Make of unit	Phosmarine	Purpose built
Brush type used during trials	406 mm wide brush head. Bristles 50 mm long, 1 mm in diameter, 36 bristles per square inch	350 mm wide brush head. Bristles 50 mm long, 2 mm in diameter, 36 bristles per square inch
RPM during typical use	400 rpm	700 rpm
Shroud		
Materials	Aluminium with PVC skirt	Stainless steel with rubber flap
Diameter (overall)	c. 600 mm	c. 1000 mm
Generator		
Make/model	Phosmarine (Hydraulic)	Purpose built
Minimum horsepower required	12	40
Pump unit		
Make/model	Stanley HP8	Casappa multi pump
Filtration system		
Make/model	Purpose built	FSI model BFNP12 316SS
Sizes available	30 to 1200 µm	1 µm upwards

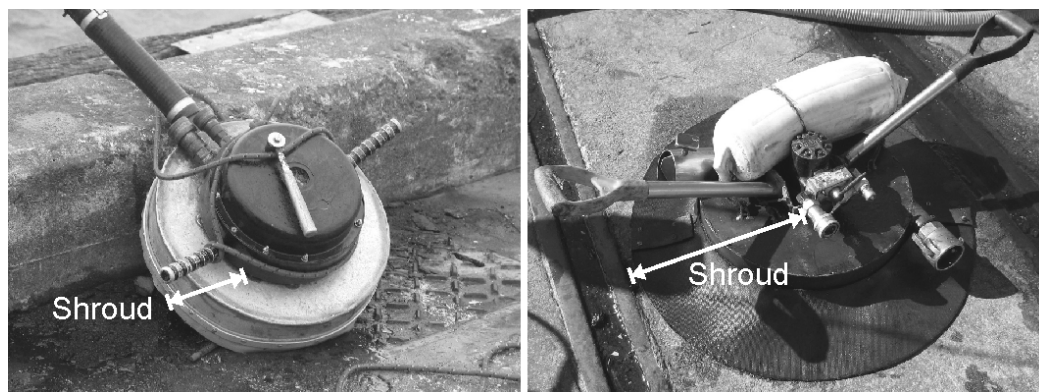


Figure 4.1 Rotating brush devices (suction/collection systems not pictured) used in the trials: System A (left) and System B (right).

4.2.2 Defouling trials on settlement plates

Experimental design

The performance of the brush systems was assessed by their ability to remove and collect fouling material from pre-fouled settlement plates over a range of fouling levels. Experimental plates were favoured over replicate vessel trials, as the uniform fouling achieved on plates within each deployment period allowed for robust statistical analyses. Each system was tested on flat and curved (radius = 15 mm) Perspex settlement plates (350 x 350 x 4.5 mm) to mimic the main surfaces encountered on a vessel hull. Plates were coated with a non-toxic paint (Plasti-kote® T-19 Red Oxide Primer) to mimic the surface colour and texture of a painted vessel hull, but had no antifouling or fouling release properties.

In April 2006, c. 200 settlement plates were suspended vertically beneath a commercial wharf in Wellington Harbour ($-41^{\circ} 16.368' \text{ S}$, $174^{\circ} 47.291' \text{ E}$) at a depth of 3-5 m. Defouling trials were undertaken every three months commencing in July 2006. During the first trial, both systems were tested on settlement plates that had three months of fouling present (i.e., fouling growth from April to July 2006). During subsequent trials, the systems were applied to plates that had been deployed since April 2006, thus providing an opportunity to trial plates with more advanced fouling communities under the assumption that fouling biomass and diversity would increase over time.

Sampling procedure

Prior to the trials, plates retrieved by divers were photographed, drained (standardised to two minutes) and weighed. Plates were then returned to the water and recessed 4.5 mm into a metal backing plate (1.8 x 0.8 m) using screws, in order to achieve a flush surface (Figure 4.2). Two metal backing plates (one for each plate shape) were used to mimic the hull of a vessel and were fixed to a wharf pile approximately 3-5 m below the surface. The plate was then subjected to the defouling trial ($n = 3$ per combination of treatment factors). Control plates to test for handling effects ($n = 3$, for each shape/deployment period combination) were subjected to the same procedure as defouled plates, minus treatment by the brush devices.

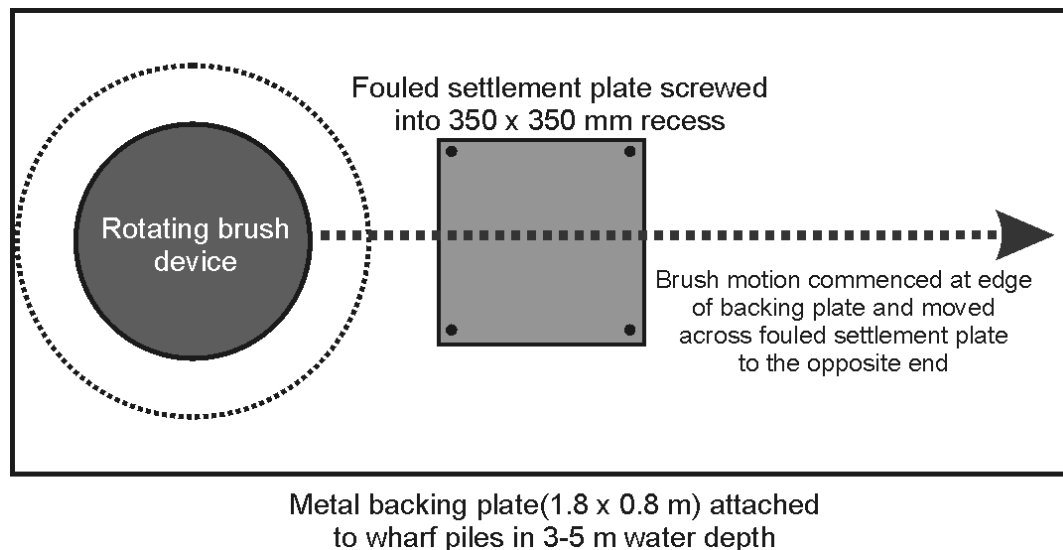


Figure 4.2 Flat and curved metal backing plates were used to replicate representative areas of a vessel hull.

For each defouling treatment, the operator (a commercial diver) started the rotating brush in motion on the metal backing plate and then slowly moved the brush system across the settlement plate (c. 30 m per minute), continuing to the opposite side of the backing plate (Figure 4.2). Both systems used the same brush grade/type throughout the project (Table 4.1). Defouled material not captured by the brush systems that would have been lost to the environment was collected using large hoop-shaped nets (60 μ m mesh size) manoeuvred by divers during the trial. All collected material was weighed

and preserved, and later determined by microscopical examination as being viable or non-viable according to the nature and extent of physical damage. The precautionary principle was adopted (i.e., organisms were considered viable) where the extent of damage was difficult to determine, in particular for species that can establish and reproduce from fragments (e.g., colonial ascidians, hydroids, bryozoans). Note that in some cases, it was not possible to determine whether taxa collected by divers were present or absent on the plates prior to treatment, given the different level of taxonomic resolution applied to the plates (see below).

The brush systems were cleaned and flushed between treatments. After treatment, each plate was again photographed, drained and weighed. Plates were then re-suspended beneath the wharf for two weeks to determine the longer term effects of treatment (compared with immediate mechanical effects), after which they were re-photographed and preserved to assist with image analyses. Images of plates two weeks post-treatment were used as the endpoint against which the efficacy of the brush systems in removing fouling biomass was measured.

Assessing treatment performance

Changes in species richness and percent cover were estimated from differences in pre-treatment (before) versus two week post-treatment photos of the settlement plates. Images were rectified in ArcMap 9.2 GIS software (ESRI, Redlands, CA, USA) and a systematic 49-point grid (7 x 7) was superimposed. Taxa present beneath each point were identified and entered into a database. A 1 cm perimeter along the edge of each plate was omitted from counting to control for possible edge effects.

Due to the inherent difficulties of identifying certain organisms to species level from photographs (e.g., colonial ascidians, bryozoans and hydroids), some were grouped to a coarser taxonomic resolution (e.g., Genus or Family), as necessary. Aggregation of different taxa based on gross morphology was considered appropriate in this study, given that the mechanical effects of the devices tested were expected to depend on gross morphological characteristics shared by species within broad taxonomic groups. Preserved control plates were used to assist with the identification of taxa and post-treatment plates were inspected to confirm whether any organisms surviving were fully intact. The number of taxa present on the entire plate was also counted to give total

richness. Tubeworm remnants were included in the % cover tally, but were not included in species richness tallies because the organisms were no longer alive.

4.2.3 Trials on a fouled vessel

The settlement plate defouling trials provided a robust method for comparing the efficacy of the two rotating brush systems on plates (flat and curved) that were designed to mimic the main surfaces of a vessel hull. As a comparison with the plate trials, the efficacy of the two systems was tested on a moderately fouled vessel coated with a copper-based self-polishing antifouling paint to provide insight into practical aspects of implementing the method at a commercial scale. The vessel trial was undertaken in April 2007 on a 47 m squid fishing boat *Pacific Wind*. Both systems were tested on six 1 m x 1 m curved regions of the steel hull toward the bow of the vessel, with photographic images taken before and after the trials. During the vessel trials, fouling levels on areas of the hull treated were less than those observed on settlement plates that had been deployed for three months during the plate trials. Scrapings were taken from several regions of the hull to identify the fouling taxa present and assist with photo identifications. During the trials, material not captured by the brush systems was collected by divers using the same methods as in the plate trials.

4.2.4 Statistical analyses

Control plates used to test for handling and other effects were not significantly different from before to after the trials ($P \gg 0.05$); therefore, control data were removed from all subsequent analyses of treatment effects. Data were tested for homogeneity of variances and normality using STATISTICA Version 7 (StatSoft Inc., Tulsa, OK, USA), and dependent variables were $\log(x+1)$ transformed where necessary. Multiple linear regression (using Gaussian and Poisson distributions) was used to determine the effect of fouling extent, plate shape and rotating brush device on reductions in fouling cover and taxa richness. The Akaike Information Criterion (AIC) was used for model selection, and the model of best fit for each analysis was validated by plotting residuals (Zuur et al. 2007).

4.3 RESULTS

4.3.1 Temporal change in fouling communities on settlement plates

Fouling percent cover was extensive on flat (mean = 98.3%, SE = 0.3%) and curved (mean = 95.6%, SE = 0.6%) settlement plates within the first three months of deployment and remained high (c. 98 - 99%) throughout the remainder of the deployment (Figure 4.3). Richness on both flat and curved plates ranged between 8 and 11 taxa, with highest mean richness values recorded on plates deployed for 12 months, indicating their more advanced stage of fouling. During the first three months of deployment, barnacles, hydroids, bryozoans (encrusting and branched), ascidians (colonial and solitary) and serpulid polychaetes colonised the plates, with barnacles being the most numerically dominant. After six months, erect bryozoans and hydroids increased in biomass and calcareous tubeworms (in particular *Galeolaria hystrix*) became more prominent. Fouling community biomass peaked after 9 months (Figure 4.4), with calcareous tubeworms and colonial ascidians (e.g., *Didemnum* sp.) becoming well established on plates. Fouling biomass on settlement plates decreased between 9 and 12 months. This effect was most pronounced with curved plates (16.6% reduction in biomass, Tukey's HSD $P < 0.001$), whereas the reduction on flat plates was not significant (6.5% reduction, Tukey's HSD $P = 0.280$). Inspection of control plates revealed that the reduced biomass was largely due to a reduction in branched bryozoans (particularly *Bugula neritina*). Over the same time period, juvenile red algal taxa became more conspicuous.

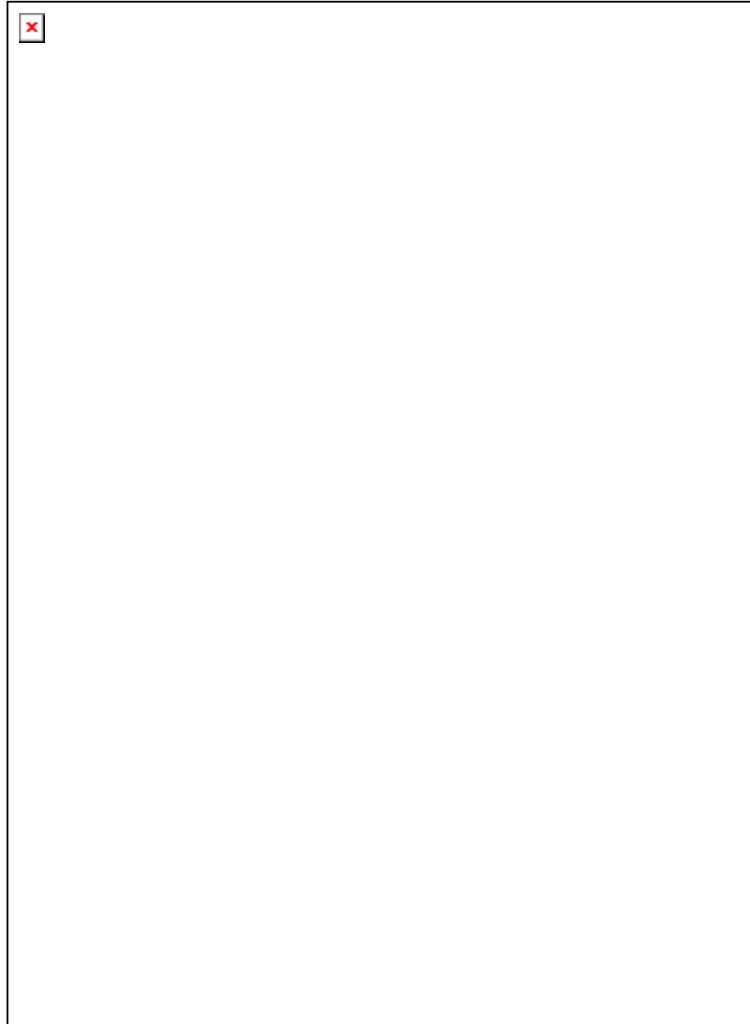


Figure 4.3 Changes in pre-treatment cover (% , \pm 1SE) and richness on (a) flat and (b) curved settlement plates with increasing deployment time.

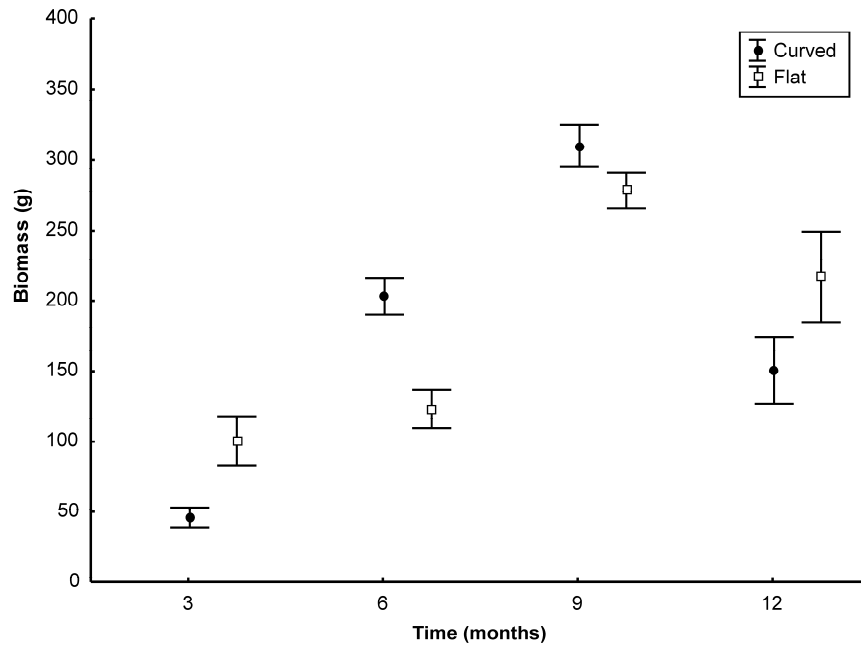


Figure 4.4 Average fouling biomass ($\text{g} \pm 1\text{SE}$) on flat and curved settlement plates with increasing deployment time (months).

4.3.2 Defouling efficacy on experimental plates

Changes in fouling cover

During the first trial (deployment time = 3 months), System A removed on average approximately 90% of fouling cover from flat and 93% from curved plates (Figure 4.5). The performance of System B was comparable, with a mean reduction of fouling percent cover of 88% and 89% for flat and curved plates, respectively. Performance on plates with 6 months of fouling was comparable to 3 months (Tukey's HSD $P \gg 0.05$); however, the efficacy of both systems generally decreased as the plates became more fouled with the 6 – 12 month deployment. In particular, fully intact and/or partially damaged calcareous tubeworms (e.g., *Galeolaria hystrix*, *Hydroides elegans* and spirorbids) remained on plates that had been deployed for a period of 9 months or more. As fouling became more advanced (i.e., deployment time 9 and 12 months), System A performed significantly better on curved settlement plates than on flat plates (ANOVA, $F_{3,32} = 6.44$, $P = 0.002$).

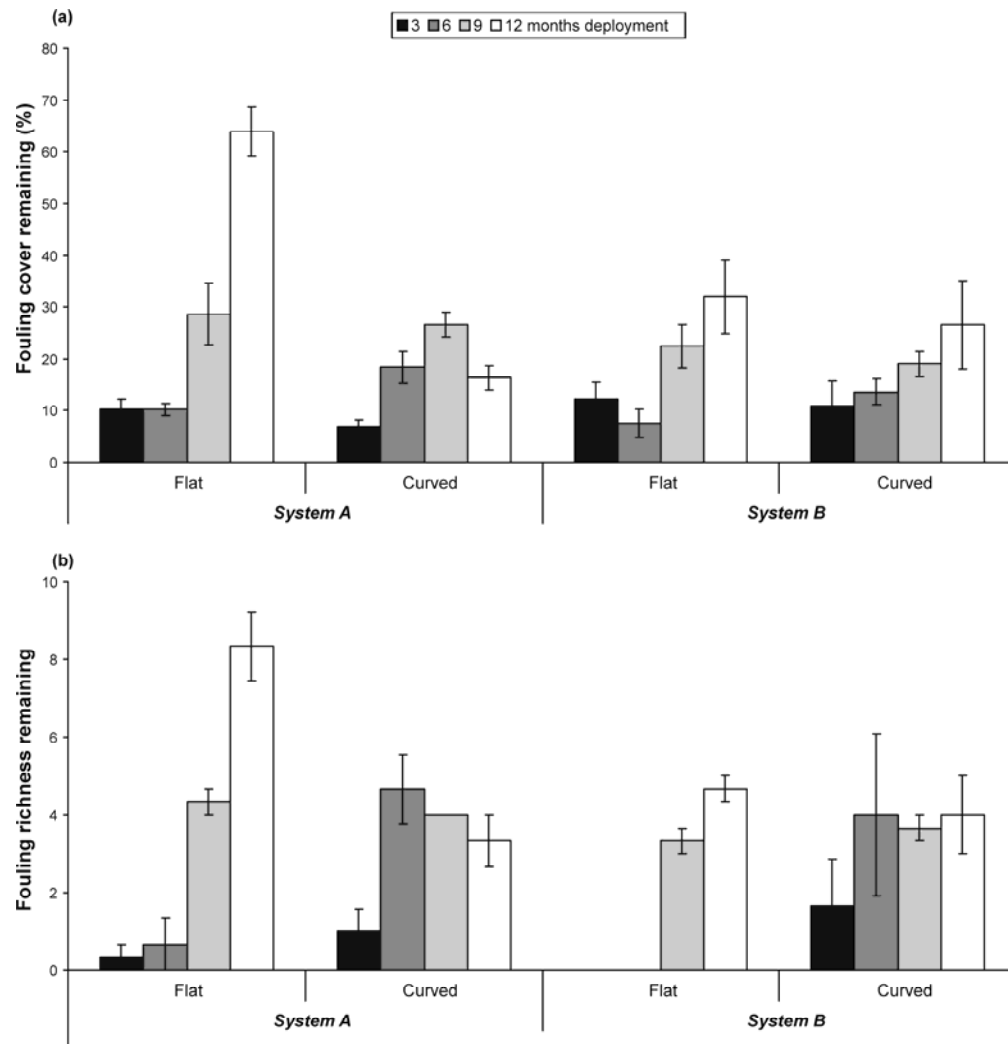


Figure 4.5 Average (\pm 1SE) fouling cover (a) and richness (b) remaining on settlement plates following defouling by two independent rotating brush systems with increasing deployment time.

With increasing deployment time, the cover of soft and/or erect (herein referred to as soft/erect) fouling organisms generally increased on plates, while the cover of hard and/or encrusting (hard/encrusting) organisms either varied (flat plates) or decreased (curved plates) (Figure 4.6). Both systems were highly effective ($> 95\%$ reduction) in removing soft/erect fouling from flat and curved settlement plates that had been deployed for 9 months or less (Figure 4.6). On average approximately 20% of soft/erect organisms remained on flat plates treated by System A that had been deployed for 12 months, compared with 100% removal achieved by System B. Both systems removed

100% of hard/encrusting fouling from plates that had been deployed for three months. Efficacy decreased as overall fouling became more advanced, with up to 61% (mean = 40.1%) of hard/encrusting taxa remaining on flat plates that were defouled by System A after 12 months (System B averaged 21.8%).

There was no strong pattern evident in the percent cover of tubeworm remnants remaining on flat settlement plates over the four deployment times tested (range = 3.4 and 19.7%), with an average of 8.2% and 12.4% for System A and B, respectively (Figure 4.7). Tubeworm remnant cover was within a comparable range following treatments on curved plates (0.7 to 16.3%), with higher levels present on plates deployed for longer times. An exception to this trend (for both systems) was plates that had been deployed for 12 months, which generally had levels less than plates that had been deployed for three months. This corresponded with an increase in the density of fully intact tubeworms remaining on settlement plates for the 12 month deployment time (i.e., impervious to treatment) following defouling (refer Figure 4.7).

Changes in taxa richness

System A removed on average 96% (SE = 3.7%) of taxa from flat plates and 89% (SE = 6.4%) from curved plates during the three month efficacy trial (Figure 4.5). System B performance was comparable, removing 100% (SE = 0%) of taxa from flat plates and 80% (SE = 15.2%) from curved. For flat plates, performance of both systems steadily decreased as fouling became more advanced (i.e., with increasing deployment time). By contrast, performance of both systems in reducing taxa richness on curved plates declined in the period of 3 - 6 months deployment, but was relatively constant thereafter (ANOVA, $F_{3,32} = 8.872$, $P = 0.002$).

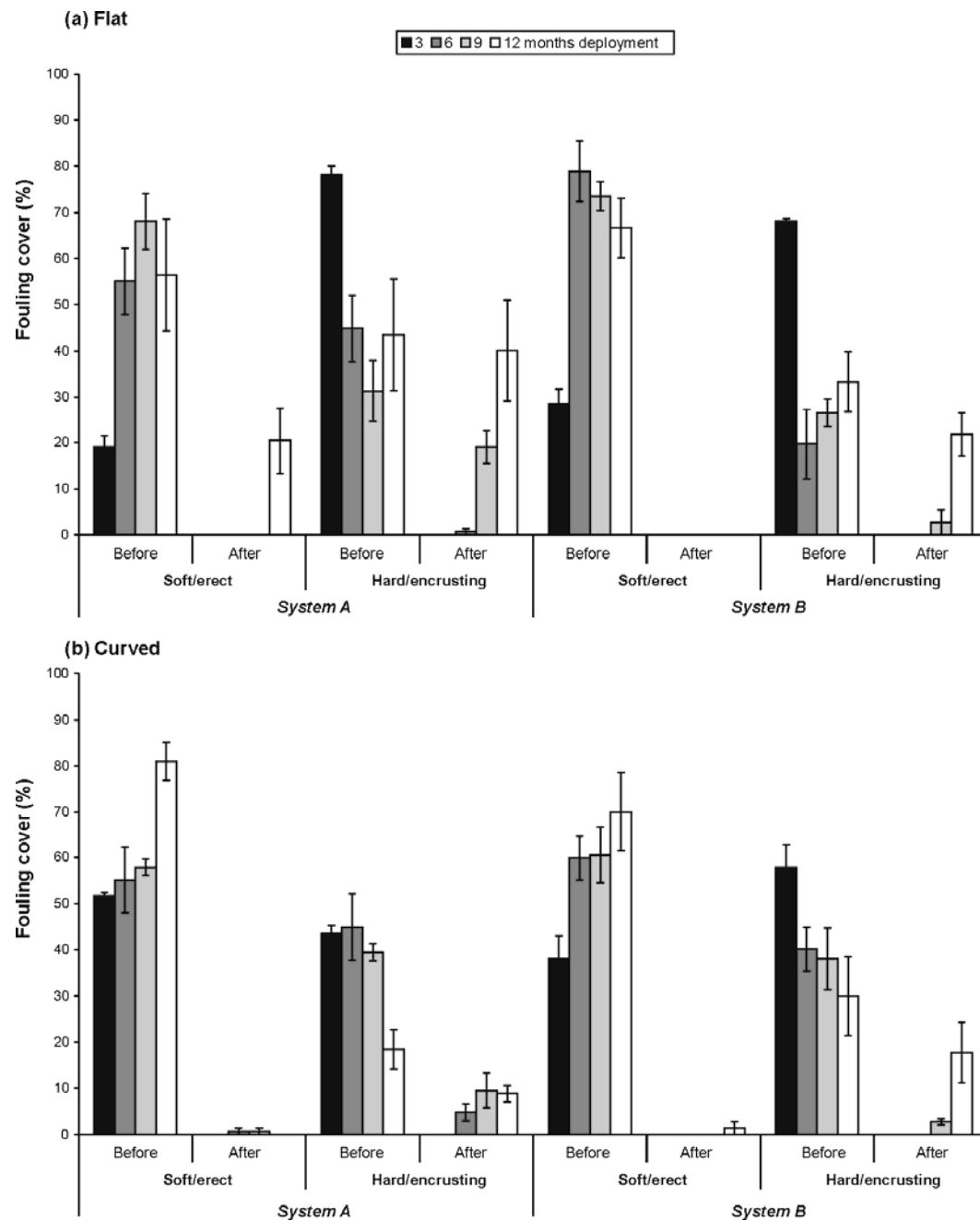


Figure 4.6 Percentage ($\pm 1SE$) of soft/erect and hard/encrusting organisms on settlement plates before and after (2 weeks post-treatment) defouling treatments with increasing deployment time.

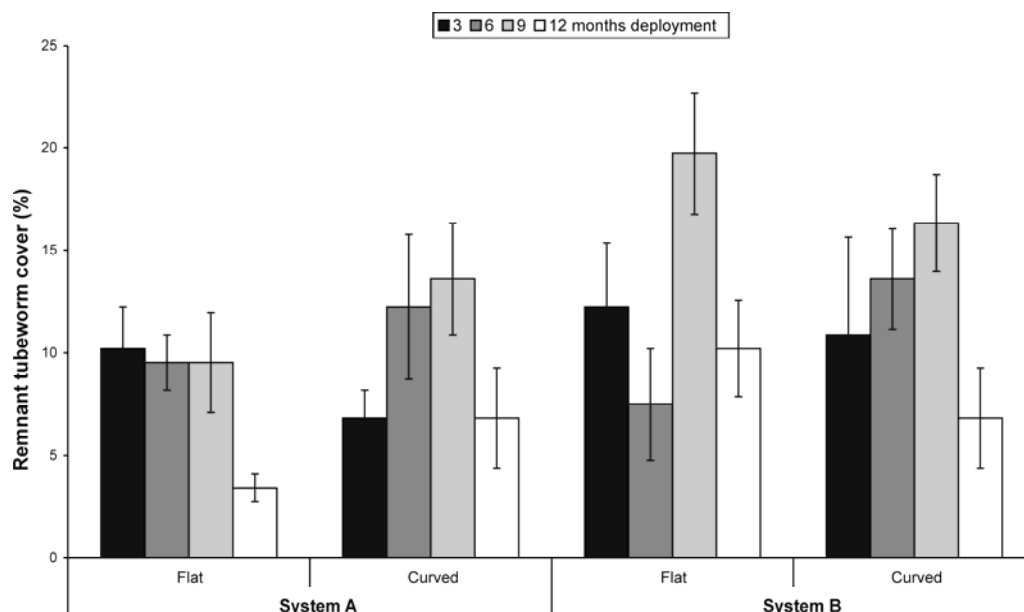


Figure 4.7 Percent cover (\pm 1SE) of tubeworm remnants remaining following defouling treatments on flat and curved settlement plates with increasing deployment time.

4.3.3 Retention efficacy and loss of viable material

Both brushes systems performed relatively well in retaining defouled material, with minor differences apparent that were not statistically significant (ANOVA, $F_{1,46} = 0.139$, $P = 0.711$). In both cases the mass of defouled material (viable and non-viable) that was not retained (i.e., lost material) represented a small proportion (mean 3.8%, range 0-12%) of total biomass removed from the settlement plates. The total amount of material lost from flat plates remained constant with increasing biomass on plates ($r = 0.054$, $P = 0.315$), whereas the amount lost from curved plates increased with greater biomass ($r = 0.702$, $P < 0.001$). Of the lost material, the mass of viable material was very low (mean = 0.09 g, range 0.01 - 0.24 g), and represented a very small proportion ($< 0.1\%$) of the total material removed from the plate. Despite the low biomass, the viable material nonetheless included a wide range of fully intact organisms (e.g., juvenile mussels, barnacles, calcareous and non-calcareous polychaete worms), as well as fragmented bryozoans, hydroids and colonial ascidians (Table 4.2). In total 27 benthic fouling taxa were represented in the viable material lost, ranging from 2 to 21 taxa across the different treatments (Table 4.2). No relationship existed between taxa

richness lost from flat ($r = 0.207$, $P = 0.333$) and curved plates ($r = 0.247$, $P = 0.243$) and pre-treatment richness.

4.3.4 Vessel trials

Fouling levels on *Pacific Wind* were low-to-moderate, and dominated by the erect bryozoan *Bugula neritina* and hydroids. Nematode and polychaete worms were also observed living amongst the fouling assemblage. Taxa that were relatively resistant to treatment effects in experimental trials on plates, such as calcareous organisms, were not present on the vessel. More advanced levels of fouling were observed on niche areas of the vessel (e.g., gratings, propeller shaft and keel); however, these regions were not amenable to treatment by the rotating brush method. Both systems removed 100% of fouling from the main areas of the hull that were treated by the brushes. However, several patches (up to 5% of the 1 m x 1 m test areas) were missed due to operator error.

On average, System A lost 2.8 g (SE = 0.8 g) of fouling material per 1 m² area of vessel treated, while System B lost 10.2 g (SE = 4.0 g). These figures represent approximately 3 to 9% of the total material defouled, which is in the same range as the trials on the fouled plates. The difference in the performance of the two systems in retaining defouled biomass was not statistically significant (ANOVA, $F_{1,10} = 3.22$, $P = 0.103$). Viable material lost to the environment comprised bryozoan fragments (e.g., *Bugula* sp., *Watersipora* sp.), ascidian fragments (*Diplosoma listerianum*), nematodes, polychaete worms and flatworms (platyhelminthes). Non-viable material comprised detritus, paint chips and various fragmented organisms.

Table 4.2 Presence (X) of viable taxa lost to the environment and subsequently collected by divers during defouling trials by two rotating brush systems on flat and curved settlement plates deployment for 3, 6, 9 and 12 months. Note: some taxa were not present on all plates treated.

Taxa	System A								System B							
	Flat				Curved				Flat				Curved			
	3	6	9	12	3	6	9	12	3	6	9	12	3	6	9	12
Algae (juv)	-	X	X	X	X	X	-	-	-	X	X	X	-	-	-	X
<i>Aplidium</i> sp.	-	-	X	-	-	X	X	-	-	-	-	-	X	-	-	-
<i>Armandia maculata</i>	-	-	X	-	-	-	-	-	-	-	-	X	-	-	-	X
Athecate hydroid	X	X	X	X	-	X	X	-	X	X	X	X	X	X	X	X
Barnacle	-	-	-	-	X	X	X	-	X	-	-	-	X	X	X	X
Bryozoan (erect)	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Bryozoan (flat)	-	X	X	X	-	X	X	X	-	X	X	X	X	X	X	X
<i>Didemnum</i> sp.	-	-	-	-	-	X	X	-	-	-	X	X	-	X	X	X
<i>Diplosoma</i> sp.	X	-	-	-	X	X	X	-	-	-	X	-	X	X	X	X
Eunicidae	-	-	-	-	-	-	X	-	-	-	-	-	-	-	X	-
<i>Galeolaria hystrix</i>	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	X
Gastropod	-	-	-	-	-	X	X	-	-	-	X	-	-	X	X	X
Gastropod egg	-	-	-	-	-	X	-	-	-	-	-	X	-	-	-	X
Hesionidae	-	-	-	-	-	-	-	-	-	-	-	X	-	-	-	-
<i>Mytilus</i> sp. (spat)	X	X	-	-	-	X	X	-	-	X	X	-	-	X	X	-
Nematode	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	X
Nereidae	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	X
Unidentified bivalve	X	-	-	-	X	X	X	-	-	-	-	-	X	X	-	X
Phylodosidae	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	X
Pomatoceros	-	-	-	-	-	-	X	-	-	-	-	-	-	-	X	-
Ribbed mussel spat	-	-	-	-	-	-	X	-	-	-	X	-	-	X	X	-
Sabellidae	-	-	-	-	-	-	X	-	-	-	-	X	-	X	X	X
Sphaerosyllus	-	-	X	X	-	X	X	-	-	-	-	X	-	X	X	X
Spirorbidae	-	-	-	-	-	-	X	-	-	-	-	-	-	-	X	-
Sponge	-	-	-	-	-	X	-	-	-	-	-	X	X	X	X	X
Syllidae	-	-	X	-	-	X	X	-	-	X	-	X	-	X	X	X
Thecate hydroid	-	-	-	-	X	X	X	-	-	-	-	-	X	X	X	-
Richness	5	5	8	5	6	17	21	2	3	6	9	12	9	15	17	19

4.4 DISCUSSION

4.4.1 Removal of fouling material by brush systems

The present study tested the efficacy of two independently developed rotating brush systems in removing and retaining fouling material from experimental surfaces (fouled Perspex plates and a vessel hull) over a range of fouling levels. High performance (c. 90% biomass removal) was achieved on experimental surfaces that had accumulated the lowest biomass (c. $0.4 - 0.9 \text{ kg.m}^{-2}$), and comprised recently settled (i.e., within 3 months) hard/encrusting and soft/erect taxa. The efficacy of both systems decreased on plates that had accumulated much greater biomass (c. $1.2 - 2.6 \text{ kg.m}^{-2}$) dominated by soft/erect taxa, but that also contained mature taxa with morphologies resistant to abrasion (e.g., calcareous tubeworms and flat oysters). For example, at least 50% of tubeworms on plates deployed for 12 months were resistant to brush treatments.

It is worthwhile highlighting that the fouling extent (cover, richness and biomass) that accumulated on experimental plates deployed for 3 months represents the upper range of that expected on commercial vessels in frequent use (e.g., Coutts and Taylor 2004; Davidson et al. 2009). Plates deployed for 6 – 12 months had fouling assemblages comparable to those described for vessels that had been idle for an extensive period (e.g., Davidson et al. 2008). As such, the efficacy of the brush systems tested, in terms of biofouling removal, is likely to be an underestimate of their performance on a “typical” vessel. Furthermore, the devices were tested on surfaces designed to mimic a self-polishing copolymer antifouling coating. Therefore, it is likely that a much higher proportion of fouling removal, particularly for advanced stages (6 – 12 months), would have been achieved if the devices had been tested on a fouling release coating.

The findings of the present study agree with the recent work of Davidson et al. (2008), who assessed the efficacy of a submersible cleaning and maintenance platform (SCAMP) on obsolete vessels that had developed extensive fouling (up to 30 cm thick). Their study found that while the SCAMP was capable of significantly reducing fouling cover and diversity, post-treatment investigations undertaken 3 days after defouling revealed a ‘substantial and diverse’ fouling community across the entire vessel. There was considerable variation in the resistance of hard and soft taxa to the mechanical disturbance of the SCAMP. However, on average organisms that attached to the hull by

cement or byssus threads were more resistant than filamentous taxa (e.g., hydroids and erect bryozoans) and those that attached to the hull with a holdfast (e.g., macroalgae).

Despite the limitations of the brush systems against calcareous taxa in the present study, their efficacy on soft/erect fouling communities was highlighted during trials on the fouled vessel, in which both systems removed 100% of fouling biomass. This result clearly reflected the nature of the treated assemblages on the vessel, which comprised erect bryozoans, hydroids and other soft-bodied taxa that were vulnerable to mechanical treatment effects. Such findings highlight the importance of taking into account the fouling community composition and the morphology of associated species when rotating brush-type tools are being considered for management of fouling.

Even if fouling species are soft-bodied, the morphology of species in the assemblage still remains an important consideration. In the present study, removal of soft-bodied organisms was at times compromised where they were interspersed among patches dominated by species with hard/encrusting morphologies. In the plate trials, soft-bodied species appeared to be protected from the abrasive forces of the brushes where dense areas of calcareous serpulid tubes prevented the stiff brush bristles contacting organisms in gaps among the tubes. Resistant fouling could possibly be removed by using different grades of brushes; however, the potential for associated changes in the efficacy of removal and collection would need to be assessed. Furthermore, in treating antifouled or sensitive surfaces (e.g., fouling release coatings) consideration would need to be given to the likelihood of damage from highly abrasive brush types (Tribou and Swain 2010).

4.4.2 Loss of viable material during treatment

The present study demonstrated that it was feasible to collect a high proportion (typically > 90%) of defouled material; however, 100% retention was not achieved during the trials. Among the lost material, there was a diverse range of viable taxa, comprising fully intact organisms (e.g., juvenile mussels, barnacles, calcareous and non-calcareous polychaete worms) as well as potentially viable fragments (e.g., hydroids, colonial ascidians). The biomass of this material was very low, but generally increased when treating curved plates with increasing biomass; whereas the material lost from flat plates was typically less and remained relatively constant throughout the trials. A potential explanation for this difference is that curved surfaces cause a partial loss of

contact with brush devices, resulting in a decrease in suction created by the systems; it was noted in the field that the leading edge of the brush unit often lost contact with the experimental backing plate when treating curved plates. The slightly greater loss of viable material from System B than System A is partially attributable to shroud design, as the leading edge of the large shroud on System B was observed to dislodge barnacles (without collection). By contrast, the shroud on System A was much smaller and did not come into contact with the fouled surface. Clearly, therefore, there are design considerations for such systems that would need to be addressed to minimise the loss of viable material for situations where there were associated ecological (e.g., biosecurity) risks.

Although the amount of defouled material lost to the environment was very low at the experimental scale, when the diversity of taxa present is considered, it is apparent that the extent of loss could be significant when the brush systems are applied at a commercial scale, especially on an entire vessel with relatively advanced fouling. Based on data from the vessel trial, it was estimated the total mass of viable organisms that would be lost if the entire hull (not including niche areas) had been cleaned to be approximately 1.2 kg for System A and 4.4 kg for System B of which < 20% would be viable. Even though the predicted loss is still reasonably small by mass, it can nonetheless represent the release of a wide range of viable taxa or fragments. A related consideration is that diver disturbance during defouling is likely to exacerbate the release of viable material. For example, observations of fouling organisms being dislodged by diver surface-supply hoses were made during the vessel trial.

Another consideration not addressed in this study is that the proportion of biofouling material lost to the environment could be higher if these devices were applied to a fouling release surface. Due to the ultra-smooth nature of fouling release coatings, biofouling communities are likely to be more readily removed from the vessel surface during hull cleaning. This may result in an increased loss of material to the environment (e.g., through contact with the cleaning device or the diver) and potentially a decrease in the extent of physical damage subjected to the fouling communities when displaced; potentially increasing the volume of viable material lost to the environment during in-water hull defouling operations.

Fouling organisms or viable fragments dislodged during manual defouling have the potential to survive and establish (Paetzold and Davidson 2010, Chapter 5). Although

the assessment in the present study was conservative and may have over-estimated the potential for survival and subsequent establishment of organisms or fragments in the lost material, the possibility that mechanical treatment could lead to the release and establishment of fouling organism is of particular interest from a marine biosecurity perspective. The ability of many non-indigenous marine species to disperse or establish after fragmentation is recognised. For example, the ability of colonial ascidians to establish from fragments is often the means by which artificial cultures are created for experimental purposes (Johnston and Clark 2007; McCarthy et al. 2007; Osman and Whitlatch 2007). Similarly, the dispersal or establishment of invasive macroalgae from fragments is documented for a number of species, including *Sargassum muticum* (Critchley et al. 1986), *Caulerpa taxifolia* (Smith and Walters 1999) and *Undaria pinnatifida* (Sliwa et al. 2006; Forrest et al. 2007). A key consideration in the establishment of defouled fragments or other viable material is the extent to which the recipient environment provides conditions that are suitable for attachment and survival. The factors that determine survivorship are likely to be highly site-specific (e.g., substratum type, predation), and are the subject of my ongoing research.

4.4.3 Ecological risks from defouling

The ecological risk of in-water defouling reflects the risk from release of material defouled during the cleaning operation combined with the residual risk posed by the reduced level of hull fouling (Chapter 3). The release risk can theoretically be eliminated through the collection of all defouled material. However, findings of the present study suggest that complete retention is unlikely even with the best designed collection system. For a vessel with non-indigenous species on its hull, the biosecurity risk of releasing viable material during defouling must be considered against the risk of alternative management strategies, including not defouling the vessel (Chapter 3). Although insufficient for quarantine purposes, the ability of collection systems to retain the majority of defouled material is certainly preferable to having no retention capability. Collection-based systems have the added benefit of reducing the mass load of organic matter and antifouling contaminants to seabed sediments beneath vessel cleaning areas (Srinivasan and Swain 2007; Dafforn 2008 et al. and references therein). Key contaminants are butyltin compounds that are still present on some vessels despite being regulated internationally since 1990 (Stewart et al. 1992; Svavarsson 2000), and copper (plus reinforcing compounds such as zinc pyrithione, Irgarol and Diuron) from

copper-based antifouling paints (Yebra et al. 2004; Almeida et al. 2007). During efficacy trials on the *Pacific Wind*, a temporary discolouration of the water was observed, due to fine antifouling paint particles removed by the brush systems. Although, several large chips (c. 2 x 0.5 mm) of paint were among the lost material collected by divers, most of the paint particles were retained by the collection systems.

In addition to the ecological risk arising from the loss of defouled material to the environment, one of the residual risks post-defouling is that, unless antifouling paints are applied immediately following hull cleaning, mechanical treatment (including in-water and land-based) may increase the susceptibility of the surface to new fouling (Floerl et al. 2005b). Fouling (including damaged calcareous tubeworms) remaining on the vessel following treatment may also provide refugia for a range of fouling taxa. For example, Floerl et al. (2004) showed that *Watersipora subtorquata* acted as a foundation species for fouling assemblages colonising toxic paint coatings. In that study, the presence of *W. subtorquata* facilitated the recruitment of an additional 22 sessile species (including barnacles, bivalves, ascidians, bryozoans and calcareous tubeworms taxa) compared with adjacent toxic patches. Thus in terms of risk management for non-indigenous species, recruitment to defouled surfaces may require consideration when this activity is undertaken in regions where pest species are known to be present, particularly during their reproductive season.

Yet another consideration is that the act of defouling may itself exacerbate biosecurity risk by stimulating the release of planktonic gametes, larvae or propagules due to physical disturbance. The potential for inducing a spawning event is not unrealistic given that physical disturbance is used to induce the release of gametes from several marine invertebrate taxa during experimental studies. For example, gametes from *Hydroides elegans* (a common fouling organism) are reliably obtained by breaking open the tube (Xie et al. 2005; Wong et al. 2006). This potentially represents a high-risk mode of gamete release, given that tubes of polychaete taxa (including *H. elegans*) were often damaged by the rotating brush systems during plate trials in the present study.

4.4.4 Utility of the brush systems tested

From a vessel operational perspective, the ability of the brush systems to eliminate or greatly reduce the cover and mass of fouling is clearly likely to have advantages. Hull fouling can reduce a vessel's speed due to a reduction in hydrodynamics and

manoeuvrability, causing increased fuel and maintenance costs (Townsin 2003; Tribou and Swain 2010). In order to maintain/optimize fuel efficiency, many vessels routinely undertake in-water maintenance, typically in the form of hull scrubbing using devices similar to those tested in this study (without the suction/collection capability). Rising fuel costs and copper-resistant organisms (Russell and Morris 1970; Ng and Keough 2003; Dafforn et al. 2008) are expected to result in a worldwide increase in the amount of in-water hull cleaning undertaken. The use of non-stick fouling release paint coatings, an alternative to biocidal paints, may also require the need for increased in-water maintenance (Finnie and Williams 2010). Given the potential increase in demand for routine in-water hull cleaning, devices that retain defouled material should be favoured over non-collecting devices due to reduced environmental risks generally.

From a marine biosecurity perspective, an effective management tool would ideally completely eliminate the risk from vessel biofouling. Based on the preceding discussion it is evident that the rotating brush systems tested in this study are not suited to this purpose. For example, complete removal of fouling is difficult (especially as fouling biomass increases), and some organisms or life-stages are inherently resistant to mechanical effects (e.g., Forrest and Blakemore 2006). Furthermore, the two systems tested were not suited to cleaning vessel niche areas (e.g., sea chests, anode straps and gratings) where a greater biomass and richness of taxa compared with the general hull area can result in a greater prevalence of NIS (Coutts and Taylor 2004; Coutts and Dodgshun 2007; Chapter 2). As a minimum therefore, for an effective biosecurity response, the application of brush systems would need to be supported by complementary methods for niche areas, or for resistant fouling on the main part of the hull.

Rotating brushes are now used routinely on merchant ships, which can be cleaned in 1-2 days. Hence, despite the limitations of rotating brush systems for eliminating biosecurity risk, they nonetheless represent a proven in-water method that are likely to substantially reduce biosecurity risks on to larger vessels having low-to-moderate levels of fouling. However, the effective management of biosecurity risks will ultimately require a broad suite of measures, including: (1) the development of specific management programmes for vessels visiting high value areas (e.g., Lewis et al. 2006), (2) education and awareness among vessel operators, (3) research to better understand the factors contributing to vessel risk and, (4) targeted surveillance programmes for vessels or vessel types identified as high risk.

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Chapter 5 Factors Affecting Survivorship of Defouled Communities and the Effect of Fragmentation on Establishment Success

PREFACE

Observations of viable fouling organisms, including fragmented colonial organisms, being released to the environment during the rotating brush trials described in the previous chapter led to this series of laboratory- and field-based trials designed to gain a better understanding of the physical and biological factors that determine post-defouling survivorship. This research fills an important gap in the scientific knowledge/evidence needed to make informed decisions on policy around in-water defouling.

Richard Piola (Cawthron Institute) provided input into experimental design and assisted with fieldwork. My thesis supervisors Barrie Forrest and Jonathan Gardner provided useful comments on a draft manuscript identical to this chapter (submitted to Journal of Experimental Marine Biology and Ecology in June 2010).

ABSTRACT

The environmental risks associated with the defouling of artificial structures (e.g., vessels, oil rigs, marina pontoons, aquaculture structures) in the marine environment are gaining international attention. This chapter presents a series of laboratory- and field-based experiments that collectively aimed to elucidate biotic and abiotic factors that influence re-establishment success of biofouling organisms and fragmented colonial organisms defouled to the seabed. Reattachment success of colonial organisms experimentally fragmented was found to be species-specific and dependent on fragment size. In both laboratory and field trials, some colonial ascidians had consistently greater reattachment success for larger size classes of fragments, while other encrusting and erect taxa showed poor reattachment capabilities. This study also revealed that sedimentation and turbidity are likely to have a strong influence on the survivorship of defouled material. Furthermore, for both high and low sedimentary environments, survivorship was found to be greater where predators were excluded. Despite risks posed by non-indigenous species (NIS), it is proposed that in-water defouling may be an appropriate management response in situations where a “do nothing” approach is potentially more detrimental. Moreover, results from this study suggest that environmental risks associated with defouling may be mitigated through appropriate defouling strategies (e.g., defouling location, frequency and method). In order to increase our predictive abilities for NIS establishment success resulting from in-water defouling, future studies should aim to further elucidate the relative importance of factors affecting survivorship of defouled material at locations where defouling is routinely undertaken.

5.1 INTRODUCTION

The term biofouling refers to the accumulation of biological growth on submerged artificial surfaces, such as ship's hulls (Minchin and Gollasch 2003; Coutts and Taylor 2004), aquaculture structures (Carlton 1992; Coutts and Forrest 2007), marina pontoons (DeFelice 1999; Webb and Keough 2000; Glasby and Connell 2001) and oil platforms (Foster and Willan 1979; Southgate and Myers 1985; Zvyagintsev and Ivin 1995; Yeo et al. 2010). Considerable efforts are made to prevent or reduce the extent of biofouling on many submerged structures, due to the detrimental impacts often associated with its occurrence. For example, fouling on ship hulls reduces the effective operating speed of the vessel (due to a reduction in hydrodynamics and manoeuvrability), resulting in increased fuel and maintenance costs (Townsin 2003; Schultz 2007). On static structures, biofouling can increase the risk of mechanical failure, through the addition of bulk and weight, as well as the enhancement of corrosion of metal by seawater (Videla 2002; Zardus et al. 2008).

Vessel biofouling has also been recognised as an important mechanism for the global transfer of marine biofouling organisms and non-indigenous species (NIS) (Gollasch 2002; Godwin 2003; Coutts and Taylor 2004; Yamaguchi et al. 2009). As a result, international ports and marinas are high risk entry points for NIS and important hubs for their further spread (Carlton 1996; Floerl and Inglis 2005; Floerl et al. 2009). The role of artificial structures associated with these transport 'hubs' (e.g., vessels, wharf piles, marina pontoons) in facilitating the establishment and spread of NIS is also recognised (Wasson et al. 2001; Glasby et al. 2007). Similarly, outside of port or marina environments, artificial structures such as aquaculture equipment and vessel moorings have been implicated in the spread of numerous NIS, including high-profile pest species such as *Didemnum vexillum* (Coutts and Forrest 2007) and *Codium fragile* ssp. *tomentosoides* (Bulleri and Airolidi 2005).

For most vessel types, the application of antifouling paints and routine vessel hull maintenance are effective in controlling biofouling accumulation, with the additional benefit of reducing unintentional biofouling transfers (Callow and Callow 2002; Coutts and Taylor 2004). However, antifouling paints applied to vessels in the marine environment can quickly become ineffective if the vessels remain idle for extended periods of time (Floerl et al. 2005; Ferreira et al. 2006). For this reason, most static artificial structures are not typically treated to prevent biofouling, and periodic

defouling may be required. For large vessels and structures that cannot be removed to land or readily dry-docked, in-water defouling may be the only available option.

In-water defouling of vessels or structures typically involves the mechanical removal of live biofouling assemblages, using a variety of methods including hand-held scrapers, mechanical rotating brushes or high pressure water blasters. Environmental risks associated with in-water defouling include the release of toxic biocidal paints and NIS to the wider environment (Chapter 3). With respect to NIS, a primary concern is whether biological material removed during the defouling process is able to survive and establish in recipient regions. Floerl et al. (2003) found that survival and viability of biofouling organisms following in-water cleaning with scrapers was significantly higher (c. 70%) than shore-based cleaning operations. Similarly, while testing novel mechanical rotating brush technology able to retain defouled material (Chapter 4), it was found that 10 - 20% of the small amount of fouling lost to the environment remained intact and/or viable. A significant proportion of this released material comprised fragmented colonial organisms (e.g., bryozoans and ascidians). This has important ramifications for the establishment and spread of NIS, as several studies have demonstrated the ability of colonial organisms to establish from fragments (O'Dea 2006; Bullard et al. 2007; Johnston and Clark 2007; Piola and Johnston 2009). However, there is a currently a paucity of knowledge regarding the importance of fragment size on establishment success for the range of colonial organisms typically associated with biofouling assemblages.

Several countries have placed restrictions on in-water defouling in recognition of the environmental risks posed from the release of contaminants associated with antifouling paints, and the potential biosecurity risks associated with such activities. In New Zealand and Australia, guidelines governing in-water defouling activities (ANZECC 1997) are under review, due in part to the lack of suitable land-based facilities available to treat fouled vessels, and recognition of advances in in-water treatment technologies and changes in paint strategies (e.g., non-toxic coatings). A better understanding of biosecurity risks associated with in-water defouling is likely to make a major contribution to the re-drafting of these guidelines, as well as improve management of biosecurity risks posed by biofouling at the border.

This chapter presents findings of manipulative laboratory- and field-based experiments that examined the survival and establishment success of a range of common biofouling taxa in post-defouling conditions. The specific aims were to: (i) investigate whether

different defouling methods (e.g., rotating brush device versus hand-held scraper) influenced fragment size; (ii) determine whether fragmented colonial organisms had the ability to survive and reattach post-defouling, and whether fragment size explained reattachment success; and (iii) identify key factors that are likely to determine initial survivorship of defouled organisms in ports and marinas (i.e., typical vessel hull maintenance facilities).

5.2 METHODS

5.2.1 Comparison of fragment sizes generated from two common defouling tools

Comparisons were made between fragmented colonial taxa defouled using two common in-water tools - mechanical rotating brushes and hand-held scrapers. Fragmentation by rotating brushes involved experimental surfaces (350 x 350 mm Perspex plates) with advanced (12 month accumulation) and low (3 months accumulation) levels of biofouling being defouled by diver-operated rotating brushes during *in situ* simulated vessel cleaning trials (Chapter 4). A high proportion (c. 95%) of defouled material was retained by the rotating brush tool and was pumped to the surface where it was sieved to 60 µm. Fragmentation by hand scraping was achieved by immersing experimental surfaces (spare plates not defouled during the rotating brush trials described above) in a 40 L container filled with seawater and defouling the Perspex plates using a metal paint scraper. Defouled material was collected from the container using a 60 µm sieve. Following defouling and fragmentation, retained material from both defouling methods was sorted and non-colonial organisms (e.g., calcareous tubeworms, bivalves) were removed. The remaining material (i.e., primarily colonial material) was sieved into five size classes (0.3 – 0.5 mm, > 0.5 – 1.0 mm, > 1.0 – 2.0 mm, > 2.0 - 5.6 mm, > 5.6 mm), oven-dried at 70°C for 72 hours and weighed to determine the relative contributions of each size class.

5.2.2 Reattachment of fragmented colonial organisms

Laboratory-based pilot study

In October 2008, colonies of the ascidians *Didemnum vexillum* and *Botrylloides leachi*, and the bryozoans *Bugula neritina*, *Bugula flabellata* and *Watersipora subtorquata* were collected from settlement plates that had been deployed for c. 6 months from floating structures in a recreational boat marina located in Nelson, New Zealand. Colonies were

placed in seawater, transported to the laboratory and dissected into several size classes (Table 5.1), based on the range of fragment sizes generated by the two defouling methods tested above. Fragments were transferred into FalconTM multi-well plates (with lids removed) and placed in a gently aerated 20 L container filled with filtered (0.32 µm) seawater inside a controlled temperature cabinet (14 °C, 12:12 day:night regime). A 50% water change was undertaken every 2 days. After 5 days, each fragment was squirted with a 5 ml plastic pipette filled with seawater to see whether it was attached (Bullard et al. 2007) and reattachment success (yes/no) calculated for each size/species combination.

Table 5.1 Size classes of colonial organisms fragmented during pilot laboratory trials.

Colonial organism	Size class (mm)		
	Large	Medium	Small
<i>Didemnum vexillum</i>	25 x 10	-	5 x 3
<i>Botrylloides leachi</i>	30 x 20	-	5 x 3
<i>Bugula neritina</i>	30	15	6
<i>Bugula flabellata</i>	30	15	6
<i>Watersipora subtorquata</i>	-	-	6 x 3

Field trial

In February 2009, fragments of *Didemnum vexillum*, *Bugula flabellata*, *Watersipora subtorquata* and *Botrylloides leachi* were collected from floating marina structures in Picton and dissected into one of three size classes: small (5 mm), medium (10 mm) and large (20 mm). Fifteen fragments from each size/species combination were placed into replicate ($n = 3$) 2 L plastic containers (Figure 5.1a). Each container had the sides and lid replaced with mesh (1 mm²) to prevent predators entering the container, but while still allowing water and food to pass through. At the base of each container was a 130 x 130 mm black, roughened Perspex plate which acted as the fragment reattachment surface. Five containers were randomly assigned and fixed to each of eight 600 x 600 mm PVC backing plates (Figure 5.1b). The backing plate was then placed on the seabed (water depth = 4 m at MLWS). Four control containers were also randomly assigned to the backing plates, each having non-fragmented representatives of each of the

four species. After 5 days, the backing plates and containers were retrieved and inspected. Each fragment was squirted with a 5 ml plastic pipette filled with seawater to see whether it was attached and species-specific reattachment success recorded.

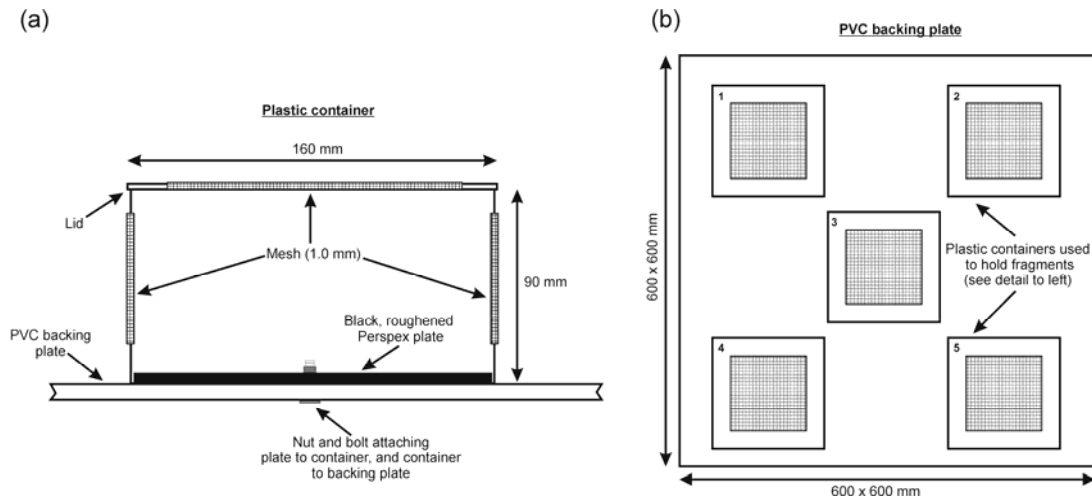


Figure 5.1 Diagram of experimental fragment transplant units, showing (a) cross-section of an experimental unit used to hold fragments near the seabed during field trials, and (b) dorsal view of the arrangement of experimental units on a backing plate for deployment in the field.

5.2.3 Survivorship of defouled communities

Preliminary field investigations showed that grazing and predation by benthic invertebrates and sedimentation were the dominant stressors influencing the ongoing survivorship of reattached fragments and/or intact colonial organisms displaced from a suspended structure to a seabed environment (G. Hopkins, unpublished data). Transplant experiments were conducted to further elucidate the role of these factors in biofouling assemblage survival.

In May 2007, 132 black Perspex settlement plates (225 x 145 mm) were deployed on a floating marina pontoon in Picton Harbour, New Zealand, at water depth of 1 m to develop biofouling communities. In February 2008, the fouled settlement plates were removed, photographed and individual plates were randomly allocated to one of four redeployment locations in central New Zealand (-41°S): Picton marina (Picton), Waikawa marina (Waikawa), Nelson marina (Nelson A) and the Port Nelson international wharf

(Nelson B). Picton and Waikawa are low sedimentary environments with high water clarity, while the two Nelson sites represent more turbid environments where sedimentation rates were expected to be relatively high. All locations had a similar assemblage of benthic predators/grazers, dominated by gastropods (e.g., *Turbo smaragdus*, *Diloma aethiops*), echinoderms (e.g., *Patiriella regularis* and *Coscinasterias calamaria*) and chitons (*Chiton glaucus*, *Sypharochiton pelliserpentis*). Plates deployed at Nelson A and B were transported from Picton (2 hours) under humid conditions in sealed containers. This transport phase was also simulated for plates deployed at Picton and Waikawa.

At each location, 27 fouled plates were labelled and attached to a metal frame in one of three orientations; facing up, facing down and vertical. These three orientations were chosen as they created the range of environmental conditions (e.g., sedimentation rates, light levels) that could be realistically encountered by taxa settling to the seabed following defouling. A total of 3 plates from each orientation were enclosed within individual stainless steel cages (mesh = 10 x 10 mm) to exclude predators. A further 3 plates from each orientation acted as cage-controls, and were enclosed within a mesh cage with the side left open to predators. The remaining 9 plates were left uncaged. Frames were positioned on the seabed at a water depth of approximately 3 m at MLWS. Control plates were deployed away from the seabed along floating pontoons at the Picton and Nelson marinas. At both sites, 6 control plates were suspended vertically 30 cm below the surface and a further 6 controls were suspended at 3 - 4 m water depth. Three of the 6 control plates at the two water depths were caged to exclude predators.

Cage treatment units were periodically inspected for predators and the cages cleaned *in situ* to remove biofouling which may inhibit water exchange. Water temperature (°C) was recorded at each site at hourly intervals for the 1 month duration of the experiment using a HOBO Water Temp Pro data logger (Onset Computer Corporation, Bourne, MA, USA). Secchi disc measurements, salinity (YSI 85 handheld field probe, Yellow Springs Instruments, Yellow Springs, OH, USA), and turbidity (USEPA Method 180.1) measurements were taken periodically at all four sites. Three sediment traps (diameter = 63 mm, length = 198 mm) were deployed at each location for the duration of the experiment to estimate relative sedimentation rates. The mass of sediment collected in the traps was later analysed for total solids (APHA 20th Edition 2540C 31.29).

At the completion of the deployment (March 2008), plates were removed, re-photographed and preserved in 70% ethanol with 5% glyoxal (with seawater). Changes in species richness and percent cover were estimated from differences in pre-treatment (before) versus post-treatment (after) photos of the settlement plates. Images were rectified in ArcMap 9.2 GIS software (ESRI, Redlands, CA, USA) and a systematic 49-point grid (7 x 7) was superimposed. A 1 cm perimeter along the edge of each plate was omitted to control for possible handling effects. Taxa present beneath each point were identified and recorded.

5.2.4 Statistical analyses

Two-way ANOVA and Tukey's *post-hoc* tests for significance were used to test for differences in reattachment success of the colonial species in relation to species and fragment size. Multiple linear regression was used to test for (i) the proportion of fragments in each of the five size classes generated by the two in-water cleaning tools (handheld scraper and rotating brush), and (ii) determine the effects of site location, orientation and the presence/absence of predators and grazers on biofouling cover and taxa richness on plates transplanted to the seabed. Biofouling cover on control plates (used to test for handling and non-treatment effects) deployed in Picton and Nelson during the community transplant experiments did not alter significantly over the duration of the experiment ($F_{1,16} = 1.225$, $P = 0.285$): data from the controls plates were therefore removed from subsequent analyses. Data from the cage-controls were also omitted from analyses because exploratory analyses revealed no significant changes in biofouling percent cover ($P \geq 0.18$) or richness ($P \geq 0.422$) over the experimental period. Data were tested for homogeneity and normality using STATISTICA Version 8 (StatSoft Inc., Tulsa, OK, USA), and dependent variables were $\log_{(x+1)}$ transformed where necessary to meet the assumptions of parametric testing. Outliers (Cook's distance > 1.0) were removed from analyses and the linear regression model refitted. The Akaike Information Criterion (AIC) was used for model selection, and the model of best fit for each analysis was validated by plotting residuals (Zuur et al. 2007).

A non-metric MDS ordination procedure, based on the Bray-Curtis similarity measure, was used to describe changes in taxon composition and dominance patterns on transplanted plates using PRIMER Version 6 (PRIMER-E Ltd, Lutton, Ivybridge, UK). Data were square-root transformed to down-weight the influence of the most dominant taxa (Clarke and Warwick 1994). The similarity percentages procedure (SIMPER) in

PRIMER was used to explore trends evident in the nMDS plots (Clarke and Warwick 1994).

5.3 RESULTS

5.3.1 Comparison of fragment sizes generated from defouling methods

The defouling method influenced the size of fragments generated from fouled settlement plates, with a strong interaction ($F_{4,160} = 3.73$, $P = 0.006$) observed between fouling level, defouling tool and fragment size (Figure 5.2). For both advanced- and low-levels of fouling, hand-held scrapers generated a greater percentage of large fragments in the > 5.6 mm and 2.0 - 5.6 mm size classes compared to the rotating brush method. Conversely, rotating brush devices generated a larger proportion of smaller-sized fragments (< 1 mm) than the scrapers (Figure 5.2). Handheld scrapers generated almost twice as many fragments in the 2.0 - 5.6 mm size class from advanced fouling assemblages than from low levels of fouling, while an opposite trend was evident for the four other size classes (Tukey's HSD, $P < 0.05$).

5.3.2 Reattachment of fragmented colonial organisms

Laboratory-based trial

Reattachment success of fragmented colonial organisms varied with taxon and size class (Table 5.2). In general, larger fragments achieved higher reattachment success (33 – 88% greater) relative to small fragments after 5 days, with the exception of *Bugula flabellata*, for which no large fragments reattached (Table 5.2). Large fragments of *Didemnum vexillum* and *Bugula neritina* had the highest reattachment success rate (50 and 66.7 %, respectively). The rapid and successful reattachment of the arborescent bryozoans *Bugula neritina* (large and small) and *B. flabellata* (small) was unexpected, and achieved by the formation of stolons, which anchored the colony to the substratum. Unidentified filamentous green algae growing on two small fragments of *Watersipora subtorquata* also attached to the well plates, effectively securing the colonies to the substratum.

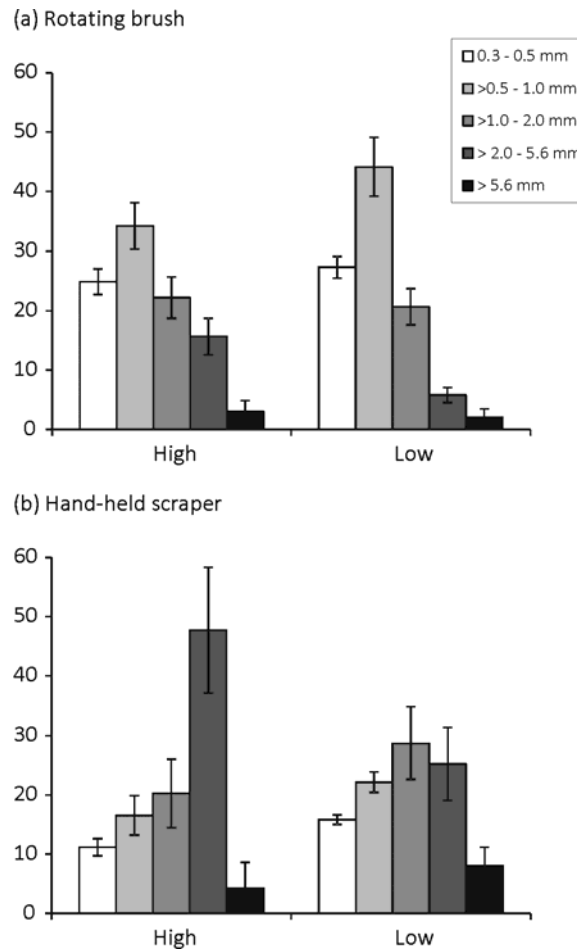


Figure 5.2 Percent biomass (mean dry weigh \pm 1SE) of fragments generated by (a) rotating brush devices and (b) hand-held scrapers during experimental defouling trials on surfaces with high (12 months accumulation) and low (3 months accumulation) levels of biofouling.

Table 5.2 Reattachment success (%) of fragmented colonial organisms after 5 days in controlled conditions (14 °C, 12:12 day:night regime). See Table 5.1 for fragment sizes.

Taxon	Size	<i>n</i>	% reattachment
<i>Didemnum vexillum</i>	Large	6	50.0
<i>Didemnum vexillum</i>	Small	12	33.3
<i>Botrylloides leachi</i>	Large	6	33.3
<i>Botrylloides leachi</i>	Small	12	0.0
<i>Bugula neritina</i>	Large	6	66.7
<i>Bugula neritina</i>	Small	12	8.3
<i>Bugula flabellata</i>	Large	6	0.0
<i>Bugula flabellata</i>	Small	12	8.3
<i>Watersipora subtorquata</i>	Small	12	16.7

Field trials

Field trials assessing fragment reattachment used the same suite of colonial organisms as the laboratory trial (minus *Bugula flabellata*; Figure 5.3). *Didemnum vexillum* and *Botrylloides leachi* had significantly greater reattachment success ($F_{2,24} = 9.96$, $P < 0.001$) than the other two taxa experimentally fragmented (*Bugula neritina* and *Watersipora subtorquata*). Highest reattached success (on average) was achieved with large fragments of *D. vexillum* ($44.4 \pm 18.2\%$) and the large and medium fragments of *B. leachi* (22.2 ± 8.9 and $24.4 \pm 4.4\%$, respectively). Only a small proportion of *Bugula neritina* fragments reattached using stolons during the field trials ($< 1\%$ across all size classes) representing a significant reduction compared to laboratory trials (66.7 %; Table 5.2). In further contrast to the laboratory findings, reattachment of small *D. vexillum* fragments was highly variable in the field ($8.9 \pm 8.9\%$) and considerably less than that observed in the lab (33.3%), while small fragments of *B. leachi* successfully reattached in the field ($13.3 \pm 3.8\%$) though not in the lab (Figure 5.3; Table 5.2). No *Watersipora subtorquata* reattachment was observed.

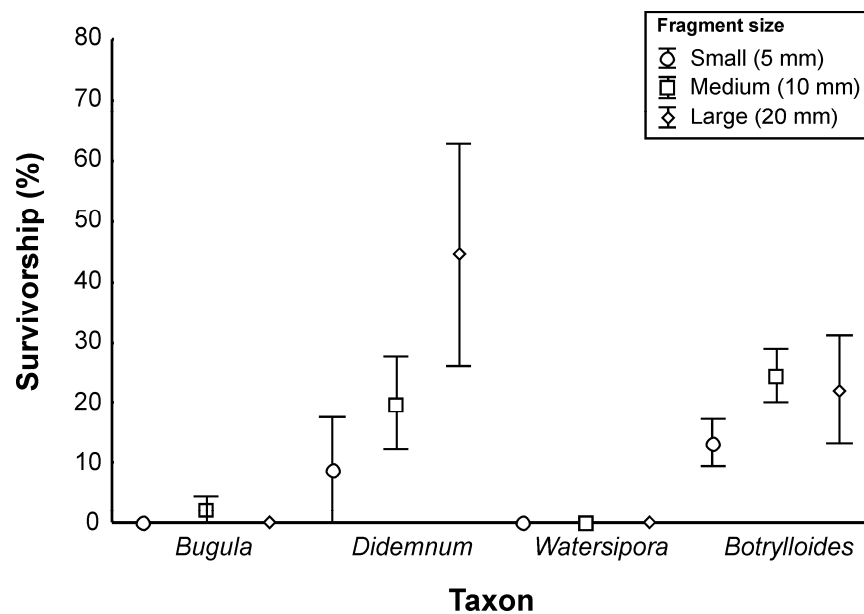


Figure 5.3 Percent survival (mean \pm 1SE) of small, medium and large fragments of colonial organisms transplanted to the seabed.

5.3.3 Survivorship of transplanted biofouling communities

Environmental characteristics of experimental sites

Sedimentation rates and turbidity at the Nelson sites (A and B) were several times greater than that measured at Picton and Waikawa (Table 5.3). Rocks and cobbles at the Nelson sites were completely covered with a layer of fine sediments and water clarity was low, particularly at low tide when resuspension of sediments by wind and currents was greatest. In contrast, rocky surfaces at the Picton and Waikawa sites were largely devoid of sediments, and changes in water clarity across tidal cycles were less evident. Temperature and salinity ranges at the four sites were comparable (Table 5.3).

Table 5.3 Environmental conditions at the four transplant sites. Salinity, turbidity, secchi, sedimentation rate (average \pm 1SE) and temperature (average and range).

Site	Sedimentation rate ($\text{g.m}^{-2}.\text{d}^{-1}$)	Turbidity (NTU)	Secchi depth (m)	Temperature ($^{\circ}\text{C}$)	Salinity (PSU)
Nelson A	708.3 ± 37.6	3.4 ± 0.8	1.69 ± 0.03	20.1 (18.3 - 21.6)	34.0 ± 0.5
Nelson B	1890.6 ± 135.4	2.6 ± 0.4	1.63 ± 0.13	19.9 (17.1 - 21.7)	34.0 ± 0.1
Picton	113.1 ± 6.0	1.1 ± 0.2	3.52 ± 0.25	17.7 (16.6 - 19.4)	34.7 ± 0.1
Waikawa	34.0 ± 3.3	0.8 ± 0.4	3.95 ± 0.32	17.6 (16.6 - 19.3)	34.7 ± 0.1

Pre-transplant biofouling community composition

Settlement plates suspended at Picton marina developed a diverse and extensive biofouling community over the 12 month deployment period. Biofouling cover on the plates was dominated by *Botrylloides leachi* (colonial ascidian), *Watersipora subtorquata* (encrusting bryozoan) and various unidentified sponges. Also present in lower densities were, calcareous tubeworms (Spirorbidae, *Serpula* sp., *Salmacina australis*, *Pomatoceros* sp., *Galeolaria hystrix*), fanworms (Sabellidae), thecate hydroids, barnacles (*Austrominius modestus*), erect bryozoa (e.g., *Bugula* spp.), ascidians (*Cnemidocarpa bicornuata*, *Aplidium phortax*, *Corella eumyota*), bivalves (*Mytilus galloprovincialis*, *Chlamys* sp., *Ostrea chilensis*) and various macroalgae (e.g., *Colpomenia* sp.).

Post-transplant survivorship

Reductions in biofouling cover were observed at all four transplant sites and for each combination of settlement plate orientation and predator exclusion (Figures 5.4 to 5.6). Caged plates (i.e., preventing predation) were generally less impacted, in particular those oriented down and vertical ($F_{2,48} = 6.54$, $P = 0.003$). The proportion of initial biofouling cover remaining also varied with respect to site and plate orientation ($F_{6,48} = 8.66$, $P < 0.001$). For example, plates orientated upwards and vertical at Waikawa had, on average, a higher proportion of cover remaining than similarly orientated plates at the two Nelson sites (Figure 5.4). However, the proportion of biofouling cover remaining on plates orientated downwards was comparable at three of the four sites (Picton, Waikawa and Nelson A; Tukey's HSD, $P > 0.05$), while Nelson B showed a significantly larger reduction in cover (Tukey's HSD, $P < 0.05$; Figure 5.4). These changes are evident in Figures 5.5 and 5.6, where reductions in the cover of mainly soft-bodied biofouling organisms (e.g., *Botrylloides leachi*, sponges) were most pronounced on plates orientated upwards and those without predator exclusion (uncaged).

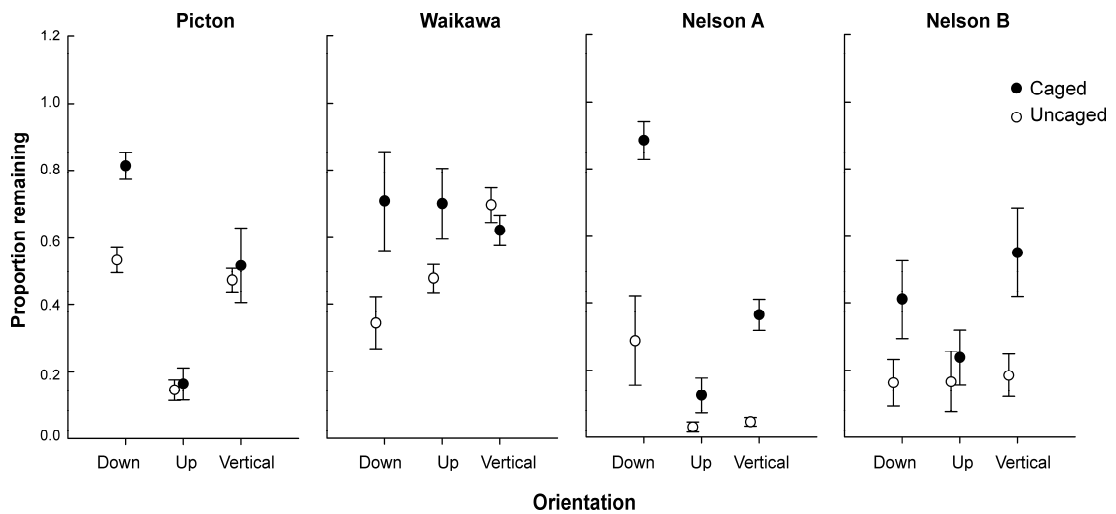


Figure 5.4 Proportion of initial fouling cover (mean \pm 1SE) remaining on plates transplanted from a floating structure to the seabed in three orientations (vertical, facing up, facing down), caged (filled symbols) and uncaged (unfilled symbols). Proportion of cover remaining on control plates (deployed at the surface and caged) ranged from 0.98 to 1.04.

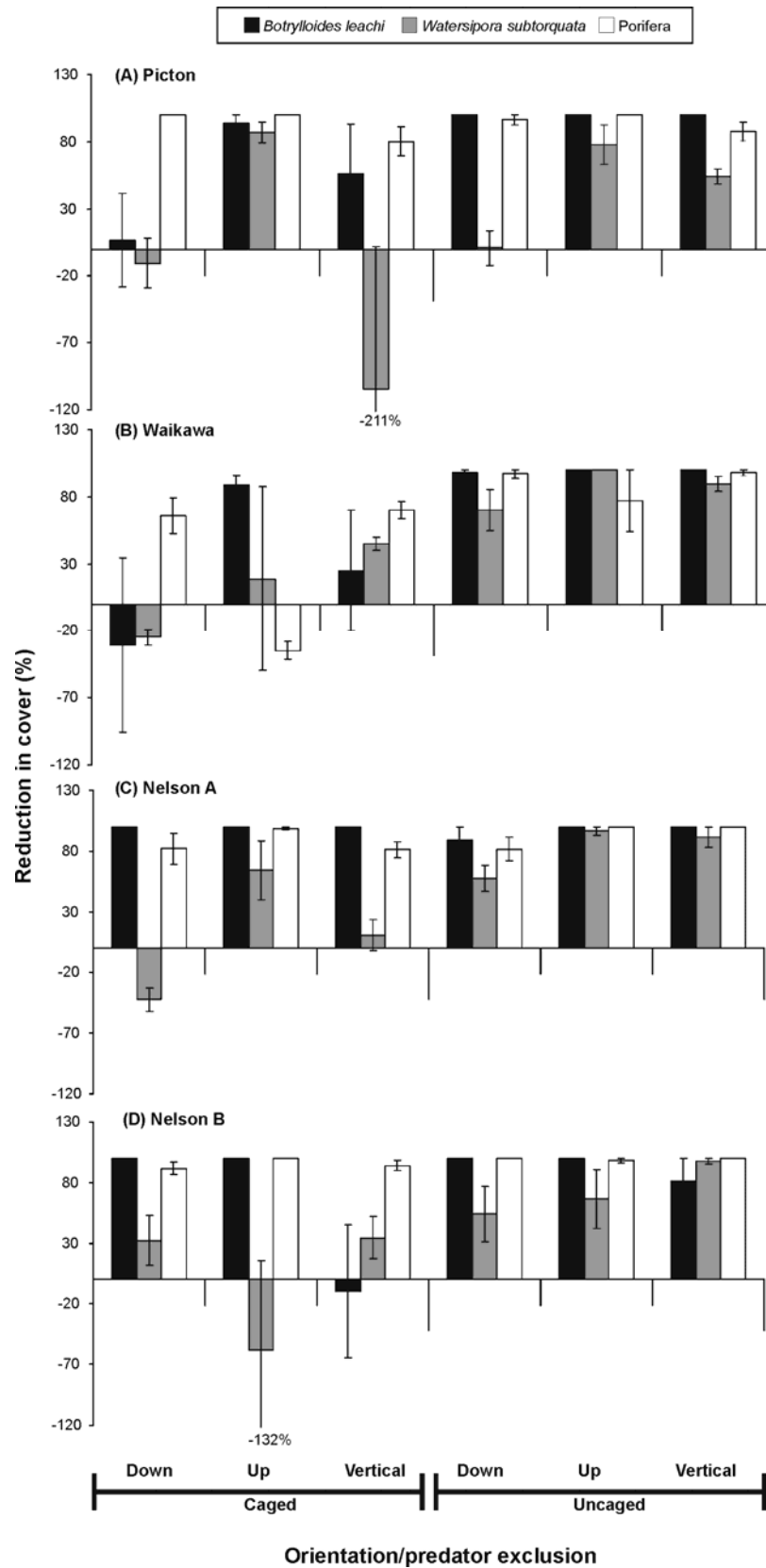


Figure 5.5 Reduction in percent cover (mean \pm 1SE) of the three most commonly encountered taxa on the pre-transplant images one month after being transplanted into a port/marina environment.

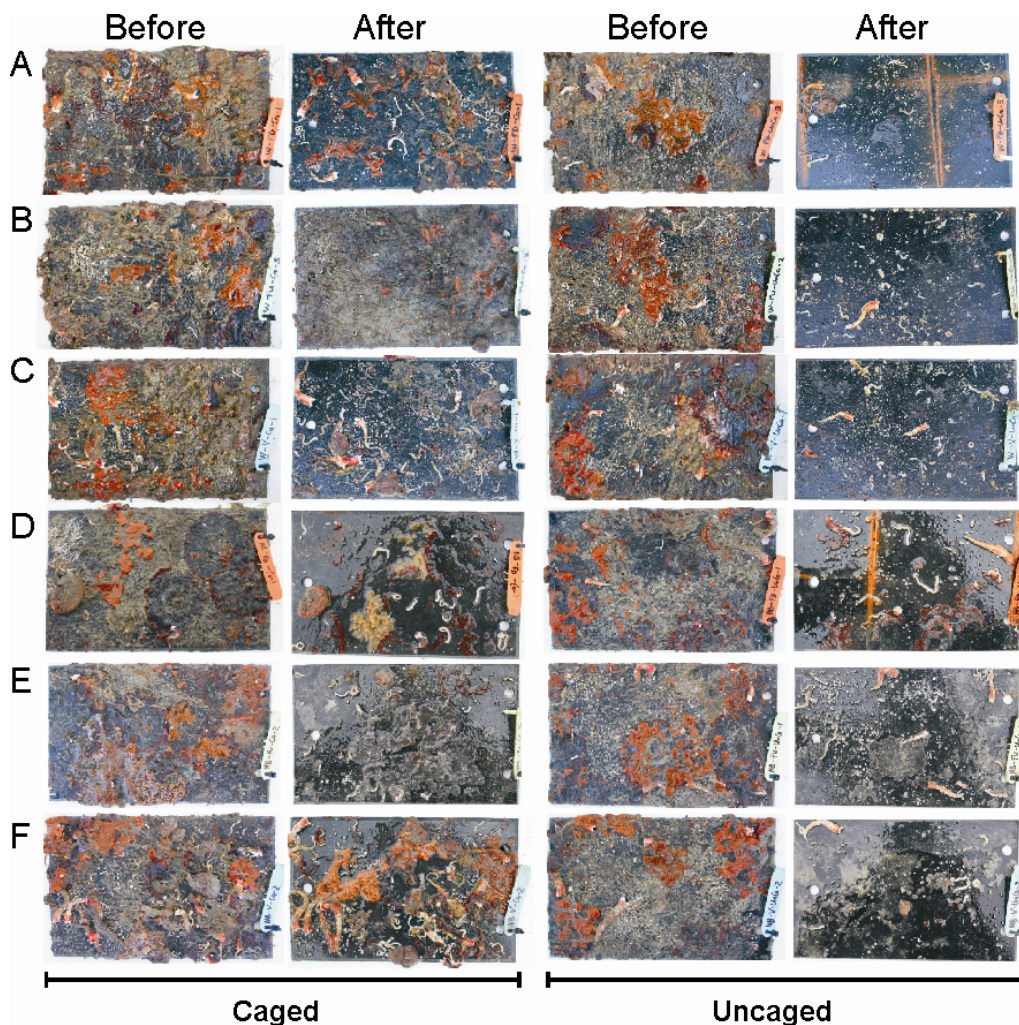


Figure 5.6 Before and after images of biofouling communities on settlement plates that were transplanted in different orientations with (left) and without (right) cages. A – Waikawa facing down, B- Waikawa facing up, C- Waikawa vertical, D- Nelson B facing down, E- Nelson B facing up, F- Nelson B vertical.

In some instances, biofouling cover of some taxa increased over the experimental period (Figure 5.5). For example, *Botrylloides leachi* and *Watersipora subtorquata* increased in cover on caged plates facing down at Waikawa. *W. subtorquata* also increased in cover on caged plates facing down and vertical at Picton, on caged plates facing down at Nelson A, and on caged plates facing up at Nelson B. No increases in biofouling cover (present in the pre-transplant image) were observed on uncaged plates at any of the transplant sites.

Changes in taxon richness on transplanted settlement plates varied significantly by site and by the presence or absence of predator exclusion cages ($F_{3,48} = 3.06$, $P = 0.037$), with larger differences between caged and uncaged at Nelson A and Waikawa (Figure 5.7). There was a greater reduction in richness for uncaged versus caged plates for almost all combinations of location and orientation, and this was most marked at Nelson A, and least at Nelson B. Plate orientation also played a significant role in determining taxon richness remaining ($F_{2,48} = 7.11$, $P = 0.002$), with mean richness declining more on plates facing upwards compared with those facing downwards and vertical (Tukey's HSD, $P < 0.05$).

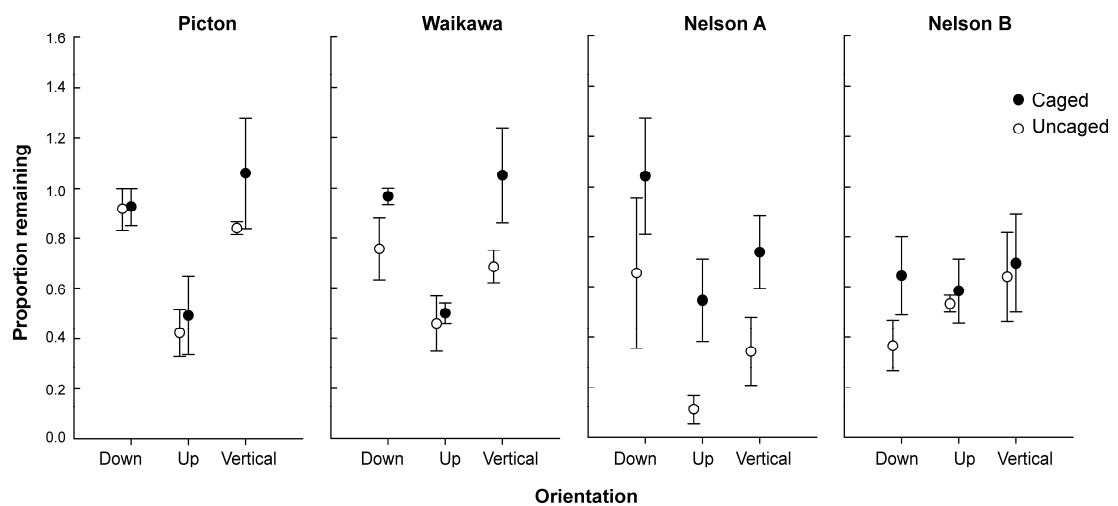


Figure 5.7 Proportion of initial taxa richness (mean \pm 1SE) remaining on plates transplanted from a floating structure to the seabed in three orientations (vertical, facing up, facing down), caged (filled symbols) and uncaged (unfilled symbols). Proportion of taxa remaining on control plates (deployed at the surface and caged) ranged from 0.77 to 1.40.

Biofouling composition on transplanted plates changed based on transplant site location, plate orientation and the presence/absence of cages, and this reflected the significant changes to biofouling cover and taxon richness described above. As expected, the nMDS plots show that the pre-treatment assemblages were generally more similar to each other than to post-transplant assemblages. Downward and vertically orientated caged treatments were generally more similar to pre-treatment assemblages than uncaged or upward orientated plates. Caged treatments also tended to group apart from the uncaged plates at the Picton and Waikawa sites. Finally, plates facing up were typically quite

similar at the turbid Nelson sites (A and B) irrespective of potential predation pressure (i.e., presence of cages).

SIMPER analyses revealed that the relative abundance of sponges, *Botrylloides leachi*, *Watersipora subtorquata* and spiorobid polychaetes played an important role in the formation of the groupings superimposed on the nMDS plots (Figure 5.8). Pre-transplant images were characterised by a high relative abundance of all four taxa, whereas plates least similar to the pre-treatment plates had none or very low densities of these taxa present. Uncaged plates at Picton and Waikawa were almost completely devoid of ‘soft-bodied’ organisms, such as *B. leachi* and sponges, while caged plates orientated downward and vertically generally retained relatively high levels of some, if not all, common soft-bodied organisms. In contrast, caged plates facing down and vertically at Nelson A and B had large losses of soft organisms.

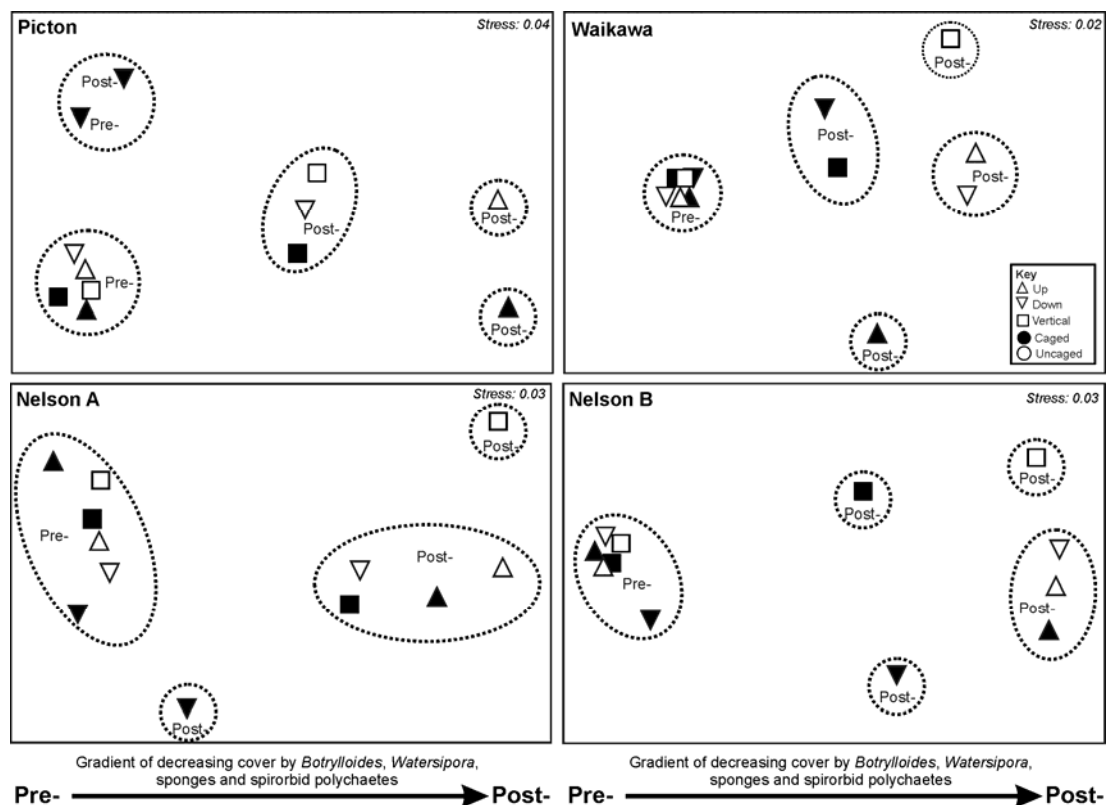


Figure 5.8 nMDS ordination showing changes in averaged ($n = 3$) fouling composition after being transplanted to the seabed (pre- = pre-treatment, post- = post-treatment) in one of three orientations (facing up, facing down and vertical) and caged or uncaged. Data were square-root transformed (2D stress for all ordinations was ≤ 0.04). Clusters (identified by dotted lines) indicate treatments having a within-group Bray-Curtis similarity of $> 75\%$.

5.4 DISCUSSION

5.4.1 Species and size-related reattachment success of fragments

Reattachment success of colonial organisms experimentally fragmented was species-specific, and for 4 of 5 taxa examined, dependent on fragment size. During field trials, the colonial ascidians *Didemnum vexillum* and *Botrylloides leachi* had consistently higher reattachment success across all three fragment size classes, while the encrusting (*Watersipora subtorquata*) and erect (*Bugula neritina*) bryozoans showed poor reattachment abilities within the 5 day experimental period. Bullard et al. (2007) reported similar differential reattachment success among four colonial ascidians (*Aplidium constellatum*, *Botryllus schlosseri*, *Botrylloides violaceus* and *Didemnum* sp.). *A. constellatum* did not reattach during their trials, although most (75 - 80%) fragments of *B. schlosseri* and *Didemnum* sp. and all fragments of *B. violaceus* reattached. Bullard et al. (2007) theorised that differences in reattachment ability of colonial ascidians are influenced by the life-history ecology of the species. For example, taxa that have a morphology resulting in a low incidence of fragmentation (e.g., *A. constellatum* – a thick and fleshy ascidian) are less likely to have evolved the ability to reattach compared with commonly fragmenting taxa (e.g., *Didemnum* sp. – a lobe-forming ascidian). This appears to be the case in the present study, where *D. vexillum*, which forms long tendrils and has a propensity for fragmentation (Lengyel et al. 2009), had higher rates of reattachment relative to *B. leachi* which a fleshy ascidian that is not lobe-forming and is less likely to be fragmented in its natural environment. Similarly, the branching morphologies of the bryozoans *B. neritina* and *B. flabellata*, which under normal conditions are known to grow stolons (or rhizomes) for anchoring the colony to substratum (Grave 1930; Walters 1992), seem more suited to reattachment following defouling compared to *W. subtorquata*, which likely has a less ‘dynamic’ growth morphology based in its more heavily calcified structure and encrusting morphology.

In general, larger fragments achieved higher rates of reattachment success. There are several possible explanations for this. For encrusting morphologies, a small fragment cut using a scalpel is likely to have a higher proportion of damaged zooids and fewer fully intact zooids than a large fragment, thereby increasing the susceptibility of the organism to mortality, and requiring a greater proportion of total colony resources be allocated to repairing damaged (i.e., cut) tissue (Bone and Keough 2005). In addition, larger colony masses would conceivably have larger energy reserves for the production of stolons or

adhesive for reattachment, albeit more would be required to hold a larger fragment in place. Fragment size also plays an important role in the growth of a colonial organism once reattached. Edlund and Koehl (1998) found that colonial ascidians detached and then reattached to eelgrass had similar linear growth rates to undisturbed colonies, with larger colonies growing faster than smaller ones. This has implications for establishment success because larger colonies generally have greater chances of survival than smaller ones (Hughes 1984; Winston and Jackson 1984), a finding confirmed by the present study.

5.4.2 Environmental factors influencing survivorship of defouled material

Predicting survivorship rates of defouled biofouling during in-water cleaning is difficult because it is likely to be influenced by many factors, including the extent of damage sustained during the defouling process, environmental conditions of the receiving environment (e.g., water depth, substrate type, water temperature, salinity, light), predation and physical disturbance (e.g., sedimentation rates, currents, wave action). The present study only compared a subset of these parameters (i.e., sedimentation and predation), and found that survivorship of biofouling on plates transplanted to port/marina environs was greater where sedimentation rates were lower. Furthermore, for both high and low sedimentary environments, survivorship was greater where predation was excluded. These findings suggest that both these parameters have a strong influence on the survivorship of defouled material.

Sedimentation rates at the two Nelson sites ($656 - 2125 \text{ g.m}^{-2}.\text{d}^{-1}$) were an order of magnitude greater than those observed at the Picton and Waikawa sites ($28 - 125 \text{ g.m}^{-2}.\text{d}^{-1}$). Deposition of sediment particles can interfere with the feeding and respiration functions of invertebrate communities, and can lead to smothering if sufficient water currents do not re-suspend and export sediments, in particular fine material such as mud (Lohrer et al. 2006; Maldonado et al. 2008). Given the low energy currents at the Nelson sites, it was not surprising that biofouling orientated upwards were covered by a 2 - 3 mm layer of sediment at the completion of the one month experiment, with associated high rates of mortality. Presumed sediment-related mortality was also observed on caged plates deployed upwards at the Picton and Waikawa sites, suggesting that even relatively low levels of sedimentation can have a significant effect on survival. Such effects could be exacerbated in port/marina environments where high concentrations of contaminants (e.g., metals, pesticides,

hydrocarbons) are often associated with the fine sediments (Förstner 1995), however this was not examined in the present study.

The role of grazing/predation in structuring community composition in the shallow subtidal and intertidal regions of the shore has been well studied (Choat and Schiel 1982; Fletcher 1987; Schiel 1988; Andrew 1993; Wright and Steinberg 2001). The two most abundant invertebrates observed grazing/predating on unprotected plates were the polyphagous turbinid snail *Turbo smaragdus* and the cushion star *Patiriella regularis*. Taxa most vulnerable to predation were the soft-bodied ascidian *Botrylloides leachi* and small sponges. Such predation effects were clearly evident as reductions in species richness, and to a lesser extent cover, on plates orientated upwards without predator exclusion at the two Nelson sites. This is consistent with observations made during earlier pilot work where plates were transplanted to seabed environments in the Marlborough Sounds, revealing cushion stars (e.g., *P. regularis*), sea urchins (e.g., *Evechinus chloroticus*) and hermit crabs (*Pagurus* sp.) as the dominant predators (pers. obs.).

Surface orientation plays an important role in determining epibiotic community composition (e.g., Todd and Turner 1986; Glasby and Connell 2001; Hurlbut 1991). However, the relative importance of pre- (e.g., larval behaviour) and post-settlement factors (e.g., sedimentation, light, predation) that alter with surface orientation remains unclear (Glasby and Connell 2001; Glasby et al. 2007). In terms of post-settlement processes (the focus of this study), orientation was found to have an indirect effect on biofouling survival. This was largely attributable to the reduced sedimentation (and subsequent increased survivorship) observed on downward-facing communities relative to upward- and vertically-orientated assemblages. Biofouling organisms on surfaces facing downwards will also experience lower light levels than plates orientated upwards or vertically. Reduced light levels are likely to provide suboptimal conditions for some fouling taxa; however it has also been demonstrated to favour others. For example, Glasby (1999) found that epibiota on experimentally shaded piles (i.e., vertical orientation) were different to non-shaded controls (upward orientated), and had higher cover of bryozoans, serpulid polychaetes, solitary ascidians and sponges.

5.4.3 Considerations for in-water defouling

In-water defouling in the marine environment represents a potentially significant source of biofouling transfer to seabed and water column environments. Clearly, there is a range of factors that affects the ability of defouled organisms to survive transfer, including the defouling method employed, the attributes of the receiving environment and the morphological characteristics of the taxa in question. At a broader level, Ruiz et al. (2000) present several hypotheses to explain patterns of marine invasion in North America. Pertinent to the present study are the complex and poorly understood interactions between ‘Propagule Supply’ and ‘Invasion Resistance’ influencing the establishment of NIS. The Propagule Supply Hypothesis holds that the (i) quantity (i.e., propagule pressure), (ii) spatial dispersion and tempo of supply, (iii) source and (iv) physiological condition of NIS are all contributors to and explanatory of invasion success (Ruiz et al. 2000). If this hypothesis proves true for in-water cleaning, habitats at most risk from NIS introductions are in locations where heavily fouled vessels from multiple source regions are frequently defouled in a confined area using devices where damage to the fouling organisms is low (e.g., hand-held scrapers). Ruiz et al. (2000) also describe the Invasion Resistance Hypothesis, where “invasion patterns result from variation in characteristics of recipient environments that prevent (or facilitate) survival and establishment of NIS.” The present study confirmed that both biotic (predation) and abiotic (sedimentation) attributes of the receiving environment provide ‘resistance’ to invasion. However, it is likely that other complex biotic (e.g., competition, food supply, disease) and abiotic factors (e.g., temperature, salinity, hydrodynamics) also play an important role (Ruiz et al. 2000).

The occurrence of unpredictable stochastic events may also facilitate or prevent the survival and establishment of defouled material, thus the interaction between chance and timing is likely to be a vital factor for establishment success (Crawley 1989). A useful example is the invasion of the Asian clam *Potamocorbula amurensis* into San Francisco Bay (Nichols et al. 1990), where a two year period of climatic extremes provided a ‘window of opportunity’ for the invasion by *P. amurensis* into resident native communities that were in an impoverished and depauperate state. With respect to in-water defouling, the multitude of chance (e.g., fragments settling on rock versus mud substratum) and timing (e.g., season, reproductive state, tide, natural and anthropogenic disturbance) events that vary and interact over small spatial and temporal scales make

predictions of establishment success very difficult in dynamic environments such as ports and marinas.

If defouled material survives and establishes, this can result in significant changes to biodiversity and community composition of recipient assemblages, and could facilitate the establishment of non-indigenous and/or pest species. Awareness of such risks has resulted in some countries restricting or completely banning the practice of in-water defouling. However, in cases where removal to land of fouled structures is not possible, the alternative management option of ‘doing nothing’ also poses risk (Chapter 3). For example, hull biofouling organisms may become accidentally dislodged, or they may be stimulated to spawn, potentially inoculating surrounding habitats (Apte et al. 2000; Minchin and Gollasch 2003). Therefore, it is arguable that in-water defouling may be an appropriate management response in situations where risks posed by biofouling removal are significantly lower to the region than if biofouling remains untreated. Given the highly site-specific nature of defouling survivorship observed in this study, it is conceivable that risks associated with in-water defouling (with particular respect to NIS) may be mitigated by intentionally choosing to defoul structures over sub-optimal recipient habitats. For example, the defouling of an oil rig in deep water in a depauperate offshore soft-sediment habitat is likely to pose a much lower biosecurity risk than if the vessel was defouled in a nutrient-rich, hard substrate near-shore environment. It should be noted, however, that logistical feasibility and cost constraints are both major considerations when undertaking such activities in offshore locations (Chapter 7).

Environmental risks may also be mitigated through the selection of appropriate defouling strategies. This study demonstrates that the defouling method employed (e.g., hand-held scrapers versus mechanical rotating brush devices) has a large influence on the size of fragments generated, which in turn may influence survival success of fragmented material. However, there exists a trade-off between the size-related viability of fragments generated, and the amount of potential inoculum released (i.e., propagule pressure) when considering which tool to use. For example, if a structure that contained species capable of reattachment from small fragments (e.g., colonial ascidians) was defouled using rotating brush devices, it is likely that a large number of small viable fragments would be generated during the defouling process. In contrast, defouling using hand-held scrapers would generate fewer fragments, but the survival potential of each fragment would likely

be greater, based on their larger size and the less destructive method of removal. Furthermore, while this study shows the survival potential of smaller fragments is less than that of larger ones, their dispersal potential (by currents and tides) would be considerably greater, potentially increasing their chance of encountering and settling in a habitat amenable to reattachment and growth.

In addition to the consideration of defouling method, biofouling community composition and maturity on artificial structures can be manipulated through defouling frequency, which in turn may reduce the diversity and survivorship of material defouled to the seabed. There are several main stages of biofouling formation: (1) the accumulation of dissolved organic matter and molecules such as proteins, protein fragments and polysaccharides, (2) the development of a biofilm, (3) colonization by marine fungi, protozoa and algal spores, and (4) colonization by marine invertebrates (WHOI 1952). Given biofilm development on artificial structures can occur within days of being immersed in seawater, cleaning a structure or vessel at this stage of the biofouling process is impractical. Defouling a structure every 2 - 3 months is likely to prevent the accumulation of the majority of fouling organisms; however some well-known early colonisers such as serpulid polychaetes and hydroids may establishment and reach reproductive maturity within this timeframe (Johnston and Keough 2002; Johnston et al. 2002). Given that some non-indigenous and/or pest species are r-strategists (i.e., early colonisation rapid growth, high fecundity; Ricciardi and Rasmussen 1998) and do well in frequently disturbed habitats (Mack et al. 2000), this clearly has consequences for NIS and their management. However, early life-stages of biofouling organisms present on frequently defouled structures are likely to be more susceptible to damage from the mechanical effects of defouling and vulnerable to subsequent environmental conditions on the seabed (e.g., sedimentation and predation).

5.5 CONCLUSIONS AND FUTURE DIRECTIONS

There is global interest surrounding the pros and cons of in-water defouling. This study demonstrates that fragmented and fully intact organisms can survive the defouling process in the short term, and if environmental conditions are suitable, become established in the longer term. This may lead to changes in local community composition, and this clearly has consequences if defouled material has NIS present. Further manipulative studies are needed to elucidate the relative importance of post-settlement factors affecting

survivorship at locations where defouling is routinely undertaken (e.g., port and marina environments). Consideration should also be given to defouling at deepwater offshore locations (e.g., for oil rigs) where social, economic and environmental risks posed by the establishment of non-indigenous species are likely to be much lower.

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Chapter 6

Collateral Effects of In-water Defouling

PREFACE

In December 2007, a semi-submersible drilling rig was defouled in Tasman Bay (New Zealand) prior to departure for Australia. The discovery of brown live mussels *Perna perna* amongst the defouled organisms prompted an incursion response led by MAF Biosecurity New Zealand (MAFBNZ). This chapter provides an example of one of the environmental risks of in-water defouling that were discussed in Chapter 3, and further reinforces the findings of the previous chapter where the potential for short-term survivorship of defouled material was demonstrated. In this chapter, I evaluate the efficacy of the dredge-based incursion response method applied to the natural seabed environment. This sort of incursion response is extremely rare in the real world, and even less frequently reported in the peer-reviewed literature. The chapter also explores the concept of using a density threshold for evaluating eradication success rather than the traditional ‘scorched earth’ approach typically applied in terrestrial and aquatic environments.

Weimin Jiang (Cawthron Institute) helped in developing models to estimate dredging efficiency and the catchability coefficient of defouled material, and my thesis supervisors Barrie Forrest and Jonathan Gardner provided useful comments on a draft manuscript based on the chapter (submitted verbatim to the Journal of Applied Ecology in June 2010).

ABSTRACT

Once established, few marine pests have been successfully eradicated in the marine environment, and there appear to be no published accounts where this has occurred for species that have been introduced into a soft-sediment habitat. The typical goal for eradication attempts is the complete eradication of the target pest; however an equally valid approach is to reduce densities to a level where establishment through successful reproduction is not possible. This chapter describes an eradication attempt following the defouling of a semi-submersible drilling rig in Tasman Bay, New Zealand. Biofouling organisms were removed *in situ* by divers and accumulated on the soft sediment below the rig in a water depth of c. 44 m. The discovery of non-indigenous brown mussels (*Perna perna*) amongst the defouled material initiated a dredge-based eradication response over a 12.6 ha target area. A total of 227 dredge tows covering c. 94% of the target area were undertaken and an estimated 35 tonnes of material defouled from the rig was dredged from the seabed and disposed of in a landfill. In this chapter, the efficacy of seabed dredging as a response tool is evaluated, and the achievement of the density-based eradication success criterion adopted for the response is also demonstrated using two methods. Firstly, the catchability coefficient (q) of defouled material was estimated using catch data, and dredge efficiency (E) was also determined. Initial and remaining estimates of the three mussel species defouled from the rig (*Perna perna*, *P. canaliculus* and *Mytilus galloprovincialis*) were then calculated using estimates of E . The reliability of these estimates was tested by simulations. Estimates of q and E were 0.0054 and 0.30 (respectively) and combined mussel densities at the completion of the incursion response were estimated to be c. 0.5 m^{-2} , well below the success criterion of 10 m^{-2} . This study demonstrates that where complete elimination of a pest is not feasible, alternative success criteria based on biological thresholds can be developed, that if achieved, can effectively mitigate biosecurity risks posed by an incursion. This study also highlights the need for further development of both vector treatment options and incursion response tools, and improved policy surrounding in-water defouling in the coastal environment.

6.1 INTRODUCTION

Eradication successes in the marine environment are scarce, with published examples limited to either intertidal and shallow subtidal environments (e.g., Culver and Kuris 2000; Miller et al. 2004; Anderson 2005), or artificial habitats (e.g., Bax et al. 2002; Wotton et al. 2004). In the above examples, successes have mainly been attributed to the early detection of the target pest and their confined distribution. However, failures (e.g., Hewitt et al. 2005; Pannell and Coutts 2007; Montgomery 2007; Hunt et al. 2009) are more common than successes, and this is often due to the highly inter-connected, expansive, relatively inaccessible and at times hostile marine environment (Forrest et al. 2006; Piola et al. 2009) and the failure of agencies to act quickly. Challenges posed by the marine environment are difficult to overcome, and appear to have hampered the development of effective incursion response methods and tools. Of the response tools available, most have been developed for modified habitats (Wotton et al. 2004, Coutts and Forrest 2007; Piola et al. 2009), because non-indigenous species (NIS) are typically more prevalent on artificial surfaces and structures rather than natural substrata (Glasby et al. 2007). However, natural habitats are also susceptible to invasion (e.g., Wyatt et al. 2005), thus further response tool development to cater for these occurrences is required.

This chapter details a rare example of a successful incursion response in a natural marine habitat, and to the best of my knowledge is the first published account of remote methods of response having been used. In November 2007, a semi-submersible drilling rig was inspected off the coast of New Zealand for NIS prior to departure for Australia (Chapter 7). The inspection identified species non-indigenous to Australia, which led to the rig being towed to a coastal embayment where it was defouled by divers using high-pressure water blasters and hand-held scraper while moored over soft sediments (Figure 6.1). Simultaneous to the defouling operation, brown mussels (*Perna perna*) and nine other species non-indigenous to New Zealand were discovered amongst the material being defouled to the seabed (Table 6.1). Many of these taxa (including *P. perna*) were found to be viable two months after the defouling event. Hence, mitigation of biosecurity risk to Australia (the recipient destination) by defouling had inadvertently exacerbated risk to New Zealand by transferring risk from the rig to the seabed.

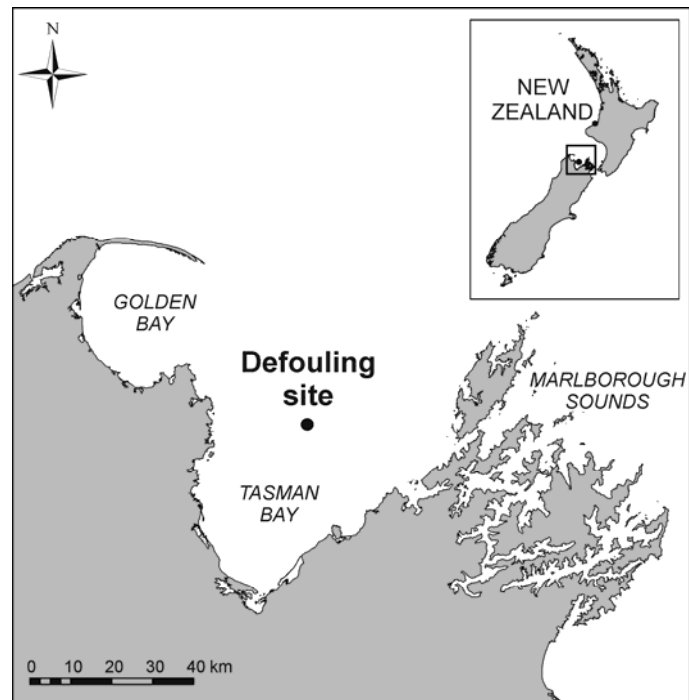


Figure 6.1 Location of the defouling site in Tasman Bay (41° S, 173° E), New Zealand. Water depth at the site ranged between 42 - 44 m.

Table 6.1 Non-indigenous taxa opportunistically sampled from a semi-submersible drilling rig defouled in-water while moored over soft sediments in Tasman Bay, New Zealand.

Taxa	Description	Native distribution
<i>Aulacomya ater</i>	Bivalve	Peru, Chile, Falkland Islands, Argentina, South Africa, Kerguelen Islands
<i>Austrobalanus imperator</i>	Barnacle	Eastern Australia
<i>Austromegabalanus cylindricus</i>	Barnacle	Southwest Indian Ocean
<i>Bugula flabellata</i>	Arborescent bryozoan	Europe
<i>Cnemidocarpa stolonifera</i>	Solitary ascidian	Australia
<i>Herdmania momus</i>	Solitary ascidian	Tropical Indo-west pacific
<i>Megabalanus coccopoma</i>	Barnacle	California and Southwest Mexico to Peru, Galapagos Islands
<i>Monanchora clathrata</i>	Sponge	Australia, Malaysia, Indonesia
<i>Mycale toxifera</i>	Sponge	South East Australia, South Africa
<i>Perna perna</i>	Bivalve	North Africa (Morocco and Algeria), South Africa, South America (Argentina northward to the Caribbean coast of Venezuela)

Following the discovery of viable fouling biota on the seabed two months after defouling activities were completed, several options to eradicate *Perna perna* (and the other NIS) were considered by MAF Biosecurity New Zealand, the Government Agency coordinating the incursion response. These included smothering the target pest with sediments (Coutts 2006), encapsulation of the seabed with plastic (e.g., Creese et al. 2004; Coutts and Forrest 2007), and underwater vacuum/suction methods (e.g., Cheshire et al. 2002; Coutts 2002; Creese et al. 2004). Given the water depth at the site (c. 44 m) and the limited resource availability, dredging the seabed using a modified commercial scallop dredge to collect and remove as much of the fouling as possible was considered the most suitable option.

The typical aim of an incursion response is the complete eradication of target pests, i.e., elimination to zero densities with limited risk of re-invasion. However, for some taxa or response methods this can be cost-prohibitive and was not technically feasible in the study because the dredge efficiency is $< 100\%$. Thus an alternative success criterion was adopted, with the aim of actively reducing target pest densities below an estimated threshold density for successful reproduction. A reproduction threshold density approach to pest eradication is not without precedent in the natural environment. For example, Culver and Kuris (2000) describe the successful eradication of the epizoid sabellid polychaete *Terebrasabella heterouncinata*, a pest to the abalone (*Haliotis* spp.) industry in California. In this example, natural densities of their native hosts (*Tegula funebris*, an intertidal snail) were reduced below the threshold for sabellid transmission. There is a strong theoretical basis to this approach. The ‘Allee effect’ allows that population growth rates can decrease at low population densities due to a variety of mechanisms; including reduced foraging efficiency, lessened defences against predators, and pertinent to the present study, reduced fertilisation success for broadcast spawners (Jang and Diamond 2007). Over the past decade, modelling approaches based on Allee effect have been applied in studies of biological invasion, deliberate introduction of biological control agents and in conservation biology (Drake 2004; Jang and Diamond 2007; Chen and Lin 2008; Guo et al. 2010).

Natural environmental conditions clearly have the potential to complement human intervention and contribute to the success of eradication programmes undertaken in terrestrial, freshwater and marine systems. However, because eradication programmes are essentially ‘uncontrolled experiments’ (Simberloff 2001), they seldom provide the

ability to accurately gauge the relative importance of the two. In the present study, the short-term nature of the incursion response and attributes of the target organism *Perna perna* (e.g., equally vulnerable to capture when alive or dead) provided a rare opportunity to evaluate the efficacy of the response method without being confounded by environmental influences or population fluctuations of the target species. Hence, this chapter aimed to evaluate the efficacy of dredging as an incursion response tool in this deep (c. 44 m water depth) soft sediment environment, and determine whether target species densities had been reduced to a level considered too low for successful reproduction to occur

6.2 METHODS

6.2.1 Dredge-based incursion response

A commercial scallop dredge (width 2.4 m, single bag, nylon mesh size = 40 mm) was used to remove the defouled material from the seabed. A 12.6 ha target area for dredging was identified based on a sidescan sonar survey within the vicinity of the drilling rig defouling site, which indicated that rig-derived material had settled within 200 m of the centre of the defouling site. Furthermore, 3 dredge tows (c. 600 m in length) undertaken outside of the 200 m radius target area confirmed that defouled material was confined to the target area. Given the potential of dredging to spread the defouled material beyond the depositional footprint, retrieval and deployment of the dredge was restricted to within 500 m of the centre of the defouling site.

Between 6 March and 24 April 2008, 227 dredge tows (speed = 2.5 – 3 knots), ranging from 88 – 1330 m in length (average = 419 m), were undertaken within the target area (Figure 6.2). During each dredge tow, the position of the vessel (with a 20 m layback for the dredge) was logged every two seconds and plotted in real time using ArcMap 9.2 GIS software (ESRI, Redlands, CA, USA). At the completion of the dredging-based incursion response, 93.5% of the 12.6 ha target area had been dredged at least once, and c. 60% up to three times (Figure 6.3). All material dredged was stored onboard the vessel, and returned to port where the contents were weighed and disposed of on land.

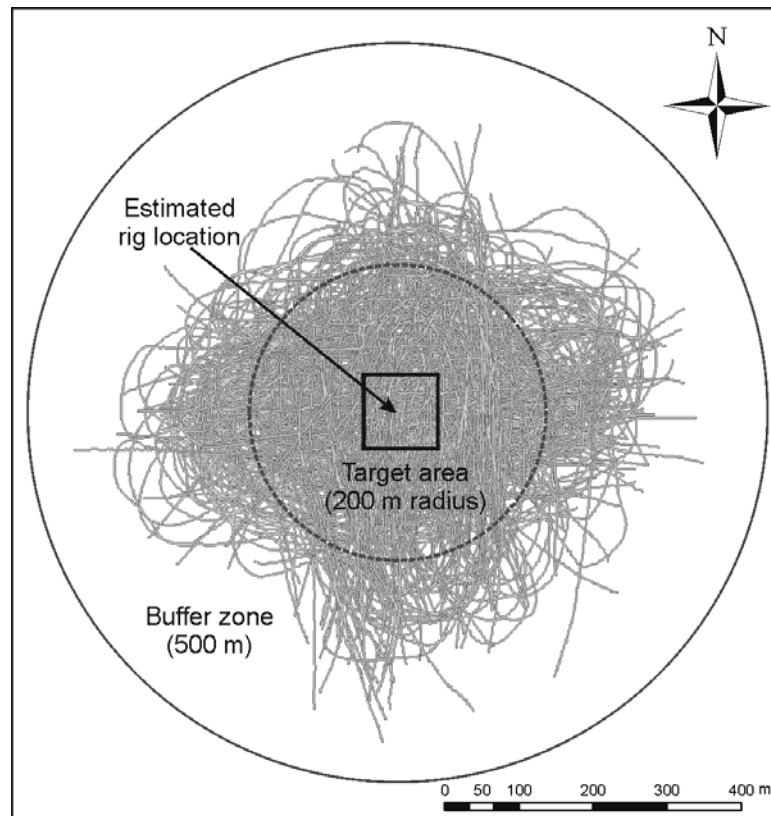


Figure 6.2 Diagram showing total dredging undertaken within the 12.6 ha target area (inner circle). Note that the box in the target area is the approximate size of the drilling rig. Dredge tow widths are drawn to scale.

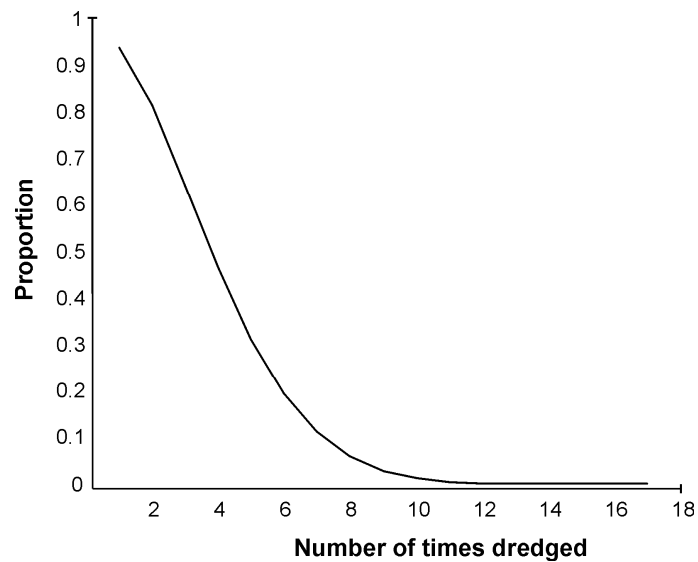


Figure 6.3 The proportion of the target area (12.6 ha) dredged at least a given number of times.

6.2.2 Determination of eradication success

Once the incursion response was initiated, staff at MAF Biosecurity New Zealand (MAFBNZ) liaised with scientists from South Africa familiar with the biology of *P. perna* to estimate the density below which fertilisation success was unlikely. Following these discussions, an eradication success criterion of < 10 mussels per m^2 within the target area was adopted, as *Perna perna* are dioecious (separate sexes) and are thought to require close proximity for successful fertilisation to occur. Given the morphological similarities between mussel species observed on the rig (*P. perna*, *P. canaliculus* and *Mytilus galloprovincialis*) and the potential for misidentification in the field, the threshold density included all three taxa. Given the low incidence of *P. perna* amongst samples collected during the inspection and the dredge-based incursion response ($< 1\%$), the inclusion of non-target mussels in the success criterion added a buffer to allow for error surrounding predictions on the likelihood of successful reproduction at lower than threshold densities. Reductions in the densities of other NIS defouled from the rig (Table 6.1) were also expected to occur given the non-selective nature of the dredging method. However, the reductions in their densities were not monitored given the low perceived risk of establishment and spread from the defouling site.

Estimations of dredging efficiency and target pest densities

Efficiency estimates exist for many dredge types used in soft sediment environments (Vølstad et al. 2000; Gedamke et al. 2005; Walter et al. 2007). However, efficiency changes with engineering factors (e.g., speed of towing vessel, fullness of catch bag), site-specific factors (e.g., water depth, substrate composition, currents) and the characteristics of the target organism (e.g., size, shape, density). Hence, in the present study, it was necessary to determine the site-specific efficiency of the dredge used so that a reliable estimate of residual mussel densities could be obtained and predictions of effort required to reduce target pest densities to a specified level could be made.

Prior to dredging activities, 100 x 125 ml and 50 x 70 ml painted containers filled with sand were randomly distributed within the target area to simulate adult and juvenile mussels. The number of containers retrieved during dredging activities was then recorded and used to determine the catchability coefficient (i.e., the proportion of the population captured by one unit of fishing effort) of fouling biomass (i.e., not specific for *Perna perna*). The Leslie model was fitted to the catch depletion data (Ricker 1975;

Hilborn and Walters 1992), with the catchability coefficient (q) estimated using linear regression,

$$CPUE_t = qN_0 - q\sum C_t \quad (1)$$

where $CPUE_t$ is the catch per unit effort at time t , q is the catchability coefficient, N_0 is the initial number of animals and $\sum C_t$ is cumulative catch at time t . Dredge efficiency (E), i.e., the fraction of animals encountering the dredge that was retained, was estimated according to Walter et al. (2007), by:

$$E = qA/a \quad (2)$$

where q is the catchability coefficient estimated from equation (1), A the total area of target zone; and a the area swept for each individual dredge. Using this equation, a dredging efficiency estimate for each individual dredge tow was obtained and the mean was used as the final estimate.

Over the first two days of dredging, a total of 27 dredges, covering about 1/3 (4.05 ha) of the target area, was conducted, and a total of 5392 kg of biomass was removed. Sub-samples were taken from 15 of the dredge tows and the proportion of each species calculated. From these sampling data, the density of a given species in the 12.6 ha target area was then estimated using,

$$D_i = C * P_i / A_{dredged} / E$$

and the corresponding biomass estimate (Bi),

$$Bi = D_i * A$$

where C is the total mass collected for the first two days, P_i is the proportion of species i , $A_{dredged}$ is the total area dredged within the target zone, E is the dredge efficiency and A is the target area. The average mass for each mussel species was calculated and this was then used to convert mass to density estimates. This approach is termed the ‘swept area method’ and is often used in fisheries science (Sparre and Venema 1998).

After 15 days of dredging, a further 20 dredge tows were undertaken within the target zone to estimate remaining mussel densities and to determine whether the density threshold of < 10 mussels per m^2 had been achieved. Each dredge tow was conducted across the width of the target area (c. 400 m in length), and all mussels dredged per tow were sorted into species and counted. Remaining density estimates for all three mussel species (i.e., *Perna perna*, *P. canaliculus* and *Mytilus galloprovincialis*) were then calculated by:

$$D_i = \frac{1}{n} \sum_j \frac{C_{ij}}{A_{ij}} / E$$

$$N_i = D_i * A$$

where n is the number of dredge tows, C_{ij} is the catch in number of species i from the j th dredge tow and A_{ij} is the corresponding area dredged, and the variance estimator is defined as,

$$\text{var}(N_i) = \left[\frac{A}{E} \right]^2 * \frac{1}{n} * \frac{1}{n-1} * \sum_{j=1}^n (D_{ij} - \bar{D}_i)^2$$

where D_{ij} is the number per unit area of species i from dredge tow j , and \bar{D}_i is the average density of species i over all dredge tows.

6.2.3 Prediction of effort required to remove x% of the initial population

Using the estimate of the catchability coefficient (q) and its associated 95% confidence intervals, it is possible to model the benefits arising from greater dredge effort than used here. To validate the power of this prediction, a similar simulation was conducted using the initial biomass/abundance estimate and the actual number of dredge tows (227). Densities after 227 dredges from the simulation were then compared to remaining densities observed at the completion of dredging activities.

6.3 RESULTS

6.3.1 Catchability coefficient (q) and dredge efficiency (E)

The catchability coefficient (estimated using a depletion model) of defouled material was 0.0054 (0.0028 - 0.0080), indicating that c. 0.5% of defouled material would be removed per unit of dredging effort (c. 418 m dredge tow). The corresponding initial population size (i.e., number of painted containers deployed at the site) was estimated to be 106 (79 - 133), which was an underestimate of the known population size (150). Dredge efficiency was estimated to be 0.30 (± 0.25) after E estimates > 1 were excluded (Figure 6.4).

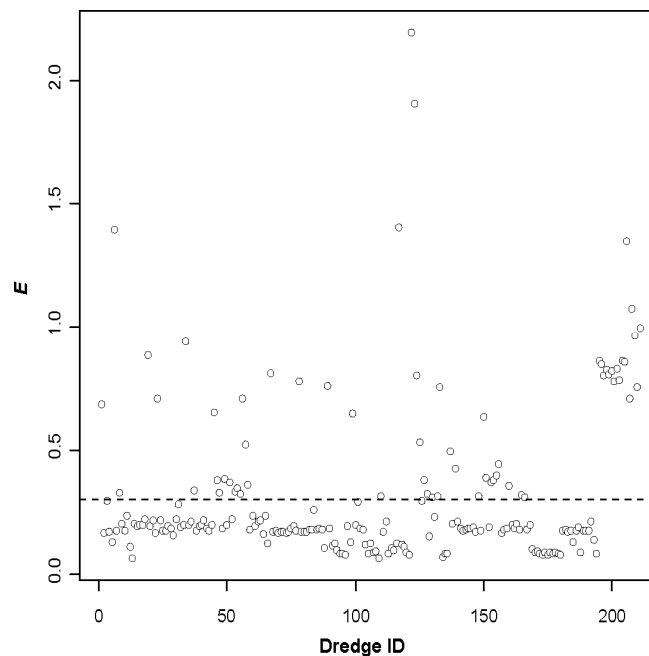


Figure 6.4 Dredge efficiency (E) estimates obtained from equation (2). The black line denotes the mean after exclusion of estimates greater than 1.

6.3.2 Reductions in biomass and densities of target pests

Total biomass (i.e., rig-derived and natural biota) within the target area at the beginning of the incursion response was estimated to be 53.2 tonnes (c. 39.9 tonnes from the rig, and 13.3 tonnes of naturally occurring biota and sediments) based on sub-sampling data.

Simulations using the catchability coefficient (q) estimated initial and remaining biomass within the target area to be 55.8 and 6.5 tonnes, respectively (i.e., a removal of 49.3 tonnes). Both estimates of biomass reduction correspond closely to the c. 50 tonnes of material actually removed from the target area and buffer zone and later disposed of at a landfill, providing confidence in these estimates.

The incursion response was successful in reducing target mussel densities below the density threshold of 10 mussels per m^2 (Figure 6.5). A total of three *P. perna* were identified amongst the 2652 mussel specimens caught during the last 20 dredge tows. The most abundant mussel species amongst the defouled material, *Mytilus galloprovincialis*, declined from 3.9 to < 0.5 individuals per m^2 , representing an 89% reduction in density. For *Perna canaliculus* and *P. perna*, densities reduced from 0.3 and 0.01 (i.e., 10^3 lower than the target density) to 0.01 and $< 0.001 \text{ m}^{-2}$, respectively (i.e., $> 95\%$ of their initial biomass was removed). Using the estimate of the catchability coefficient, the effort needed to remove a given proportion of the population was calculated (Figure 6.6). Based on this model, it was estimated that 71 (47 - 138) dredge tows would be required to remove 50% of the initial population, whilst 232 (156 - 453) dredge tows would be needed to achieve a 90% reduction in population size. As Figure 6.6 illustrates, the reliability of catchability coefficient estimates is important, given the relatively broad range (i.e., 95% CI) in mussel densities remaining at the actual level of dredging effort (i.e., 227 dredge tows).

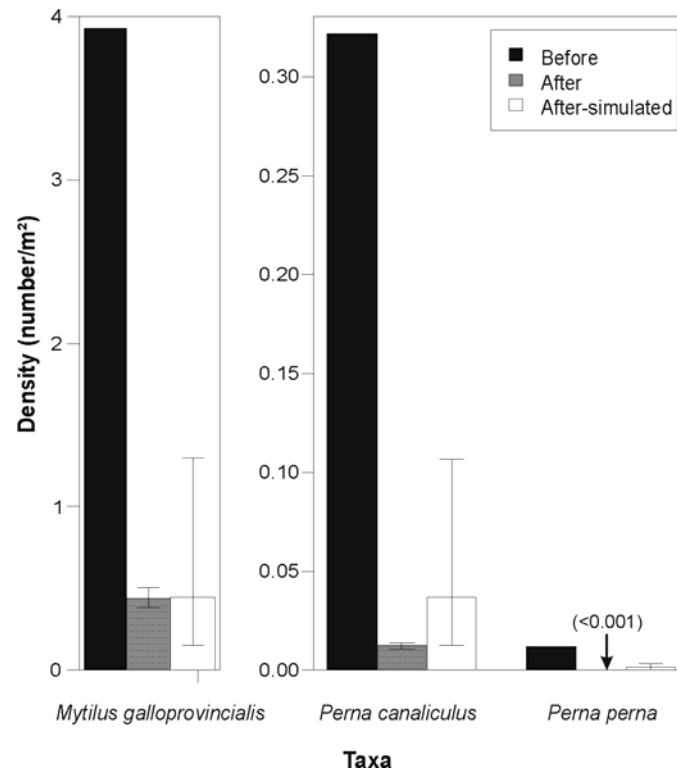


Figure 6.5 Before and after mussel density estimates obtained using the estimate of *E*. Remaining densities simulated using the catchability coefficient are provided for comparison.

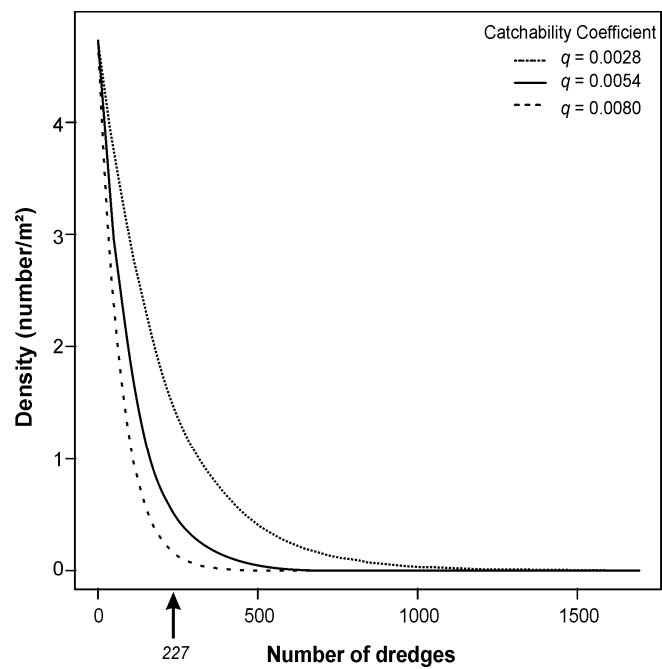


Figure 6.6 Estimated number (mean \pm 95% CI) of dredge tows (c. 418 m in length) required to reduce mussel densities to a specified level. The black arrow indicates the actual number of dredge tows ($n = 227$) undertaken at the completion of the incursion response.

6.4 DISCUSSION

6.4.1 Efficacy of the dredge-based incursion response and reliability of density estimates

The commercial scallop dredge used in the incursion response was successful in removing a high proportion of fouling taxa (including NIS) from the seabed. Large reductions (c. 90 - 95%) of target and non-target taxa were achieved, with approximately 53 tonnes of rig (c. 70%) and resident (c. 30%) material removed from the 12.6 ha target zone. Final mussel density estimates predicted based by the swept-area method (0.46 m^{-2}) were comparable to simulations using the catchability coefficient (0.50 m^{-2}), providing confidence that residual mussel densities at the site were well below the target of 10 m^{-2} .

Two methods were applied to determine whether the endpoint success criterion had been achieved. Firstly, the catchability coefficient estimate was obtained through a depletion experiment using containers filled with sand. An underlying assumption of this experiment was that the containers had the same vulnerability to fishing gear as the defouled material on the seabed. However, the shape of the containers used differs from the shape of the animals and this may have biased the catchability coefficient estimate. Another potential source of bias may arise from the differing spatial distributions of the containers and defouled material. In this experiment, the single containers were randomly distributed over the target zone, but it is possible that the *Perna perna* (and other mussel taxa) had a patchy (i.e., clumped) distribution.

Our estimate of the total population size of containers (using a depletion model) was 106 (70 - 133), an underestimate of the known value of 150. This could be due to containers being buried beneath sediments during the dredging process (and thus become unavailable to the fishing gear), or containers not being detected due to the large volume of material being processed on the vessel (i.e., observation error). Nonetheless, the current estimate of the catchability coefficient is comparable to other fisheries (e.g., Walter et al. 2007). In fact, the estimate of dredge efficiency (E) of 0.3, which was obtained based on the catchability coefficient estimates, was comparable to the dredge efficiencies of 0.3 – 0.4 estimated for scallop fishing gear used in the local scallop fishery (pers. comm. Grant Roberts, commercial fisherman). Even with a much

lower estimate of E (e.g., 0.1), the target density of < 10 mussels per m^2 would have been achieved (Figure 6.7).

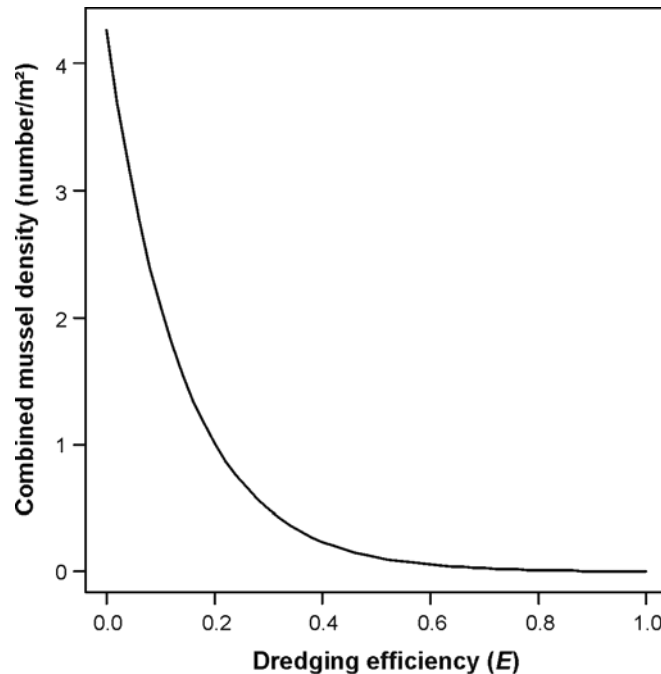


Figure 6.7 Final combined mussel densities (m^{-2}) estimated using the theoretical range of E values.

Estimates using the simulation approach, the second method to assess whether the endpoint criterion had been achieved, were more conservative than density estimates obtained from the swept area method. Density estimates from actual catch data (i.e., using the swept area method) are likely to be an underestimate due to the likely burial of some of the mussels in sediments, as discussed above for containers. Moreover, the modelling approach provides useful information regarding the effort required to remove the population to a target level and thus can be applied to similar projects. In this study, the predictive model based on the ‘catchability’ of the defouled material indicated that over 500 dredge tows (i.e., more than double the actual effort) would have been required to reduce mussel densities to near-zero. Clearly, however, obtaining a reliable estimate of the catchability coefficient (q) is important, as highlighted in Figure 6.6, where predicted mussel densities range between 0.2 and 1.5 m^{-2} (based on the 95% CI).

6.4.2 Density-based versus complete eradication endpoint criteria

For most incursion responses where eradication is the goal, the target endpoint is the elimination of pests to zero densities and the prevention of reintroduction/reinvasion (Piola et al. 2009). As noted above, a large number of dredges would have been required to achieve close to 100% density reductions in target pests. In fact, negligible decreases in pest densities would have been achieved per unit of effort beyond c. 500 dredge tows (i.e., diminishing returns). Therefore, an alternative approach would almost certainly have been required to achieve complete removal of *P. perna* at the defouling site.

If an endpoint success criterion had been developed at the beginning of the incursion response rather than near the end, repetitive dredging of the defouling site may not have been undertaken. This is because the risk of reproduction by individuals of the defouled material was considered to be negligible. While ‘doing nothing’ may seem irresponsible and high risk, there is a widely accepted belief that most invaders fail to establish. This view has been presented more formally as the “tens rule,” i.e., 10% of NIS are ‘released’ into the wild, of which 10% establish, and 10% of established species spread and become a pest (Lodge 1993; Williamson and Fitter 1996). However, despite the seemingly low odds of successful establishment, the response undertaken was considered justified for several reasons. Firstly, even though initial densities of *P. perna* were much lower than the endpoint criterion, there was inherent uncertainty surrounding the reliability of this target, particularly given that a mean density value did not account for ‘clumping’ of mussels on the seabed. Another point considered was the relatively low cost and high chance of success of eradicating the target pest in the confined location compared with the high costs associated with ongoing management if it became established. Thus, given the environmental, social and commercial values at risk in the region, the precautionary principle was adopted and an incursion response initiated.

6.4.3 Factors contributing to the successful eradication of *Perna perna*

It is highly likely that *Perna perna* was successfully eradicated from this site and therefore from New Zealand. Even if conditions were suitable for survival and reproduction, measured and modelled final densities were at least two orders of magnitude lower than the predicted density considered necessary for successful reproduction. In fact, the final density estimate of $< 0.001 \text{ m}^{-2}$ is equivalent to one

mussel per 0.2 hectare of seabed, thus it is highly unlikely that successful reproduction will occur unless there was significant ‘clumping’ (i.e., isolated patches of higher than average densities). Clumping of mussel taxa was rarely observed in the dredge contents, and *P. perna* were sporadically encountered as individuals (not clumps) across the tows.

While successful eradication attempts in the marine environment are scarce in the published literature (e.g., Culver and Kuris 2000; Bax et al. 2002; Miller et al. 2004; Wotton et al. 2004; Anderson 2005), a similar approach to the present study to mitigating biosecurity to the seabed environment following defouling is described by Wells and Jones (2003). That study describes a large self-propelled cutter suction dredge that was fouled by several non-indigenous and invasive mussels, gastropods, crabs and acorn barnacles upon arrival in Geraldton, Western Australia. In order to mitigate biosecurity risks, divers removed fouling from the vessel while it was at berth. A small dredge was then used to remove the defouled material from the seabed. A re-survey undertaken 1 year after the eradication attempt failed to detect any of the non-indigenous taxa identified on the vessel on wharf piles adjacent to where the vessel was defouled. There are presently no plans by MAF Biosecurity New Zealand to undertake a follow up survey at the oil rig defouling site in Tasman Bay, nor is there any targeted surveillance for *Perna perna* colonisation on natural or artificial habitats in the region.

The eradication attempts described by Wells and Jones (2003) and in the present study highlight some of the major challenges associated with marine incursion responses. Coutts and Forrest (2007) list the key elements needed for an incursion response to be successful: (i) sufficient resources to fund a programme to its conclusion; (ii) effective control procedures for the target organism; (iii) a knowledge of invader attributes (e.g., dispersal ability, reproductive biology) that determine ease of population reduction and potential for re-invasion; (iv) prevention of re-invasion through management of spread; and (v) an ability to detect and remove all target pest organisms, or at least reduce pest densities to levels that cannot sustain a viable population. Arguably, all elements were met in the present study, resulting in a successful incursion response. However, equally important to the successful response was the early detection of the incursion and the quick decision by biosecurity managers and the drilling rig operators to act. Simberloff (2003) provides several terrestrial examples where early intervention involving “brute force” (i.e., rather than detailed population biology research) resulted in the successful

eradication of a pest that was initially limited to a small area. However, two potential hazards associated with a rapid response involving “scorched earth” approaches were identified: (i) the economic cost associated with a failed attempt, and (ii) collateral damage to non-target organisms and the environment. Furthermore, if the incorrect response method is adopted, other hazards may include the unintentional spread of the target pest beyond the initial incursion area, and the facilitation of further invasions by NIS due to degradation or modification of the environment (Altman and Whitlatch 2007; Piola and Johnston 2008; Dafforn et al. 2009).

6.4.4 Wider considerations for dredging as a response tool

Dredging of the seabed was selected as the best available tool to mitigate biosecurity threats posed by the defouling of *Perna perna* over soft sediments (c. 60:40 split of mud:sand-sized particles) in relatively deep water (c. 44 m). However, it is conceivable that defouling may occur over habitats less suitable for dredging (e.g., reef habitat) or target organisms may not be amenable to removal by dredging (e.g., infaunal taxa). Another potential concern is that dredging may in fact exacerbate biosecurity risk in some situations. For example, taxa capable of growing from small fragments may be spread in higher than original numbers if disturbed by dredging equipment. Additionally, the physical stress imposed by dredging may stimulate some taxa to spawn (this is dependent on state of gonad development), and although adults are removed by the dredge, the newly formed larvae have been liberated into the receiving environment and are capable of dispersal.

Collateral effects of the incursion response method also need to be considered against the potential damage posed by the target pest species (Myers et al. 2000). The deleterious effects of dredging to marine benthic habitats and communities are well described (Currie and Parrie 1994; Kaiser et al. 2006), resulting from direct physical damage and removal of biota, and the resuspension of sediments. However, despite more than half of the target area being dredged three times, one month later there was no detectable change to the sediment grain size distribution, organic content and infauna community composition compared with reference sediments (G. Hopkins, unpublished data). However, this was not entirely unexpected given that the dredge used had no ‘teeth’ and was designed to skim over the sediment surface rather than disturb the sediment profile. Hence, it is likely that infaunal disturbance was minimal despite intensive dredging, as opposed to the less plausible explanation that the one month

period following dredging activities was sufficient for a full recovery of infauna within sediments.

While short-term effects to sediments appear to be negligible, the dredge used during the incursion response was clearly efficient at removing both rig-derived and naturally occurring epibiota (e.g., the flat oyster *Ostrea chilensis*) at the defouling site (G. Hopkins, pers. obs.). Removal of epibiota by dredging has received a lot of attention in the literature, and has been linked to reductions in species diversity (Thrush et al. 1995) and increased near-bottom turbidity (Wilber and Clarke 2001). In particular, sensitive habitats such as biogenic reefs (e.g., bryozoan corals) are likely to be susceptible to long-term damage from dredging activities (Cranfield et al. 2003; Kaiser et al. 2006). Therefore, dredging might not be an appropriate incursion response method in areas where there are high value habitats sensitive to dredging impacts, particularly if alternative and more selective methods are feasible (e.g., removal by divers). Nonetheless, it is arguable that the permanent establishment and spread of a pest justifies a scorched earth approach where the method has localised and reversible effects and no other alternatives are available.

6.5 CONCLUSIONS

This study demonstrates that any incursion response in the marine environment can be successful when key elements are met. In particular, decision-makers were quick to respond to the localised incursion, a suitable method was employed, the rig operators were willing to fund the response to a satisfactory endpoint, and experts knowledgeable about the biology of the target pest were consulted so that meaningful success criteria could be developed. Given the limitations of the response method and environmental challenges associated with the defouling site, the complete eradication of the target organism was not considered feasible or necessary. Instead, the primary success criterion of the response was tailored to the biology of the target organism, with the aim of reducing *Perna perna* densities on the seabed to levels where the establishment of a self-sustaining population was considered unlikely. However, it is acknowledged that while the density-threshold approach was appropriate in the present study, there are situations where complete eradication to zero densities using a method or tool would be required for a response to be successful (e.g., for target species able to reproduce asexually). It is important to reiterate that the need to respond to *P. perna* in the first

instance arose from defouling a drilling rig in the absence of a more biosecure method to manage biofouling. Hence, this study highlights that in-water cleaning methods that do not destroy or collect fouling material upon removal are capable of transferring biosecurity risk from a vessel/structure to the seabed. Therefore, there needs to be further development of both vector treatment options and incursion response tools, and improved policy surrounding in-water defouling in the coastal environment.

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Chapter 7

Challenges Associated with Pre-border Management of Biofouling on Oil Rigs

PREFACE

The events surrounding the defouling of an oil rig in coastal waters and the resulting incursion response (Chapter 6) have led to heightened awareness in New Zealand of biosecurity risks posed by these structures, such that future rig transfers into the country are likely to come under much greater scrutiny. This chapter draws together the various themes in preceding chapters of this thesis, including; understanding and characterising risk (Chapter 2), potential risks from management of biofouling (Chapter 3), the efficacy of in-water treatment tools (Chapter 4), the potential for transferring biosecurity risks to the seabed when in-water tools are used (Chapter 5), and the efficacy of incursion response tools in mitigating risks from an incursion to the seabed (Chapter 6). In this chapter, I use two case studies to highlight existing challenges associated with the pre-border management of biofouling risks, and provide a framework for future oil rig transfers into a new bioregion.

My thesis supervisors Barrie Forrest and Jonathan Gardner provided useful comments on a draft manuscript identical to this chapter (submitted to Marine Pollution Bulletin in June 2010).

ABSTRACT

The potential for oil rigs to transport diverse, reef-like communities in high biomass around the globe makes them high risk vectors for the inadvertent spread of non-indigenous species (NIS). However, at present the risk of NIS transfers with their movements is largely unmanaged at a national level; exceptions to this include countries such as Australia and New Zealand where pre-border management has recently been implemented. This chapter describes two case studies where a suite of pre-border management approaches was applied to semi-submersible drilling rigs. In the first case study, a drilling rig was defouled in-water prior to departure from New Zealand to Australia. Risk mitigation measures were successful in reducing biosecurity risks to the recipient region, but they resulted in the unintentional introduction of the non-indigenous brown mussel (*Perna perna*) to New Zealand when the rig was defouled in-water by divers. In the second case study, lessons learned from this high-profile incursion resulted in a more structured approach to pre-border management being applied to an oil rig travelling to New Zealand from Trinidad and Tobago, via Australia (the second case study). In these two case studies, numerous challenges associated with the pre-border management of oil rigs are identified, including: (i) logistical and safety issues associated with inspecting sub-surface structures at offshore locations, (ii) high costs to the rig operators due to unscheduled delays during NIS inspections and potential risk reduction measures (e.g., in-water defouling, dry-docking), (iii) determining what NIS to look for and the associated difficulties of identifying a broad range of taxa in the field, (iv) the worldwide shortage of suitable land-based facilities to service fouled rigs, (v) the ongoing absence of biosecure, in-water methods to treat/remove fouling, and (vi) a lack of knowledge surrounding the efficacy of alternative treatment options (e.g., prolonged exposure of biofouling to air, encapsulation in plastic). This primary purpose of this chapter is to highlight the need for the development of effective in-water treatment options for oil rigs. In the longer term, improvements in antifouling paint coatings and rig design have the potential to reduce colonisation rates of these large structures and make it easier for risk-mitigation methods to be implemented.

7.1 INTRODUCTION

The potential for spread of non-indigenous species (NIS) by fouling on vessels and marine structures is well recognized (Gollasch 2002; Minchin and Gollasch 2003; Coutts & Taylor 2004; Davidson et al. 2009; Yeo et al. 2010). There is a risk that if NIS are present, they may be introduced into a recipient region through spawning (Apte et al. 2000), accidental dislodgement (Acosta and Forrest 2009), or intentional removal but unintentional introduction (Chapter 6). Given the cost and logistical difficulties in managing NIS once established (e.g., Bax et al. 2001; Finnoff et al. 2007), it is preferable to put resources into preventing their inadvertent introduction with human activities.

Recent studies highlight the need to improve pre-border management of biofouling risks associated with large structures from the oil and gas industry. In particular, a common theme emerging from biofouling surveys undertaken on oil rigs is the presence of advanced reef-like communities having a large biomass (Wanless et al. 2009; Yeo et al. 2010; Chapter 6). This is because the application of antifouling paints to structures such as oil rigs is rarely undertaken, because existing paint technologies are reasonably ineffective on structures that remain idle for extended periods of time (Floerl et al. 2005; Ferreira et al. 2006). Therefore, close to 100% of submerged structures on a rig are likely to be colonized by a diverse range of foulers within months of being cleaned. This is in contrast to the much lower levels of fouling encountered on other vessel types (Coutts and Taylor 2004; Davidson et al. 2009). Furthermore, given the much lower hydrodynamic forces (and associated indirect effects such as reduced feeding) experienced at the speed at which rigs are towed (c. 5 knots) compared with a merchant vessel travelling at > 20 knots (Coutts et al. 2010; Yeo et al. 2010), fouling survivorship during transfers between drilling sites (and bioregions) is likely to be higher on a slow moving towed rig than on a faster moving vessel.

There are estimated to be 6 submersible drilling rigs, 186 semi-submersible drilling rigs, 451 jack-up drilling rigs, 163 floating-production-storage-offloading (FPSO) structures, and 94 drill barges or ships in operation worldwide (Yeo et al. 2010, and references therein). While such structures only account for a small proportion of international vector traffic, their ability to accumulate high levels of fouling biomass and diversity is of concern. While there are only a few documented examples where fouling on such

structures has lead to the incursion of a NIS into the recipient environment (e.g., Ferreira 2003; Chapter 6), there are numerous examples where NIS have been found on a rig and hence have the potential to be transported to a non-native recipient region (Benech 1978; Foster and Willan 1979; Ferreira et al. 2006; Galil 2008; Yeo et al. 2010).

There is a broad range of pre-border management options available for reducing risks from potential vectors of NIS, including: (1) pre-departure inspections for NIS, (2) risk profiling to identify high risk vectors (e.g., based on factors such as operational and maintenance histories, likely infection mechanisms and intended movements in the recipient region), (3) the development of border standards for biofouling extent and vector hygiene (e.g., time since last dry-docking), and (4) biofouling treatment or removal to mitigate unacceptable risks. Each of these pre-border management options has its own limitations and challenges when applied to rigs, mainly due to the large size of these structures and associated issues with fouling removal and treatment if NIS are discovered. In this chapter, two case studies are used to highlight these challenges, and provide a framework for future rig transfers between bioregions that can equally be applied to a range of vessel types arriving at the border. The first case study details a decision to defoul a rig in-water to mitigate biosecurity risk before transport to Australia but which led to the incursion of a potential pest to New Zealand waters. As a result of this incident a subsequent proposal to transport a rig from Australia to New Zealand resulted in the implementation of a systematic risk-based approach to mitigate biosecurity threats that has become the standard for rigs entering New Zealand territorial waters. This latter example is described in case study 2.

7.2 CASE STUDY 1: PRE-BORDER MANAGEMENT RESULTING IN AN INCURSION

7.2.1 Background

In 2004, a semi-submersible drilling (Rig A) arrived in New Zealand from South Africa, where it had been dry-docked and completely defouled. Over the next three years, the rig drilled at several offshore locations along the New Zealand coast and in Bass Strait, Australia. In November 2007, the rig operators prepared to tow the rig from New Zealand to Portland (Australia) for refurbishments. Because the rig was going into port, there was the requirement by the Department of Sustainability and Environment

(Victoria, Australia) for the rig to be inspected for target NIS prior to arrival. The target list comprised *Perna canaliculus* (green-lipped mussel), *Undaria pinnatifida* (Asian kelp), *Didemnum vexillum* (colonial ascidian), *Asterias amurensis* (Northern Pacific sea star), *Maoricolpus roseus* (New Zealand screw shell), *Grateloupia turuturu* (red alga) and *Charybdis japonica* (Asian paddle crab).

7.2.2 Rig inspection while in source region

On 28 November 2007, the rig was inspected for the target taxa by divers and using a remotely operated vehicle (ROV) while it was moored off the west coast of the North Island, New Zealand (Figure 7.1). The inspection was undertaken while the rig was de-ballasted, therefore only the two 9 x 11 x 100 m (height x width x length) pontoons were submerged. Specimens collected from the rig were preserved and later identified to the lowest practical taxonomic level. Considerable fouling (c. 150 mm thick) was observed on the rig pontoons, with a total of 23 species identified from samples collected haphazardly by divers from representative areas of the structure (Table 7.1). This is by no means an exhaustive list of taxa on the structure, but nonetheless provides a good indication of the dominant taxa present. Fouling biomass was dominated by mussels (mainly *Mytilus galloprovincialis* and *Perna canaliculus*), but a diverse assemblage of other taxa was also present, including ascidians, bryozoans, colonial anemones, tubeworms, barnacles and crabs (Figure 7.2). Approximately one-third of species identified were non-indigenous to New Zealand, including five taxa that had not been previously described in New Zealand waters (Table 7.1).

Table 7.1 Taxa sampled from Rig A and their present biosecurity status in New Zealand (* = first record).

Taxa	Description	Biosecurity status in New Zealand
<i>Aulacomya atra</i>	Bivalve	Alien*
<i>Austrobalanus imperator</i>	Barnacle	Alien
<i>Austromegabalanus cylindricus</i>	Barnacle	Alien*
<i>Balanus trigonus</i>	Barnacle	Indigenous
<i>Bugula flabellata</i>	Arborescent bryozoan	Alien
<i>Cnemidocarpa bicornuta</i>	Solitary ascidian	Indigenous
<i>Cnemidocarpa nisiotis</i>	Solitary ascidian	Indigenous
<i>Cnemidocarpa stolonifera</i>	Solitary ascidian	Alien*
<i>Corynactis</i> sp.	Colonial anemone	Indeterminate
<i>Dicathais orbita</i>	Whelk	Indigenous
<i>Herdmania momus</i>	Solitary ascidian	Alien
<i>Megabalanus coccopoma</i>	Barnacle	Alien
<i>Microcosmus</i> sp.	Solitary ascidian	Indeterminate
<i>Monanchora clathrata</i>	Sponge	Alien
<i>Mycale toxifera</i>	Sponge	Alien*
<i>Mytilus</i> sp.	Bivalve	Indeterminate
<i>Notomegabalanus decorus</i>	Barnacle	Indigenous
<i>Perna canaliculus</i>	Bivalve	Indigenous
<i>Perna perna</i>	Bivalve	Alien*
<i>Plagusia chabrus</i>	Crab	Indigenous
<i>Pyura spinosissima</i>	Solitary ascidian	Indigenous
<i>Schizoporella errata</i>	Arborescent bryozoan	Cryptogenic
<i>Scruparia ambigua</i>	Arborescent bryozoan	Cryptogenic

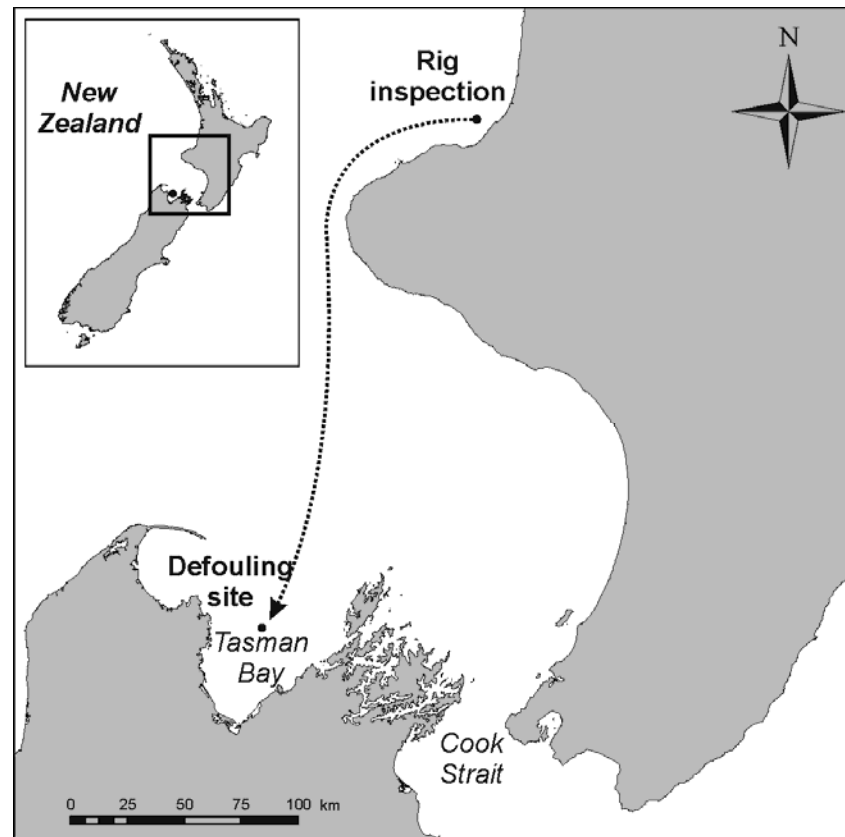


Figure 7.1 Rig A was inspected off the New Zealand coast (38° S, 174° E) for a list of target taxa which are non-indigenous to Australia. Upon the discovery of New Zealand green-lipped mussels (*Perna canaliculus*), it was towed to Tasman Bay (41° S, 173° E) where it was defouled in-water by divers.



Figure 7.2 A sub-sample of sessile and mobile fouling organisms collected from the hull of the semi-submersible drilling rig (Rig A), including the endemic New Zealand green-lipped mussel, *Perna canaliculus* (far left) and the blue mussel, *Mytilus galloprovincialis* (far right).

7.2.3 Risk mitigation

Following the discovery of *Perna canaliculus*, the rig was towed at low speed (c. 5 knots) to Tasman Bay (Figure 7.1) where it was defouled in-water over a period of 6 weeks by divers using high-pressure water blasters and hand-held scrapers. Material removed from the rig was not retained and thus accumulated on the seabed immediately below the rig. During the defouling operation, a single brown mussel (*Perna perna*) and nine other non-indigenous species were identified in samples collected from the rig during the initial inspection (Table 7.1), which had presumably colonised the rig while in South Africa and/or Australia. Concerns regarding the potential effects of *P. perna* on the aquaculture industry and natural environment led to a full incursion response coordinated by MAF Biosecurity New Zealand and fully funded by the rig operators. The response involved repetitive dredging of the defouling site using a modified commercial scallop dredge over a period of 7 weeks in order to reduce *P. perna* densities to a level at which the establishment of a self-sustaining population was considered unlikely (Chapter 6).

Sub-samples (15 replicates, each of 15 L) collected from the dredge material over the first two days of the incursion response revealed that the majority of taxa were intact and alive. Nonetheless, some mortality was observed (evident by the presence of shell material and barnacle plates) and this was thought to be due to the physical damage encountered during the defouling process (i.e., damage from high pressure water blasters and hand-held scrapers), and post-defouling environmental conditions (Chapter 5). The high survivorship of mussels (c. 80-90%), compared with barnacles (c. 50%) and soft-bodied organisms was attributed to their attachment biology. Mussels attach by byssal threads, which when broken have little affect on the overall health of the organism. In contrast, sessile taxa such as barnacles cement or glue themselves directly to surfaces, thus their removal can result in substantial tissue damage (for soft-bodied taxa such as colonial ascidians) or structural damage (e.g., the removal of basal plates of barnacles) that may lead to immediate mortality or increased susceptibility to predation (Chapter 5). Observations from Chapter 5 also suggest that the longer-term survival of taxa that endured the initial defouling disturbance and subsequent environmental conditions is unlikely, due to predation and sediment-induced mortality. This was not assessed because the discovery of NIS prompted a full-scale incursion response - the defouling site was repetitively dredged.

7.3 CASE STUDY 2: PRE-BORDER RISK ASSESSMENT AND PATHWAY-BASED MITIGATION

7.3.1 Background

In April 2009, a semi-submersible drilling rig (Rig B) was transported from Trinidad and Tobago to Australia onboard a heavy-lift vessel (HLV). Following several months of refurbishments in Port Phillip Bay, the rig drilled at several sites in Bass Strait before being towed to New Zealand (February 2010). Heightened awareness of potential biosecurity risks posed by oil rigs following the defouling of Rig A prompted the operators of Rig B to consider biosecurity risk mitigation options well in advance of departing from both source regions (i.e., Trinidad and Tobago, and Australia).

7.3.2 Risk mitigation

Biofouling removal and pre-arrival inspection

A range of mitigation measures was implemented to ensure that the rig posed little or no risk to Australia and New Zealand's marine biosecurity (Hopkins and Forrest 2009a). The bottom and ends of the two rig pontoons were defouled in-water by divers in Trinidad and Tobago before the rig was loaded onto a heavy-lift vessel (HLV). This was undertaken because these areas of the rig would be inaccessible for treatment once the rig was onboard the HLV, and was considered appropriate given that the rig had not left the region since last being defouled. Additionally, prolonged exposure to air (c. 45 days) during the journey from Trinidad and Tobago to Australia was expected to kill off most of the remaining external biofouling (Figure 7.3). However, as an additional safeguard, the external surfaces of the rig were also mechanically defouled prior to arrival in Australia, except for some small areas that could not be accessed (e.g., around brackets that held the rig to the HLV). Sea chests were cleaned back to bare metal and new antifouling paint applied. All ballast water in the rig was completely cleaned out when loaded onto the HLV; this included sediments in the bottom of the pontoons. Rig B was inspected for viable biofouling organisms upon arrival in Port Phillip Bay while still onboard the HLV (Hopkins et al. 2009). Of the small amount of fouling that was observed on the rig, no living material was found.

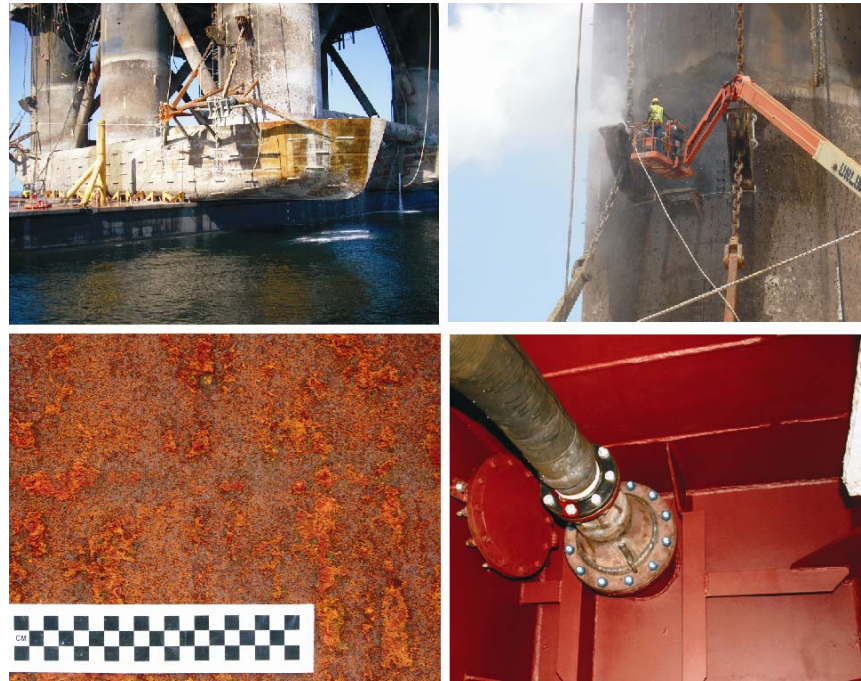


Figure 7.3 Rig B onboard a heavy-lift vessel (HLV). Pontoons were defouled either in-water by divers or by surface high pressure water blasting (top). During an inspection upon arrival in Australia, > 95% of sub-surface structures were devoid of fouling (bottom left). The six sea chests were defouled and a new coat of antifouling paint applied while in transit on the HLV (bottom right).

Monitoring the rig for colonisation by NIS

A preliminary assessment was undertaken to address biosecurity risk to New Zealand subsequent to the arrival of the rig in Australia. The preliminary assessment identified the potential for the rig to be colonized by NIS (both larvae and mobile adult life-stages of invertebrate taxa) while moored in Port Phillip Bay during the refurbishment period (Hopkins et al. 2009). Many invasive marine species have established in Port Phillip Bay (Currie and Parry 1999; Hewitt et al. 1999, 2004), including five species defined as Unwanted Organisms under New Zealand's Biosecurity Act 1993: the Mediterranean fan worm (*Sabella spallanzanii*), the Northern Pacific sea star (*Asterias amurensis*), the European shore crab (*Carcinus maenas*), the clubbed tunicate (*Styela clava*) and the Asian kelp (*Undaria pinnatifida*). *A. amurensis* and *C. maenas* are the only listed Unwanted Organisms present in Port Phillip Bay that have not already become established in New Zealand. *Sabella* has been recorded in two ports in New Zealand and is actively managed.

To detect the potential colonisation of the rig by these taxa and other species non-indigenous to New Zealand, substrata suitable for settlement by a diverse range of invertebrate taxa were suspended from the rig during refurbishments in Port Phillip Bay (Hopkins et al. 2009). During this monitoring, none of New Zealand's Unwanted Organisms were detected. In fact, throughout the deployments there was consistently low recruitment by a limited number of taxa; namely *Electroma georgiana* (a small bivalve), *Pomatoceros* sp. (calcareous tubeworm), *Obelia dichotoma* (hydroid), *Austrominius modestus* (barnacle), *Coscinasterias muricata* (11-armed sea star), and *Pyura stolonifera* and *Ciona intestinalis* (ascidians). Several unidentified polychaetes (bristle worms) and small crustaceans (amphipods, copepods and ostracods) were also encountered.

Prior to departing for Bass Strait in southern Australia, the rig was inspected by divers for adult crabs and sea stars (i.e., adult mobile pests), as the preliminary risk assessment also identified the potential for adult NIS to also colonise the rig. No adult taxa were detected on the rig pontoons or in the sea chests. This was not surprising given that the mooring configuration of the rig while in Port Phillip Bay was such that no part of the sub-surface structure was connected to habitats (i.e., the adjacent seabed or wharf) where sea stars (e.g., *Asterias*) and crabs (e.g., *Carcinus*) occurred.

7.3.3 Formal risk assessment

Semi-submersible drilling rigs have the same suite of potential mechanisms for marine pest transfer as that present on merchant vessels (i.e., ballast water, external fouling and sea chests). However, despite the presence of suitable mechanisms, biosecurity risks to a recipient region from rig activities can only exist in a situation where they lead to the transfer of a pest species that subsequently becomes established. For pest establishment to take place a chain of events must occur, starting with the infection of the structure by NIS in its source region, followed by the survival of NIS during transit to the recipient region, and finally its subsequent release and establishment (Figure 7.4). Each of these events was formally assessed with respect to New Zealand Unwanted Organisms present in Port Phillip Bay and taxa identified during pest monitoring on the rig.

Of the taxa identified during invertebrate larvae monitoring, colonisation of the rig by *Electroma georgiana* and *Pyura stolonifera* was of most concern, as both species had been described as reaching high densities in natural and/or artificial habitats. However,

aspects of the biology of both species were considered likely to result in a low biosecurity risk to New Zealand. Firstly, *E. georgiana* forms a very weak byssal thread attachment to natural (e.g., seagrass and macroalgae) and artificial structures (e.g., wharf pilings), and as such is easily dislodged by wave action or strong current (pers. comm., G. Edgar, Aquenal Pty Ltd.). This inability to attach firmly probably explains why this species has not previously been recorded in hull fouling studies in New Zealand (e.g., Coutts and Taylor 2004; Chapter 2). Additionally, even if low densities were present on the rig (e.g., in hydrodynamically-protected niche areas) and survived transport to New Zealand, it was considered unlikely that *E. georgiana* would be able to reproduce successfully and produce enough larvae to colonise coastal habitats in sufficient numbers to form a self-sustaining population. *Pyura stolonifera*, a known pest species, was also unlikely to be present on the rig in high densities. This was because only one individual was found on settlement plates during monitoring, suggesting that colonisation of the rig was unlikely to have occurred during its time in port. Furthermore, this species is reported to have very limited larval duration (i.e., 2-4 hours before settlement, Clarke et al. 1999), and therefore a limited ability to disperse to suitable rocky shore habitat from any of the offshore drilling sites (tens to hundreds of kilometres from land).

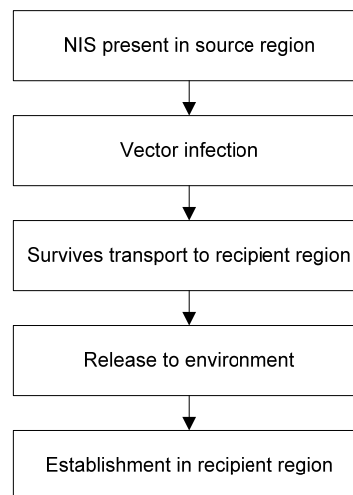


Figure 7.4 The chain of events in the invasion process that must occur for the activities of a rig to lead to biosecurity risks to recipient region.

Given the uncertainty surrounding the ability of *E. georgiana* to remain attached to the rig during transit, a video inspection of the rig pontoons was undertaken by ROV two months after the rig had travelled to Bass Strait (> 200 km from the refurbishment location), immediately prior to departure for New Zealand. No *E. georgiana*, *P. stolonifera* or any of the target pests were observed on the rig pontoons, providing assurance that the scenario of very low risk portrayed in the risk assessment was an accurate reflection of actual events.

7.4 DISCUSSION

7.4.1 Challenges associated with pre-border management

The pre-border management of biofouling pathways/vectors is in its infancy, and management approaches applied to some vessels are only a recent development. As such, there remain many challenges to overcome. In the first case study, mitigation measures were successful in reducing biosecurity risks to the recipient region (Australia). However, it was never determined that risks were completely eliminated. In fact, given the logistical difficulties of in-water defouling, it is likely that *Perna canaliculus* and other New Zealand species remained on the rig, especially in niche areas. Furthermore, pre-border management attempts (i.e., in-water defouling) resulted in an incursion to New Zealand by several NIS, including the brown mussel *Perna perna*. Lessons learned from this experience led to the more structured approach of risk assessment and mitigation, described in the second case study. The approach applied in that case was well received by regulatory agencies in both New Zealand and Australia, and received buy-in from the rig operators despite it adding significant cost and inconvenience to their normal operating procedures. While this example demonstrates that pre-border management of oil rigs is achievable under existing rules and regulations, it is likely that further policies will be put in place, both at a regional and global scale, to manage biofouling risks posed by international movements of vessels, rigs and other structures.

Several countries, including New Zealand and Australia, are in the process of developing border standards for biofouling. At a global scale, the International Maritime Organisation (IMO) was requested in July 2007 to consider a request from several member States and observers for vessel biofouling to be included in the agenda of the IMO work plan. However, given the operational profile and maintenance

schedule of rigs, they will almost always be fouled by a diverse range of taxa unless they have recently (i.e., within several months) been dry-docked or defouled (e.g., onboard a heavy-lift vessel or using in-water methods). Thus, if rigs are required to meet border standards based on fouling extent (rather than fouling composition or the presence of NIS), it is likely that some form of fouling removal or treatment (e.g., air exposure, chemical methods) will be required prior to arrival in a new bioregion. This will have serious ramifications for rig operators, given the large costs associated with unscheduled delays and the limited options to remove or treat biofouling on a structure of this size (see Section 7.4.2).

In many cases, rigs operate at offshore locations that are tens to hundreds of kilometres from suitable habitat for biofouling organisms. For some taxa (e.g., many colonial ascidians), this distance is well beyond their natural dispersal range (Clarke et al. 1999; Smith et al. 2007), while for others (e.g., bivalves) dispersal from this distance is possible. For example, the feeding larvae of the New Zealand endemic green-lipped mussel (*Perna canaliculus*) found on Rig A can spend 3-5 weeks being passively dispersed by ocean currents until a suitable settling substrate is encountered. Despite the large potential for dispersal distances (e.g., hundreds of kilometres), it has been found for other mussel taxa (*Mytilus galloprovincialis*) that wind driven currents coupled with frequent reversals can limit dispersal ranges to tens of kilometres (e.g., McQuaid & Phillips 2000). Thus, there are conceivably situations where the requirement to meet border standards will arguably be overly-restrictive (i.e., if there are NIS present that are not capable of establishment or spread from the rig location).

It is important to realise, however, that there may be situations where an offshore drilling rig needs to move to more sheltered waters closer to land at short notice (e.g., due to structural failure or unscheduled maintenance). Moving a rig to within dispersal range of coastline habitats undoubtedly increases the likelihood of incursion via spawning. In situations like this, there may be a requirement for the rig to be inspected for NIS, and if discovered, treated so that risks are mitigated. However, as highlighted in the first case study, caution is required over in-water methods as there is the possibility that risk mitigation may result in the transfer of biosecurity risk elsewhere. An additional risk is that posed by supply craft that service rigs: these may act as a mechanism for transferring risk from the rig to the coastline. However, given that supply craft are typically well maintained, travel at high speed and are in frequent use,

risks associated with supply vessels are likely to be comparable to similar-sized craft arriving at the border (e.g., small merchant vessels). Furthermore, given their operational profile and routine maintenance, these craft are more likely to meet stringent border standards.

If stringent border standards are imposed on oil rigs (e.g., such as that currently proposed by MAF Biosecurity New Zealand), identifying structures that fail to meet the criteria is likely to be straight forward. For example, video footage of submerged structures would be sufficient to characterise fouling extent, and unless the rig has been defouled within the last 1-2 months, an assumption that marine invertebrates are present on the rig can be made with confidence. However, if border standards are ‘relaxed’ for rigs, and risks are assessed on a case-by-case basis (i.e., the approach applied for Rig B), some form of risk assessment will be required to identify high risk situations. Risk assessment aimed to identify high risk pathways of NIS also has its own challenges and limitations. For example, while there have been numerous studies documenting species compositions on rigs (e.g., Bram et al. 2005; Wanless et al. 2009), it is difficult to reliably predict the presence of NIS without explicit knowledge of taxa in the region and their ability to disperse to the rig location. For short term deployments, knowledge of seasonal breeding cycles of target NIS would also be required, and in many cases this information may not exist.

For rigs considered high risk, it is likely that some form of confirmation of NIS would be required before expensive and time-consuming mitigation measures were implemented (e.g., dry-docking or in-water defouling). Recent experience from undertaking the biosecurity inspection of Rig B highlighted the large amount of preparation work required to get onto the rig (e.g., hazard management plans, helicopter underwater evacuation training, etc), particularly given that divers were being used to collect specimens for taxonomic verification. In addition to inspection costs, the rig had to cease drilling and surface activities while divers were in the water, which can represent a large opportunity cost for the drilling company. Weather conditions at offshore locations can also result in lengthy delays (four days were lost during the inspection of Rig A) and can make diving conditions difficult. There is also the issue of being able to identify all taxa collected by divers during the inspection, which can be particularly difficult for rigs that have not been dry-docked recently or have visited multiple bioregions prior to inspection. In most cases where a large number of samples

are collected from a broad range of groups of taxa, samples will need to be preserved and sent to specialist taxonomists.

An alternative approach to identifying all taxa on a rig during an inspection is to check for the presence of a limited number of NIS (i.e., target species) that are considered to pose a high biosecurity risk to a country or region. A major advantage of this approach is that inspectors can become familiar with key identifying features of target NIS, and are therefore able to quickly confirm the presence of high risk taxa while in the field. However, as highlighted in the second case study, there are some issues associated with this approach. Settlement plates suspended from Rig B were colonised by a small bivalve (*Electroma georgiana*) while moored in Port Phillip Bay for refurbishments. *Electroma* was not on the target pest list, and in fact was not familiar to the field team. Further investigation revealed that this species is endemic to southern Australia. While very little is published about the biology of *Electroma*, discussions with scientists familiar with the species revealed that it can behave invasively and can be a nuisance fouler on aquaculture structures (Hickman et al. 2005). This example highlights that while a rig (or any other vector) may not harbour target NIS, it may be fouled by species capable of ‘pesty’ behaviour in new habitats. Such species may not have previously been identified, either taxonomically or for such behaviour.

7.4.2 Biofouling treatment options for rigs

There are presently limited treatment options available upon the discovery of high risk NIS on rig structures. In the absence of a biosecure in-water cleaning option, removal of biofouling on land is the most desirable alternative. Land-based treatment provides the ability to prevent defouled material from re-entering the marine environment through the installation of barriers such as filters and containment tanks (Woods et al. 2007). Also, residual risks can be eliminated through the treatment of niche areas, and the enhanced elimination of microscopic life-stages, for example through passive desiccation. However, there are limited dry-docking facilities available capable of servicing a semi-submersible rig in Australasia: the nearest facility to New Zealand is in Singapore. Dry-docking such a structure is a major logistical and financial undertaking. To put it into perspective, to transport the rig from New Zealand to a Singapore, perform hull cleaning and hull re-coating, and return to New Zealand, it is estimated that the rig would be out of service for at least 2 months (assuming HLV transport), the

commercial cost of which would likely make the use of the rig in Australasia economically unviable.

The second case study demonstrated that prolonged air exposure during heavy-lift vessel (HLV) transfers has the potential to mitigate biosecurity risk. However, there is virtually no information on the efficacy of the method (e.g., how many days exposure is required for all taxa to die?) and how it varies with environment conditions (e.g., humidity during HLV transport) or the extent of fouling. For example, an emersion tolerance of several weeks to months has been described for some invasive macroalgae under high humidity conditions (Sant et al. 1996; Schaffelke & Deane 2005; Forrest & Blakemore 2006). The 45 day journey from Trinidad and Tobago to Australia appeared sufficient to eliminate residual fouling that was not removed by high pressure water blasters. However, if HLV transport is adopted as a risk mitigation measure, further work is needed to provide confidence around the emersion period required to ensure complete die-off across the broad range of taxa commonly found on rigs. Furthermore, the influence of biofouling biomass, humidity during transport and location on the rig (e.g., shaded versus unshaded) need to be considered. The availability and costs associated with HLV transport may also be prohibitive for some rig transfers, especially when considering moving rigs relatively short distances in remote locations such as Australasia. Thus there remains the need for alternative mitigation methods to be developed.

Upon discovery of NIS on vessels/structures, operators are often likely to be limited to in-water defouling methods. High pressure water blasters and handheld scrapers have proven to be effective in-water tools for the removal of fouling from a range of structures (including oil rigs). However, the major drawback of this method is that biofouling risks are transferred to the seabed if the defouled organisms survive the physical disturbance, as in the first case study. If in-water defouling is considered to be the only available option to mitigate biosecurity risk, a sensible strategy would be to ensure that defouling is undertaken over sub-optimal habitats in the source region where fouling has accumulated to minimise survivorship risks (Chapter 5). However, logistical feasibility and cost constraints are both major considerations when undertaking such activities in offshore locations.

It is conceivable that in-water defouling methods could be modified to enable the capture of dislodged material (e.g., Chapter 4). However, this is likely to require

substantial infrastructure (e.g., a large fine-meshed net suspended beneath the rig during defouling) or modification of the defouling devices to pump defouled material to the surface (e.g., onto a barge). A similar approach was developed during the defouling of a barge (the *Steel Mariner*) in Picton during an attempted eradication of *Didemnum vexillum*, for which a vacuum pump was used to suck defouled material to a barge moored alongside the vessel (Coutts and Forrest 2007). In Chapter 4, the efficacy of two different rotating brush devices that had been modified to retain defouled material was evaluated. However in both instances, 100% retention of defouled material was not achieved. Furthermore, the inability of these methods to treat biosecurity risks posed by an entire vessel (e.g., niche areas, microscopic stages) limits their use in situations where complete elimination of risk (i.e., quarantine/elimination of a pest) is necessary. Other methods (e.g., dry-docking or slipping, where feasible) are likely to achieve a more biosecure outcome.

A potentially effective and biosecure in-water method is plastic encapsulation. Vessels in many size categories (e.g., yachts, barges, merchant-size vessels), as well as artificial structures (e.g., wharf piles, moorings, fish farming cages) which are fouled with pest species have been treated *in situ* by encapsulating them in plastic wrapping (Coutts and Forrest 2005, 2007; Denny 2007; Pannell and Coutts 2007). The method relies on the development of anoxic conditions in the encapsulated water and, if necessary, mortality can be accelerated through the addition of non-persistent chemical agents (e.g., acetic acid and bleach; Coutts and Forrest 2005). A major advantage of encapsulation methods is that risk organisms are contained once the wrap is in place. However, the practicality and efficacy of this concept being applied to a floating marine structure of the dimensions of a semi-submersible drilling rig is yet to be established by any detailed research or experimentation. Further work is also required to clarify the factors that influence mortality rates (e.g., temperature, fouling biomass) so that treatment guidelines can be developed.

7.5 CONCLUSIONS AND FUTURE DIRECTIONS

The difficulty in dealing with marine pest species after they arrive is highlighted in a number of tried and failed management attempts: the preferred approach is to reduce the risk of the initial introduction of potential pest species. Furthermore, given the limited options available to treat large structures such as rigs, it is clearly preferable to evaluate risks prior to arrival in the recipient region. The approach applied to Rig B provides a

useful framework for assessing biosecurity risks posed by high risk vectors. In the New Zealand case, the low number of slow-movers arriving each year (Campbell 2004) theoretically makes it feasible to assess vessel risks on a case-by-case basis prior to their entry into the country, and to implement appropriate pre-border mitigation strategies (Chapter 2). A logical starting point in evaluation of risk would be a desktop assessment to develop a risk profile for high risk vessels intending to visit, based on factors such as voyage and maintenance history, likely infection mechanisms and intended movements in the recipient region. Such information can be used to assess biosecurity risk in a systematic and structured manner, and identify the need for inspection and mitigation.

Present methods for removing/treating biofouling on large structures such as oil rigs are not completely biosecure. While mechanical removal (in and out of the water) can greatly reduce risk, some risks are likely to remain, especially if the structure is towed as opposed to transported on another ship. Where mitigation is necessary, enhancement of existing and development of novel treatment tools are required for situations where removal to land (or HLV) is not feasible. In the longer term, improvements in antifouling paint technologies for structures that remain idle for extended periods may result in much lower colonisation rates by biofouling. Improvements in rig design may also reduce the susceptibility of submerged surfaces to fouling (e.g., less surface area available for colonisation), as well as make in-water risk mitigation methods such as encapsulation easier to implement.

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Chapter 8

General Discussion and Conclusions

8.1 SYNTHESIS OF RESEARCH FINDINGS

Collectively, research undertaken in this thesis: (i) demonstrated that slow-movers and towed structures can pose a high biosecurity risk if left unmanaged, (ii) identified a range of ecological and other issues that arise when undertaking in-water treatment to mitigate risks, (iii) evaluated the efficacy of New Zealand's most biosecure in-water defouling tools, (iv) assessed the efficiency of a response tool applied to the natural environment, and (v) highlighted through case studies the many challenges associated with the pre-border management of biofouling.

In New Zealand, the perception of slow-moving vessels and towed structures as being high risk vectors of non-indigenous species (NIS) has undoubtedly been reinforced by a number of high profile incursions in recent years (Box 8.1). In contrast to vessels associated with these high-profile incursions, slow-movers surveyed upon arrival in New Zealand had generally low-to-moderate levels of biofouling and a low incidence of NIS (Chapter 2). Biofouling was mainly restricted to the dry-docking support strips (DDSS) and niche areas where antifouling paint was in poor condition or absent, an observation consistent with other recent biofouling studies (Coutts and Taylor 2004; Davidson et al. 2009; Piola and Conwell 2010). Despite these observations of low fouling, and in the case of Chapter 2 low biosecurity risk, it is nonetheless evident that there is considerable potential for significant biosecurity risk to arise from the international movement of slow-movers. These findings highlight the need for improved hygiene standards, especially for niche areas of a vessel. This could be achieved in several possible ways; including: (i) the development and application of paint coatings effective in areas of a vessel not subjected to laminar flows, (ii) improved hull painting techniques to eliminate DDSS when a vessel is in dry-dock, and (iii) the development of treatment tools designed specifically for the niche areas of a vessel.

Improvements in vessel hygiene and pre-border management of vectors will undoubtedly reduce the risk of NIS transfers; however more marine incursions are inevitable due to New Zealand's 'leaky' borders (Hewitt et al. 2004). As outlined in Chapter 3, there are limited biosecurity management options available for vessels discovered to be fouled by potential NIS; these include turning the vessel away from the border, treating the vessel on land (e.g., dry-docking, haul out facilities), treating the vessel using in-water methods, or leaving the vessel unmanaged. For these options to be applied successfully, biosecurity managers and inspection staff at the border will require an improved understanding of: (i) treatment efficacy of in-water and land-based methods, (ii) resources required, their availability and associated costs, (iii) restrictions and limitations (e.g., consents for discharges to the environment), and (iv) the feasibility of applying these options to the range of vessel types arriving at the border, especially when considering time available for treatment (i.e., the residency time). The appropriateness of various management options can be theoretically evaluated against specific criteria (e.g., feasibility of implementation, efficacy of treatment, implications for trade) using various decision making approaches (e.g., multi-criteria decision analysis, logic trees, Burgman 2005). Results from these analyses could then be used to support decisions around mitigating risks posed by vessels (or structures) discovered to be fouled at the border.

Box 8.1 Recent incursions involving slow-movers

- In 2001, a barge was discovered in the Marlborough Sounds with an estimated hull fouling biomass of 26 tonnes, comprising 6 algal and 70 animal taxa (Coutts 2002). Among this assemblage was an estimated 1400 kg of an ascidian (*Didemnum vexillum*), which has subsequently spread and become a fouling pest to aquaculture in the region (Coutts and Forrest 2007).
- In December 2007, ten NIS were identified amongst c. 35 tonnes of biofouling defouled from an oil rig in Tasman Bay (Chapters 6 and 7).
- In November 2009, a barge moored in Waitemata Harbour that had recently arrived from Port Phillip Bay (Australia) was discovered to be heavily fouled by the Mediterranean fanworm *Sabella spallanzanii*; a species designated as an Unwanted Organism under New Zealand's Biosecurity Act 1993 (pers. comm., P. Stratford, MAF Biosecurity New Zealand).

There are few in-water treatment options available for fouled vessels, and for most their efficacy in mitigating biosecurity risk has not been experimentally evaluated. To

address this lack of knowledge, the efficacy of New Zealand's most biosecure in-water defouling tools (diver-operated rotating brush devices) were evaluated in Chapter 4. The two devices tested proved effective in removing low-to-moderate levels of fouling from flat and curved experimental surfaces; however performance was generally poorer as fouling became more advanced (e.g., calcareous tubeworms were resistant to treatment). As identified in Chapter 3, one of the potential risks posed by in-water defouling is the transfer of biofouling from the vector to the environment. However, unlike most other commercial rotating brushes, these devices tested had been modified to capture defouled material $> 60 \mu\text{m}$ in size, and on average $> 95\%$ of defouled material was collected and retained. The majority (c. 80%) of fouling not captured by the devices was crushed by the brushes (i.e., non-viable); however a wide range in types of viable organisms (e.g., barnacles, hydroids, etc) was lost to the environment. In addition to their inability to remove and retain all biofouling material from experimental surfaces, the devices were also considered unsuitable to treat niche areas of vessel due to their large size and the shape of the brush heads. This is considered a major limitation given that higher levels of fouling are typically found in these areas of a vessel (as shown in Chapter 2). Clearly, therefore, these devices (as tested) would not be suitable for treating a heavily fouled vessel or structure (e.g., an oil rig), particularly if NIS were known to be present.

There is a risk that biofouling organisms may survive and establish on the seabed if they are not retained or destroyed by the defouling method. The potential for this occurring was experimentally demonstrated in Chapter 5 where intact biofouling communities were transplanted to the seabed in contrasting port/marina environments. While exposure to sediments and predation affected post-defouling survivorship of transplanted organisms, a broad range of taxa was able to tolerate the transition to the seabed and the associated change in biotic and abiotic environmental conditions. This is interesting, given that artificial fixed and floating structures can have novel epibiotic assemblages compared to adjacent natural rocky habitats (Glasby 1999; Connell 2000). This could potentially be explained by the short-term nature of the transplant experiments and that the fouling organisms may not persist in the longer term (i.e., there may have been a lag in the amount of predation pressure in response to the elevated food source). However, an alternative hypothesis (not explored in this thesis) is that adult life stages of biofouling organisms may be more resilient to environment pressures (e.g., predation, sedimentation) present than their propagules.

In Chapter 6, a high proportion of mussels and their epibionts (including taxa found to be vulnerable to predation and sedimentation in Chapter 5) were viable three months after being defouled from an oil rig to muddy sediments in Tasman Bay. This was thought to be due to their attachment to mussels which kept them away from the muddy seabed and the absence of grazers and predators (e.g., gastropods) that are typically found in shallow subtidal rocky communities where the transplant experiments in Chapter 5 were undertaken. Collectively, therefore, these findings clearly highlight the need to characterise biofouling composition and extent before in-water defouling is undertaken on vessels or structures that potentially have NIS present.

The ability of colonial taxa to survive fragmentation is well established in the published literature (e.g., Bullard et al. 2007; Johnston and Clark 2007). However, possible implications for biofouling management, in particular risks posed by fragmented colonial organisms being dislodged to the seabed have received little scientific attention. The ability of several globally ubiquitous colonial biofouling taxa to reattach to artificial surfaces following fragmentation was confirmed in both laboratory- and field-based experiments (Chapter 5). In the field experiments, reattachment success was evaluated in the absence of grazers, because unless the fragments were contained, it was considered likely that tidal currents and other disturbances (e.g., vessel wake) would move them off the Perspex substrate. However, refugia from predation are uncommon in typical port and marina environments, and as such, fragmented colonial organisms are likely to be as susceptible to predation as the fully intact taxa in the whole community transplant experiments undertaken in Chapter 5.

In situations where NIS survive defouling and subsequent environmental conditions, or alternatively colonise natural and artificial habitats via spawning, there are presently limited response tools capable of eliminating biosecurity risks. This was illustrated in Chapter 6, where the defouling of an oil rig in Tasman Bay is described. Following the discovery of viable *Perna perna* and nine other NIS amongst the defouled material, several options were considered (e.g., encapsulation, smothering with sediments) to mitigate potential biosecurity risks. Given the water depth at the site (c. 44 m) and resource availability, dredging the seabed using a modified commercial scallop dredge was considered to be the most suitable option. For most incursion responses, the overall aim is to reduce target pests densities to zero, but in this example an alternative success criterion was developed based on the reproductive biology of the target pest. *Perna*

perna are dioecious (separate sexes) and are thought to require close proximity for successful fertilisation to occur. A target density of < 10 mussels per m^2 was chosen based on discussions with experts familiar with the biology of this species. Interestingly, a similar approach is now being considered by scientists managing the incursion of the Mediterranean fanworm (*Sabella spallanzanii*) in Lyttelton Harbour, New Zealand (pers. comm., P. Stratford, MAF Biosecurity New Zealand). This approach to invasive species management has a strong theoretical basis (i.e., ‘Allee effect’) and in the past decade, models based on the Allee effect have been applied in studies of biological invasion, deliberate introduction of biological control agents and in conservation biology (Drake 2004; Jang and Diamond 2007; Chen and Lin 2008; Guo et al. 2010).

Despite the dredge-based approach adopted being successful in reducing target pest densities below an estimated threshold for successful reproduction, this event reinforces the widely held belief that it is better to put resources into prevention rather than the cure (Ruiz and Carlton 2003; Hewitt et al. 2004; Finnoff et al. 2007). Prevention was the approach taken by operators of a rig transported from Trinidad and Tobago to New Zealand, via Australia (Chapter 7), where a broad range of pre-border mitigation measures (i.e., biofouling removal, stripping of ballast water, antifouling of sea chests, monitoring and inspection for potential colonisation by NIS in Australia, formal assessment of residual risks) were implemented. This template can also be applied to other high risk vectors on a case-by-case basis (e.g., slow-movers) where this is feasible. For vectors such as merchant vessels that represent a large proportion of arrivals and typically have a short residency period in New Zealand, an alternative approach will be necessary, such as border standards for vessel hygiene and fouling extent. To this end, New Zealand and Australia are in the process of developing border standards for biofouling, however it is yet to be determined what criteria will be developed and how these will be implemented at the border. Even in the case of slow-moving vessels and towed structures, it is recognised that the template will be easier to apply in some circumstances more than others (e.g., Rig B in Chapter 7). Furthermore, in many situations, the only practical or affordable solution to address biofouling risks will involve *in situ* methods (e.g., in-water defouling, encapsulation). Thus, provided below are some additional considerations of biosecurity risks associated with in-water defouling not addressed by the thesis research and some perceived future issues when present tools are applied to novel paint coatings.

8.2 *IN SITU* METHODS TO MITIGATE PRE-BORDER RISK

Of the available *in situ* methods, in-water defouling is the most common approach to routinely manage biofouling on vessels and artificial structures of all sizes in the marine environment. Concerns regarding the release of toxic paints and NIS to the environment have resulted in this activity being controlled or banned in some countries (e.g., Australia and New Zealand), although in many parts of the world in-water defouling is undertaken without restrictions in place. Despite awareness of biosecurity risks associated with in-water defouling, there are few devices in commercial use that have been designed to mitigate the release of defouled material to the seabed (e.g., Bohlander 2009). However, there are currently several alternative methods to biofouling treatment under development. For example, in Chapter 3, encapsulation methods (i.e., wrapping a vessel or structure in plastic) were discussed, and while they have proven effective in treating small vessels and structures (Pannell and Coutts 2007), an attempt to wrap a navy frigate (approximately the same length as a rig pontoon) was unsuccessful (Denny 2007). Hull cleaning systems have been designed to retain defouled material (Bohlander 2009), but unlike the devices tested in Chapter 4, their efficacy is yet to be evaluated experimentally. Given the current shortage of land-based facilities for biofouling removal and the absence of paint technologies capable of preventing fouling on all areas of a vessel and structures that remain idle for extended periods, there is likely to be a reliance on in-water defouling methods for the foreseeable future.

8.2.1 When is in-water defouling appropriate?

Despite the potential biosecurity risks identified above, there are conceivably situations where in-water defouling is likely to pose little or no incremental biosecurity risk to a region. In such cases, in-water defouling may be appropriate if other management options (e.g., removal to land) are not available or feasible, particularly if unmanaged risks (i.e., no treatment) are likely to be greater than in-water defouling risks, as discussed in Chapter 3. For example, if land-based facilities are not available and locally-fouled vessels leave a region unmanaged, there is the potential for spread of local taxa (e.g., natives, pest species and NIS) beyond their present distribution. In contrast, vessels or structures that have not left a region since being cleaned and have been fouled by local taxa are likely to pose a low biosecurity risk to the region if

defouled in-water. Other situations where biosecurity risks posed by in-water defouling are likely to be low include:

- The defouling method retains close to 100% of defouled material (see Chapter 4). This is considered a realistic outcome for commercially cleaned vessels; but in the short-term is unlikely to be the case for small recreational craft where fouling removal is typically undertaken by the owners.
- Biofouling has been treated and is no longer viable. For example, drilling rigs are often transported long distances onboard a heavy-lift vessel, exposing biofouling communities to desiccation stress (Chapter 7). Encapsulation techniques have also been used to ‘suffocate’ biofouling communities on vessels and structures (Coutts and Forrest 2007; Denny 2007), as described above.
- The vessel or structure is defouled over sub-optimal habitat (e.g., at a deepwater offshore location) to minimise survivorship, and habitats suitable for colonisation and establishment are beyond the dispersal capacity of biofouling organisms.

Ideally, vessels or structure would be regularly defouled (e.g., monthly), thus preventing the development of an advanced biofouling community. Furthermore, larval life-stages establishing on a surface over a short period are likely to be susceptible to damage by the defouling method and have a lower rate of establishment success once defouled. Technologies are presently being designed for this purpose, including heat delivery systems that can treat an entire vessel, including the niche areas (A. Coutts, pers. comm.). However, to my knowledge there are no systems operating commercially that have had their performance experimentally determined. Finally, despite the low biosecurity risks associated with the above scenarios, wider environmental risks of treatment also need to be considered; including the release of toxic biocides to the environment (e.g., copper, tributyl-tin) and organic material to the seabed.

8.3 CONCLUDING REMARKS

Biofouling risks to New Zealand can be reduced through improved pre-border management of vectors. For slow-movers such as barges and oil rigs, this can be undertaken on a case-by-case basis given their low numbers. For other vessel classes where there are a large number of arrivals each year, alternative approaches such as the

development of border standards, risk assessment and targeted inspections will be required. In terms of risk mitigation, research undertaken as part of this thesis demonstrates that in-water defouling is not appropriate in situations where NIS are present unless the method is capable of destroying or retaining all of the defouled material or environmental conditions are considered unsuitable for establishment and spread. It is acknowledged that there are situations where in-water defouling (without collection) poses a low incremental biosecurity risk, such as the defouling of surfaces that have only been colonised by resident taxa. However, in these situations environmental risks posed by the release of biocides from antifouling paints and the deposition of organic material to the seabed need to be considered.

In situations where, despite pre-border management efforts, a marine pest incursion occurs and becomes well established, there are presently few tools for effective management. Although partnerships are being developed to address marine biosecurity at the regional level in New Zealand, including a ‘Top of the South’ (Nelson, Tasman, Marlborough) partnership, there is no over-arching national vector management strategy to contain the domestic human-mediated spread of established pests or any other non-indigenous species that arrive in New Zealand. Hence, it is inevitable that domestic transport vectors (e.g., inter-regional vessel movements) will continue to spread established species and new non-indigenous species throughout New Zealand. In this respect, the stringent management approaches being implemented or proposed for international border control would ideally also be implemented post-border to reduce the domestic spread of NIS with vessel movements and other activities.

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