Managing Risks from Invasive Marine Species:

Is Post-Border Management Feasible?

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Is Post-Border Management Feasible?

by

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PREFACE

This thesis uses the Asian kelp *Undaria pinnatifida* as a model organism to explore key issues in marine biosecurity in relation to the post-border management of marine pests. The thesis comprises a combination of previously published work (Chapters 3 - 5), and research in progress at the time of enrolment in 2003 (Chapters 4-7). The Chapters are discrete pieces of work that do not represent a narrowly-defined area of research but nonetheless explore closely related subject material. The Victoria University Statute for the Degree of Doctor of Philosophy allows theses to be formulated on this basis. The requirements for such theses are outlined in Section 4(b)((ii) of the Statute, and in Section 3.2.1 of the 2002 PhD Handbook. The key requirement is that the thesis must be an integrated report that describes how the chapters relate to a unified theme.

In this respect, the greater picture considered is the feasibility of managing marine pests after they have established populations in New Zealand, with aspects of the biology and management of *Undaria* providing the unifying themes on which the chapters are wholly or partly based. The focus shifts from consideration of *Undaria* specifically (e.g., impacts and dispersal characteristics in Chapters 3 and 4, based on work conducted over 1997 - 2001), to the use of *Undaria* as a case study organism in the work initiated after 2001 (Chapters 5 - 9). The nine chapters that make up the thesis comprise 5 refereed publications, and extracts from Cawthron technical reports for which I have been primary author, with introductory and general discussion chapters making up the balance.

A preface is included at the beginning of each technical chapter (Chapters 3-8) that describes whether and where the work has been published and, for multi-authored publications, the contribution made by key co-authors. Chapter 2 provides an overview of *Undaria* and its distribution in New Zealand based on knowledge to date, which supersedes the previously published work in Chapters 4 and 5. Hence, in both Chapter 2 and the General Discussion in Chapter 9, cross-referencing is used to assist with clarity and tie the thesis together. By contrast, Chapters 3-8 have largely been extracted verbatim from published work, and are thus self-contained with no cross-referencing. Because the chapters represent discrete pieces of work, I have included reference citations at the end of each, rather than compile a single list at the end of the thesis. Where the chapters have originated from papers published in refereed journals, the abstract of the paper has been included for completeness.

ABSTRACT

Non-indigenous marine species are a major threat to marine environments and economies globally. This thesis examines whether management of pest organisms post-border (i.e, after they have established in New Zealand) is feasible in the marine environment, using the non-indigenous Asian kelp *Undaria pinnatifida* as a model organism. Background information on *Undaria* in Chapter 2 recognises the paucity of information on *Undaria*'s impacts. Hence, Chapter 3 investigates ecological effects from *Undaria* in a low shore rocky habitat. Although negligible effects were described, the uncertainty in extrapolating findings to other places and times means that the precautionary principle should be applied by managers, and 'worst-case' impacts assumed.

Chapter 4 investigates mechanisms for *Undaria*'s natural dispersal, and describes strategies based on spore release and sporophyte drift that may lead to spread over scales of metres to kilometres. This work highlights the importance of human transport vectors (especially vessels and aquaculture) in the post-border spread of *Undaria* at regional and national scales. Hence, a case study in Chapter 5 describes aquaculture activities that could be vectors for spread of *Undaria* in New Zealand, and presents criteria for identifying present and future high risk pathways.

Chapters 6 and 7 describe methods to reduce the accidental transport of *Undaria* and other biofouling pests with aquaculture, with a focus on mussel farming. Treatments based on water blasting, air drying and freshwater immersion provide low cost options for equipment such as floats and rope. For treatment of mussel seed-stock, immersion in dilute (4%) acetic acid (the active ingredient in vinegar) is identified as a method that could eliminate *Undaria* and other soft-bodied fouling organisms without resulting in an unacceptable level of mussel mortality.

Chapter 8 proposes a risk-based framework for setting post-border management priorities based on the feasibility, benefits and costs of risk reduction. This chapter elucidates how knowledge generated from research in Chapters 2-7 can be used in a biosecurity risk management context. It shows that effective management post-border is possible even when pest organisms become relatively well established, and that the benefits gained from even limited successes have the potential to greatly outweigh the consequences of uncontrolled invasion. However, as unwanted species become

increasingly widespread, management will become increasingly focussed on the protection of specific values.

Chapter 9 extends some of the ideas proposed in Chapter 8, and considers a broad post-border management framework for marine pests. A comprehensive system should consist of vector management, surveillance, and incursion response that targets particular pests or suites of functionally similar species (e.g., biofouling organisms), coupled with generic vector management approaches that aim to reduce human-mediated transport of all organisms at a national scale. New Zealand's geographic isolation and low population, hence relatively low level of vector activity, makes the management of human-mediated pathways of spread entirely feasible in many circumstances. Hence, while there are clearly many challenges in the post-border management of marine pests, this is nonetheless a realistic goal, and probably moreso in New Zealand than in any other country in the world.

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Chapter 8 originated from efforts to develop management options for *Undaria*, and the draft manuscript summarizing this work benefited from the input of Chad Hewitt (Ministry of Fisheries), Geoff Hicks (Department of Conservation), and John Hellstrom (Biosecurity Limited). The development of the framework was funded by the New Zealand Foundation for Research, Science and Technology Marine Invaders Programme, Contract CAW X0202. I also thank Bill Lee and Rob Allen (Landcare Research) for the opportunity to submit this work as a Chapter in their book.

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General Introduction

1.1 INTRODUCTION

1.1.1 Background

The natural or human-mediated introduction of non-indigenous species has been a familiar part of human history in terrestrial and freshwater environments. Invading species range from microbes to vertebrates, and can greatly influence the functional and structural properties of ecosystems (Mooney and Drake 1989; Allen and Lee 2006). By comparison with terrestrial and freshwater ecosystems where non-indigenous species are often conspicuous and their impacts well documented, knowledge of exotic marine species, both in New Zealand and world-wide, is relatively poor. In the last two decades, however, but especially since the mid-1990s, there has been considerable scientific interest in the occurrence and consequences of invasion by non-indigenous species in marine environments, both in New Zealand (e.g., Forrest et al. 1997; Hayward 1997; Cranfield et al. 1998) and in many countries and continents world-wide (e.g., Morton 1987; Carlton 1989; Griffiths et al. 1992; Carlton and Geller 1993; Eno 1996; Furlani 1996; Galil 2000; Lewis et al. 2003; Castilla et al. 2005; Garcia-Bethou et al. 2005; Wonhom and Carlton 2005; Colautti et al. 2006).

Despite this recent groundswell of interest, it has long been recognised that human activities in the marine environment, and especially trans-oceanic movements of vessels, have been a major pathway for the inadvertent spread of marine organisms well beyond their natural dispersal ranges (e.g., Carlton 1985; Chilton 1910; Elton 1958; Skerman 1960). There are only a few examples documenting natural movements of marine organisms across oceanic barriers, for example, those with long-lived planktonic larvae (Scheltema 1971) or rafting ability (Winston et al. 1996; Waters and Roy 2004). Recent literature suggests that the rate at which non-indigenous species are being transported around the globe by human activities, and establishing adventive populations outside their natural range, is steadily increasing (Ruiz et al. 1999; Harris and Tyrrell 2001; Hewitt et al. 2004; Grosholz 2005). Among other things, this reflects

a greater frequency of vessel movements, changing patterns of shipping trade that open up new source regions (Taylor et al. 1999; Kolar and Lodge 2001; Perrings et al. 2005), and changing environmental conditions that allow the successful invasion of species that have previously failed to establish (Dukes and Mooney 1999; Harris and Tyrrell 2001; Diederich et al. 2005; Grosholz 2005; Nehls et al. 2006).

Non-indigenous marine species are now considered a major threat to marine environments globally, and moreso than the threat from the plethora of other human activities (e.g., waste discharge, habitat reclamation) whose impacts have traditionally received considerably greater attention. Although positive commercial and even ecological benefits of some non-indigenous species are recognised (e.g., Galil 2000; Sinner et al. 2000; Hayes and Sliwa 2003; Wonham et al. 2005), the primary focus of scientists, regulatory agencies, and other stakeholders is on invasive species as a threat to ecological and socio-economic values (e.g, Hewitt et al. 2004). In the US and Canada alone, the projected economic impact from a few of the more notorious marine pest species has been estimated to be in the order of approximately 2 billion dollars per year (Pimentel et al. 2000; Colautti et al. 2006).

1.1.2 Marine biosecurity in New Zealand

A number of marine introductions to New Zealand have been intentional, such as the saltmarsh cordgrass *Spartina* spp. (Swales et al. 2005), which was originally introduced for its perceived beneficial role in reclaiming and stabilizing tidal flats in estuaries. However, for most (at least 148) of our non-indigenous taxa the initial introduction has occurred inadvertently with transoceanic vessel movements, primarily via ballast water and hull fouling (Hayward 1997; Cranfield et al. 1998). While accidental introductions of non-indigenous species to New Zealand via shipping have been reported since at least Chilton (1910), historically this appeared to be regarded with little concern. For example, the non-indigenous Pacific oyster *Crassostrea gigas* has been present in New Zealand since at least 1971 (Dinamani 1971) and perhaps as early as 1958 (Dromgoole and Foster 1983), and has been cultivated in northern New Zealand since the mid-1970s. Concerns regarding negative impacts, such as effects on coastal amenity value from high oyster settlement in natural habitats, have arisen only relatively recently (Hayward 1997).

The increasing profile of marine pests in New Zealand was largely precipitated in the late 1990s by media attention regarding the Asian kelp *Undaria pinnatifida* (hereafter referred to as *Undaria*). This species was initially discovered in Wellington in 1987 (Hay and Luckens 1987), but received little public coverage as a pest until a decade later. At around this time, biofouling pest issues also began to emerge within New Zealand's aquaculture industry. This included a population explosion of the non-indigenous solitary tunicate *Ciona intestinalis*, which at that time had already been present in New Zealand for several decades, which resulted in the decimation of mussel crops in parts of the Marlborough Sounds. New Zealand now has eight non-indigenous marine species listed under the Biosecurity Act 1993 as 'unwanted'. In addition to *Undaria*, and the clubbed tunicate *Styela clava* which was discovered in New Zealand in 2005, the list comprises six organisms not yet recorded in New Zealand, namely: the European green crab, *Carcinus maenas*; the northern Pacific sea star, *Asterias amurensis*; the Mediterranean fanworm, *Sabella spallanzanii*; the green macroalga, *Caulerpa taxifolia*; the Chinese mitten crab, *Eriocheir sinensis*; the Mediterranean fanworm, Sabella spallanzanii and the Asian clam, *Potamocorbula amurensis*. Biosecurity New Zealand is also assisting the Marlborough Sounds aquaculture industry and local authority group with the management of the colonial ascidian fouling pest *Didemnum vexillum*.

As a consequence of public awareness and media exposure surrounding *Undaria*, and more recently a range of other terrestrial pests (e.g., the varroa bee mite), freshwater algae such as 'Didymo' (*Didymosphenia geminata*) and marine fouling pests (notably *Styela clava* and *Didemnum vexillum*), the term 'biosecurity'[1] is now entrenched in the public psyche. November 2004 also saw the establishment of Biosecurity New Zealand as part of the Ministry of Agriculture and Forestry, which brought central government responsibility for biosecurity across marine, freshwater and terrestrial systems under one organisation (previously central government responsibility for marine biosecurity came under the Ministry of Fisheries). However, despite general acknowledgement of the threats posed to New Zealand's environmental, economic, social and cultural values by present and potential marine pests, there has nonetheless been widespread uncertainty among scientists, government agencies and marine user groups regarding how to deal with such issues. Without a structured approach to setting priorities,

[1] Biosecurity was defined by Hewitt et al. (2004) as the management of risks posed by introduced species to environmental, economic, social and cultural values.

management efforts to date have been largely ad hoc, lacked focus, and lacked 'buy in' from affected stakeholders. This situation highlights the need for a better approach to identification and assessment of marine biosecurity risks, and establishment of management priorities that will enable limited budgets to be used most efficiently and effectively.

1.1.3 Approaches to managing biosecurity risks

Broadly, there are two main stages within which marine biosecurity risks and risk management can be considered. These are the 'pre-border' stage involving the trans-oceanic transport and delivery of non-indigenous species from an overseas source region, and 'post-border' events involving the establishment, spread, impacts and management of high risk pests (Forrest et al. 1997, Figure 1.1).

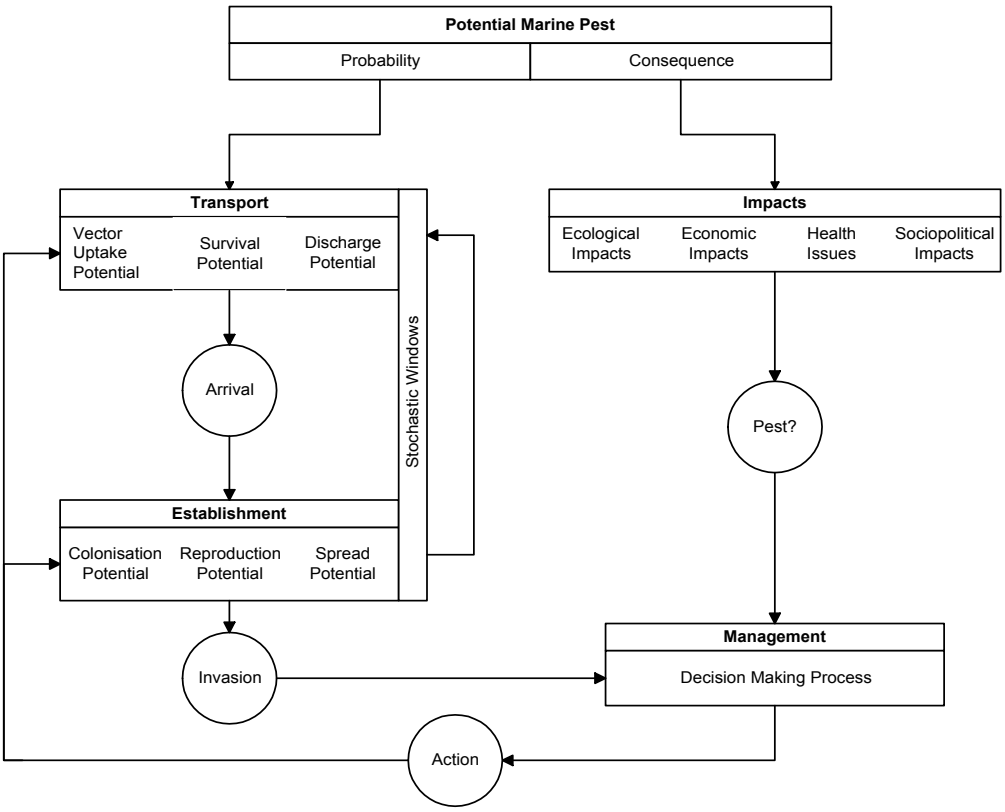

Figure 1.1 Components of risk assessment for potential marine pests (source: Forrest et al. 1997).

In Figure 1.1 it is recognised that biosecurity risk has two components: the likelihood of events that lead to the establishment of a pest organism in a new country, and the consequences of establishment. The likelihood of establishment occurs as a chain of events starting with the pre-border uptake of a pest organism by a human transport vector in the overseas source region, followed by the establishment and subsequent spread of the pest post-border. The latter two phases involve the initial colonisation of the pest organism, the establishment of a reproductively viable population, and subsequent spread beyond the point of first incursion via both natural dispersal and domestic transport vectors. Decisions to manage the consequences of pest incursion and spread (e.g., based on actual or perceived impacts) can lead to a change in these likelihoods.

Given both the technical and financial constraints in controlling non-indigenous species post-border, it is clearly preferable to prevent the initial introduction of pest species as a first line of defence (e.g., Bax et al. 2001; Meyerson and Reaser 2002; Simberloff 2003; Branch and Steffani 2004; Hewitt et al. 2004). Accordingly, New Zealand puts considerable effort into terrestrial border control and inspection procedures to intercept potential pest organisms, and thus protect its highly valued environments and resources. Similarly, in the marine environment, there has been considerable effort globally to identify risks associated with international vessel traffic (Carlton 1985; Coutts et al. 2003; Coutts and Taylor 2004; Verling et al. 2005) and develop treatment solutions for transport mechanisms such as ships' ballast water (Mountfort et al. 1999; Oemcke et al. 2004). Despite such efforts, effective or affordable management tools are still lacking, with the associated recognition that New Zealand's 'leaky' borders make continued incursions of pest species inevitable (Wotton and Hewitt 2004). This situation raises the question as to whether post-border management, which has a track record of successes in freshwater and terrestrial systems in New Zealand and elsewhere (e.g., Genovesi 2005; Allen and Lee 2006), might also be feasible in the marine environment?

A recent synthesis of biological invasions in New Zealand by Allen and Lee (2006) provides a number of examples where the efficacy of post-border pest management in terrestrial and freshwater systems has been demonstrated. These include successes in the restoration and recovery of native vegetation through control of introduced herbivores such as rabbits, goats and deer (Coomes et al. 2006), control programmes for non-indigenous predators (e.g., stoats, rats) of native birds or their eggs (McLennan

2006), and spraying programmes for invasive aquatic and terrestrial weeds (e.g., Swales et al. 2005). Among the more high profile recent examples have been the use of 1080 poison to control possum populations, trials with copper-based agents to control 'Didymo' in rivers, and a successful eradication of the painted apple moth in Auckland through an aerial spraying campaign.

The above examples, among many others, highlight the wide range of control strategies possible in freshwater and terrestrial systems. They include aerial and ground spraying/release of biocides, mechanical control (e.g., of aquatic weeds), installation of predator-proof fencing, creation of pest-free island habitats, commercial harvest for introduced mammals (e.g., goats, deer and pigs; Parkes 2006), and release of biological control agents (e.g., for insects). These methods range from those that are publicly acceptable (e.g., mechanical removal, hunting) to those that are highly controversial (e.g., biological control, use of poisons such as 1080). In contrast with freshwater and terrestrial systems, the marine environment is highly inter-connected and expansive, relatively inaccessible, and can be a hostile system to work in. Intuitively, it is apparent that many of the methods developed for freshwater and terrestrial systems are unlikely to be directly transferable to the marine environment. Hence, the goal of this thesis is to evaluate the feasibility of managing established marine pests, and examine the long standing view that management of marine pest incursions post-border will largely be futile (e.g., Sanderson 1990; Brown and Lamare 1994; Thresher and Kuris 2004).

1.2 SCOPE AND CONTENT OF THESIS

1.2.1 Content of thesis chapters

Using *Undaria* as a model organism, and with reference to other case studies, this thesis will demonstrate that options for management post-border are often limited, but nonetheless feasible in some circumstances. To provide a context for this work, in the next section I outline the rationale for using *Undaria* as a case study organism, then in Chapter 2 provide background information on the biology of *Undaria* and the history of its management to date in New Zealand. Chapter 3 then describes a field assessment of the ecological impacts of *Undaria* in low shore rocky habitats of Lyttelton Harbour. The impetus for this work was driven by the fact that at the time it was initiated in 1998, very little was known about the actual effects of *Undaria*, a situation that had a number of implications for management as described in Chapter 2. As a wider contribution to

marine biosecurity, the *Undaria* investigation in Chapter 3 is also used to explore issues associated with measuring the ecological effects of invasive marine species, and in particular to highlight the limitations in applying traditional environmental sampling designs to studies of invasion impacts.

The themes of Chapters 4 to 7 relate to the natural mechanisms and human vectors for *Undaria*'s spread around New Zealand, and whether management of human-mediated pathways is feasible. Chapter 4 describes an experimental evaluation of the importance of natural dispersal mechanisms in *Undaria*'s spread, by comparison with human-mediated pathways. Chapters 5 to 7 then focus on the management of human-mediated spread, using the marine farming industry as a case study. This focus reflects the fact that aquaculture activities in New Zealand are recognised as an important post-border vector for *Undaria* and a number of other pest species, yet work on *Undaria* risks and management has primarily focused on vessel-related pathways (e.g., Stuart and McClary 2004).

Chapter 5 describes a desktop assessment of the aquaculture pathways that are likely to be important in the spread of *Undaria*, and presents criteria for identifying present and potentially high risk pathways. Chapters 6 and 7 then consider whether it is possible to manage transfers of *Undaria* and other biofouling pests on high risk aquaculture pathways. These are technical chapters that describe experimental evaluations of methods to treat key aquaculture vectors. While *Undaria* is used as a model organism from which treatment criteria are developed, Chapter 7 also examines the wider applicability of this work to other biofouling pests and to management applications beyond aquaculture.

Chapter 8 describes a risk-based model for setting priorities for the management of marine pest species, using lessons learned from experience with *Undaria* to exemplify some of the key issues pertinent to marine systems. To a large extent, Chapter 8 elucidates how the information generated from research conducted under Chapters 2-7 can be used in a biosecurity risk management context or, conversely, how application of the model can be used to identify management-oriented research needs. Since Chapter 8 was originally developed as a book chapter, and hence is a stand alone piece of work, the General Discussion in Chapter 9 is used to expand on some of the issues touched on, and to tie the various elements of the thesis together within the context of a comprehensive post-border management framework.

1.2.2 The utility of *Undaria* as a model organism

By comparison with many pests, much is known about the basic biology of *Undaria* because it is an aquaculture species in some Asian countries, and has also been the subject of aquaculture or wild harvest research in New Zealand (e.g., Gibbs and Forrest 1999; Gibbs et al. 2000). Such knowledge has been invaluable for the management-oriented research described in this thesis and elsewhere. Knowledge from *Undaria* aquaculture research, for example, provided valuable guidance on appropriate spore release and culturing techniques for the experimental work described in Chapters 4, 6 and 7. *Undaria* is also a useful case study species because it has a range of actual and potential effects on different environmental values, and has the interesting feature of potentially being both a pest and a product. For example, it has the potential to adversely affect both economic and ecological values, it is a conspicuous species that alters the natural character value of coastal areas, and is one of the few fouling pests that can also be highly invasive in natural ecosystems (Sinner et al. 2000). On the other hand, *Undaria* has recognised benefits as a commercial species because it is both edible and has a range of pharmaceutical and industrial properties (e.g., Suetsuna and Nakano 2000; Apoya et al. 2002).

Undaria is arguably one of the more easy invasive species to manage (although not the easiest), primarily because it is benthic, conspicuous, has a limited depth range, and a relatively short dispersal phase. Hence, successes and failures with *Undaria* management provide a useful benchmark as to what may or may not be feasible for other pests. Already the knowledge gained and lessons learned from *Undaria* management, and the logic that has been applied in consideration of management options, have been invaluable for the management of other marine pests. For example, the logic behind the evaluation of options for *Undaria* management by Sinner et al. (2000) and the risk model in Chapter 8, was applied to an evaluation of management options for the invasive ascidian *Didemnum vexillum* in the Marlborough Sounds (Sinner and Coutts 2003). It has also been applied by a Technical Advisory Group tasked with providing guidance to Biosecurity New Zealand on a strategy for managing the clubbed tunicate *Styela clava*. In summary, therefore, the lessons learned and knowledge gained from the management of *Undaria* can be used to provide insights into the feasibility of post-border management for marine pest organisms more generally.

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Chapter 2

Overview of *Undaria* and its management in New Zealand

PREFACE

This chapter describes background information on *Undaria* as the model organism on which the chapters in this thesis are based. Key biological attributes and life-cycle characteristics of *Undaria* are described in Section 2.1, as this knowledge is pivotal to understanding the seaweed's management, and is therefore highly relevant to subsequent chapters. Section 2.1 also provides information on *Undaria*'s present and potential distribution within New Zealand, and an overview of the key human-mediated vectors for its spread. Note that the natural dispersal potential and ecological impacts of *Undaria* are not detailed in Section 2.1, because they are separate themes addressed in Chapters 3 and 4 respectively. In Section 2.2 an account is given of the various steps taken regarding the management of *Undaria* in New Zealand, since this background information is relevant to the risk management framework described in Chapter 8, and also to the General Discussion in Chapter 9.

2.1 OVERVIEW OF *UNDARIA* AND ITS DISTRIBUTION IN NEW ZEALAND

2.1.1 Background to the origin and biology of *Undaria*

Undaria is a laminarian (kelp) native to cold temperate coastal areas of Japan, Korea and China (Akiyama and Kurogi 1982). It is an edible species, known as Wakame in Japan, and is extensively cultivated for commercial sale (Hay and Luckens 1987). In addition to New Zealand, populations of *Undaria* have become established in recent decades along the Mediterranean and Atlantic coasts of Europe (e.g., Perez et al. 1981; Floc'h et al. 1988; Curiel et al. 1998; Cecere et al. 2000), in Britain (Fletcher and Manfredi 1995), Argentina (Casas and Piriz 1996), eastern Australia and Tasmania (e.g., Sanderson 1990; Campbell and Burridge 1998), Mexico (Aguilar-Rosas et al. 2004) and the western US (Silva et al. 2002). *Undaria* sporophytes in New Zealand can reach approximately 1.5 m in length (e.g., Hay and Villouta 1993; Figure 2.1), with mature specimens easily distinguished from New Zealand native kelps by the convoluted spore-producing sporophyll at the base of the stipe (Figure 2.2). New Zealand has three different morphotypes of *Undaria* (Hay and Sanderson 1999), which are: (i) the relatively large *northern* type, characterised by an elongated sporophyll often extending the full length of the stipe to the base of the blade; (ii) the *naruto* variety, which is a smaller plant with a relatively short stipe and a large flaccid sporophyll that sometimes spreads out onto the basal part of the blade; and (iii) the *nambu* type, which is intermediate between the *naruto* and *northern* forms. Different localities around the country may have only one or all morphotypes, presumably reflecting separate introductions.

Undaria is an annual species, with a life-cycle that alternates between microscopic spores and gametophytes, and the visible kelp stage or sporophyte (Figure 2.2). Within its native range, the life-cycle has a strongly defined seasonality (e.g., Akiyama and Kurogi 1982); sporophytes grow through winter and mature in early-mid spring, release millions of asexual spores as sea temperatures increase, and die-off in summer and autumn. Following settlement, spores germinate into microscopic male and female gametophytes. As sea water temperatures drop, fusion of egg and sperm produced by the gametophytes gives rise to the next season's sporophytes. Hence, spores released from a single sporophyte can seed a new generation of *Undaria* (Hay and Luckens 1987).

Figure 2.1 *Undaria* from Lyttelton Harbour.

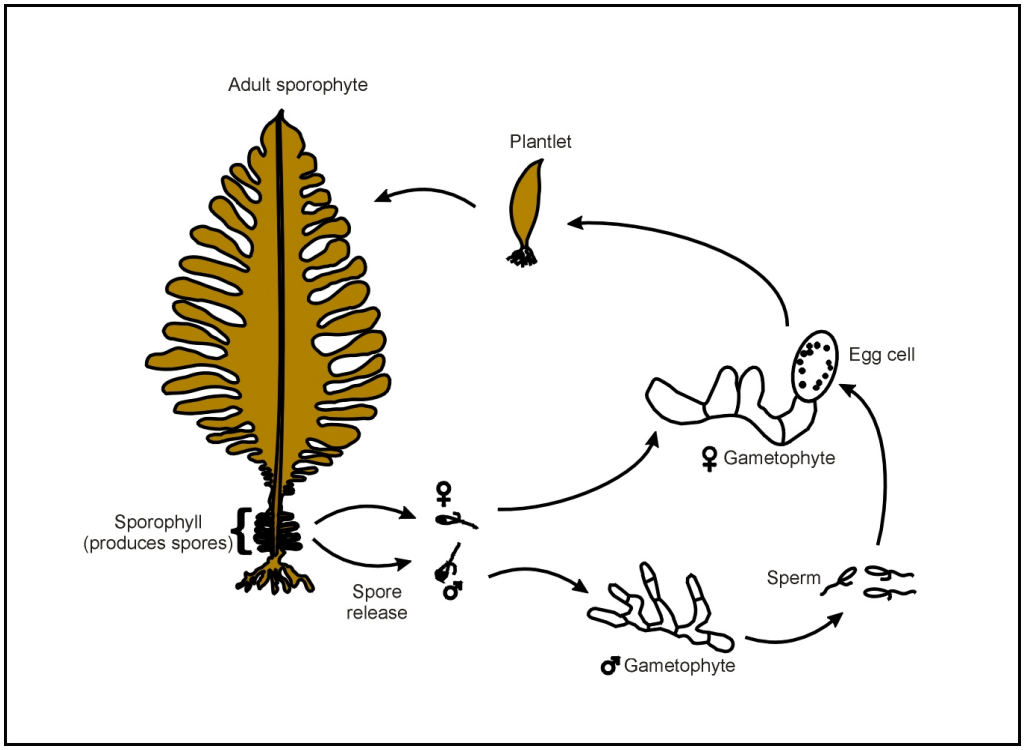

Figure 2.2 Life cycle of Undaria showing visible (brown) and microscopic (no shade) phases.

In New Zealand, *Undaria*'s annual life-cycle is less clearly defined than in its native range. Although the bulk of the sporophyte population dies off during summer, at many sites in the South Island mature sporophytes may be present over most of the year (e.g., Hay and Villouta 1993; Stuart 1997). A lack of strongly defined seasonality has also been observed in a number of other countries where *Undaria* has established (e.g., Fletcher and Farrell 1999; Castric-Fey et al. 1999) and probably reflects the less severe seasonal range in seawater temperatures in those localities compared with *Undaria*'s native range. For example, annual variation of about 10 °C is typical in New Zealand (Greig et al. 1988), with summer seawater temperatures of 20 ^{o}C or less in central New Zealand being suitable for sporophyte development and growth (Akiyama 1965; Saito 1975). By contrast, sea surface temperatures in some areas of Japan range from less than 0 °C in winter to 27 °C in summer (e.g., Funahashi 1973), with the latter being beyond sporophyte tolerances.

2.1.2 *Undaria*'s recorded distribution in New Zealand

The known distribution of *Undaria* in New Zealand, along with the year it was first recorded, was last reported in the published literature by Forrest et al. (2000) as shown in Figure 4.1 of Chapter 4. Subsequently, *Undaria* has established populations in: Half Moon Bay, Stewart Island (2000); Wainui Bay, Nelson (2001); Karitane, Otago (2002); Kaikoura (2002); the Firth of Thames (2002); Waitemata Harbour (2004); Tauranga Harbour (2005); and New Plymouth (2005). Hence, since its first discovery in 1987, *Undaria* has become established in many sheltered harbour areas and a few semi-exposed localities (e.g, Moeraki) along New Zealand's east coast south of Waitemata Harbour. While the seaweed is clearly quite widespread, available knowledge nonetheless suggests that large tracts of natural coastline remain uninfested. It is important to recognise, however, that even for a high profile species like *Undaria*, the recorded distribution may under-represent the true situation because there are no comprehensive surveillance regimes in place outside of the main harbours.

Within its present distributional range *Undaria* can be found from the low intertidal zone down to approximately 18 m depth (Hay and Villouta 1993). It can grow on a range of artificial surfaces including rope, wood, bottles, floating pontoons, and plastic (e.g., Hay and Luckens 1987; Hay 1990; pers. obs.). On natural shores, *Undaria* occurs in a wide range of habitats including: on stable rocky reefs; in mobile cobble habitats; on mudstone; and within primarily soft sediment habitats where it can attach to hard

surfaces such as shell (Hay and Luckens 1987; Sanderson 1997; pers. obs.). *Undaria* can also grow on seagrass (while a small sporophyte), the shells of abalone and other bivalves (Campbell and Burridge 1998), on invertebrates such as sea tulip stalks, and epiphytically on other seaweeds (pers. obs.). Its invasiveness can vary widely in space and time, as noted in Chapter 6, hence populations can range from dense infestations to sporadic, low density stands (Figure 2.3).

Figure 2.3 High density infestation of *Undaria* in the low intertidal zone of Lyttelton Harbour (left), and a low density subtidal population near Picton (right).

2.1.3 Environmental constraints on *Undaria*'s distribution in New Zealand

The geographic boundaries of *Undaria*'s further spread in New Zealand will primarily be determined by sea water temperature (e.g., Sanderson and Barrett 1989). In this regard, sea surface temperatures around much of the New Zealand coastline largely fall within the limits for *Undaria*'s survival and reproduction, although sea surface temperatures along the northern coast from Cape Reinga to East Cape are higher than 'optimal' for the sporophyte (Sinner et al. 2000). This could explain why *Undaria* took many years to establish in the Hauraki Gulf region around Auckland when, based on the relatively high amount of vessel traffic to the region (e.g., Gust et al. 2005; Dodgshun et al. 2004), one would presume that it was transported there on numerous occasions.

The distributional predictions for *Undaria* made by Sinner et al. (2000) were based on broad patterns in sea surface temperature, hence are relatively crude given that variation will occur over time (including long-term changes) and locally within harbours or shallow bays. It should also be recognised that there are many examples in the bioinvasion literature, including for *Undaria* (Floc'h et al. 1988) and the highly invasive green alga *Caulerpa taxifolia* (Chisholm et al. 2000), where the adventive distributions of pest species have extended beyond that anticipated from seawater temperatures in their native range.

Factors such as salinity and wave exposure will also add layers of complexity to the assessment based on temperature alone. For example, the intolerance of *Undaria* sporophytes to low salinity water (Bardach et al. 1982; Chapter 6) suggests that the seaweed is likely to be excluded from coastal areas with a high freshwater input, which may explain its apparent absence from New Zealand's river-dominated southern and western ports such as Riverton, Greymouth and Westport. In terms of wave-exposure, *Undaria* in New Zealand is present from sheltered to semi-exposed areas, but is not known to be present on highly wave-exposed coasts. In Tasmania, however, low densities of *Undaria* can be found in association with the kelp *Durvillaea potatorum* at shallow depths on moderately wave-exposed coasts, with high density *Undaria* stands in adjacent deeper areas of high water clarity where wave exposure is less (Hay and Sanderson 1999). The Tasmanian experience suggests that exposed New Zealand coastlines will not necessarily provide a barrier to *Undaria*'s ultimate spread.

2.1.4 The role of human vectors in *Undaria*'s potential distribution

In terms of understanding the rate and pattern of *Undaria*'s future spread, within the environmental constraints on its distribution, it is important to understand that the seaweed has a limited capacity for natural dispersal; in the order of 100s of metres to a few kilometres per year via spores released from fixed stands or from detached drifting sporophytes (Chapter 4). This means that human-mediated transport vectors are important in the dispersal of *Undaria* at inter-regional or greater scales - this is reflected in the haphazard spatio-temporal pattern of the seaweed's spread around New Zealand and its distribution primarily around hubs of human activity.

A summary by Hewitt et al. (2004) identifies at least 20 present-day pathways that could be important in the domestic translocation of non-indigenous marine species from

their initial points of incursion. Coastal vessel traffic and aquaculture are widely recognised as important vectors for the domestic spread of non-indigenous species generally, (e.g., Forrest and Blakemore 2002; Coutts et al. 2003; Coutts and Taylor 2004; Floerl and Inglis 2005), and are particularly important for *Undaria*. Key vectors for *Undaria*'s spread are vessels of all sizes (commercial and recreational), primarily via ballast water or hull fouling (Hay 1990; Fletcher and Manfredi 1995; Casas and Piriz 1996; Floc'h et al. 1996), and marine farming gear and seed-stock (Perez et al. 1981; Bourdouresque et al. 1985; Stuart 1997; Forrest and Blakemore 2003). Fouling transfers associated with vessel movements and marine farming activities appear to be particularly significant (e.g., Hay 1990; Forrest and Blakemore 2003), as illustrated in Figure 2.4. Other potential vectors include:

- Transport of sporophytes as a food for fishery species such as sea urchins and abalone (Campbell and Burridge 1998; Dr C. Sanderson, pers. comm.).
- Entanglement or fouling of *Undaria* on equipment associated with vessels such as anchors, lobster pots, nets, ropes and floats (Sanderson 1997).
- Vessel bilge water contaminated with spores.
- Diving gear such as wet suits contaminated with spores, and fouled catch bags (Dr C. Hewitt, pers. comm.).
- Fouled flotsam, such as marine farm floats that have been lost from farms or vessels and washed ashore (pers. obs.).
- Commercial or scientific cultivation of *Undaria*.

Depending on the particular vector, therefore, *Undaria* can be transported as a visible sporophyte, or as microscopic gametophytes or spores, and sometimes via all of these life-stages. However, transport of mature sporophytes arguably represents the greater risk (Chapter 4); a single mature sporophyte carried on a boat visiting an uninfested (*Undaria*-free) area for a short period (i.e., minutes to hours), for example, has the potential to release millions of spores and establish a new population. By contrast, where microscope life-stages are transported they primarily constitute a risk if they are released from the transport vector, or the vector (e.g., an infected marine farm structure) remains in an uninfested region for a sufficient duration that *Undaria* completes it's life cycle and releases spores. Available data suggest that the duration required would be approximately 1-3 months depending on whether gametophytes or immature sporophytes were introduced (Campbell and Burridge 1998; Gibbs et al. 1998).

Figure 2.4 *Undaria* fouling vessels (top), a mussel farm float (bottom left) and mooring ropes (bottom right).

Clearly, therefore, the ability of *Undaria* to colonize a wide variety of artificial surfaces, it's multiple modes of dispersal, and propensity to colonize vessels and floating structures like marine farms (Hay 1990; Floc'h et al. 1996; Fletcher and Farrell 1999), makes it well-suited to human-mediated transport. Hence, the rate and pattern of *Undaria*'s future spread to suitable habitats is likely to be primarily determined by

spatio-temporal patterns of vector movement from infested to uninfested areas. *Undaria* is likely to radiate out from present reservoirs, locally by natural spread and both locally and regionally by human-assisted mechanisms.

It is probable that, over some unknown time-frame (e.g., perhaps hundreds of years) and in the absence of management measures, *Undaria* will spread naturally to most suitable (i.e., where environmental constraints are not limiting) mainland habitats. However, where natural barriers prevent coastal transport (for example extensive soft-sediment areas along exposed shores of the South Island's west coast), *Undaria*'s spread may depend almost entirely on the assistance of human vectors.

2.2 HISTORY OF *UNDARIA*'S MANAGEMENT

Management of *Undaria* in New Zealand does not appear to have been considered during the 10 year period following its discovery in 1987, reflecting the general lack of interest and awareness among government agencies and the general public at that time regarding marine biosecurity issues. Over this period, there was nonetheless scientific interest and concern regarding *Undaria* and its potential impacts (e.g., Hay 1990; Hay and Villouta 1993; Parsons 1994), with a more widespread interest in *Undaria* and its management subsequently emerging. This was precipitated in March 1997 when *Undaria* was reported on a marine farm in Big Glory Bay, Stewart Island. At this time, the kelp had not been recorded further south than Otago Harbour, and it was recognised that a widespread infestation in Big Glory Bay would increase the likelihood that it would be spread by human vectors to high value conservation areas. These included other parts of Stewart Island, as well as Fiordland and the sub-Antarctic Islands. Hence, following consultation with scientists, the Department of Conservation (DoC) recommended that eradication of the Big Glory Bay population be attempted.

Although there was no clear line of responsibility for management, in August 1997 Cabinet agreed to funding of $0.163 million for DoC to conduct an eradication campaign. In subsequent years, increases in funding were approved, amounting to approximately $2.2 million over the 5 years from 1998/99 to 2002/03. This allowed the programme to expand beyond Big Glory Bay to include surveillance of vessels in southern New Zealand ports coupled with public awareness campaigns, and also provided sufficient funds to manage a population of *Undaria* that was discovered in

1999 in nearby Bluff Harbour. These steps recognised that prevention of re-incursion to Big Glory Bay was fundamental to the success of the eradication programme. Overall, therefore, a comprehensive management regime was put in place on a scale that does not appear to have been attempted anywhere else in the world.

With the eradication programme in southern New Zealand in progress, consideration of *Undaria* management at a national level was initiated in 1999. This arose because of a perceived threat from *Undaria* to natural ecosystems and associated fisheries, and because it was considered a potential biofouling pest to the marine farming industry. Hence, the Government directed the Ministry of Fisheries to develop a national strategy for the long-term management of *Undaria*. It was expected that such a strategy would maintain the benefits of the southern eradication programme, and provide a national framework for containment and control of *Undaria* in other areas. This national phase began with stakeholder consultation and a report on options for managing *Undaria* by Sinner et al. (2000). Two key themes emerged from the Sinner et al. (2000) report that had significant implications for *Undaria*'s management, the first relating to *Undaria*'s potential impacts and the second to the feasibility of its widespread control.

In the first instance, while it was clear that the presence of *Undaria* threatened that natural character of much of the New Zealand coastal environment, there was no clear evidence that *Undaria* caused significant economic or ecological impacts. In relation to the expected commercial impacts from fouling, for example, the assessment by Sinner et al. (2000) suggested that *Undaria* had a certain 'nuisance' value to marine farmers, but was being managed along with other biofouling pests for little additional cost. Furthermore, the likelihood that major ecosystem effects would occur was being debated by scientists at that time and opposing views emerged; without clear evidence either way, arguments both for and against adverse effects are equally plausible, as described by the Sinner et al. (2000) report. As such, management decisions by government agencies would need to be driven by perceived threats rather than actual knowledge.

The second key theme to emerge from the Sinner et al. (2000) report was that *Undaria* management on a large scale was not feasible. In the 13 years that had elapsed since *Undaria*'s first discovery, the seaweed had become well-established around the New Zealand coastline. Eradication of established populations or widespread containment of *Undaria*, for example through national vector management, was clearly not feasible.

Moreover, the lack of compelling evidence for a significant ecological or economic threat resulted in a lack of interest from some stakeholders whose co-operation was essential to a management programme. Hence, while Sinner et al. (2000) evaluated a range of management options, they were all based around protecting geographically discrete high value areas (HVAs) from the effects of *Undaria*. Such an approach would provide for management at a smaller and more feasible scale than the entire New Zealand coastal environment. That work formed the basis of a spatially-explicit values-driven approach to setting management priorities, which is formalised within the context of the risk management model presented in Chapter 8.

Although the Sinner et al. (2000) report made no firm management recommendations, the most feasible approach the authors put forward was to attempt to keep *Undaria* out of uninfested HVAs for which management of human vectors was feasible and which were sufficiently remote that they were not vulnerable to the spread of *Undaria* via natural dispersal. In particular, they recommended that management focus on the most pristine HVAs (Fiordland and the sub-Antarctic Islands), whose remoteness would make effective surveillance or incursion response difficult. In such instances, it was considered that prevention of introduction was more feasible than detecting and responding to new incursions.

Following the release of the Sinner et al. (2000) report, the Ministry of Fisheries developed a national strategy for *Undaria* management (although did not pursue a formal National Pest Management Strategy under the Biosecurity Act 1993), and in 2003 the Ministry of Fisheries sought funding for the strategy from Cabinet. Cabinet agreed in principle to a limited programme of vector management and population control to protect a few pristine HVAs (primarily those referred to above) from *Undaria*, with a final decision for ongoing funding depending on the 2004 Budget process. Leading up to this time, however, a significant *Undaria* population had been discovered on a shallow subtidal reef at Half Moon Bay, Stewart Island (adjacent to Big Glory Bay). This required diversion and dilution of funds from the southern New Zealand management efforts, and essentially reflected a failure of that programme to prevent incursions with the controlled area.

Hence, at a Technical Advisory Group meeting in 2003 it was recognised that eradication of *Undaria* from Stewart Island was no longer feasible, and recommendations were made that efforts should focus more on containment of existing

southern New Zealand populations to prevent the seaweed's spread to the key HVAs (pers. obs.). However, the final Cabinet decision following the 2004 Budget process was not to fund any management programmes specific to *Undaria*, which meant an end to the efforts in southern New Zealand. The only *Undaria* management would be covered as part of a general vector management programme being implemented for vessels travelling to the subantarctic islands.

The government's decision caused sufficient concern among some stakeholders that regional councils together with Biosecurity New Zealand formed a national forum to further consider the need for and efficacy of *Undaria* management; as the seaweed spreads there continues to be interest (especially at a regional level) in the management of localised populations that threaten areas of high conservation value. Hence, Biosecurity New Zealand convened a Technical Advisory Group meeting in June 2005 to again consider *Undaria*'s actual or potential impacts and the feasibility of its management. Based on the outcomes of this meeting it was intended that the costs and benefits of managing *Undaria* would be evaluated against other biosecurity priorities. At the time of writing no further decisions have been made regarding the future of *Undaria* management in New Zealand, although small-scale management efforts are being undertaken within particular regions.

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Chapter 3

Ecological Impact of *Undaria*

PREFACE

This chapter is a case study of invasion consequences. It is built around the ecological impacts of *Undaria* in low shore rocky habitats of Lyttelton Harbour, and constitutes the first rigourous evaluation of *Undaria*'s effects in New Zealand. I also use the work as a platform to present a novel view of the limitations in applying traditional environmental sampling designs to studies of invasive species impacts. This work has been published in a refereed journal and is presented below in identical form. The citation for the original publication is:

Forrest BM, Taylor MD. 2002. Assessing invasion impact: survey design considerations and implications for management of an invasive marine plant. Biological Invasions 4: 375-386

My co-author and thesis supervisor Dr Mike Taylor had input into a number of facets of the project, but especially into the survey design and through participation in the field programme. The taxonomic and data analyses, and preparation of the manuscript are primarily my work, with editorial input from Dr Taylor that included assistance with SAS analyses. The wider discussion regarding sampling design issues, which sets the publication apart from a manuscript that only presents a study of ecological effects, stems from my background and experience in environmental pollution assessment studies.

ABSTRACT

We use a three-year study of sheltered low shore assemblages colonised by the non-indigenous Asian kelp *Undaria pinnatifida* to explore survey design issues for assessing the ecological impacts of invasive species. The weight of evidence overall suggested little impact from *Undaria* on low shore assemblages, with control-impact contrasts that could plausibly be interpreted as impacts probably reflecting natural causes. We demonstrate that the potential for reaching incorrect conclusions regarding the impacts of invasive species using control-impact designs is greater than when such designs are used to assess traditional forms of anthropogenic impact. We suggest that a before-after-control-impact (BACI) framework is essential, but recognize that such an approach has a number of limitations. In particular, there is no assurance that the before-after impact site will be invaded at all, or to the extent that provides worst-case impact information for coastal managers. We discuss possible ways of assessing invasive species impacts, but suggest that the uncertainty inherent in extrapolating impact information to other places and times means that the precautionary principle should be applied, and 'worst-case' impacts assumed, until the level of scientific uncertainty is reduced. Such an approach should only be applied, however, after an evaluation of the feasibility, costs and benefits of managing a particular pest in relation to other priorities for invasive species.

3.1 INTRODUCTION

The nature and severity of impacts caused by invasive species, and the relative effects of one species over another, will be key considerations in setting management priorities for them. Comparative studies (e.g., Findlay et al. 2000), local-scale field surveys (e.g., Windham 1999), long-term data sets (e.g., Howe et al. 1997) and various experimental approaches (e.g., Olsen et al. 1991; Floc'h et al. 1996) have all been used to describe the effects of invasive species and identify mechanisms that may lead to significant impacts. For many invasive species, however, and for invasive marine species in particular, unequivocal evidence of impacts is generally lacking, even for those considered a significant threat (Blossey 1999; Parker et al. 1999). Rather, the literature for many such species is primarily dominated by accounts of only their occurrence and spread. While evidence of impacts remains equivocal and largely speculative, rational management decisions cannot be made, and dissenting views from scientists are likely (Peterson 1993; Blossey 1999).

The Asian kelp, *Undaria pinnatifida,* typifies this situation. *Undaria* is a large (1-2 m length) canopy-forming species that can reach high densities in both artificial and natural habitats (e.g., Hay and Villouta 1993). It is considered a potential fouling nuisance (Sanderson 1997; Fletcher and Farrell 1999), and a threat to natural

ecosystems and associated fisheries, for example through displacement of native species via the development of 'mono-specific' *Undaria* stands (Sanderson and Barrett 1989; Miller et al. 1997; Stuart 1997; Battershill et al. 1998). While its basic biology (summarised in Sanderson and Barrett 1989), spread (e.g., Hay 1990; Sanderson 1990; Fletcher and Manfredi 1995; Casas and Piriz 1996; Forrest et al. 2000), population dynamics (Hay and Villouta 1993; Brown and Lamare 1994; Castric-Fey et al. 1999), and physiology (Campbell et al. 1999) are quite well understood, information on impacts is limited, and often speculative and polarised (e.g., Rueness 1989; Parsons 1994; Battershill et al. 1998; Miller et al. 1997; Stuart 1998; Walker and Kendrick 1998; Sinner et al. 2000).

Battershill et al. (1998), for example, made spatial comparisons of ecological assemblages in areas with *Undaria* at different infestation levels, with those dominated by native *Carpophyllum* spp. They suggested that significant ecological changes to the *Carpophyllum* sub-canopy community resulted from *Undaria*'s establishment, and concluded that *Undaria* may displace multi-species macroalgal communities characterised by *Carpophyllum*. In contrast, Hay and Villouta (1993), with reference to the same general locality, suggested that *Undaria* colonised bare areas outside beds of native *Carpophyllum*, rather than the beds themselves. Similarly, Hay and Sanderson (1999) considered that there was very little evidence that *Undaria* displaced native brown seaweeds in several New Zealand harbours where it had been established for many years.

In the climate of uncertainty regarding *Undaria*'s impacts, a precautionary approach to the seaweed's management in New Zealand has been advocated by some regional and central government agencies. In contrast, many private stakeholders (e.g., vessel operators, marine farmers), for whom *Undaria* management costs (e.g., for regular hull de-fouling) could be significant, are reluctant to be drawn into a management strategy when adverse effects have not been documented and hence the benefits of management are unclear.

The example of *Undaria* thus highlights a considerable need for defensible information on impacts. In the studies referred to above, the lack of a pre-invasion baseline, and hence the associated uncertainty regarding the level of ecological change caused by *Undaria*, clearly contributed to the dissenting opinions on impacts and the need for management. The limitations of control-impact surveys in studies of the effects of

anthropogenic pollution have been recognised for some time, and the advantages of establishing baselines and inferring impacts based on before-after control-impact (BACI) designs and their variants have been widely promoted (e.g., Green 1979, 1993; Stewart-Oaten et al. 1986; Underwood 1991, 1992, 1993, 1994).

This paper describes a three year investigation of rocky low shore assemblages in a sheltered New Zealand harbour, and examines the efficacy of BACI and control-impact designs in assessing *Undaria*'s impacts. We also consider the utility of these survey designs in assessing the effects of invasive species generally, and identify a number of areas where their application has significant limitations when compared with their more traditional use in anthropogenic impact studies.

3.2 MATERIALS AND METHODS

3.2.1 Study sites and sampling

Our investigations were conducted in the low neap-spring tide zone at four sites in Lyttelton Harbour, New Zealand (Figure 3.1), in algal-dominated habitats consisting of stable boulders and bedrock. A combination of small tidal range (~ 2 m), moderate shore slope, and poor water clarity, confined *Undaria* to a narrow band (typically 1-3 m wide) in this zone. Sites consisted of: one infested locality (Cass Bay) where *Undaria* was already established; one uninfested locality (Diamond Harbour) which became infested during the study (as we had anticipated); and two uninfested control locations (Control 1 and Control 2) that were isolated from known vector pathways and beyond the likely range of natural spread via spore dispersal (Forrest et al. 2000).

In its native range *Undaria* is an annual species exhibiting a strong seasonal hiatus between the sporophyte which is dominant in spring, and the microscopic gametophyte that is present over late summer and autumn/fall during sporophyte senescence (Akiyama and Kurogi 1982). While such a marked seasonality is less evident in New Zealand, larger sporophytes are nevertheless more prevalent during late winter and spring (Hay and Villouta 1993), suggesting some potential for a seasonal difference in impact. To account for such possibilities, surveys at each of the four sites were carried out in spring (September-November) and autumn/fall (March-May), for the three years from spring 1997 to autumn 2000.

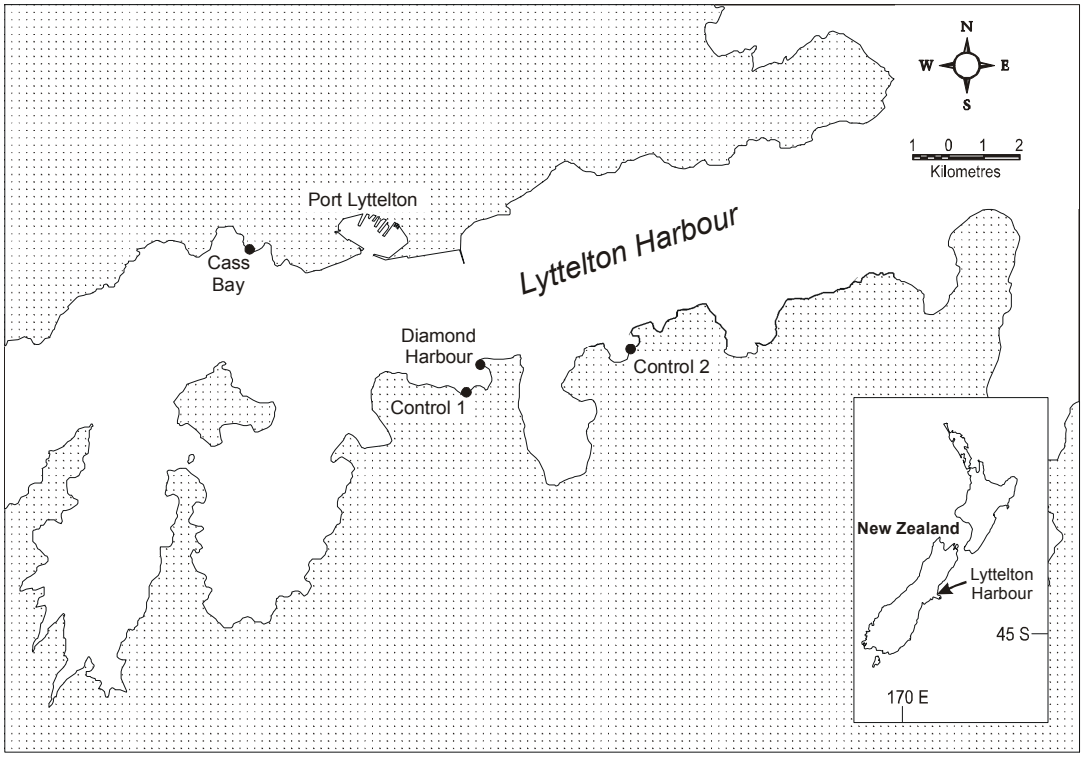

Figure 3.1 Map of Lyttelton Harbour, New Zealand, showing the four study sites.

Sampling was undertaken using transect and quadrat methods. Two long-shore transects (50 m length) were sampled at each site: one along the neap tide level corresponding to *Undaria*'s upper limit on the shore and one at the level of low spring tide where *Undaria* was most prevalent. Point-sampling on each transect was conducted at 80 randomly generated distances. Macroalgae, sessile invertebrates, or bare rock falling beneath each of the 80 points were recorded. Between the two transects (i.e., spanning the neap-spring tide zone) eight quadrats (0.25 m^2) with 80 mm grid spacings were placed at pre-determined random distances. The number of *Undaria* within each quadrat was determined, and macroalgae, sessile invertebrates, or bare rock falling beneath each of the 49 intercept points formed by the grid were recorded.

The time constraints of low shore sampling and the limited number of suitably low spring tides meant that only the canopy level of substratum cover could be sampled using this method. Changes to sub-canopy assemblages are nevertheless of interest in terms of assessing the ecological effects of *Undaria*, hence we also recorded (presence/absence) the conspicuous taxa in the quadrats that were not detected by the

point count method. Point counts generated from both the transect and quadrat sampling were later converted to percent cover. Taxon richness data were derived from the total number of different taxa recorded within quadrats irrespective of sampling method (point intercept, counts and presence/absence). Taxonomic identification in the field was made to species level where practicable, but voucher specimens collected as necessary.

3.2.2 Statistical analyses

A control-impact inference structure was based on planned comparisons between Cass Bay and the control sites, since Cass Bay was infested with *Undaria* from the outset. At Diamond Harbour, where *Undaria* was first recorded in spring 1998, there were two 'before' sampling times (spring 1997-autumn 1998) and four 'after' sampling times (spring 1998-autumn 2000). Thus, the inference structure was based on a BACI design and used the following planned comparisons: 'before' at Diamond Harbour versus 'after' at Diamond Harbour; 'before' at the two control sites versus 'after' at the controls; 'before' at Diamond Harbour versus 'before' at the controls; and 'after' at Diamond Harbour versus 'after' at the controls. Hence this BACI structure at Diamond Harbour also provided a 'control-impact (after)' contrast for direct comparison with the Cass Bay situation.

For univariate analyses (ANOVA and Pearson correlation), data were entered into SAS (SAS/STAT 1997) and log(X+1)-transformed (where necessary) prior to analysis to satisfy the independence and normality of error terms assumptions of the general linear model. Data were analysed using the MIXED procedure with site, sampling time and their interaction term included as main fixed effects. Quadrat and transect (spring and neap tide) were declared random effects nested within site, and evidence for quadrat effects and serial correlation (AR 1) were investigated using the restricted maximum likelihood (REML) method.

Multivariate analyses of quadrat data (pooled within each site and survey) were undertaken with the software package PRIMER V5, to examine spatio-temporal patterns in community composition. The dataset was derived by weighting each taxon by the number of quadrats in which it was recorded for any one site and survey, thus providing a measure of relative abundance on a 0-8 scale. For example, *Undaria* was recorded in six out of eight quadrats in spring 1997 at Cass Bay so is scored as six. Using this dataset, a 2-dimensional non-metric multi-dimensional scaling (nMDS)

ordination was produced from a Bray-Curtis similarity matrix. Using group average clustering, site groups that formed at a 60% Bray-Curtis similarity threshold were superimposed on the nMDS ordination pattern (Clarke 1993). The SIMPER procedure (Clarke 1993) was used to identify the major taxa contributing to the site groups, and one-way ANOSIM (Clarke 1993) used to examine the control-impact and BACI contrasts described above. Bray-Curtis similarity measures for pairwise combinations of sites were examined to describe temporal trajectories in site similarity.

3.3 RESULTS

3.3.1 *Undaria* infestation levels and impacts on canopy species

Temporal changes in *Undaria* infestation levels did not follow any consistent seasonal pattern, in contrast to our expectations. The percent cover of *Undaria* in quadrats (Figure 3.2) and along transects (Figure 3.3) was greatest at both Cass Bay and Diamond Harbour in spring 1998, and steadily declined thereafter. Maximum percent cover levels, as recorded from transects, were approximately 45% and 19% for the two sites respectively. The density of *Undaria* was notably high at Cass Bay (~130 sporophytes m^{-2}) in Spring 1998 but was otherwise less than half of this value, with higher density patches characterised by numerous small or immature sporophytes rather than mature-sized plants.

Native canopy species (defined in this study as *Sargassum sinclairii*, *Ecklonia radiata* and *Macrocystis pyrifera*) covered up to 40% of the substratum and consisted primarily of *Carpophyllum maschalocarpum*, although juvenile *Ecklonia radiata* and *Macrocystis pyrifera* were sometimes more dominant (Figs 6.2 and 6.3). As for *Undaria*, a greater canopy cover was generally recorded along transects than in quadrats (e.g., spring 1997). In part this will reflect the placement of the spring tide transects in the lowest accessible part of the intertidal zone where algal cover was very high compared with the area between spring and neap where the quadrats were positioned.

There was no evidence for displacement of the native canopy by *Undaria*, with planned contrasts of percent cover between the controls and each of the infested sites largely suggesting a ‘no impact’ result (Table 3.1). The quadrat percent cover results are equivocal, however, owing to significant random effects. The most interesting contrast was the significantly lower native canopy cover at Diamond Harbour compared with

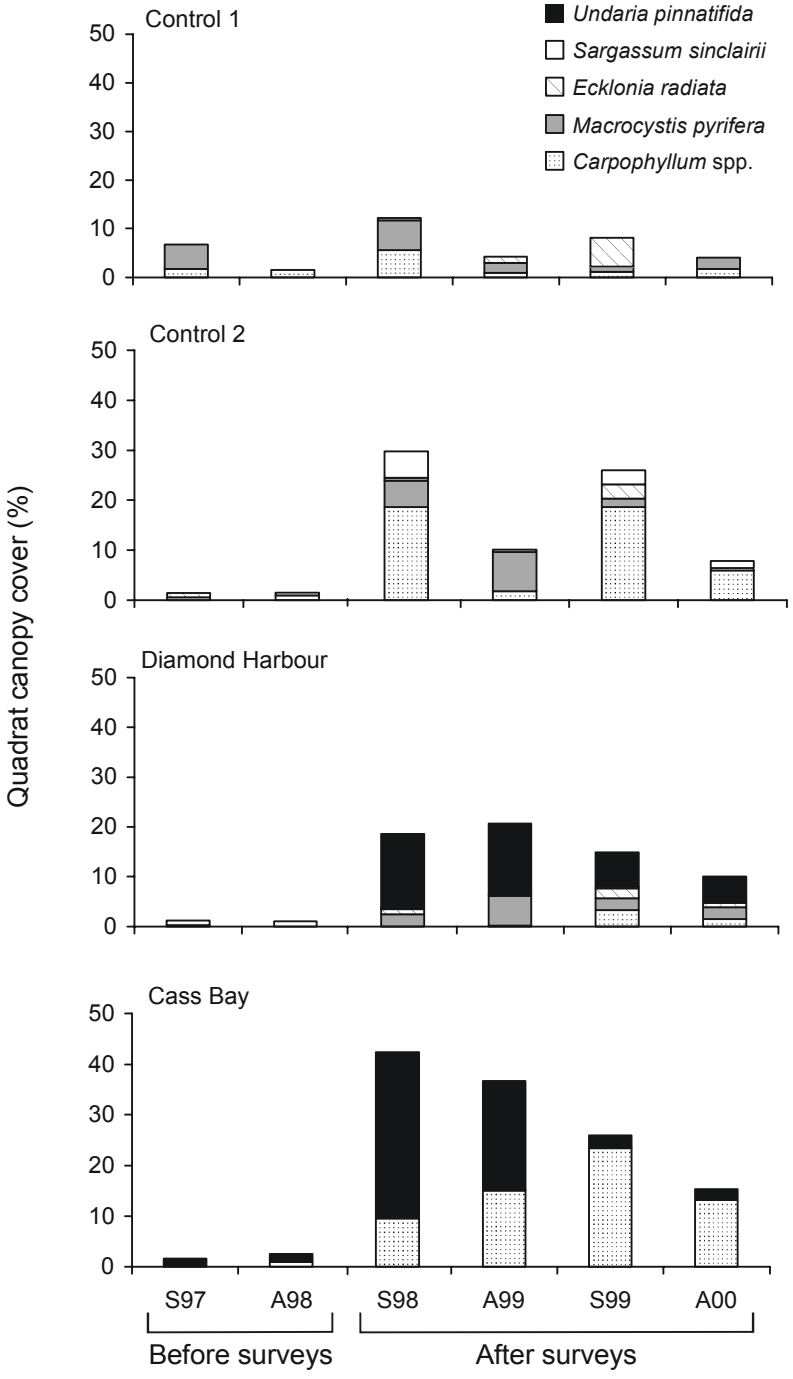

Figure 3.2 Mean percent cover of *Undaria* and other canopy-forming seaweeds within quadrats (0.25 m2) over the six surveys from spring 1997 (S97) to autumn/fall 2000 (A00). *Undaria* was first recorded at Diamond Harbour in Spring 1998.

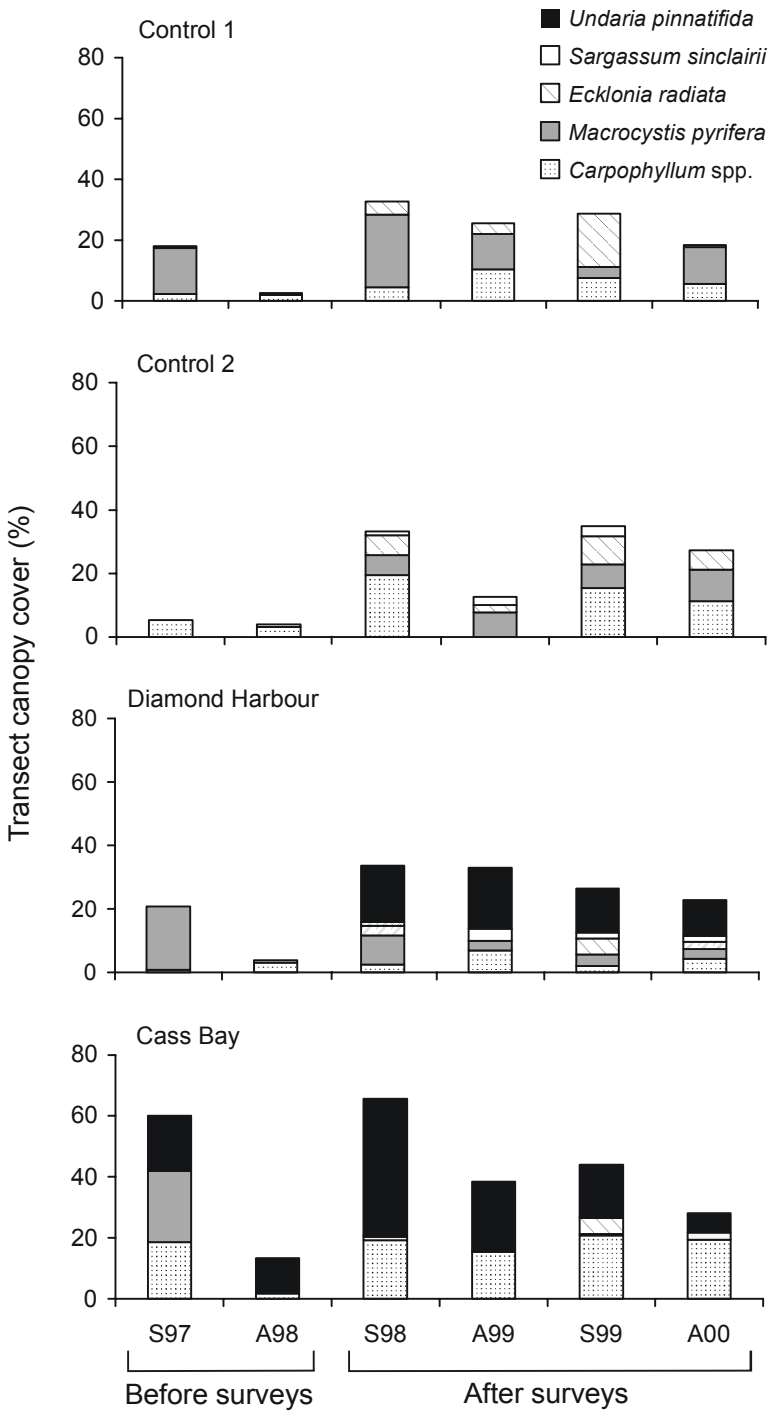

Figure 3.3 Percent cover of *Undaria* and other canopy-forming seaweeds along transects (data pooled over neap and spring tide level) over the six surveys from spring 1997 (S97) to autumn/fall 2000 (A00). *Undaria* was first recorded at Diamond Harbour in Spring 1998.

Table 3.1 Summary of mixed model analyses of variance for the control vs impact and BACI designs. P values are shown with numerator/denominator degrees of freedom for fixed effects and 95% confidence limits [L_1, L_2] for random effects. AR 1 refers to serial correlation within random effects.

Survey design	Canopy cover (quadrats)	Canopy cover (transects)	Algal richness (quadrats)	Faunal richness (quadrats)
Control-impact				
Fixed effects:				
Time	<0.001, 5/104	<0.001, 5/15	<0.001, 5/104	<0.001, 5/104
Site	0.244, 2/21	0.942, 2/3	0.013, 2/21	<0.001, 2/21
Time*Site	0.02, 10/104	0.127, 10/15	0.372, 10/104	<0.001, 10/104
Cass Bay vs Controls	0.634, 1/21	0.831, 1/3	0.020, 1/21	0.160, 1/21
Random effects:				
Quadrat/Transect(Site)	0.017, [0.01, 0.11]	0.127, [-0.08, 0.03]	0.135, [-0.19, 1.37]	0.963 [-0.74, 0.78]
AR 1 Quadrat/Transect(Site)	0.005, [-0.54, -0.09]	0.517 [-0.61, 1.19]	0.975, [-0.28, 0.27]	0.159 [-0.41, 0.07]
BACI				
Fixed effects:				
Time	<0.001, 5/104	<0.001, 5/15	<0.001, 5/104	<0.001
Site	0.032, 2/21	0.879, 2/3	0.060, 2/21	<0.001
Time*Site	0.062, 10/104	0.096, 10/15	0.839, 10/104	0.001
Diamond Hbr vs Controls (before)	0.534, 1/104	0.321, 1/15	0.159, 1/104	0.454, 1/104
Diamond Hbr vs Controls (after)	0.019, 1/104	0.216, 1/15	0.616, 1/104	0.011, 1/104
Before vs after: Diamond Hbr	0.019, 1/104	0.114, 1/15	<0.001, 1/104	0.001, 1/104
Before vs after : Controls	<0.001, 1/104	<0.001, 1/15	<0.001, 1/104	0.002, 1/104
Random effects:				
Quadrat/Transect(Site)	0.044, [0.0, 0.08]	0.544, [-0.07, 0.13]	0.106, [-0.11, 1.13]	Note 1
AR 1 Quadrat/Transect(Site)	0.094, [-0.44, 0.03]	0.3, [-0.3, 0.96]	0.624, [-0.34, 0.21]	0.767 [-0.24, 0.18]

Note:

1 Variance estimate = 0, however p value and confidence limits not calculable

control quadrats 'after' *Undaria* arrived. While displacement of the native canopy could be inferred from this spatial pattern, such an interpretation contrasts with the observation that the cover of native canopy species at Diamond Harbour significantly increased from 'before' to 'after' *Undaria*'s arrival (Table 3.1, Figure 3.2). In fact, Pearson correlation revealed a weak positive association ($r = 0.28$, $p = 0.06$) between the cover of *Undaria* and the native canopy at Diamond Harbour, rather than a negative effect. There was little association between these variables at Cass Bay ($r = 0.19$, $p = 0.20$).

3.3.2 Impacts on taxon richness

The richness of both macrofaunal and macroalgal taxa showed a high degree of year to year variation, although the sites showed similar temporal trends (Figure 3.4). Mean richness levels for macrofauna and algae were reasonably low, ranging from approximately 3-18 and 4-11 taxa per site respectively. There were no significant control-impact or BACI contrasts that would be consistent with the displacement of either macrofaunal or algal species by *Undaria* (Table 3.1). While algal richness at Cass Bay was significantly ($p < 0.05$) less than the control sites in the overall control-impact contrast (Table 3.1), this result does not appear to reflect an impact of *Undaria*, since algal richness was greater at Cass Bay than the controls on a number of occasions, including spring 1998 when the percent cover of *Undaria* was greatest. In fact, Pearson correlation provided evidence for a positive association between macroalgal richness and *Undaria*'s percent cover ($r = 0.24$, $p = 0.10$) and density ($r = 0.33$, $p = 0.02$) at the Cass Bay site. Similarly, algal richness at Diamond Harbour exhibited a strong positive correlation with *Undaria*'s percent cover ($r = 0.39$, $p = 0.006$) and density ($r = 0.49$, $p = 0.0004$).

3.3.3 Impacts on assemblage composition

The grazing snail *Turbo smaragdus* was common at all sites, but substratum cover outside the primary canopy was dominated by macroalgae - notably articulated corallines, *Ralfsia verrucosa*, *Cystophora* spp., *Hormosira banksii*, and *Gelidium caulacantheum*. The cover of bare rock and sessile macrofauna outside the primary canopy was typically < 20%, and was particularly low (or zero) at most sites in spring 1998. This not only reflected the arrival of *Undaria* at Diamond Harbour and the marked increase in its percent cover at Cass Bay, but also a far greater cover of other macroalgae at all sites in spring 1998 compared with other times. As was the case with the univariate measures above, the multivariate analyses of low shore assemblage composition provide no evidence of an ecological impact that could be attributed to *Undaria*'s invasion.

The nMDS site/survey ordination discriminated five groups of sites having a within-group Bray-Curtis similarity of approximately 60% (Figure 3.5). The infested Cass Bay site formed a distinct group for all six surveys. In spring 1998, each of the Diamond Harbour and two control sites formed individual clusters, while for all other surveys these sites formed a single group (hereafter referred to as the Diamond Harbour/Control

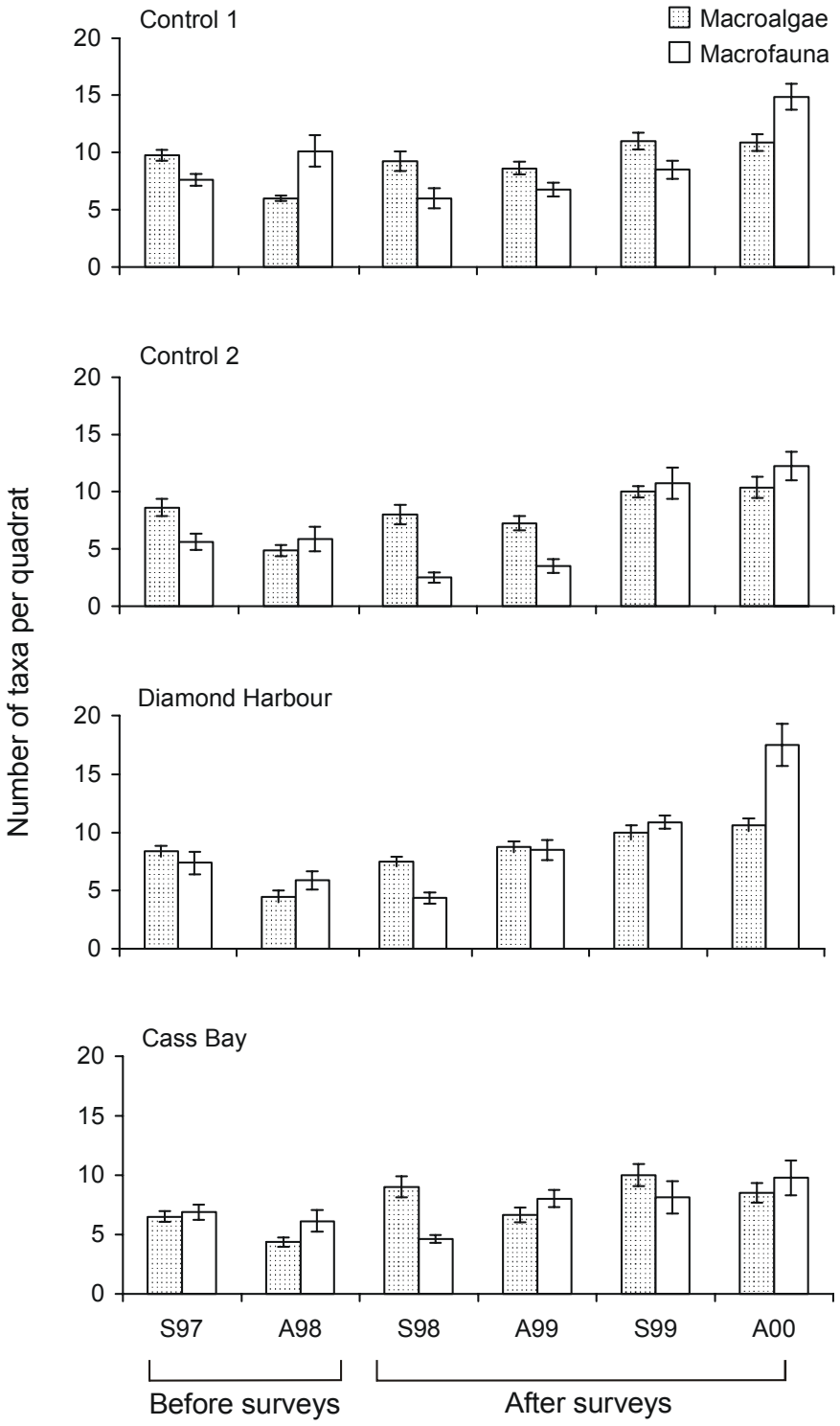

Figure 3.4 Mean number (± 1SE) of macroalgal and macrofaunal taxa within quadrats (0.25 m^2) over the six surveys from spring 1997 (S97) to autumn 2000 (A00). *Undaria* was first recorded at Diamond Harbour in Spring 1998.

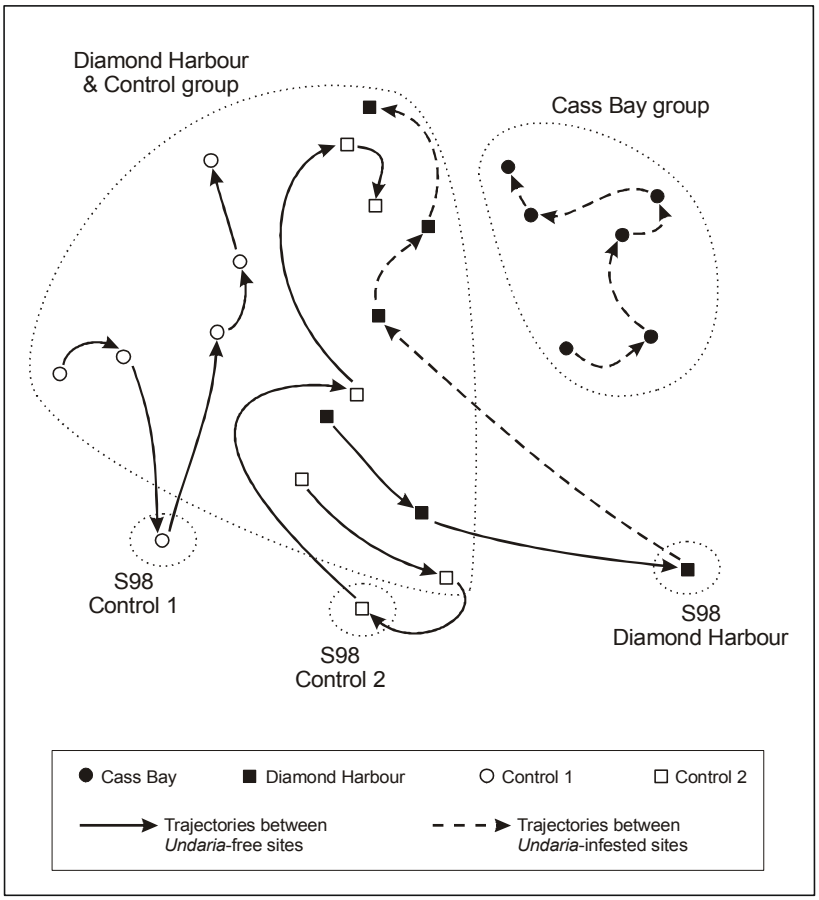

Figure 3.5 nMDS ordination (stress = 0.21) showing trajectories in assemblage composition at each of the four sites over the six surveys from spring 1997 to autumn 2000. The cluster analysis overlay indicates five groups of sites (encircled by a dotted line) having a within-group Bray-Curtis similarity of approximately 60%. *Undaria* was first recorded at Diamond Harbour in Spring 1998.

group). One-way ANOSIM revealed significant differences in composition between Cass Bay and the controls (R = 0.535, $p < 0.05$), but all BACI contrasts at Diamond Harbour were non-significant (R = -0.036 - 0.25, $p > 0.05$). Hence, from the two infested sites, opposing conclusions could be drawn from the ANOSIM results regarding the impacts of *Undaria*.

SIMPER analysis revealed that the Cass Bay group was primarily discriminated from the Diamond Harbour/Control group by the relative dominance of *Undaria* and to a

lesser extent *Gelidium*, and the relative paucity of *Cystophora* (Table 3.2). However, *Undaria*'s contribution to the average measure of dissimilarity between the two groups was low (~ 5%). As such, the ordination pattern that resulted when *Undaria* was omitted from the data was strikingly similar to that shown in Figure 3.5, indicating that *Undaria*'s presence in the analysis does not mask other spatio-temporal patterns in the assemblage.

Table 3.2 Summary of SIMPER analysis showing individual and cumulative contribution of the 10 most important taxa (rank 1 = most important) to average measures of dissimilarity between the Diamond Harbour/Control group compared with the other site groups shown in Figure 3.5.

Taxon	Cass Bay group		Spring 1998, DH		Spring 1998, C1		Spring 1998, C2	
	Rank	Percent	Rank	Percent	Rank	Percent	Rank	Percent
Asparagopsis armata			1	6.1	1	4.3		
Aulacomya ater maoriana			6	(3.3)			1	(5.0)
Bryozoa (encrusting)	8	(2.4)			10	(2.3)		
Carpophyllum maschalocarpum	7	2.8					5	3.1
Ceramium spp.			5	3.4				
Chiton pelliserpentis							6	(3.2)
Cladophoropsis herpestica					7	2.6		
Cnemidocarpa bicornuata	9	2.4						
Codium dimorphum	10	(2.3)					10	(2.6)
Colpomenia spp.			8	(2.6)				
Cystophora distenta							8	2.9
Cystophora scalaris	3	(3.5)	9	(2.5)	4	(3.7)		
Ecklonia radiata			7	2.7	3	3.9		
Elminius modestus	6	(2.5)	10	(2.4)	5	(2.7)	7	(3.0)
Gelidium caulacantheum	2	3.6			8	(2.5)		
Hormosira banksii			4	(3.8)	2	(4.3)	2	(4.7)
Micrelenchus sp.	4	2.9						
Myriogramme denticulata	5	2.8	2	4.5			3	3.9
Mytilus galloprovincialis					6	2.6		
Ralfsia verrucosa					9	(2.3)		
Sargassum sinclairii							4	3.4
Trochus viridis							9	2.7
Undaria pinnatifida	1	4.9	3	4.0				
Average dissimilarity (%)		48.4		58.9		48.6		49.7
Cumulative percent contribution		29.9		35.4		31.2		34.4

Note: Numbers outside brackets indicate situations where group discrimination was based on the specified taxon being less dominant in the Diamond Harbour/control group, whereas numbers inside brackets indicate the opposite. DH = Diamond Harbour, C1 = Control 1, C2 = Control 2.

The site ordination trajectories (Figure 3.5) and the temporal trend of Bray-Curtis similarity scores for pairwise comparisons of sites (Figure 3.6) show a convergence in site similarity over time. While the time of greatest divergence of Cass Bay from the controls occurred when *Undaria* was most abundant there in spring 1998, this was also a time when dissimilarity among the two controls was relatively high. A marked spatial separation of Diamond Harbour was also evident at this time (Figure 3.5), coinciding with *Undaria*'s first appearance there. Despite the fact that *Undaria* was reasonably prominent (up to 22% cover), however, more important determinants of the dissimilarity in spring 1998 were the dominance of the rhodophytes *Asparagopsis armata* and *Myriogramme denticulata* (Table 3.2). Hence, differences among sites in spring 1998 appeared to be a general phenomenon, rather than a pattern solely attributable to the proliferation of *Undaria* at infested sites.

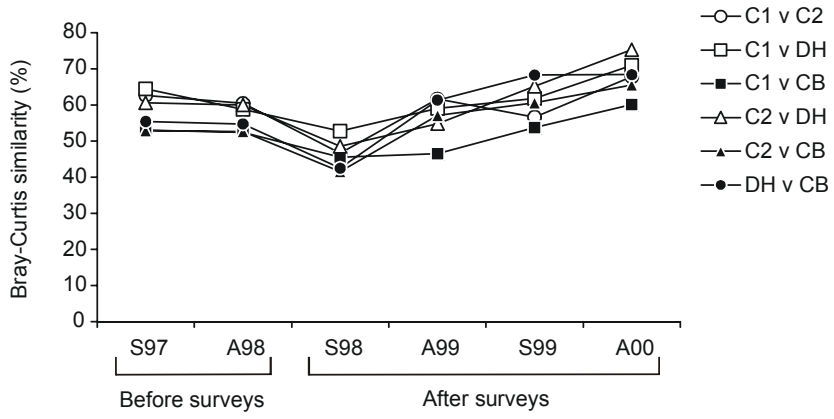

Figure 3.6 Trajectory of Bray-Curtis similarity values for pairwise combinations of the four sites over the six surveys from spring 1997 (S97) to autumn 2000 (A00). C1 = Control 1, C2 = Control 2, DH = Diamond Harbour, CB = Cass Bay. *Undaria* was first recorded at Diamond Harbour in Spring 1998.

3.4 DISCUSSION

Our three year study of low shore assemblages in a sheltered New Zealand harbour has provided no evidence of significant ecological impacts from the invasion of *Undaria*. While impacts could be inferred from the differences between the infested Cass Bay site

and the controls, our findings suggest that these differences reflect underlying spatio-temporal variation rather than effects from *Undaria*.

These results, and apparent effects such as the positive association between *Undaria* cover and algal richness, contradict what might be predicted, but are nonetheless plausible in this situation. For example, the increased canopy cover resulting from *Undaria*'s infestation could enhance sub-canopy low shore algal populations by providing greater shelter from dessication at low tide, as has been discussed in other studies (e.g., Leonard 1999; de Figueiredo et al. 2000). The fact that *Undaria*'s first appearance at Diamond Harbour and its proliferation at Cass Bay were associated with significant changes in the low shore assemblage (especially the algae) at all sites, suggests that *Undaria* was responding to the same favourable environmental variables as other species, thus tracking as opposed to causing the changes observed.

The lack of clear evidence of ecological impacts at *Undaria*-infested sites may partly reflect the fact that these areas already had an assemblage of canopy-forming species, albeit not spatially dominant. Although *Undaria* provided an addition to this, its level of infestation would not have altered the physical structure of the habitat to the extent that might be expected from the formation of an enclosed canopy (e.g., Jenkins et al. 1999; Leonard 1999). It follows that dramatic changes to the structure and function of the resident assemblage would not necessarily be expected. Greater apparent ecological impacts from *Undaria* (Battershill et al. 1998) and marsh plants (e.g., Daehler and Strong 1996; Posey 1988), have been described where the invasions have occurred in relatively barren habitats. Battershill et al. (1998), for example, suggested that there was an increase in sub-canopy species diversity inside *Undaria* patches at shallow subtidal sites that had previously been largely devoid of native macroalgae.

Our conclusion of no appreciable impact, especially for Cass Bay, is weakened by the absence of 'before' data. In contrast, the pre-infestation baseline for Diamond Harbour greatly strengthened the inference we could make about *Undaria*'s impacts at that site. If, for example, the last four surveys at Diamond Harbour were analysed in isolation as part of a control-impact study, a plausible conclusion would have been that the cover of native canopy-forming algae at that site was 'reduced' by *Undaria*. The inherent assumption that underlies this conclusion (and seems quite reasonable) is that four surveys (i.e., two years) of control site data are representative of the natural range in levels of native canopy cover. In fact the native canopy cover significantly increased at

both Diamond Harbour and the control sites from 'before' to 'after' the arrival of *Undaria*.

Our study of *Undaria* has thus reaffirmed the importance of a number of key survey design elements that have been widely promoted for studies of anthropogenic impacts, including the need to establish baselines, and incorporate temporal and spatial replication of control and impact sites. In reality, however, many studies of anthropogenic impact default to less ideal designs. The multiple control vs single impact site approach, for example, is still relatively common in pollution monitoring but can nevertheless provide convincing evidence for (or against) ecological effects (e.g., Smith 1994; Chapman et al. 1995; Roberts and Forrest 1999; Hindell and Quinn 2000).

This more simplistic approach could have been highly misleading in our study of *Undaria*, raising a question as to the necessary survey design requirements for investigating the ecological impacts of *Undaria* or in fact marine invaders generally. If it is assumed that worst-case impacts are of primary interest to managers, then control-impact designs are an appealing prospect, since a site (or multiple sites) of greatest infestation can be targeted and results produced within a short time-frame. The weak inference structure provided by control-impact designs is clearly an issue with invasive species studies, however, especially where infestation levels are patchy as was the case for *Undaria* in this study. When the underlying causes of patchiness are unknown, the validity of any assumption that the control sites are invadable at all, or to the same degree as the impact sites, is questionable. Temporal replication, coupled with an evaluation of ecological changes associated with changing infestation levels over time does not adequately solve this problem. In the same way that the level of invadability may change spatially, it may also change over time as a result of external factors that similarly drive changes in the associated community. In both cases, questions of invadability and ecological impacts are confounded.

Where control-impact designs include temporal replication there are also practical issues to consider. Ensuring that control sites remain uninvaded for the duration of a study may be problematic, since it requires that they be selected from areas beyond the predicted dispersal range of a given invader (unless regular removal of new arrivals is an option). Where this leads to wide spatial separation between the impact and control sites (e.g., the invader has a lengthy planktonic larval stage), the controls are more likely to be subject to different environmental conditions and thus differ markedly from

the impact locations at the outset, or follow different trajectories over time. In the current study, a few *Undaria* plants were discovered in the vicinity of both controls towards the end of the programme, but the founding populations either disappeared again or had not established along our transects before the completion of the study.

While baseline data for potentially infested sites appears critical in invasive species studies, the *a priori* prediction of areas of future worst-case infestation may be particularly difficult, even with a good understanding of invasion processes. In the present study we successfully identified appropriate controls and an area of future infestation using knowledge of *Undaria*'s natural and human-mediated dispersal mechanisms. However, we were probably unsuccessful in describing the seaweed's worst-case effects at a harbour scale, since a subsequent infestation at a nearby reef appeared considerably more significant than at our two infested study sites.

In light of such limitations, it is clear that the current study would have benefited greatly by the inclusion of *Undaria* removal experiments from plots within heavily infested sites. By also including heavily infested plots that were not cleared, this approach would have circumvented the question of the invadability of *Undaria*-free areas both spatially and over time. A spatial comparison of the assemblages of cleared plots with uninfested control plots, and evaluation of their trajectories over time, would have provided a valuable insight into the invadability hence utility of the controls.

A complementary approach, though one that may raise ethical concerns, would be to artificially introduce an exotic species (e.g., perhaps one already established in the general locality) to sites where a baseline had been established. Success is not guaranteed with such approaches, however. Floc'h et al. (1996), for example, inoculated the seabed with *Undaria* spores in areas from which native algae had been cleared, but few sporophytes appeared. Such results are not inconsistent with our own observations or artificial inoculation studies in and around the study area and elsewhere in New Zealand (authors, unpubl. data). While *Undaria* possesses a number of the characteristics of a 'classic' invader (e.g., Fletcher and Manfredi 1995) its invasion patterns do not always reflect this. Even though a single *Undaria* sporophyte can in theory seed a new population, it is not a foregone conclusion that this will happen, or that conditions in the recipient habitat will favour the formation of high density canopy-forming stands.

Clearly, therefore, *Undaria*'s infestation levels and associated effects are likely to vary from place to place, and for reasons that may never be well understood. Hence even with compelling evidence of impacts (or lack of) from one general area or habitat, as *Undaria* spreads to different habitats and invades different assemblage types, the severity of its impacts may change. In terms of managing invasive species like *Undaria*, this caveat must always be kept in mind. Hence while defensible approaches to describing impacts can be developed, and information gathered accordingly, coastal managers and other stakeholders must seriously question the extrapolation of such information to other places and times. On this basis, we suggest that it is necessary to apply the precautionary principle to the management of pest species, and assume 'worst-case' impacts, until the level of scientific uncertainty is reduced. Such an approach should only be applied, however, after an evaluation of the feasibility, costs and benefits of managing the pest in question in relation to other priorities for invasive species.

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Chapter 4

Natural Dispersal Mechanisms of *Undaria*

PREFACE

This chapter presents research into natural dispersal mechanisms in *Undaria*, comprising three pieces of research that contribute to understanding in this field. Two of these (a laboratory investigation of spore viability, and a descriptive study of *Undaria* spread at a field site) were primarily conducted by the second author (with input from myself and others) and contributed to an MSc thesis (Brown 1999; University of Otago). Subsequently I undertook further statistical analyses of the Brown (1999) data and a field-based experimental study of spore dispersal, which tied this earlier work together. From these separate studies, I have developed a synthetic view of natural dispersal mechanisms in *Undaria*, which was subsequently produced as a multi-authored publication as follows:

Forrest BM, Brown SN, Taylor MD, Hurd CL, Hay CH. 2000. The role of natural dispersal mechanisms in the spread of *Undaria pinnatifida* (Laminariales, Phaeophyta). Phycologia 39: 547-553

The text of this chapter is taken verbatim from that publication.

ABSTRACT

The Asian kelp *Undaria pinnatifida* (Laminariales, Phaeophyceae) was first recorded in New Zealand in 1987 and has since spread via shipping traffic and other vectors to a number of ports and harbours. Here we report the results of laboratory and field studies devised to assess the potential for natural dispersal of *Undaria* from a founding population. Under laboratory conditions, > 90% of *Undaria* spores were viable in seawater for at least 5 days, with some viable after 14 days. Spores artificially released into a tidal current resulted later in sporophytes appearing on artificial surfaces positioned 10 m down-current of the release point. Field monitoring of a founding population within the Marlborough Sounds, New Zealand, suggested that natural populations spread at least 100 m yr^{-1}. Reasons for the differences between the dispersal distances of the artificially released spores (10 m) and natural populations (100 m) are discussed. We propose that spore dispersal from fixed stands of *Undaria* results primarily in short-range spread (metres to hundreds of metres), with dispersal of fragments or whole sporophytes facilitating spread at scales of hundreds of metres to kilometres.

4.1 INTRODUCTION

Undaria pinnatifida (Harvey) Suringar, a laminarian seaweed native to cold temperate coastal areas of Japan, Korea and China (Akiyama and Kurogi 1982), was first discovered in Wellington Harbour, New Zealand (Figure 4.1), in 1987 (Hay and Luckens 1987). The appearance of a different morphotype at Timaru in the same year suggests separate transoceanic introductions (Hay and Villouta 1993). *Undaria*'s subsequent spread to approximately 15 additional ports and harbours around New Zealand (Figure 4.1) highlights the importance of human-assisted transport between regions. Concern has been expressed over the spread of *Undaria* because of its potential impacts on important natural ecosystems and fisheries, and the possibility that it could become a fouling pest. As such, a proposal for a national pest management strategy for *Undaria* (Sinner et al. 2000) is presently under consideration. In order to aid management, an understanding of the seaweed's potential for spread is essential.

The wide variety of human vectors by which *Undaria* could be inadvertently transported have been discussed previously and include: vessel traffic of all types (Hay 1990; Sanderson 1990; Casas and Piriz 1996; Fletcher and Manfredi 1995; Floc'h et al. 1996); transfer of contaminated mariculture stock and equipment (Pérez et al. 1981; Boudouresque et al. 1985); and less obvious mechanisms such as fishing nets and boat anchors (Sanderson 1997). Hay (1990), for example, provides compelling evidence for the spread of *Undaria* within New Zealand by vessel movements between ports and harbours. The risk of *Undaria* being introduced to a new location via vessels and other

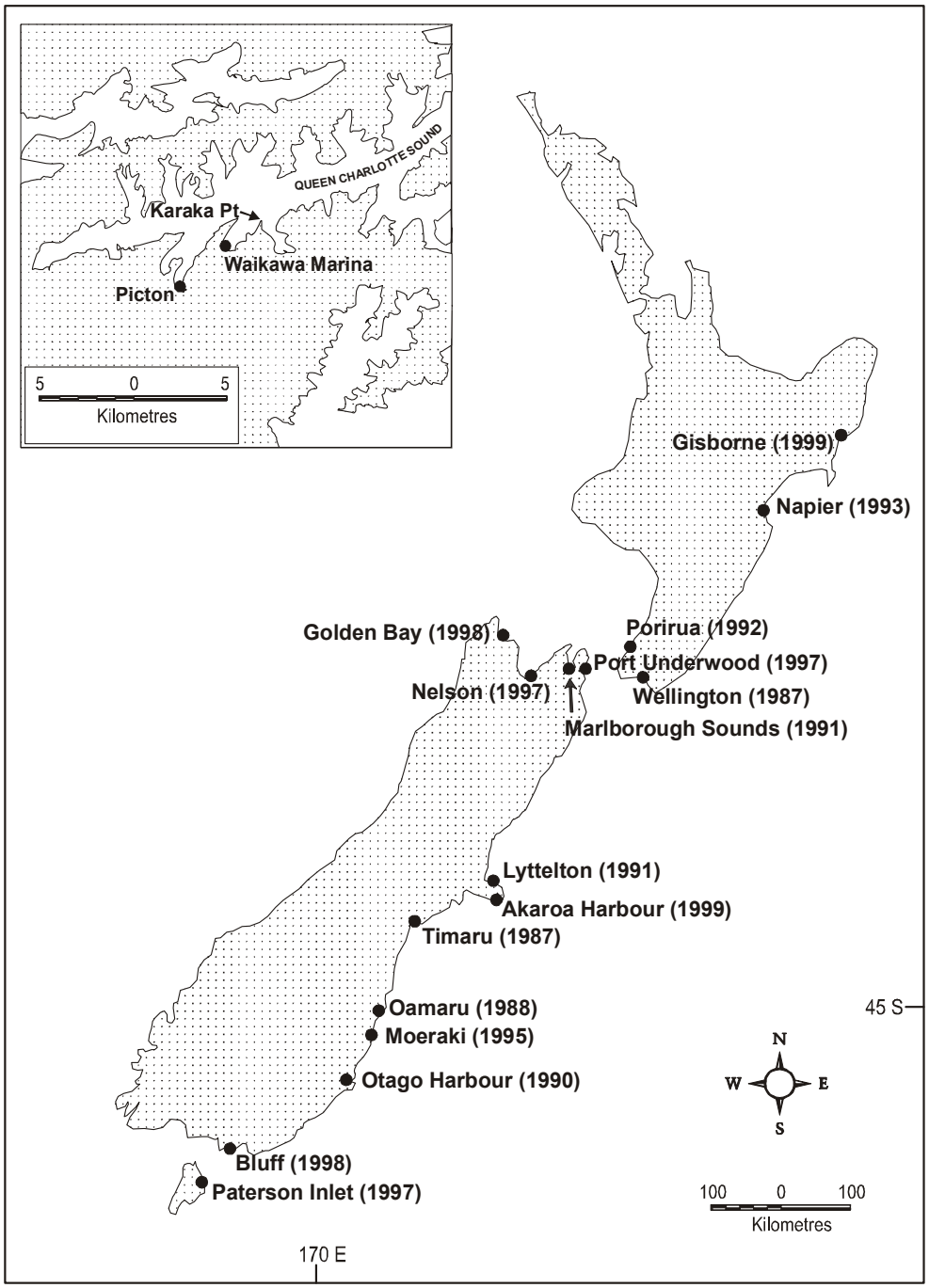

Figure 4.1 Regional distribution of *Undaria* in New Zealand, indicating the year it was first recorded at each location. The inset shows the location of Karaka Point in the Marlborough Sounds where field work was conducted.

vectors will depend on the stage of the life-cycle transported, survivorship *en route*, and attributes of the vector such as the length of time it remains in a recipient region. For example, when a fouled vessel remains in a suitable recipient region for a short time (e.g., days), the transport of mature sporophytes would be expected to pose a greater risk than transport of gametophytes.

Although the role of human mechanisms in the transport of *Undaria* is well recognized, the relative importance of natural dispersal is not well understood. Natural dispersal occurs following the release of motile spores from the sporophyte. The distance over which spores travel before settling will largely be determined by their viability and behaviour and the speed of ambient water currents (Hoffman and Camus 1989; Santelices 1990; Norton 1992; Reed et al. 1992). Descriptive studies of *Undaria* suggest that spores settle within metres of the parent sporophyte (Suto 1950; Arakawa and Morinaga 1994a). However, the reported annual spread of *Undaria* populations ranges from hundreds of metres to several kilometres (Hay 1990; Casas and Piriz 1996; Sanderson 1997; Brown 1999). This raises the possibilities that: (1) spore dispersal is greater than suggested by Suto (1950) and others; (2) natural dispersal mechanisms may be operating in addition to spore release, e.g., drifting sporophytes; and (3) human-assisted dispersal of *Undaria* is also important at local scales.

The objective of this study was to further understand the role of natural dispersal in the spread of *Undaria*. We describe a laboratory experiment conducted to determine how long spores remain viable when kept artificially suspended in seawater, and a field experiment that describes the distance of spore dispersal from a point source. We also monitored the range extension of *Undaria* from a discrete population where natural spore dispersal was hypothesized to be the primary means of spread.

4.2 MATERIALS AND METHODS

4.2.1 Spore viability

Ten *Undaria* sporophylls were collected from Nelson (see Figure 4.1) on 21 August 1998. The sporophylls were rinsed in chilled seawater (filtered to 0.35 μm), sterilized for 1 min in 0.5% commercial bleach, then rinsed again, using a method modified from Moigne et al. (1991). Sporophylls were wrapped in damp paper towels and left in the dark for 4 h. To stimulate spore release, the sporophylls were reimmersed in 10 separate glass beakers, each filled with 400 ml of filtered seawater. Each of the

resulting spore solutions was filtered through a mesh sieve (11 μm) to remove mucilage and other debris. Following microscopical determination of spore concentration, each solution was diluted with filtered seawater to a concentration of approximately 500 spores ml^{-1} and 800 ml of the resultant solution was transferred to each of 10 one-litre conical flasks. In order to agitate the spore solution and prevent settlement, flasks were placed on a rotary shaker (at 150 rpm) in a culture room at 18°C, with a 12 : 12 h light : dark regime and a photon flux density (PFD) of 90–100 μmol m^{-2} s^{-1} provided by two Osram 18W cool-white fluorescent bulbs.

At intervals of 2 h, then 1, 3, 5, 7, 10 and 14 days after preparation of the spore solutions, a 2.5 ml aliquot of solution was removed from each of the 10 conical flasks and transferred to a separate well in a sterile 12-well Falcon™ (Beckton Dickson and Co., New Jersey) tissue culture dish in the same culture room. Each well contained 2.5 ml of double strength standard seaweed medium, modified from the F2 medium of Guillard (1975). After three days, when all propagules had stopped swimming, the number of spores per square centimetre that had settled on the bottom of each well in the Falcon™ dish was counted. Counts were repeated after a further 10 days. Propagules were considered viable if they had undergone cell division after the 10-day period (i.e., after a total of 13 days in culture in the Falcon™ dish).

Percent viability was determined from the number of viable propagules after 13 days in culture, as a proportion of the 3-day count. Normal probability and scatter plots of residuals for each time period were examined to check assumptions of normality and homogeneity of variance of error terms. Data were arcsine($\sqrt{x}$) transformed to meet these assumptions. Changes in percent viability over time were analysed using repeated measures ANOVA in Systat 7.0 (Systat 1997). Percent viability could not be assessed quantitatively for spores kept suspended in the conical flasks for more than seven days, because the propagules formed clumps that prevented accurate counting of individuals. Further detail is described in Brown (1999).

4.2.2 Spore dispersal

Spore dispersal was investigated in a field experiment initiated on 11 August 1998 in the Marlborough Sounds (Figure 4.1), in a location where *Undaria* was well established. Spores were released from a point source in a measured water current. The distance and pattern of spore dispersal were inferred from the appearance (four months

later) of sporophytes on ropes of the kind used to catch mussel spat, which were positioned at eight distances up to 200 m down-current of the release point.

The spore settling apparatus was positioned in a 'ladder' array, consisting of 2 × 200 m parallel floating ropes that were spaced 2 m apart using wooden battens and aligned parallel to the prevailing tidal current (Figure 4.2). The ladder was anchored at its up-current end to a moored pontoon, which was used as a working platform. The wooden battens were suspended above the water surface, using polystyrene blocks, so that they did not interfere with spore dispersal. From eight of the battens – corresponding to distances of 1, 2, 5, 10, 25, 50, 100, and 200 m down-current of the spore release point – triplicate 2 m lengths of weighted mussel spat rope were hung at 0–2 m and 2–4 m depths. The level of replication (three ropes per depth per distance) was chosen to minimize the likelihood that the settlement ropes would interfere with water flows and hence alter spore concentrations in down-current areas. The deeper ropes were included to examine the hypothesis that spores would sink to progressively greater depths with increasing distance from the spore source. Following spore release, vertical profiles of temperature and salinity were measured at 0.5 m depth intervals at the up-current end of the ladder, to determine the presence of any significant water stratification that could affect the vertical pattern of spore dispersal.

To induce spore release, 50 partially dehydrated *Undaria* sporophylls (totalling 1.94 kg wet weight) were added to each of two bins containing 40 litres of ambient seawater. After 10 min of manual stirring, the resulting spore solutions were subsampled for microscopical determination of spore concentration. The sporophylls were then removed from the fish bins and the spore solutions were poured onto the water surface at the up-current end of the ladder over a period of 5 min. Care was taken to ensure a uniform distribution of the spore solutions across the 2 m width of the ladder (Figure 4.2). An estimated 2.7×10^{10} spores were discharged in total.

A 'holey sock' drogue was deployed on each side of the ladder (adjacent to the spore release point) at the time of spore release, to gauge the direction and speed of surface water movement, thereby providing an estimate of the speed and direction of spore movement. Each drogue consisted of a cylindrical nylon tube (2 m long) reinforced with stainless steel rings and attached at the top to a small spherical float. This design is known to follow current patterns accurately (Sombardier and Niiler 1994).

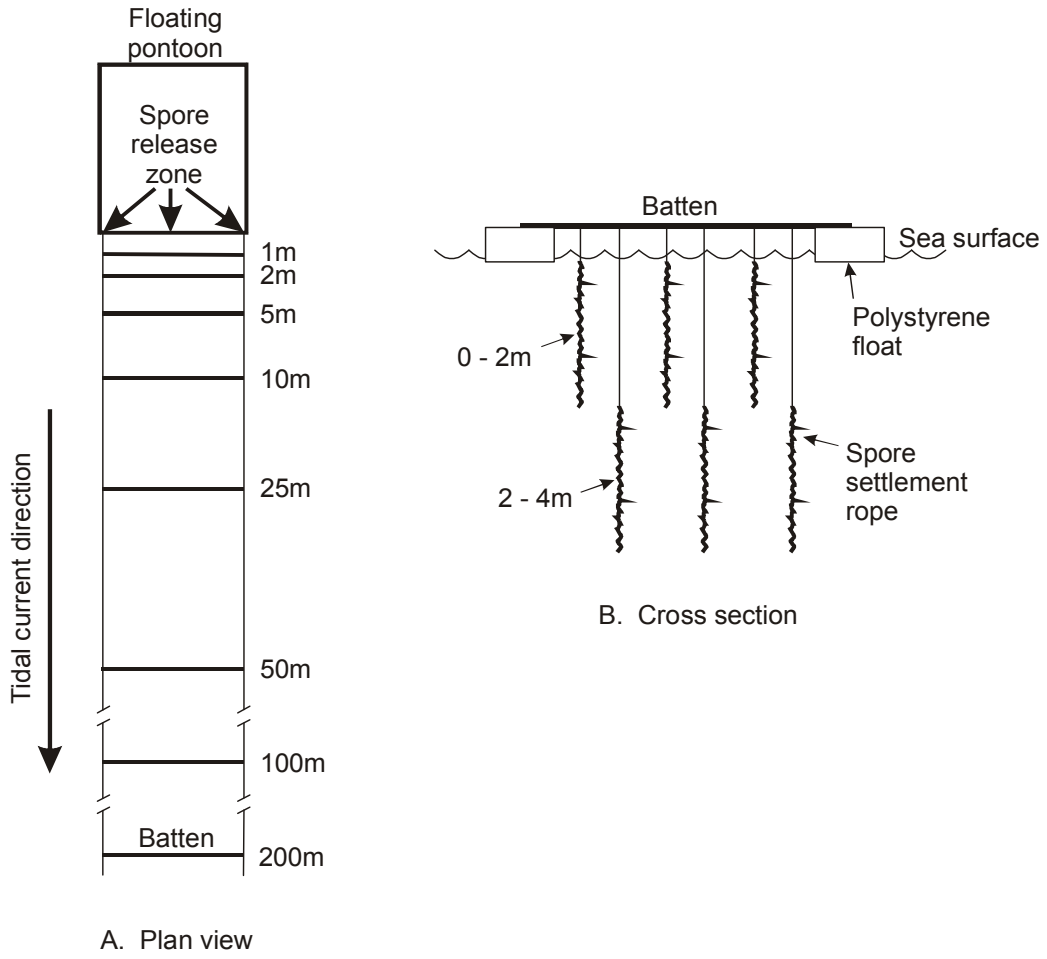

Figure 4.2 Schematic layout of the ladder array used in the spore dispersal experiment (not to scale). A. Plan view showing 2 m wide battens. Each of the eight distances indicates metres from the spore source. B. Cross section of a 2 m wide batten on polystyrene floats. Two metre ropes were used as settlement surfaces for *Undaria* spores and were suspended in triplicate at depths of 0–2 m and 2-4 m.

One hour after the completion of spore release, the spat ropes were moved from the ladder and hung around the sides of the pontoon (spaced at ~ 0.5 m) for settled spores to grow into visible sporophytes. At the same time, we included four unseeded spat ropes and four ropes artificially seeded in the spore solution. The unseeded ropes were included as a control for any subsequent seeding from wild *Undaria* in the wider embayment. The seeded ropes were included to assess whether *Undaria* could be successfully ongrown on ropes suspended off the pontoon, in the event that we failed to

detect any sporophytes on the experimental ropes, and to evaluate the importance of water depth during growth of the sporophytes.

Sporophytes appearing on the ropes were counted in December 1998, four months after the spore release experiment was conducted. No sporophytes were recorded on ropes from the 2–4 m depth interval, and beyond 25 m from the release point the drogue tracks suggested that the path travelled by spores deviated from the alignment of the ladder. Hence, only those sporophyte counts made for experimental ropes from the 0–2 m depth interval at distances of 1–25 m from the release point are included in graphical displays and statistical analyses. These data required $\log(x + 1)$ transformation to meet assumptions of normality and homogeneity of variance of error terms. One-way ANOVA with Tukey's HSD was carried out using Systat 7.0 (Systat 1997) to test for differences in sporophyte counts at different distances from the spore source. Pearson correlation coefficients were calculated using Systat 7.0 to further explore the relationship between *Undaria* counts (transformed) and distance.

4.2.3 Spread from discrete populations

To compare results from our laboratory and field experiments with the rate of natural spread of an *Undaria* population at a local scale, we monitored the annual change in population density and distribution at Karaka Point, a wave-sheltered locality in the Marlborough Sounds (Figure 4.1). We assumed that natural dispersal from the existing population at Karaka Point (which had probably been established for several years) would be the primary means of *Undaria*'s spread, although not necessarily the only means, since natural or human-assisted dispersal must have introduced the seaweed to the area initially. The nearest recorded *Undaria* population was approximately 2 km from the study site.

At the beginning of the monitoring programme, *Undaria* sporophytes were distributed around the promontory of Karaka Point. Habitats ranged from bedrock to cobbles and boulders among soft sediments. Surveys of the promontory and adjacent areas were conducted by SCUBA in November 1997 and November 1998 (Brown 1999). A stratified random sampling design was used, with strata based on habitat characteristics. Within each stratum, counts of *Undaria* were made within 12 × 1 m^2 quadrats placed randomly along the 1 m isobath within rocky areas.

In strata where sporophytes were sparse in 1997 and which could not be reliably sampled using quadrats, total counts were made over the entire area. In 1998, an increase in sporophyte density in these areas meant that random quadrats were used instead of total counts. In order to illustrate the main trends in the spread of populations, sporophyte counts were pooled and assigned to one of four density categories as follows: > 0–1, > 1–5, > 5–10, and > 10 sporophytes m^{-2}.

4.3 RESULTS

4.3.1 Spore viability

Mean spore viability was > 90% during the first 5 days in suspension, and 68% on day 7 (Figure 4.3). There was no significant difference in spore viability over time (F = 0.985, p = 0.43, ν = 4, 32). Although viability could not be assessed quantitatively after 7 days because of clumping within the conical flask cultures, propagules were still capable of forming gametophytes after 14 days.

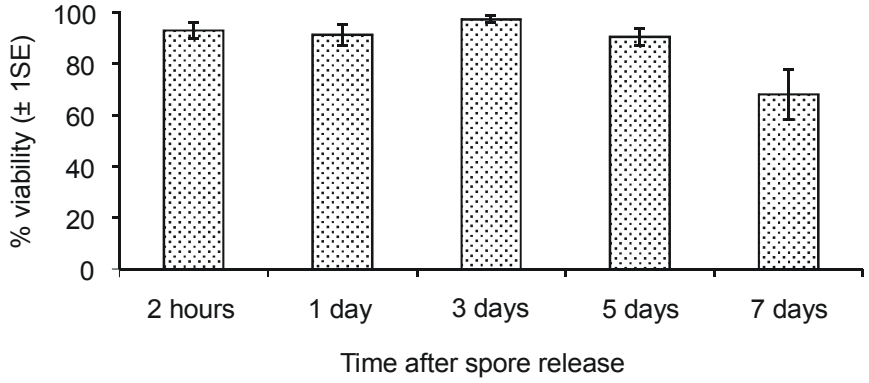

Figure 4.3 Mean percentages (± SE, n = 10) of *Undaria* propagules remaining viable following suspension in seawater for different periods following spore release.

4.3.2 Spore dispersal

For the 0–2 m depth interval, a mean of 12 sporophytes per rope was recorded 1 m from the spore release point, decreasing to approximately two per rope at 10 m (Figure 4.4). Most sporophytes were clustered at the water surface, with none recorded deeper than 0.75 m. No sporophytes were recorded on the ropes 25 m from the spore release point.

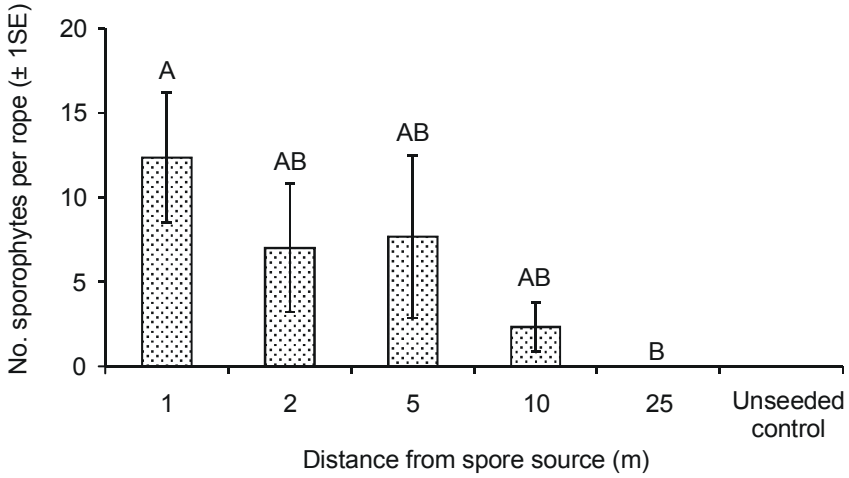

Figure 4.4 Mean number (± SE) of *Undaria* sporophytes on settlement ropes at five distances from the spore source at the 0–2 m depth interval (n = 3). Columns sharing a letter indicate groupings from the ANOVA that were not significantly different from each other (Tukey's HSD test, $p > 0.05$).

There was a significant difference in the number of sporophytes between distances (F = 3.939, $p = 0.036$, $\nu = 4, 10$), although the only significant pairwise comparison of mean values was between the 1 m and 25 m distances ($p = 0.028$). There was a highly significant decrease in sporophyte numbers with increasing distance from the spore release point ($r = -0.749$, $p < 0.01$, $\nu = 13$).

Unseeded control ropes had no visible sporophytes. Seeded control ropes had > 1000 sporophytes per rope with no apparent change in density with depth. Salinity levels gradually increased from 28.6‰ at the water surface to 30.5‰ at 2 m depth. The water temperature was uniformly 12.5°C across this depth range. Drogue speeds were similar down both sides of the ladder, with a mean value of 7.9 cm s^{-1}.

4.3.3 Spread from discrete populations

In 1997 at Karaka Point, the greatest *Undaria* densities (> 10 sporophytes m^{-2}) occurred in two main stands on the western and eastern side of the promontory, with lower densities around the tip of the promontory (Figure 4.5). These established stands were largely maintained through to 1998, although sporophyte numbers increased in some

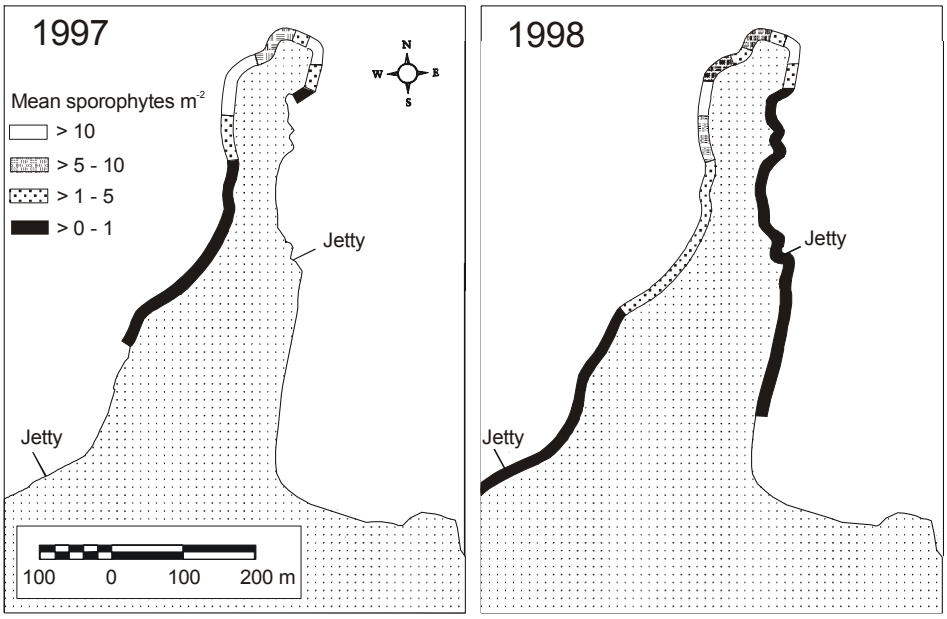

Figure 4.5 Distribution of *Undaria* at Karaka Point in November 1997 and November 1998. Density categories are based on mean values within each sampling stratum. See text for details.

strata and decreased in others. In 1997, densities south of the main western stand decreased from > 10 sporophytes m^{-2} to < 1 sporophyte m^{-2} within a distance of only 65 m. One year later, densities of 1–5 sporophytes m^{-2} occurred up to 260 m south of the main western stand (Figure 4.5), and scattered individuals (< 1 sporophyte m^{-2}) extended at least 300 m further than the 1997 population boundary.

The pattern of spread on the eastern side of the promontory was similar to that on the western side. Although the zone of intermediate density (1–5 sporophytes m^{-2}) immediately south of the main eastern stand did not change appreciably between 1997 and 1998, low densities of < 1 sporophyte m^{-2} were present 500 m further south than the 1997 population boundary (Figure 4.5).

A more detailed consideration of changes in sporophyte density and dispersal distance to the south of the promontory must take into account two further factors. First, the extent of rocky habitat decreases to the south, and *Undaria* plants within the low density (< 1 sporophyte m^{-2}) area were clustered where the habitat was most suitable. Thus low

densities likely reflect habitat availability as well as dispersal distance. Second, sporophytes appeared in 1998 on two private boat jetties within the survey area (see Figure 4.5). On the western side of the promontory, densities on the jetty were greater than on adjacent seabed. The possibility that the jetty populations reflect human-assisted spread of *Undaria* confounds our interpretation of natural dispersal. This point is considered further below.

4.4 DISCUSSION

Our observations that, under laboratory conditions, the majority of *Undaria* spores are viable in seawater for at least five days, with some viable after 14 days, are consistent with the duration of spore viability for other members of the Laminariales and other seaweed species (Hoffmann and Camus 1989; Santelices 1990; Reed et al. 1992). There is evidence that spore viability varies seasonally (Hoffmann and Camus 1989); hence it might be expected that the viability of *Undaria* spores would have differed if our experiment had been carried out at a different time of year. However, research carried out at a nearby location showed that germination of *Undaria* spores (following laboratory-induced spore release) was high (> 80%) for most of the year (Brown 1999), suggesting that seasonal differences may not be significant in our study area.

In the dispersal experiment, sporophytes appeared no further than 10 m from the spore source, providing direct support for previous suggestions of short-range spore dispersal for *Undaria* (Suto 1950; Arakawa and Morinaga 1994a). The decrease in sporophyte numbers with increasing distance from the spore source is probably due to spore dilution by the ambient water mass. Increased dilution of propagules as a function of time and distance greatly reduces the likelihood that individual spores, and hence gametophyte germlings, will settle next to another of the opposite sex (Norton 1992). Drogue speeds suggested that it would take only a few minutes for a spore to travel from the spore source to ropes located 25 m away. Thus it is highly unlikely that spore settlement and subsequent development is limited by viability or a decrease in the ability of spores to attach over time (Suto 1950), to the extent that would be necessary to explain the decrease in sporophyte numbers with distance.

The aggregation of sporophytes at the water surface on the experimental ropes appears to reflect a real pattern of spore dispersal rather than an artefact resulting from the differential development and growth of sporophytes (e.g., as a result of greater light

near the water surface), since the seeded control ropes had uniformly high densities of sporophytes over their 2 m length. Surface aggregation could be explained by limited sinking of the spores during their short travel time, given that the spore solution was added to the water surface, or by entrapment of the spores by water surface tension (which we observed in the laboratory). Alternatively, since *Undaria* spores are motile and swim at speeds of 3-8 mm s^{-1} in laboratory cultures (Suto 1950), it is possible that they actively moved towards or remained at the water surface; however, we consider this a less likely explanation of sporophyte aggregation.

Results from the surveys at Karaka Point showed expansion of the population boundaries by hundreds of metres over the twelve months between surveys. This is of the same order as the rates of spread of 110 and 140 m yr^{-1} observed by Brown (1999) in two isolated *Undaria* populations sampled elsewhere in the Marlborough Sounds over the same period. It is also consistent with our unpublished observations of *Undaria*'s spread in other parts of New Zealand, where there is no identifiable human transport factor. The pattern of decreasing sporophyte density with increasing distance south of Karaka Point suggests spread by natural means. While the presence of *Undaria* on the jetties on each side of Karaka Point in 1998 may reflect inoculation with spores from a visiting vessel, it may also reflect the seaweed's propensity for growing on artificial or suspended structures (Hay 1990; Floc'h et al. 1996). Even if the distribution at Karaka Point in 1998 were partly due to human-assisted introduction to the jetties, the results would still suggest increments of natural spread in excess of 100 m yr^{-1}.

4.4.1 Comparison of laboratory and field investigations

Even though Karaka Point is subject to relatively weak tidal currents, this study has shown that *Undaria* spores can remain viable sufficiently long to disperse and settle over distances of at least a few hundred metres. In fact, given suitable hydrographic conditions, the duration of spore viability could facilitate coastal dispersal over distances of kilometres and possibly tens of kilometres. However, our spore dispersal experiment suggests that dilution is likely to be an important factor in determining the outer limit of population spread from a discrete spore source. This is consistent with other studies of spore dispersal in seaweeds (Hoffman 1987; Reed et al. 1988; Kendrick and Walker 1995) and is supported by the pattern of decreasing sporophyte density from the populations at Karaka Point.

By increasing the number of propagules, the effects of dilution may be reduced and new sporophytes may appear further from the spore source, as was shown for *Macrocystis* by Anderson and North (1966). In this regard it may seem counterintuitive that the distance of new *Undaria* sporophytes from the spore source at Karaka Point was an order of magnitude more than in our dispersal experiment, where an artificially enhanced number of spores was released. There are a number of possible explanations for the magnitude of this difference. In the experiment the once-only release of spores into a tidal current would provide only one chance for them to attach, upon contact with a settlement surface, before being swept away. In contrast, spore release may occur year-round in some natural populations of *Undaria* in New Zealand (Hay and Villouta 1993; Brown 1999) and spores may have multiple opportunities to settle. Also, while water currents during the experiment (approximately 7.9 cm s^{-1}) were suitable for spore adhesion, they were higher than the optimum of approximately 3 cm s^{-1} reported for *Undaria* by Arakawa and Morinaga (1994b). In the natural situation, quiescent water (e.g., while the tide is turning) would reduce any negative effects of water movement on settlement (Vadas et al. 1992) and may act as a cue for spore release (Pearson et al. 1998). Such factors increase the likelihood that, in the natural situation, there will be a suitable 'window of opportunity' for the successful development of sporophytes.

The orientation and amount of available settlement surface may also explain the magnitude of the difference between the natural spread of populations at Karaka Point and the dispersal experiment. Spores may settle at greater densities on a horizontal surface (as in the natural situation) than on a vertical surface (as in our experiment), as has been described for *Undaria* (Arakawa and Morinaga 1994b) and other seaweeds (Reed et al. 1988). Furthermore, the experimental settlement ropes comprised only 5% of the cross-sectional surface area within the spore dispersion path. If instead 100% of the cross-sectional area was available for settlement, the recorded sporophyte numbers (Figure 4.4) could be multiplied by a factor of 20 (assuming a similar trend in density with distance), suggesting that sporophytes could have appeared at least 200 m from the spore release point before dilution became limiting. This is of the same order as the spread described for Karaka Point.

4.4.2 Multiple strategies for the natural spread of *Undaria*?

Even though spore dispersal alone may explain the scale of spread at Karaka Point, other natural mechanisms, such as the drift of sporophytes or fragments that release

spores, may be equally important. Unattached sporophytes and sporophyte fragments were noted in the Karaka Point survey area during the course of this study, particularly in early summer when sporophytes began to senesce and were easily displaced by waves generated naturally and by vessel traffic. In areas of strong water current in Tasmania, drifting sporophytes are believed to facilitate the dispersal of *Undaria* over scales of up to 10 km (Sanderson 1997).

Our findings also suggest another potential mechanism of longer range dispersal. High densities of spores near the water surface (e.g., because of surface tension) may lead to clumping, as was observed in the laboratory. Clumping during dispersal could increase the likelihood of male and female gametophytes maturing in close proximity, promoting fertilization and the initiation of a new sporophyte generation; Santelices (1990) has discussed the potential benefits of dispersal of aggregated seaweed propagules. However, while clumping of spores may circumvent problems caused by the physical 'dilution' of propagules, other factors may become limiting during long-range dispersal. For example, grazing in the water column may reduce propagule densities, or propagules may lose their ability to attach over time. Suto (1950) notes that *Undaria* spores lose the ability to attach after several hours, although attachment was evident after fourteen days in the present study.

Hence, *Undaria* may exhibit multiple dispersal strategies, as has been noted for many other macroalgae, including invasive species such as *Sargassum muticum* (Hoffmann 1987; Norton 1992). Spore dispersal in *Undaria* is probably a key mechanism for short-range (metres to hundreds of metres) spread from fixed stands. Short-range dispersal would maintain established stands of sporophytes and increase densities adjacent to such stands, as observed at Karaka Point. Dispersal via whole sporophytes or fragments, and possibly via spore clumping, is likely to be particularly important in range extensions of *Undaria* over scales of hundreds of metres to kilometres, with episodic or chance events potentially leading to spread at even greater scales (Reed et al. 1988). Subsequent short-range dispersal of spores around the more distant and scattered 'frontier' individuals at the boundary of the population would establish a new sporophyte stand by gradual infilling and enhance the propagule supply for further spread. The multiple dispersal strategies for *Undaria* described here may play an especially significant role in facilitating rapid spread within regions where human-mediated transfer of the seaweed is limited.

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Chapter 5

Aquaculture Pathways for *Undaria*

PREFACE

This chapter describes aquaculture pathways that have the potential to spread *Undaria* around New Zealand. This information was collected for the Ministry of Fisheries marine biosecurity group (now part of Biosecurity New Zealand) prior to 2002 and is extracted from relevant sections of the following Cawthron technical report:

Forrest BM, Blakemore KA. 2002. Inter-regional marine farming pathways for the Asian kelp *Undaria pinnatifida*. Cawthron Report 726, Cawthron Institute, Nelson, New Zealand. 26p

This work, along with information on other pathways for the spread of pest species around New Zealand (from Dodgshun et al. 2004), will be published in an abridged form in 2007 in a Department of Conservation technical series report. My co-author assisted with this work by obtaining information from regional councils and other agencies on water space allocated for marine farming in their regions.

The purpose of the project was to identify high risk marine farming transfer pathways where vector control measures (as described in Chapters 6 and 7) might help to avoid or reduce the spread of *Undaria*. Since the time the work was undertaken, the situation with respect to *Undaria* (e.g., geographic distribution) has changed, hence much of the analysis and specific recommendations regarding pathway management no longer apply. However, because it was intended at the time of thesis enrolment that the Forrest and Blakemore (2002) report would be the basis of a chapter (given that previous work was allowed; see preface to thesis on p. ii), I have extracted much of the text verbatim from that report. An addendum is included at the end of the Chapter (Section 5.5) to clarify the present situation and the describe extent to which the analysis, recommendations and criteria for management remain relevant.

ABSTRACT

This work describes the distribution of the Asian kelp *Undaria pinnatifida* in relation to marine farming areas in New Zealand, the types of marine farm activities that might transfer *Undaria*, and the principle pathways along which these activities occur. In broad terms, pathways where management of *Undaria* spread is most desirable are those from any *Undaria*-infested marine farming area to any present or future marine farming area that is *Undaria*-free (or where *Undaria* is under control), and where management measures are not undermined by the natural spread of the seaweed or by its uncontrolled spread via non-marine farming vectors (e.g., recreational vessels).

For areas like the Marlborough Sounds, we suggest that within-region management of *Undaria* pathways will largely be futile - while there may be parts of the Sounds that are *Undaria*-free, such areas cannot be identified with current knowledge and without considerable ongoing effort to monitor the seaweed's distribution. We also assume that most localities suitable for *Undaria* at a regional scale will be vulnerable to infestation via natural dispersal or non-marine farming vectors. Hence our discussion of marine farming pathways and their management focuses on broad regions only. We identify three current mussel farming-related pathways where the efficacy and feasibility of managing vectors should be further evaluated.

We also suggest that *Undaria* management should be considered on a case by case basis where any of the following situations arise through future industry development: (i) The infestation of *Undaria*-free marine farming areas whose current pathways lead to uninfested areas or areas where *Undaria* is managed; (ii) The development of new pathways from infested areas to existing marine farming areas that are currently uninfested; and (iii) The development of new pathways from infested areas to new marine farming areas that are currently uninfested. In considering management of these and any other pathways, it should be recognised that measures will only be effective if they have the support of affected marine farmers and other vector owners/operators. This support may be more easily gained for management measures that are applied equally across all vectors and have generic biosecurity benefits.

5.1 INTRODUCTION

This work describes aquaculture pathways that have the potential to spread *Undaria* around New Zealand and has been undertaken as part of government-funded investigations into management options for the Asian kelp *Undaria pinnatifida* in New Zealand. We: (i) summarize the type and location of existing and proposed marine farming activities in New Zealand; (ii) describe the recorded distribution of *Undaria* with respect to these marine farming locations; (iii) describe, in general terms, the type of transfer activities that occur within the marine farming industry and their potential to translocate *Undaria*; (iv) discuss the regional scale across which vector management might be feasible, and identify the main pathways of marine farming activities operating at this scale around New Zealand; and (v) discuss key marine farm pathways where

Undaria management measures are desirable or require further consideration. Particular emphasis is given to those pathways that lead from *Undaria*-infested to uninfested areas, with a focus on the links between the main aquaculture regions, for reasons described below.

5.2 MARINE FARMING AREAS IN RELATION TO *UNDARIA* DISTRIBUTION

5.2.1 Background

Marine farming activity is concentrated in a number of regions around New Zealand (Figure 5.1). The main crops are Greenshell™ mussels (*Perna canaliculus*) and Pacific oysters (*Crassostrea gigas*), with long-line mussel farming being by far the dominant sector. Other established sectors include farming of sea-cage salmon (*Oncorhynchus tshawytscha*) and paua (*Haliotis iris*), with a number of other small-scale or experimental species under evaluation. The development of the industry includes allocation of new coastal water space in excess of 10,000 hectares, mainly for mussels. Some of the proposals involve developments in parts of the New Zealand coastline (e.g., Bay of Plenty, Hawke Bay, South Westland) where there is no aquaculture at present (Figure 5.1).

The reported distribution of *Undaria* around New Zealand (Figure 5.2) shows that the seaweed is established in most ports and harbours along the east coast from Gisborne to Stewart Island. A comparison of Figure 5.1 and Figure 5.2 reveals that *Undaria* is established within all of the main marine farming regions within this geographic range, and has also been recorded in the Firth of Thames. Information pertinent to the main marine farming regions is as follows:

- **The Firth of Thames (area C of Figure 5.1):** *Undaria* was reported on a mussel farm in the eastern Firth of Thames in May 2002, and infected culture lines were removed. Our current understanding is that this area is *Undaria*-free.
- **Golden Bay (area E):** *Undaria* is established on mussel farms in Wainui Bay in the south of Golden Bay and Collingwood in the north, but is not thought to be present in natural habitats.
- **The Marlborough Sounds (area F):** *Undaria* is widespread throughout the Sounds from Croisilles Harbour in the west to Port Underwood in the east.

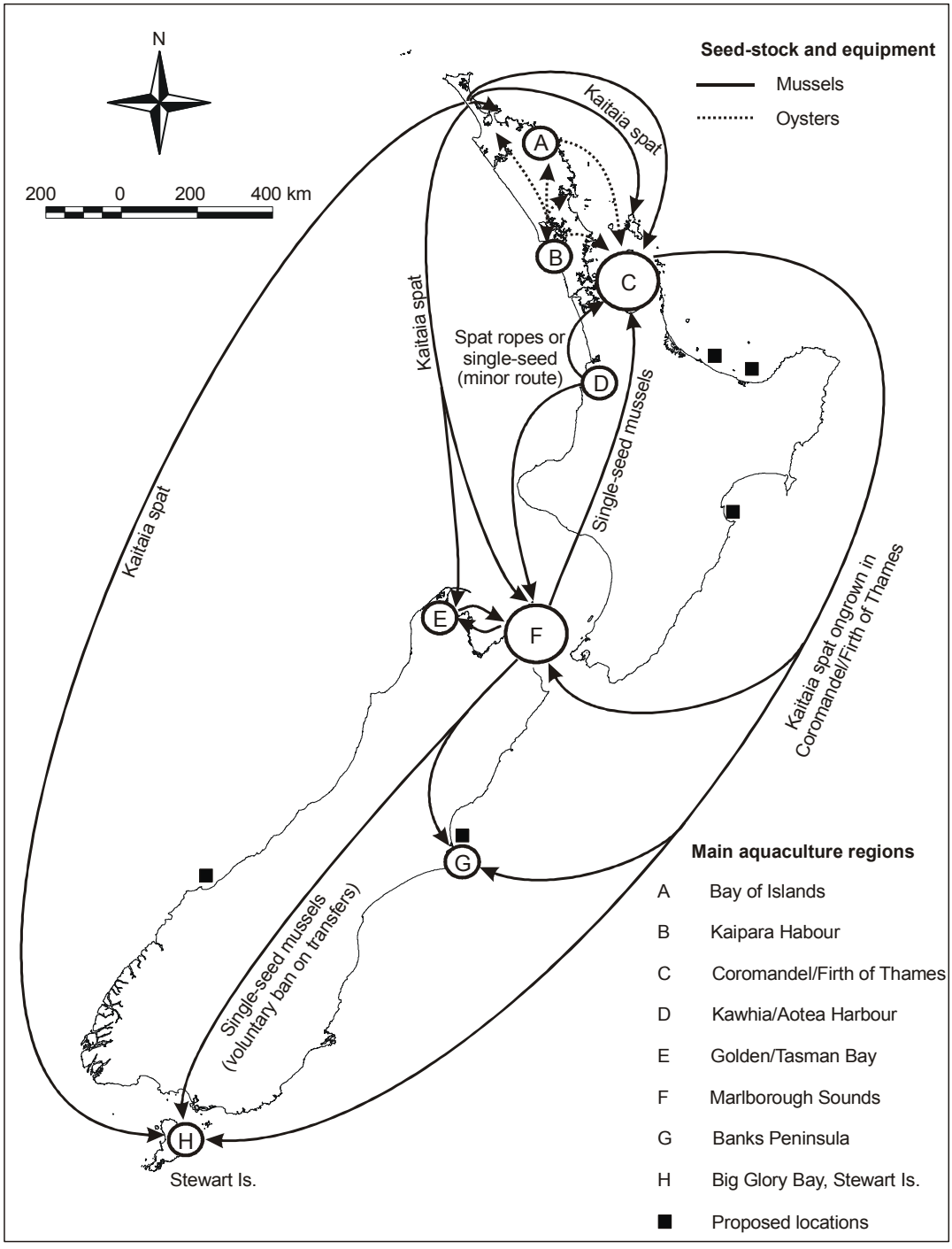

Figure 5.1 Existing (A – H) and proposed marine farming regions, showing the main pathways of equipment/vessels, Kaitaia mussel spat, seed-mussels and oysters around New Zealand. Bubble size for areas C and F indicates the greater intensity of aquaculture in these regions relative to other parts of New Zealand (Figure collated from Figs 1 and 4 in Forrest and Blakemore 2002).

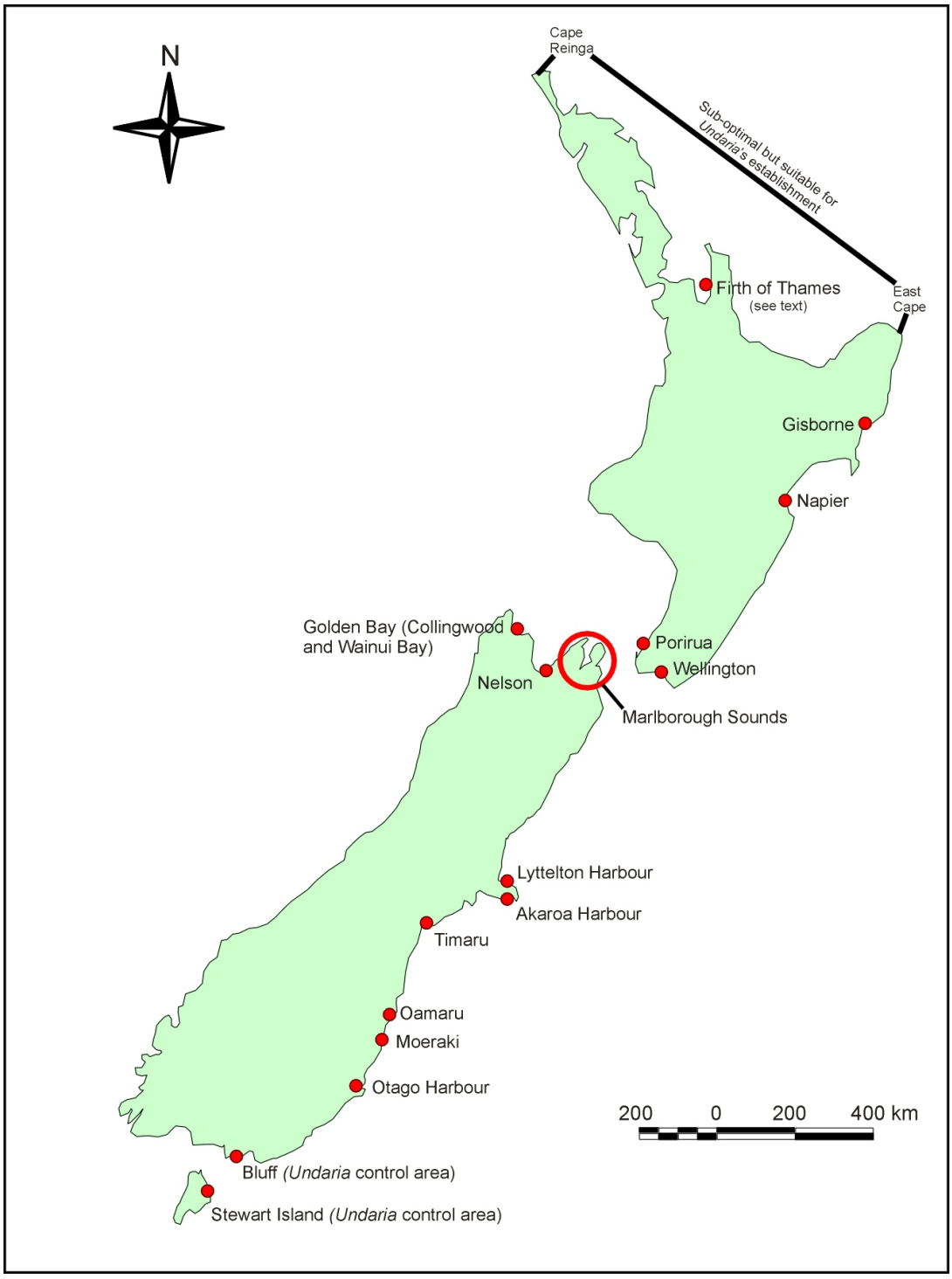

Figure 5.2 Recorded distribution of *Undaria* in New Zealand showing the northern region where sea surface temperatures are higher than optimal for the visible sporophyte stage. See text Section 5.2.1 regarding the Firth of Thames.

- **Banks Peninsula (area G):** *Undaria* is present in Akaroa Harbour on the south side of the Peninsula, and is widespread in Lyttelton Harbour (Forrest and Taylor 2002).
- **Bluff Harbour and Big Glory Bay (area H):** A programme to eradicate *Undaria* started in 1997 in Big Glory Bay (Stewart Island), and later extended to Bluff Harbour. Despite the efforts to date, *Undaria* still occurs at low densities throughout the controlled area, and has also been discovered in Half Moon Bay on Stewart Island.

It is important to recognize that only limited surveillance (in eight New Zealand harbours) for *Undaria* is carried out at present, as part of a wider MFish-funded programme that targets a number of potential marine pests. Given *Undaria*'s current distribution, this level of surveillance will provide little extra information on the seaweed's spread, especially in relation to marine farming regions or other high value areas (HVAs). It is entirely conceivable therefore, that *Undaria* is even more widespread than indicated in Figure 5.2. With this is mind, *Undaria* has not yet been reported from New Zealand's west coast, or along most of the northeast coast between East Cape and Cape Reinga, with the exception of the Firth of Thames. Sea surface temperatures in the Firth of Thames and elsewhere along the northeast coast are regarded as higher than optimal for the visible sporophyte stage of *Undaria*, but nevertheless suitable for its establishment (Sinner et al. 2000). The latest finding in the Firth of Thames confirms this, suggesting that *Undaria* may eventually establish (in suitable habitats) along the entire northeast New Zealand coastline and parts of the west (e.g., in sheltered harbours).

5.3 MARINE FARM PATHWAYS FOR *UNDARIA*

5.3.1 Overview of marine farm activities as a vector for *Undaria*

Inter-regional activities within the marine farming industry may include the movement of shellfish seed-stock and associated materials (e.g., ropes, frames, seaweed), vessel movements, post-harvest transfer of shellfish to processing facilities, and associated waste disposal practices. *Undaria* has the potential to be translocated with many of these activities. To appreciate this point it is important to recognize that *Undaria* is an annual plant, with a life-cycle that alternates between a microscopic gametophyte stage and the visible plant or sporophyte. Both life-stages are adept biofoulers, and *Undaria*

can become established on a wide range of natural and artificial surfaces, especially floating structures including marine farms (e.g., long-line mussel farms, salmon cages), marina pontoons, moorings, and vessel hulls.

The transfer of *Undaria* by fouled vessels and structures is reasonably well documented (e.g., Hay 1990), and shellfish seed-stock and marine farming equipment also appear to be a key vector (e.g., Perez et al. 1981; Stuart 1997). The microscopic gametophyte life-stage is particularly problematic in relation to marine farm vectors, not only because it is 'invisible', but also because it appears reasonably tolerant of current industry handling practices and to the range in environmental conditions (e.g., temperature extremes) likely to be encountered during routine inter-regional transfer activities (e.g., Sanderson and Blackburn 1994; Forrest and Blakemore, in prep.). While aquaculture activities are a significant vector for *Undaria*, it is also relevant to note that marine farms and other floating structures provide ideal habitats for the seaweed to grow on. Hence, such structures are likely to be relatively easily inoculated by other vectors as well, for example visiting vessels that are fouled with mature spore-producing plants.

5.3.2 Existing marine farming transfer pathways and management priorities

Information on aquaculture pathways was obtained from discussions with 16 marine farmers representing a range of industry sectors, with additional feedback on the *Undaria* management issue obtained from at least 14 additional marine farmers via the New Zealand Mussel Industry Council Ltd (NZMIC). In describing marine farming vector pathways, we have focused primarily on transfers that occur between broad regions that either have *Undaria* (e.g., Marlborough Sounds) or are thought to be *Undaria*-free, rather than within regions. With exceptions discussed below, we consider that attempting within-region management of *Undaria* in areas that already have infestations will be pointless in many cases. In large marine farming areas like the Marlborough Sounds, for example, *Undaria* is widespread on farm structures and natural habitats in the area. While there may be some embayments that are *Undaria*-free within this large region, attempting to prevent the spread of *Undaria* to such areas would be difficult and largely futile. This is because:

- One could often expect many if not most localities suitable for *Undaria* at the regional scale to be vulnerable to infestation via the seaweed's natural dispersal mechanisms (Forrest et al. 2000), or via non-marine farming vectors. The latter would include, for example, high-risk but difficult-to-manage vectors such as

moored recreational vessels. The potential for marine farmer management efforts to be undermined by natural spread or other vectors, would probably make it difficult to gain support for regional-scale initiatives in many cases.

- In most large regions like the Marlborough Sounds, the detailed distribution of *Undaria* is unknown. This means that relevant sub-regions for management cannot be identified with existing knowledge, and without an enormous and ongoing effort to monitor the 'within-region' distribution of the seaweed. The costs vs benefits of going to such an effort would need to be carefully considered, and balanced against other priorities for *Undaria*, for example preventing its spread to large regions that are currently uninfested.

Hence, limiting the discussion of pathways to broad regions seems intuitively sensible, and also makes pathway information gathering manageable and more focused. The discussion below is divided in sections covering mussel farming, oyster farming, and other industry sectors. We present the main pathways that have been revealed through discussions with key marine farmers or industry representatives. The activities of most small operators, or within some of the small marine farming regions, may not be well represented. The enormous effort required to capture the complete picture, even within any one region, cannot be justified at this stage.

At the outset of the project our intention was to characterize the main pathways according to the volume, frequency and seasonality of transfers. However, it became apparent during our liaison with industry representatives that movements are highly dynamic and often unpredictable. Among other things, the extent and type of movements are dictated by regional shellfish spat and seed supply/demand, which changes from year to year and from one region to the next. The pathways described below represent the situation over the last 1-2 years.

5.3.3 Mussel industry

Transfer patterns

The main inter-regional pathways in relation to mussel farming are summarised in Figure 5.1 along with oyster farming pathways. Within the mussel industry 'Kaitaia spat' typically comprises approximately 70% of industry needs. This refers to spat that is sourced from Ninety Mile Beach northwest of Kaitaia, and which is attached to

seaweed naturally deposited on the beach in the area. Kaitaia spat is moved to all of the farming regions, and poses no direct risk with respect to *Undaria* transfer since the source region is thought to be *Undaria*-free.

Inter-regional spat transfer on ropes or frames appears to have been relatively common within the mussel industry historically, and still occurs between some regions (e.g., between Golden Bay and the Marlborough Sounds). Recently, however, the NZMIC developed a voluntary code of practice identifying three geographic marine farming zones[2] and requiring that mussel spat moved between these zones be declumped, thoroughly washed, transferred as single seed (typically referring to mussels > 20 mm length), and visually free of blue mussels, *Ciona intestinalis* (a sea squirt), and *Undaria*. Blue mussels and *Ciona* are particularly problematic bio-foulers in some marine farming regions. While this code may go some way to reducing the transfer of the target species between the three zones, recent investigations suggest that there is likely to be high survival of *Undaria* gametophytes on seed mussels following the declumping and washing processes (Forrest and Blakemore, in prep.).

Inter-regional movements of service vessels are relatively infrequent and intermittent and, where they do occur, follow the same pathways described for spat and seed mussels in Figure 5.1. The greatest inter-regional vessel activity appears to occur between the Marlborough Sounds and Golden/Tasman Bays, otherwise movements are mainly within regions. Post-harvest processing and waste disposal practices appear to occur primarily within areas already infested with *Undaria* or involve treatment processes that would minimize any risk. In Bluff (where *Undaria* is currently managed), for example, some mussels from the Marlborough Sounds are processed, but hot water and infra-red sterilisation are part of the production system. Furthermore, process wastewater is discharged to the local sewerage system, solid wastes are land-filled, and the bulk bags in which the mussels are transported are sterilised. There may be exceptions to these general patterns, but a more thorough evaluation at present is beyond the scope of this work.

[2] The three zones are: northern New Zealand (north of Mahia Peninsula including the Firth of Thames and Coromandel); southern New Zealand (south of Kaikoura); and a central zone between these two (which includes the Marlborough Sounds and Golden/Tasman Bays).

Implications for management of current mussel farming transfer pathways

It is worthwhile discussing transfers from *Undaria*-infested areas where management options should be considered. In general terms, management may be worthwhile when either of the following two criteria apply:

- The **recipient region is *Undaria*-free** and spread via **natural dispersal is unlikely**, or possible only over 'long' time scales, and the risk of human-mediated spread is low and/or **vector management is feasible** (e.g., all significant vectors can be effectively managed). Clearly, vector operators must be supportive of any management measures that are proposed, otherwise they are unlikely to be successful.
- *Undaria* is **present in the recipient region but subject to eradication or control efforts**, and minimising or preventing further introductions is important to the success of such efforts.

Based on these criteria, a key point that can be taken from a comparison of Figure 5.1 and Figure 5.2 is that many inter-regional mussel farming pathways from *Undaria*-infested areas involve transfers to regions where *Undaria* is already established but not controlled. Assuming that no control efforts will be initiated in these infested regions, then there is little or no purpose in attempting to manage *Undaria* introductions on mussel farming pathways. This reasoning arises from our viewpoint that the local spread of the established populations, and the risk of secondary regional/national spread from them, will likely far outweigh any additional risk posed by continued *Undaria* introduction on marine farming or other vectors. There are three current mussel farming pathways, however, that meet the above criteria to some extent, and where the efficacy of management warrants discussion. These are described below.

1. Marlborough Sounds to Stewart Island: The management of this pathway is desirable given that vector control is considered an important component of *Undaria* eradication/control efforts in Big Glory Bay, Half Moon Bay, and Bluff Harbour. Part of the current management programme involves vessel monitoring in southern New Zealand ports to minimize the risk of further introductions of *Undaria* to Stewart Island and other southern areas of high conservation value. Marine farmers have contributed

to this goal in a number of ways, including adhering to a voluntary ban on the importation of spat or seed mussels from the Marlborough Sounds.

Should the *Undaria* eradication/control programme be discontinued, it could be argued that there would be little benefit in continued management of *Undaria* on incoming vectors. We note, however, that the marine farmers in Big Glory Bay are interested in preventing the inoculation/introduction of marine species other than *Undaria*, and are interested in tools for reducing vector risks for marine pests in general, irrespective of any decisions made regarding *Undaria*. They have emphasised, however, the need to strike a balance between the risks versus the cost to the industry. For example, they are currently in a position of having an insufficient supply of spat or seed mussels to meet their requirements, reflecting a combination of the voluntary ban on movements from the Marlborough Sounds and a shortage of Kaitaia spat.

2. Marlborough Sounds to the Firth of Thames: Assuming that the Firth of Thames region remains uninfested, management of pathways from the Marlborough Sounds should be considered. A key vector is likely to be the transfer of single seed mussels, but equipment (e.g., ropes) and vessel movements have also occurred during recent mussel farming development in the region. An uncontrolled *Undaria* infestation in the Firth of Thames would greatly enhance the seaweed's potential to spread along the northeast coast, and even to the Northland west coast if current oyster farm pathways (see Figure 5.1 and Section 5.3.4) became infected. *Undaria*'s natural spread to the Firth of Thames from the nearest recorded population in Gisborne is unlikely because of the numerous dispersal barriers present, such as long stretches of wave-exposed or sandy coastline. As such, reducing the risk of *Undaria* infestation to the region will primarily rely on management of human-mediated pathways.

Any decision regarding management of pathways from the Marlborough Sounds should be made only after a thorough evaluation of regional vector movements generally, along with their relative risks and the extent to which they can be managed. The latter should include wide consultation with affected parties to determine the feasibility and implications of management. Given the recent *Undaria* population recorded in the Firth of Thames, and its subsequent removal, it is also clearly important that the infestation status in this region is closely monitored. A widespread infestation, if

uncontrolled, would almost certainly make management of vectors from *Undaria* donor regions largely futile, unless the vector controls targeted marine pest species generally.

3. Marlborough Sounds to Golden Bay and Tasman Bay: Marine farm operators in Golden Bay (at Collingwood and Wainui Bay) have developed voluntary *Undaria* management plans, which include measures such as removing *Undaria* from long-line anchor warps. *Undaria* does not yet appear to have reached adjacent areas, which is especially interesting at Wainui Bay given the close proximity of the rocky shoreline to the infested structures. Based on our recent work (authors, unpubl.), we suggest it is likely that *Undaria* (and other seaweeds) in these areas are grazed by marine animals (snails, sea urchins, etc) to an extent sufficient to limit their colonisation and establishment in natural habitats. The *Undaria* management plans for the marine farms may help considerably in this respect by limiting the supply of colonising spores to the adjacent shoreline. Hence the current or proposed marine farmer management efforts are probably well worthwhile, given the proximity of the farms to HVAs such as the Abel Tasman National Park (ATNP) coastline.

By also managing the pathways to the Golden Bay farms from the Marlborough Sounds (if feasible), less effort may be required for on-site management. There is arguably little point in managing these pathways, however, if the marine farmer management plans are not widely supported, since there are significant vectors unrelated to marine farm activities (e.g., high risk recreational vessels) that remain unmanaged at present. Until recently, locally funded control measures for *Undaria* were in place in Port Nelson (a significant point of vector departure to the ATNP coastline), but this funding was withdrawn in the absence of any clear government direction on *Undaria* management nationally.

If any long-term *Undaria* management plans were implemented for pathways to Golden/Tasman Bay, and the ATNP coastline, then the risk presented by all significant vectors would need to be considered. With respect to marine farm activities outside of Wainui Bay and Collingwood, *Undaria* transfer risks to the ATNP coast are probably negligible in comparison with other vectors (e.g., recreational vessels with *Undaria* on their hulls that visit the ATNP area directly). Other than vessel movements, the other main marine farming activity is the seasonal deployment of spat catching equipment (ropes, frames) in the 'ring road' sites of Tasman and Golden Bay. This is likely to be

low risk with respect to *Undaria*'s spread, because the spat catching equipment typically remains out of the water for several months before deployment and is not left in the water for long enough that any *Undaria* present could mature. Furthermore, the 'ring road' sites are far enough offshore to be beyond *Undaria*'s natural dispersal capability.

5.3.4 Oyster industry

The oyster farming industry is primarily located north of Auckland, with relatively minor activity in the Coromandel area and Marlborough Sounds. Kaipara Harbour on the west coast north of Auckland provides approximately 70% of the spat supply to farms in the northeast harbours and Coromandel area. The Kaipara spat is transferred on wooden sticks year-round. The detail of the spat movements is simplified in Figure 5.1, but the general west-to-east transfer direction is indicated. The remaining 30% of spat are locally caught, with some produced at a land-based hatchery near Nelson and transported to the northern areas. The seawater intake for the hatchery is presently *Undaria*-free, with the nearest *Undaria* populations being approximately 10 km along the coast in Port Nelson.

In addition to the west-east movement of Kaipara spat, there are weekly transfers of adult oysters back to Kaipara Harbour from some of the east coast sites. There are also weekly movements of oysters from the Bay of Islands to sites in the Coromandel. Intermittent movements of oysters may also occur in response to degraded water quality in growing areas. For example, in response to degraded water quality at a Bay of Islands growing site, the oyster stock was recently moved to Kaipara, Mahurangi and Parengarenga Harbours. As far as we can ascertain, there are currently no inter-regional movements of oysters from areas where *Undaria* has been reported.

There appear to be no movements of oyster farm service vessels between the growing regions. All oyster processing occurs locally within the growing areas, with on-site discharge of wastewaters and land-filling of solid wastes. The industry has no current management plans to address bio-fouling or other pests. Some heat treatment of transferred Kaipara spat was initiated to kill cysts of the toxic microalga *Gymnodinium catenatum* following its discovery on the northwest coast in September 2000. This is not carried out at present, in part because it was considered impractical by some growers.

5.3.5 Other industry sectors

Sea-cage salmon farming is undertaken in Stewart Island and the Marlborough Sounds. A different company operates within each region and there are generally no transfers between the two. Where cages have been transferred historically, they have been completely refurbished (water/sand blasted and repainted) before re-deployment. The salmon stock used to supply the sea-cages is produced in freshwater hatcheries.

Approximately 25 land-based hatcheries (e.g., for paua) are scattered around the coastline and most have sea water intakes and discharges, but are in areas thought to be *Undaria*-free. In most cases, transfers from one hatchery to another are unlikely to constitute a significant risk with respect to *Undaria*, even where transfers are between infested and uninfested areas. This is because only microscopic stages have the potential to be transferred, and would clearly not have the opportunity to develop into mature *Undaria* within a hatchery system. The likelihood of gametophytes or microscopic plantlets being dislodged within a hatchery system, being discharged, and then reattaching in the natural environment is suggested to be remote.

Undaria is used as a food source within some land-based hatchery systems (e.g., paua hatcheries/farms) but as far as we are aware this only occurs at a local scale within infested regions. Potentially, the most significant hatchery-related pathways are those such as described for oyster spat in Section 5.3.4, where hatchery production is moved to sea-based systems for ongrowing. We are unaware of any current examples of this that would be high risk with respect to *Undaria*, and to ascertain the level of risk would require an evaluation of the practices of all hatchery operations.

5.3.6 Potential pathways

Future pathways may emerge that would require consideration of *Undaria* management on a case by case basis. There are three main categories that can be envisaged. These are described below with relevant examples.

(i) The infestation of *Undaria*-free marine farming areas whose current pathways lead to uninfested areas or areas where *Undaria* is managed and vector control is important

For example, if *Undaria* established in the Firth of Thames/Coromandel area, the potential for secondary spread from the Firth area would warrant evaluation, but would

clearly be a complex undertaking. Based on Figure 5.1, for example, voluntary controls on pathways of spat or seed mussels from the Firth of Thames to Stewart Island would need to be considered if *Undaria* was still subject to eradication/control in Stewart Island.

With respect to *Undaria*'s potential for spread north of the Firth of Thames via mussel farm activities, we are aware of transfers from the Firth to Waiheke Island near Auckland. A more in-depth assessment may reveal other pathways within this northern area where management would need to be considered. Further analysis of pathways to Great Barrier Island may be particularly worthwhile, given that this area is relatively isolated in geographic terms. A detailed analysis for such areas could not be justified in the present work, since the northern region generally is not yet regarded as infested, and because of the effort involved. Great Barrier, for example, has eight consented mussel farms each held by a different person or company.

Similarly, if *Undaria* was discovered in any of the oyster growing areas of northeast New Zealand (which we assume to be currently uninfested), unmanaged oyster transfers to Kaipara Harbour could result in the spread of *Undaria* to the relatively isolated harbours of New Zealand's northwest coast. These areas are probably not particularly vulnerable to infestation from other sources at present. This scenario assumes that *Undaria* would survive within oyster growing areas, but this is not certain, since tidal elevation may be a limiting factor. Many oyster cultivation racks are positioned at the level of an extreme low water neap tide to avoid 'mud-worm' infestation problems. *Undaria* is generally regarded as a subtidal species, but can grow as high as the neap tide level in the South Island (e.g., Nelson, Lyttelton). It is possible, however, that warmer air temperatures, hence greater dessication during periods of low tide, could prevent intertidal establishment in the northern oyster growing areas.

(ii) The development of new pathways from infested areas to <u>existing</u> marine farming areas that are currently uninfested

There are some marine farming areas from which *Undaria* has not been reported, and which are currently self-contained or have no incoming pathways from infested areas. These include, for example, Aotea Harbour (area D of Figure 5.1), which occasionally supplies small quantities of spat or seed mussels to the Firth of Thames and Marlborough Sounds. In the Aotea (and adjacent Kawhia) Harbour area, the natural

spread of *Undaria* is unlikely because of dispersal barriers (i.e., large stretches of wave-exposed soft-sediment habitats where *Undaria* would be unlikely to establish), hence human activities provide the most likely means of introduction.

(iii) The development of new pathways from infested areas to <u>new</u> marine farming areas that are currently uninfested

We have not attempted to ascertain in any detail the potential transfer pathways to any of the proposed areas shown on Figure 5.1. There are a number of interesting aspects to some of the current applications that are worthy of discussion, however. One comment is that many of the large blocks that are proposed (e.g., Bay of Plenty, Hawke's Bay) are several kilometres from the coastline, and situated over soft sediments in relatively deep water. Based on present evidence regarding *Undaria*'s natural dispersal capability and habitat requirements, the proposed new blocks are effectively isolated 'islands' to which *Undaria*'s spread will only be possible via infected vectors. Vector management to prevent initial infestation might be warranted in some of these areas if:

(a) The offshore farm was likely to be a significant reservoir of Undaria for secondary spread to HVAs: For most of the offshore sites, secondary spread from the structures to the natural environment would not likely be significant given the relatively deep water and soft sediments over which the proposed blocks are located. Clearly, however, the importance of secondary *Undaria* spread would need evaluation on a case by case basis, including consideration of the risk of secondary vector transfers to HVAs.

(b) All significant vectors could be identified and effectively managed: There would be little point focusing on marine farming vectors if other significant vectors remained unmanaged. In this regard, the proposed offshore block at Napier provides an interesting example. Proposed consent conditions for the Napier site require a Biosecurity Plan to be developed specifying, among other things, the development of:

"management practices to ensure that no spat, mussels, equipment, vessels or organisms known to be harbouring harmful, toxic or nuisance organisms, including but not restricted to *Undaria pinnatifida,* are transferred to the mussel farm…".

Compliance with such conditions will almost certainly come at considerable cost to the marine farmers. It is important, therefore, that the risks posed by other vectors are acknowledged, and comparable management measures implemented for them. This is especially important given that *Undaria* is already established in the nearby Port of Napier, and the presence of marine farms in the area will likely provide a focal point for recreational fishers (as is the case in other marine farming regions). A visit by even one *Undaria*-fouled vessel from the Port (or elsewhere) may be enough to seed the infestation of the farm structures, undermining efforts made by marine farmers to comply with their consent. We are aware of similar consent requirements being proposed by regulatory authorities in other *Undaria*-infested regions, highlighting an urgent need for guidance on the *Undaria* issue so that consent/permit conditions or regional management initiatives are sensible, and consistent with national management directions.

(c) *Marine farmers themselves considered Undaria to be of sufficient nuisance that it was worth trying to prevent initial infestation*: Based on current general views within the industry, it seems unlikely that the nuisance value or economic cost of managing *Undaria* would itself provide the incentive for many marine farmers to see any net benefit in preventing initial infestation, although there are likely to be some exceptions (e.g., smaller growers in areas where *Undaria* is a particular nuisance). Some sectors of the marine farming community have already been active in contributing to *Undaria* management generally, both at the local and national level. Examples of this include: the voluntary management approaches described above for Wainui Bay, Collingwood and Big Glory Bay; the NZMIC code of practice for mussel seed transfer; and current policy development by the New Zealand Marine Farming Association which seeks to raise industry awareness of the *Undaria* issue in order to minimize the seaweed's spread with marine farming activities.

5.4 CONCLUSIONS

This document describes the distribution of *Undaria* in relation to marine farming areas in New Zealand, the types of marine farm activities that might transfer *Undaria*, and the principle pathways along which these activities occur. In broad terms, pathways where management of *Undaria* spread is most desirable are those from any *Undaria*-infested

marine farming area to any present or future marine farming area that is *Undaria*-free (or where *Undaria* is under control), and where management measures are not undermined by the natural spread of *Undaria* or by uncontrolled spread via non-marine farming vectors (e.g., high risk recreational vessels). For regions like the Marlborough Sounds, we suggest that *Undaria* management will largely be futile. While there may be parts of the Sounds that are *Undaria*-free, such areas cannot be identified with current knowledge and without considerable ongoing effort to monitor the seaweed's distribution. We also assume that most localities suitable for *Undaria* at a regional scale will be vulnerable to infestation via natural dispersal or non-marine farming vectors.

Furthermore, we suggest that *Undaria* management should be considered on a case by case basis where any of the following situations arise through future industry development: (i) The infestation of *Undaria*-free marine farming areas whose current pathways lead to uninfested areas or areas where *Undaria* is managed; (ii) The development of new pathways from infested areas to existing marine farming areas that are currently uninfested; (iii) The development of new pathways from infested areas to new marine farming areas that are currently uninfested. In considering management of these and any other pathways, it should be recognised that measures will only be effective if they have the support of affected marine farmers and other vector owners/operators. This support may be more easily gained for management measures that are applied equally across all vectors and have benefits beyond *Undaria*. For example, because aquaculture industry members consider *Undaria* bio-fouling effects to be far less significant that impacts from pests such as blue mussels and sea squirts (e.g., *Ciona intestinalis*), they are more supportive of management practices that have generic biosecurity benefits.

5.5 ADDENDUM

As noted in the preface to this Chapter, much of the information above was extracted verbatim from a report by Forrest and Blakemore (2002), but changes since the time the report was produced mean that much of the analysis and recommendations no longer apply. The rapidity with which this occurred is itself of interest; it highlights the dynamic nature of biological invasions, and hence the need to ensure that risk management plans and processes can be adapted to changing circumstances.

A critical change since the Forrest and Blakemore (2002) report is that *Undaria* has been found in a number of additional coastal locations, as described in Chapter 2. Of relevance to management of marine farming pathways, the seaweed is now well-established in the Firth of Thames and Waitemata Harbour, Auckland (see Chapter 2). As such, its spread to other parts of the Hauraki Gulf is almost certain in the absence of management; arguably the intensity of vessel activity in this region, especially recreational vessels (see Dodgshun et al. 2004) would make management of human-mediated spread difficult and probably futile. For such reasons, the recommendations in Section 5.3.3 to manage mussel farm pathways to the Firth of Thames are no longer relevant with respect to *Undaria*. Similarly, the withdrawal of funding from the southern New Zealand *Undaria* management programme means that the seaweed is likely to become widely established in Bluff Harbour and on Stewart Island in the absence of regionally-led management. Again, this makes any attempt to manage *Undaria* pathways to these regions largely futile.

In both of these cases it should be noted, however, that marine farmers have interest in the management of marine pests other than *Undaria*; in fact *Undaria* is perceived as little more than a nuisance compared with more significant fouling pests such as *Styela clava* and *Didemnum vexillum*. Clearly, therefore, marine farmers have a strong incentive not to promote the spread of such organisms via their own practices, irrespective of management actions in relation to *Undaria*. Furthermore, despite many of the specific recommendations made in this report now being redundant, the general criteria in Section 5.3.6 for determining whether and where pathway management may be worthwhile are still relevant to *Undaria*, and also have a wider relevance to other pest species. For example, vessel-mediated spread within the Hauraki Gulf could eventually lead to *Undaria* and other unwanted pests like the tunicate *Styela clava* establishing in oyster growing areas of the east coast of northern New Zealand. The subsequent infection of oyster crops could lead to the associated transfer of these pests with crop movements. Management of such risks would be desirable for transfers to regions that are currently pest-free (e.g., the Kaipara Harbour) and for which spread via natural dispersal mechanisms and other vectors is unlikely.

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Chapter 6

Reducing the Spread of *Undaria* with Aquaculture Transfers

PREFACE

This chapter describes an evaluation of methods to reduce the spread of *Undaria* with the inter-regional movement of equipment and seed-stock within the New Zealand mussel industry. This work was published with the following citation:

Forrest BM, Blakemore KA. 2006. Evaluation of treatments to reduce the spread of a marine plant pest with aquaculture transfers. Aquaculture 257: 333-345

The chapter is identical to the above citation. The original work on which the chapter and publication is based was detailed in a Cawthron technical report as follows:

Forrest BM, Blakemore KA. 2003. Evaluation of methods to reduce inter-regional spread of the Asian kelp *Undaria pinnatifida* via marine farming activities. Cawthron Report 773, Cawthron Institute, Nelson, New Zealand. 38p plus appendices

This report contains greater detail on methods, more comprehensive results, and appendices detailing the various culturing methods and experimental end-point criteria that are referred to. This is my own work, with my co-author providing technical support for the laboratory trials that are described.

ABSTRACT

The role of aquaculture and other human activities in spreading non-indigenous marine organisms is well recognised. This paper assesses the feasibility of various 'environmentally friendly' treatments for key inter-regional transport vectors (equipment and seed-stock) within the New Zealand mussel farming industry, focusing on control of an internationally recognised seaweed pest, *Undaria pinnatifida*. The effects on *Undaria* of high pressure water blasting, natural air drying, and freshwater immersion at ambient (10 and 20 ^{o}C) and hot (35-55 ^{o}C) temperatures are described, and the tolerance of mussel seed-stock to the freshwater and hot water treatments is investigated.

Water blasting was completely effective in removing *Undaria* gametophytes from shells at pressures ≥ 2000 psi for 2 sec. *Undaria* survived natural air drying for up to 2 d at ambient humidity (55–85% relative humidity; RH) and > 8 wk at high humidity (> 95% RH). In freshwater, gametophytes survived immersion for 1–2 d, but plantlet mortality occurred within < 10 min. *Undaria* survival in hot water across the 35–55 ^{o}C range was tens of minutes to a few seconds. Using these findings as guidelines, these treatments would be relatively easily applied to sterilize equipment such as farm floats and rope, with the preferred method selected based on cost, practicality and other constraints.

Removing *Undaria* from mussel seed-stock is more problematical because of the importance of identifying treatment conditions that do not compromise mussel health. Mussels were not adversely affected when immersed in freshwater for a 2 d duration sufficient to ensure complete *Undaria* mortality. Hence, mussels could potentially be treated in freshwater at reasonable cost while being transported between aquaculture regions. However, to ensure an effective treatment the water would need to be exchanged during the transport phase in order to maintain salinities at ≤ 1 psu. Our findings also suggested that exposure to hot water at 55 ^{o}C for approximately 5 sec would achieve complete *Undaria* mortality while maintaining a level of mussel survival comparable to untreated seed-stock. This method is likely to involve greater costs than freshwater immersion, and requires field validation to confirm both the efficacy against *Undaria* and to identify a method of implementation that ensures mussel survival is not compromised by the combined stresses of treatment and inter-regional transport.

6.1 INTRODUCTION

The role of human activities in facilitating or exacerbating the spread of non-indigenous marine organisms is well recognised, and in New Zealand particular attention has been given to the feasibility of managing the spread of the Asian kelp *Undaria pinnatifida* (e.g., Sinner et al. 2000; Wotton et al. 2004). This is a conspicuous species, which is internationally regarded as a fouling pest on marine farms and other structures (Sanderson 1997; Fletcher and Farrell 1999) and a threat to the ecology and natural character of high value coastal areas (Sanderson and Barrett 1989; Hay and Villouta 1993; Fletcher and Manfredi 1995; Battershill et al. 1998; Walker and Kendrick 1998).

Undaria has a limited capability for natural dispersal (Forrest et al. 2000), with its spread at inter-regional scales greatly exacerbated by vessel movements (Hay 1990; Fletcher and Manfredi 1995; Casas and Piriz 1996; Floc'h et al. 1996) and marine farming practices such as transfers of equipment and seed-stock (Perez et al. 1981; Boudouresque et al. 1985; Stuart 1997; Forrest and Blakemore 2003).

In New Zealand, longline cultivation of the green-lipped mussel *Perna canaliculus* (marketed as the Greenshell™ mussel) is the dominant form of marine aquaculture. Farm development and routine operations can involve the transfer of equipment (especially rope) and seed-mussels (15–60 mm shell length) between the main aquaculture regions. The industry has in place a number of management practices to reduce the incidental spread of pest species with such transfers. For *Undaria* and other biofouling pests the main process includes stripping seed-mussels from the crop rope and subjecting them to a vigorous declumping, washing and screening process to remove biofouling organisms. Both the seed-mussels and rope are stored in large (~1 m^3) bags for transport.

Management of *Undaria* is made particularly difficult by the fact that the seaweed has an annual life-cycle that alternates between a microscopic gametophyte stage and the visible sporophyte stage. This means that when a population of *Undaria* apparently dies off and is no longer visible to the eye, it is almost certainly still present in its microscopic gametophyte form or as a small sporophyte (plantlet). Recent research has shown that existing industry management measures are not completely effective against such life-stages, and that secondary treatment methods would be required to further reduce inter-regional transfer risks (Forrest and Blakemore 2003).

This paper describes research into secondary treatments for aquaculture equipment and seed-mussels. We describe laboratory-based work intended to highlight the relative advantages and limitations of different treatment approaches, with the preferred approach dictated by the particular needs and constraints of each aquaculture company. Despite a number of studies demonstrating the efficacy of various chemical treatments against bio-foulers (Burridge and Gorski 1998; Gunthorpe et al. 2001; McEnnulty et al. 2001), the focus of our study is on methods that are ‘environmentally friendly’, reflecting the desire of the industry to protect its ‘clean green’ brand.

We first assess the survival of *Undaria* to high pressure water blasting and natural air drying. These are inexpensive methods with potential application to mussel farm equipment (e.g., floats and/or rope), but would not be practical as treatments for seed-mussels without major changes to standard industry operating procedures. For a seed-mussel treatment to be acceptable to the industry, it must cause minimal disruption to operations and not adversely affect the stock. For this purpose, we evaluate the efficacy of ambient and hot freshwater immersion, and undertake a preliminary evaluation to identify the limiting factors that are likely to arise in the field-scale application of these treatments.

6.2 MATERIALS AND METHODS

6.2.1 Experimental end-points for *Undaria* and seed-mussels

Treatment experiments with *Undaria* used cultures of microscopic gametophytes and plantlets ≤ 20 mm length, because these are key life-stages potentially transferred by mussel industry practices (Forrest and Blakemore 2003). Culturing procedures for gametophytes followed methods described in Forrest et al. (2000). An initial spore concentration of 5000 ml^{-1} produced gametophyte densities (allowing 1 h for spore settlement) of ~25–50 mm^{-2} after 2–4 wk, which were suitable for quantitative mortality assessments.

After 1 wk of culturing post-treatment, gametophytes were considered dead if they: had lost their brown pigmentation, had a discontinuous cytoplasm or 'necrotic' appearance, and did not fluoresce under an epi-fluorescent microscope (WG filter). Plantlet cultures were initiated on weathered nylon rope (6 mm diameter) in laboratory aquaria (Gibbs et al. 1998) then transferred after 1 month to a field site for ongrowing. Triplicate 75 mm sections of the culture rope, each containing ~50–100 plantlets, were used as the experimental units. Treatment efficacy was determined qualitatively according to whether the rope sections had plantlets that were dead, alive, or both, after ongrowing in the field for 1 wk post-treatment.

Our experimental approach with seed-mussels involved determining both short-term and long-term treatment effects. Absolute mortality measures, such as cessation of ciliary movement (Edwards et al. 2002) were not considered suitable end-points in that a functionally healthy seed-stock is critical to the industry. Further, short-term

measures of functional impairment such as shell gaping tests (e.g., Guderley et al. 1994; Rajagopal et al. 1995) provided equivocal results. As an alternative, we assessed short-term effects as the percentage of mussels (from 5 replicate batches of 20 mussels each) that reattached via their byssus after being held in 4 litre buckets in a seawater facility for 24 h post-treatment. This approach had functional relevance to the industry (if a mussel cannot reattached then it will fall off the culture rope), was robust (control mussel reattachment was consistently ≥ 90%) and gave us the ability to rapidly evaluate a wide range of potential treatments. To validate this method, and provide an indication of long-term effects, mortality over a 6 month cultivation period in a hatchery was also evaluated.

6.2.2 Water blasting and air drying treatments for *Undaria*

High pressure water blasting

The efficacy of high pressure water blasting was assessed against gametophytes only, because pilot investigations revealed that plantlets were considerably more susceptible to mechanical damage. Gametophytes were cultured for 2 wk on moderately fissured shells of the bivalve *Paphies subtriangulata*. This substratum was used as a surrogate for the complex materials on which *Undaria* may be transferred via aquaculture practices (e.g., rope and floats), and provided a light-coloured surface that was suitable for direct counting of gametophytes. Trials were conducted at 1000, 2000 and 3000 psi, using a water blasting pump capable of producing 3000 psi at 15 litre min^{-1}. With the jet nozzle positioned 100 mm from the shell (held in a clamp), two exposure times (1 and 2 sec) were tested and two types of jet nozzle compared; a turbo nozzle that emitted a rotating stream of high pressure water and a standard nozzle that produced a direct stream of water in a 15 degree arc. To evaluate treatment effects, gametophyte mortality was assessed according to the criteria described above. Percent survival was calculated from the density of living individuals (80 x magnification) in a pre-defined grid on each shell 1 wk post-treatment by comparison with a pre-treatment count.

Natural air drying

Air drying experiments with *Undaria* were conducted at 10 and 20 ^{o}C in constant temperature (± 1 ^{o}C) cabinets; the choice of treatment temperatures being within the seasonal range encountered around much of New Zealand. The experiments involved a comparison between ambient (55–85% RH) and high (> 95% RH) humidity. The latter

was included on the basis that high humidity conditions would typically be created during the transfer of equipment (especially rope) and would likely lead to enhanced survival (e.g., Sant et al. 1996; Schaffelke and Deane 2005).

Gametophytes were cultured on sterile 24-well Falcon™ tissue culture plates, with treatments and controls for a given exposure time assigned in alternating columns (6 columns x 4 rows). Control columns were filled with fresh growth medium pre-heated to 10 or 20 ^{o}C as appropriate. The ambient humidity treatment was applied by leaving the lid off each designated 24-well plate, and the high humidity treatment applied by placing the lid on each plate. For both temperatures, exposure times for each treatment ranged from 1–72 h at ambient humidity and 1–8 wk at high humidity. Percent survival of gametophytes within 6 randomly selected treatment and control wells was determined quantitatively from pre- vs post-treatment counts as described above. Instances where gametophyte percent survival exceeded 100% reflected the growth of small gametophytes that were not recorded during the initial count.

Plantlet treatments using triplicate 75 mm sections of the culture rope were conducted in 1 litre clear plastic pots, and applied in a similar manner to that described above for gametophytes. Controls for each exposure time consisted of pots filled with UV-sterilised seawater filtered to 35 μm and pre-heated as appropriate. Exposure times were similar to those described for gametophytes, with treatment effects determined as described above.

6.2.3 Freshwater and hot water treatments for *Undaria* on seed-mussels

Freshwater immersion

Freshwater effects on *Undaria* were assessed at 10 and 20 ^{o}C using an identical experimental set-up and assessment approach to that described above for air drying. Freshwater treatments for gametophytes were applied by filling the treatment columns in the 24-well tissue culture plates with tap water pre-heated to 10 or 20 ^{o}C, and for plantlets by filling 1 L pots. Controls consisted of sterilised pre-heated seawater as described above. Exposure times for gametophytes ranged from 1–48 h but for plantlets were as short as 10 min, reflecting their greater sensitivity to freshwater effects. A logistic regression procedure (Allison 1999) in Systat 9 (Systat 1999) was used to model the mean survival of gametophytes (the most resilient *Undaria* life-stage) for the freshwater treatments.

Freshwater immersion experiments with seed-mussels followed the same general approach and experimental design as described for *Undaria*, with each replicate batch of mussels (16–36 mm shell length) held in 1 litre pots of aerated tap water (treatments) or filtered sterilised seawater (controls) pre-heated to 10 or 20 ^{o}C as appropriate. The relatively large volume of water meant that treatment salinity was maintained at ≤ 1 psu during the experiment. Exposure times ranged from 1–5 d for each temperature. After 24 h post-treatment, mussel attachment was assessed, and mean attachment modelled using logistic regression. Follow-up work compared freshwater effects on two mussel size classes (small, 19–30 mm; and large, 30–45 mm) for the 10 ^{o}C treatment only, and included assessment of both 24 h attachment and 6 month survival. Treatment effects were examined using three-way ANOVA with planned comparisons between treatments and controls for mussel size and exposure time. An arcsine square-root transformation was applied to satisfy assumptions regarding normality and homogeneity of variances.

Hot water immersion

Hot water immersion was considered for mussel industry companies needing a relatively fast-acting treatment. Experimental temperatures of 35, 45 and 55 ^{o}C were selected on the basis of existing literature indicating likely efficacy against *Undaria* (e.g., Mountfort et al. 1999; Webb and Allen 2001), and because such temperatures would not present an occupational hazard. Gametophytes cultured in sterile plastic pots (35 mm diameter, 5 replicates) were immersed in tap water pre-heated to 35, 45 and 55 ^{o}C (± 1 ^{o}C) for exposure times of 1–60 min, 5 sec to 2 min, and 1–15 sec respectively. Plantlets and mussels (15–50 mm length) were treated in a similar way to that described for gametophytes, with similar exposure times used. All experiments included cooling in filtered sterilised seawater at ambient temperatures immediately after treatment. Gametophyte survivorship and mussel attachment for each temperature were modelled using logistic regression.

Identification of seed-mussel treatments and issues for field implementation

The primary focus of the freshwater and hot water treatment investigations was to identify the optimal combination of treatment and exposure time that led to an acceptable level of mussel survival (defined here as ≥ 90%) while achieving an adequate level of *Undaria* mortality (defined here as 100%). To facilitate the identification of treatment conditions where these criteria were met, the logistic models developed from the gametophyte and mussel data were used to predict the level of

gametophyte mortality at exposure times resulting in 90% mussel attachment. For the hot water treatments, the logistic model was also used to predict exposure times that would result in mussel attachment at levels < 90% to illustrate the trade-off between the two conflicting goals of treatment (i.e., the need to maximize both *Undaria* mortality and mussel survival).

From this work we further investigated the efficacy of treatments based on immersion of mussels in freshwater for up to 48 h, and hot water treatment at 55 ^{o}C for 5 sec. Laboratory-based experiments were used to evaluate key limitations that would arise when these treatments were scaled up to a typical seed-mussel transfer situation in which the mussels are transported in large (~1 tonne) bags, and are often out of the water for 24–36 h before re-seeding.

For freshwater, the preferred treatment involved transport of mussels in bins of water so that the treatment took place concurrently. A key limitation would be the need to minimize the volume of freshwater to reduce transport costs. In such a situation, pilot work identified that the salinity within the bins could reach 8 psu following the 48 h immersion period required to kill *Undaria* (Forrest and Blakemore 2003). Hence, in order to evaluate the potential efficacy of this method under field conditions, we qualitatively assessed gametophyte survival at salinities of ≤ 1 , 2, 4, 6 and 8 psu for exposure times of up to 48 h, to compare with our freshwater (≤ 1 psu) treatment findings. For this purpose, gametophyte survival in treatments relative to controls was ranked as: 0 = all dead, 1 = most (> 75%) dead, 2 = similar numbers alive vs dead (25–75% alive), or 3 = most (> 75%) alive.

For the selected hot water method, a key consideration was whether seed-mussels could survive the short and long-term effects not only of the treatment, but also the combined stress resulting from treatment in combination with air exposure during inter-regional transport. We investigated this using the same general methods as described above. The transport phase was simulated by holding the mussels in covered bins for 36 h. Experiments were conducted that compared the individual effects of treatment and transport, and the combined effects resulting from treatment followed by transport and vice versa. One-way ANOVA with a post hoc Tukeys HSD test was used to examine the effects of treatment on mussel attachment and survival, using an arcsine square-root transformation to satisfy assumptions for parametric analysis.

6.3 RESULTS

6.3.1 Water blasting and air drying effects on *Undaria*

The turbo nozzle was completely effective in removing gametophytes from a shell substratum at 2000 psi for 2 sec and 3000 psi for 1 and 2 sec, with a pressure of 1000 psi being > 90% effective (Figure 6.1). Under all treatment conditions the direct water jet from the non-turbo nozzle was considerably less effective. While the mean effectiveness of gametophyte removal was > 60% in all cases, there was little clear or consistent difference in mean values in relation to either pressure or exposure time, and survival was highly variable among replicates. For all treatments where gametophytes survived, they were primarily lodged within shell fissures or indentations (e.g., around the valve margin).

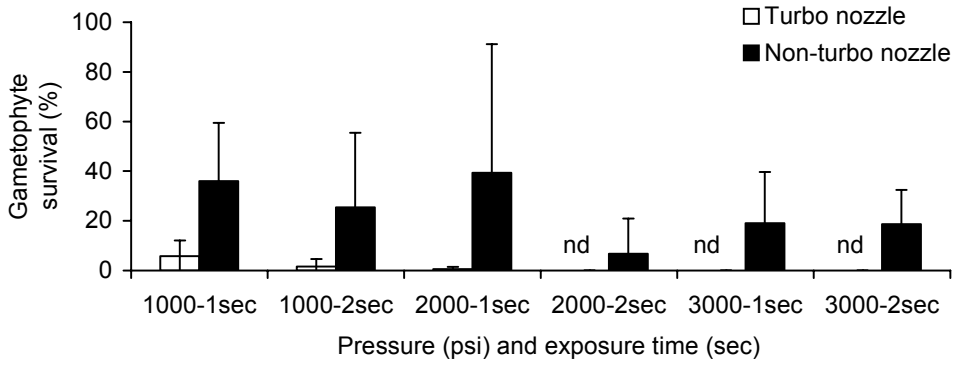

Figure 6.1 Mean (± SD, n = 5) *Undaria* gametophyte survival on shell cultures subjected to high pressure (1000–3000 psi) water blasting for one (1 sec) and two (2 sec) seconds. nd = none detected.

The survival of gametophytes and plantlets subjected to natural air drying under ambient and high humidity conditions is summarised in Table 6.1. Complete gametophyte mortality was achieved after ambient air drying for 2–3 d at 10 ^{o}C (Table 6.1). While in an initial experiment 100% mortality was achieved after 2 d (Figure 6.2), in a second experiment a single surviving gametophyte was present after this time but dead at 3 d. At 20 ^{o}C gametophyte survivorship was reduced, with an exposure time of 12 h being sufficient to achieve 100% mortality in two consecutive experiments (Table 6.1). The survival of plantlets exposed to ambient humidity was comparable to

Table 6.1 Summary of lethal exposure times for gametophytes and plantlets subjected to natural air drying under ambient (55–85% RH) and high humidity (> 95% RH) conditions at 10 and 20 °C.

Treatment	Temperature (°C)	Lethal Exposure Time	
		Gametophytes	Plantlets
Air drying (ambient humidity, 55–85% RH)	10	2–3 d	3 d
	20	12 h	1 d
Air drying (high humidity, > 95% RH)	10	> 8 wk	8 wk
	20	6 wk	3 wk

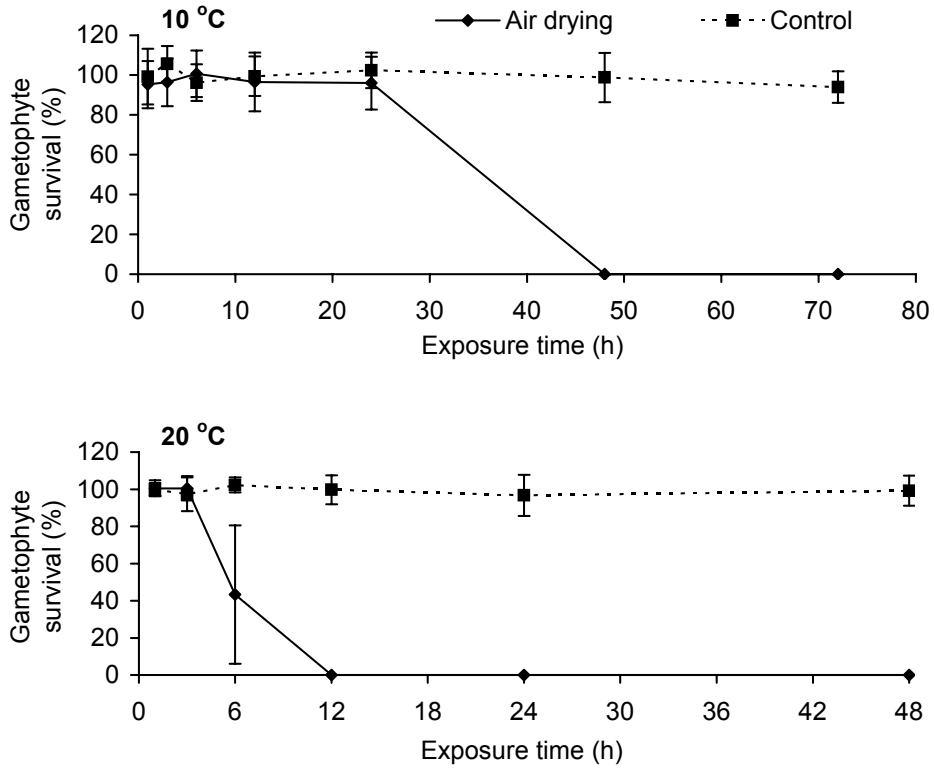

Figure 6.2 Mean (± SD, n = 5) survival of *Undaria* gametophytes subjected to natural air drying (ambient humidity, 55–85% RH) at 10 and 20 °C. Note different time scales on axes.

gametophytes. Plantlet mortality was 100% after 3 d at 10 °C and after 1 d at 20 °C, although most plantlets were dead after 12 h at the latter temperature.

In contrast to the ambient humidity results, *Undaria* survived for several weeks at high humidity (Table 6.1). At 20 °C, a 6 wk exposure was 100% lethal to gametophytes. In the 10 °C treatment an end-point was not achieved, with live gametophytes still present after the maximum exposure period of 8 wk, at which time approximately half of the treated and control wells contained a few survivors. The competency of both treated and control gametophytes after this time is questionable, however, because attempts to initiate plantlet development from the cultures were unsuccessful. The tolerance of plantlets to high humidity exposure was less than for gametophytes. A mortality of 100% was achieved after 8 wk at 10 °C and after 3 wk at 20 °C, although at these temperatures it was estimated that < 5% of plantlets were alive after 4 and 2 wk respectively.

6.3.2 Efficacy of freshwater immersion for *Undaria* on seed-mussels

The mortality response of gametophytes after freshwater immersion (Figure 6.3) was similar to that for air drying. At 10 °C, 100% mortality occurred after 2 d in consecutive experiments, with > 80% mortality after 1 d. At 20 °C, gametophytes were all dead after 1 d, with > 90% mortality after 12 h. Logistic regression closely modelled the trend shown for gametophytes in Figure 6.3, and indicated that complete gametophyte mortality could be achieved with freshwater immersion for 43 and 22 h at 10 and 20 °C, respectively (McFadden's rho-squared ≥ 0.50). *Undaria* plantlet survival in freshwater was considerably less than for gametophytes, with a 10 min immersion time sufficient to kill all plantlets at both treatment temperatures.

Mussels survived 5 d of freshwater immersion at 10 °C, with attachment in both treated and control batches close to 100%. Interestingly, there was a decrease in attachment at 1 and 2 d in both treatments and controls (Figure 6.4), which was attributable to the largest mussels in each batch. Logistic regression analyses showed that the overall survival trend for the treatment was not significantly different (Chi-square P = 0.25) from the null model for which 100% survival is assumed. At 20 °C, attachment of treated and control mussels was close to 100% for the first 3 d, but declined on subsequent days in the treatments. The logistic model for the 20 °C data achieved a relatively poor fit (McFadden's rho-squared = 0.28), suggesting that a 2 d treatment

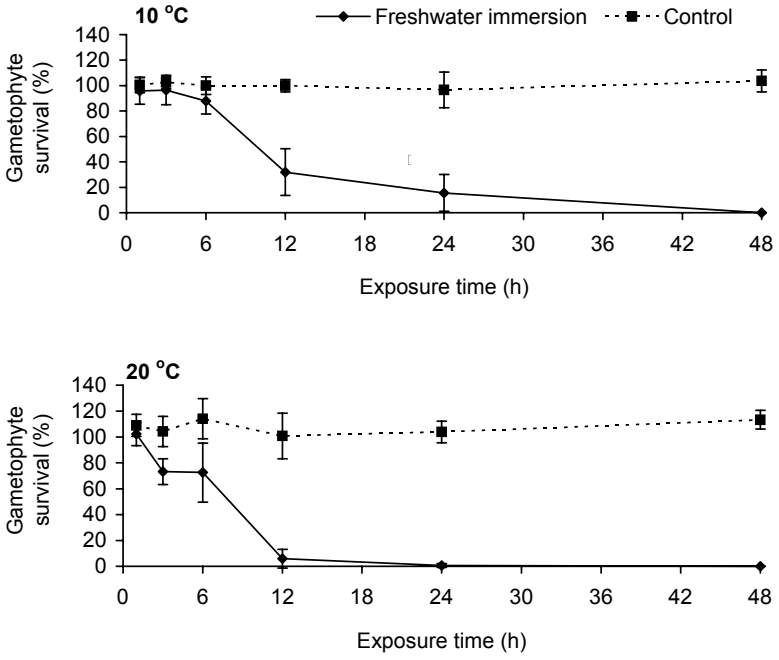

Figure 6.3 Mean (± SD, n = 5) survival of *Undaria* gametophytes after immersion in freshwater at 10 and 20 °C.

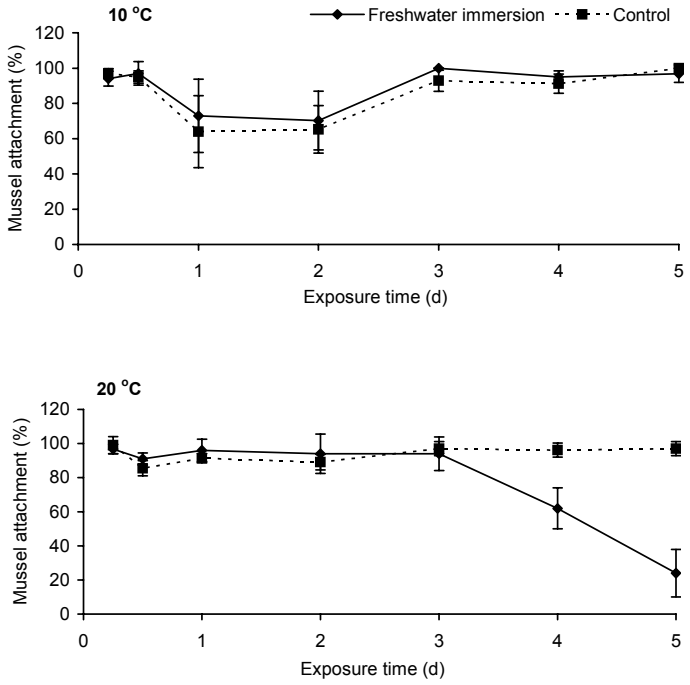

Figure 6.4 Mean (± SD, n = 5 batches of 20 mussels) mussel attachment after immersion in freshwater at 10 and 20 °C.

would reduce mussel survival to 90% when our experimental observations indicated otherwise.

The comparison of two mussel size classes revealed a level of attachment close to 100% in all cases except for large mussels immersed for 2 d (Figure 6.5), consistent with the findings above. Attachment of these mussels was significantly less than for small mussels in both treatments($P < 0.001$) and controls ($P < 0.05$). However, the treatment vs control difference for large mussels immersed for 2 d was not significant ($P = 0.74$) and, more importantly, the survival of all treated mussels after 6 months was close to 100%. Hence, long-term mussel health following freshwater immersion was better than the short-term attachment measure suggested.

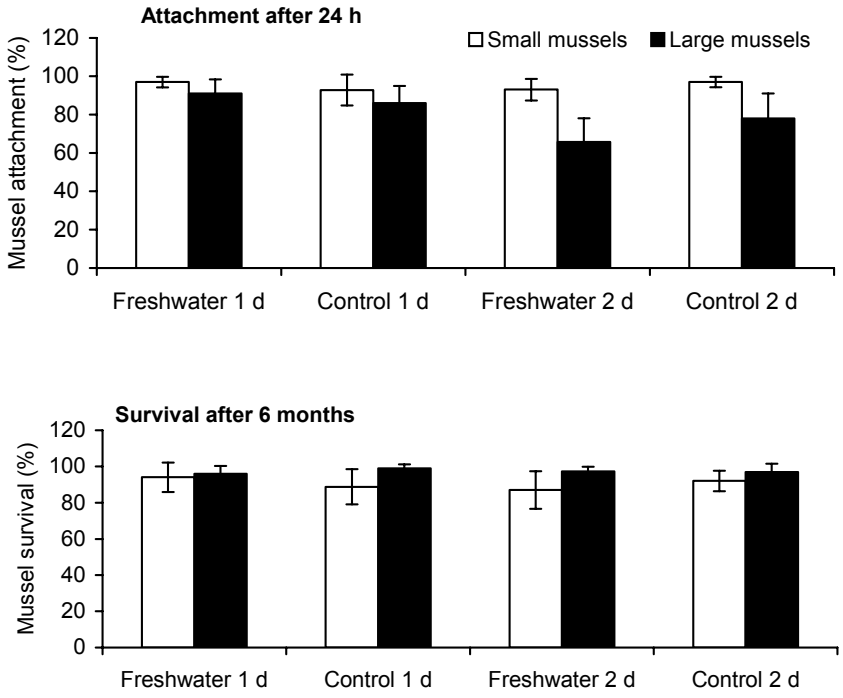

Figure 6.5 Mean (± SD, n = 5 batches of 20 mussels) attachment and survival of small (19–29 mm) and large (30–45 mm) mussels, after immersion in freshwater at 10 ^{o}C for one (1d) and two (2d) days.

A comparison of the results from the mussel experiments with the findings for *Undaria* (Figure 6.3) suggests that a 2 d freshwater immersion will meet the seed-mussel survival criterion of ≥ 90%, while at the same time achieving 100% *Undaria* mortality.

By contrast with the freshwater (≤ 1 psu) results, however, *Undaria* gametophyte survival was greatly enhanced at salinities of ≥ 2 psu (Table 6.2). In fact after 2 d at 8 psu, gametophyte survival was comparable to seawater controls in 80% of replicates.

Table 6.2 Semi-quantitative assessment of gametophyte survival at different levels of salinity and exposure. Data are the range of n = 20 estimates for each treatment and a sterile seawater control.

Salinity (psu)	Exposure time (h)			
	12	24	36	48
≤ 1	0–2	0–1	0–1	0
2	2–3	0–2	0–1	0–1
4	3	2–3	1–2	1–2
6	3	3	2–3	2–3
8	3	3	3	2–3
Control	3	3	3	3

Note 1: Survival ranked as: 0 = all dead, 1 = most (> 75%) dead, 2 = similar numbers alive vs dead (25–75% alive), or 3 = most (> 75%) alive

6.3.3 Efficacy of hot water immersion for *Undaria* on seed-mussels

Exposure times that resulted in complete gametophyte mortality at 35, 45 and 55 ^{o}C were 10 min, 45 sec and 5 sec, respectively (Figure 6.6). Mean survival in some of the controls was highly variable, reflecting a greater mortality in some of the cultures because of contamination by protozoa. Assuming mortality in the treatments was primarily attributable to heat effects, logistic regression analyses indicated that complete mortality at 35, 45 and 55 ^{o}C could be achieved at exposure times of 16 min, 47 sec and 4 sec respectively. The 35 ^{o}C result is notably more conservative than our experiments indicated, despite a good fit of the logistic model (McFadden's rho-squared ≥ 0.50). Plantlet survival in hot water was considerably less than for gametophytes, with exposure times of 30, 5 and 1 sec required to achieve 100% mortality at 35, 45 and 55 ^{o}C, respectively.

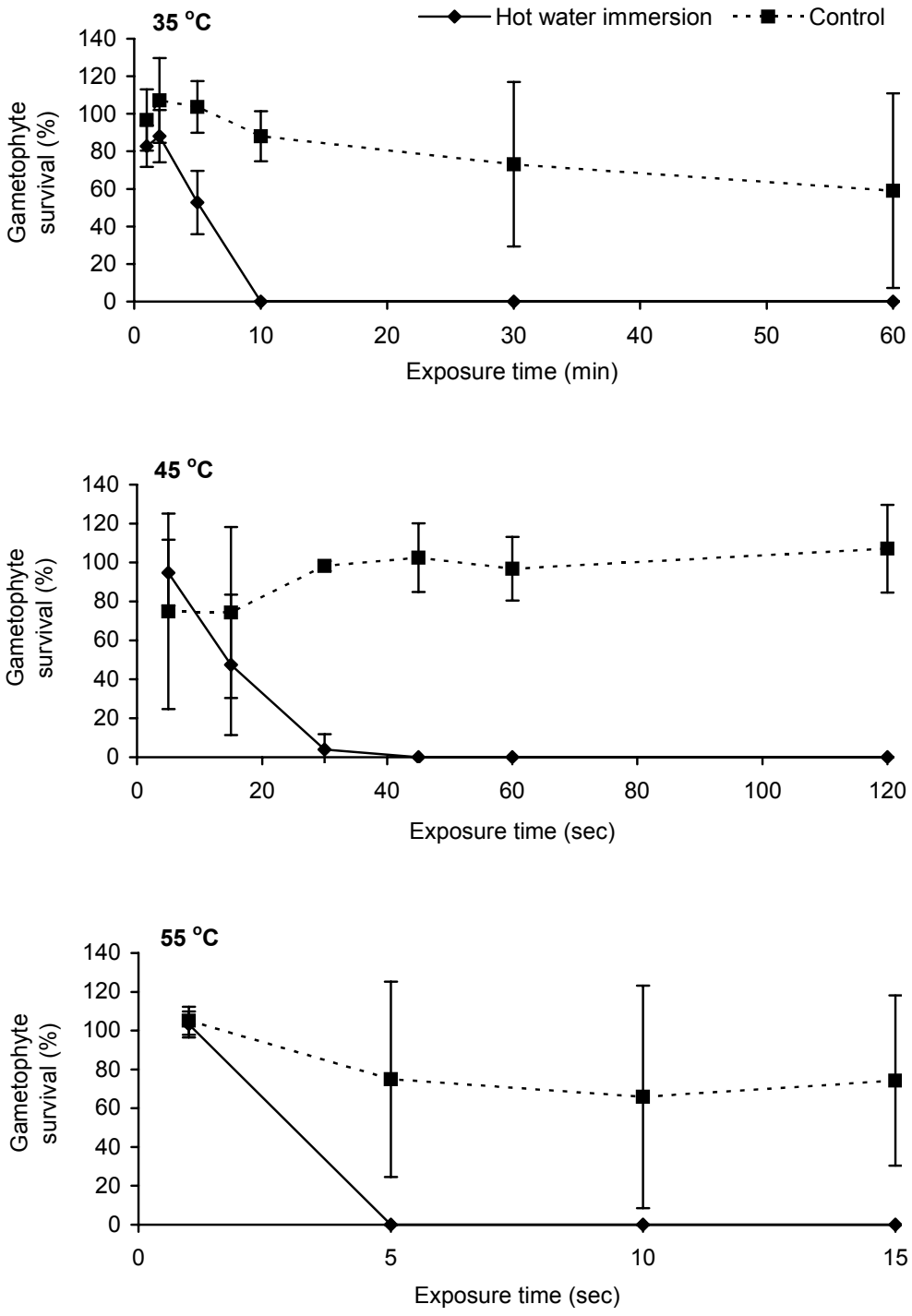

Figure 6.6 Mean (± SD, n = 5) survival of *Undaria* gametophytes after immersion in hot water at 35–55 °C. Note different time scales on axes.

Mussel attachment across the 35–55 °C range is shown in Figure 6.7, with logistic regression providing models of reasonable fit (McFadden's rho-squared ≥ 0.32). Results from the application of the logistic models for both *Undaria* and mussels (Table 6.3) indicate that treatments severe enough to achieve complete gametophyte mortality at 35 and 45 °C are likely to have an adverse effect on mussel health. Conversely, treatments benign enough to maintain at least 90% mussel attachment will require immersion times that may not result in adequate *Undaria* mortality (e.g., 68% mortality for a 6 sec exposure at 35 °C). By contrast, model predictions indicate that at 55 °C, complete *Undaria* mortality can be achieved while maintaining post-treatment mussel attachment at a 90% minimum.

Further investigation of the effects of a 55 °C treatment (for 5 sec), combined with a simulated 36 h transport phase, indicated that mussels could adequately survive the combined stresses of treatment and transport (Figure 6.8). Mussel attachment 24 h post-treatment was 90–98% in treated batches, even though it did not exceed the 90% threshold in the controls. Survival after 6 months was 71–81% in mussels exposed to both the treatment and the transport phase (or vice versa), which was comparable to mussels subjected to the treatment or transport phase in isolation. While this did not meet the 90% acceptance criterion, this level of survival was not significantly different ($p \geq 0.374$) to the controls (Figure 6.8).

Table 6.3 Association between mussel attachment (across the 75–90% range), *Undaria* gametophyte mortality and exposure time, as derived from logistic models applied to survivorship data for the 35, 45 and 55 °C hot water treatments.

Mussel attachment (%)	*Undaria* mortality (%)	Exposure time (min)	*Undaria* mortality (%)	Exposure time (sec)	*Undaria* mortality (%)	Exposure time (sec)
	35 °C		**45 °C**		**55 °C**	
90	67.59	6	97.44	32	> 99.99	6
85	98.51	12	98.92	36	> 99.99	8
80	99.89	16	99.44	39	> 99.99	10
75	99.99	20	99.67	42	> 99.99	11

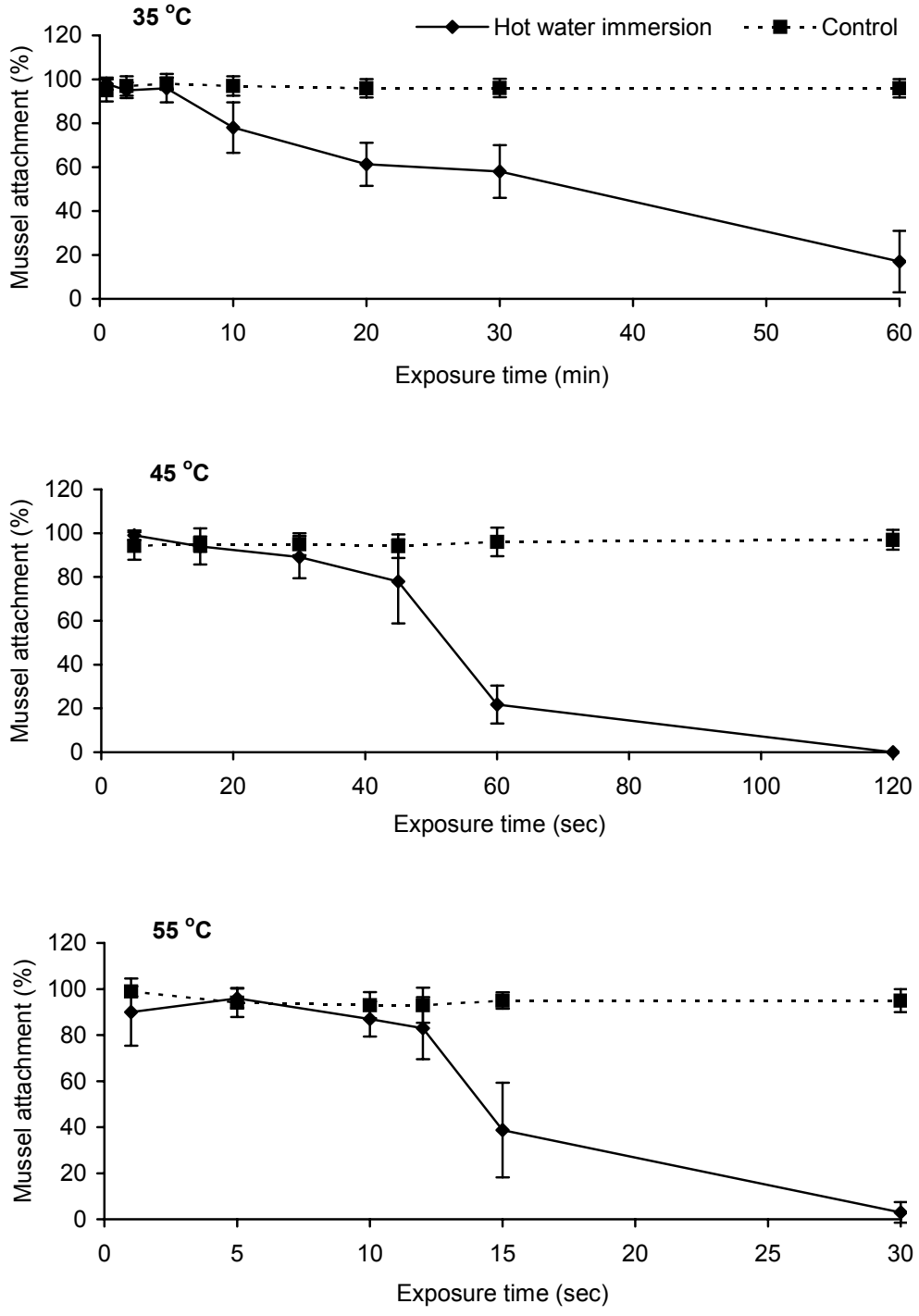

Figure 6.7 Mean (± SD, n = 5 batches of 20 mussels) mussel attachment after immersion in hot water at 35–55 °C. Note different time scales on axes.

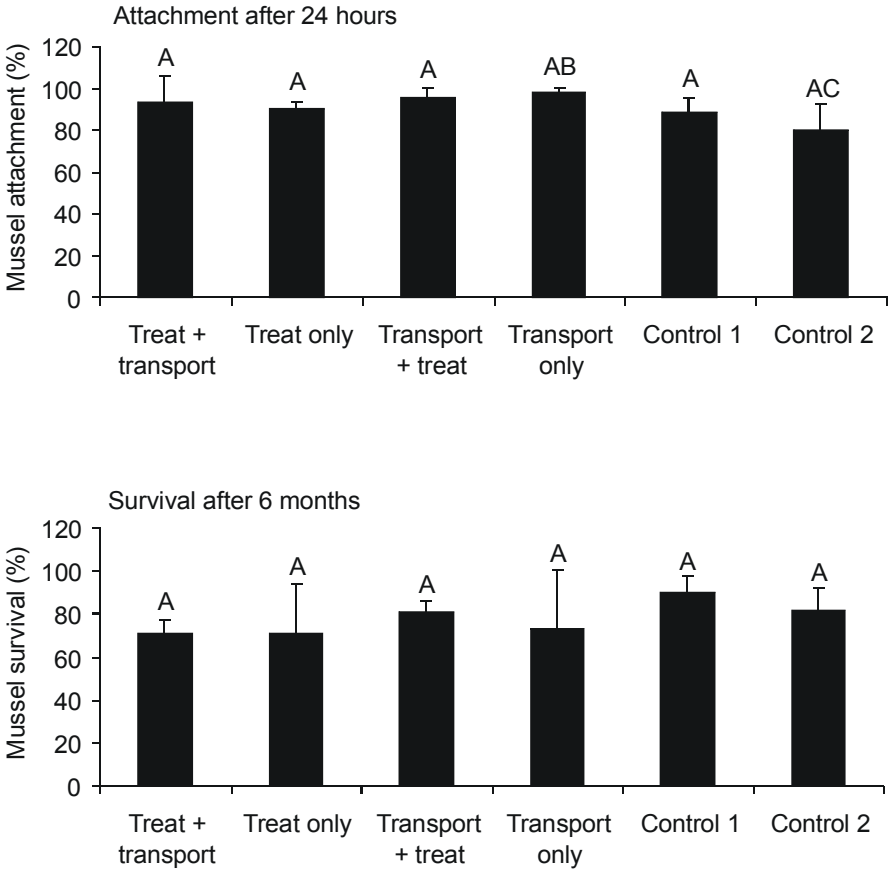

Figure 6.8 Mean (± SD, n = 5 batches of 20 mussels) mussel attachment and survival after immersion in hot water (55 ^{o}C, 5 sec) in combination with air exposure for 36 h to simulate inter-regional transport. A significant ($p < 0.05$) difference in attachment between transport and control 2 is indicated.

6.4 DISCUSSION

6.4.1 Water blasting and air drying as treatments for *Undaria* on equipment

Water blasting at pressures easily achievable with standard equipment has potential application in removal of microscopic life-stages of *Undaria* from aquaculture vectors. The fact that the method removed gametophytes from fissures within *Paphies subtriangulata* shells suggests that further investigation into its efficacy and potential applications would be worthwhile, especially as a tool for cleaning structures such as long-line floats. One of the advantages of this method is that it is likely to be less species-specific in its mode of action than approaches based on physiological tolerance

to desiccation, osmotic stress, or heat stress. As such, it is likely to be effective against bio-fouling organisms in addition to *Undaria*, potentially leading to wider pest management benefits.

The natural air drying trials indicated that at humidity levels representative of typical conditions around New Zealand (~65 to 85% RH), *Undaria* can survive for at least 2 d at 10 ^{o}C, and less when it is warmer and hence evaporation greater. These observations are largely consistent with previous laboratory studies (e.g., Sanderson and Blackburn 1994), although in a field situation air drying effects would probably be more rapid than in a laboratory (e.g., because of combined effects from sunshine, wind, etc). This is consistent with the fact that *Undaria* in New Zealand is not found higher on the shore than about a low neap-tide level, where it would be exposed to air for only 4 h during a spring low tide (Hay and Villouta 1993; Brown and Lamare 1994; Forrest and Taylor 2002).

By contrast with ambient humidity conditions, when high humidity conditions are present it is apparent that *Undaria* survival is greatly enhanced, and can probably extend to at least two months. Hence to minimize the risk of *Undaria* transfer on aquaculture ropes that are stored in bags (which is the most common practice), the bags would need to be kept out of the water for at least this length of time before being re-deployed. Survival for a similar duration under high humidity, and the marked contrast with ambient humidity conditions, has also been reported for other macroalgal pests including *Caulerpa taxifolia* (Sant et al. 1996) and *Codium fragile* ssp. *tomentosoides* (Schaffelke and Deane 2005), and is recognised as a major contributing factor in the human-mediated spread of these species.

6.4.2 Freshwater immersion

The effect of freshwater immersion on gametophytes was similar to the air drying trials in that the pattern of survivorship and total exposure times required to achieve complete *Undaria* mortality at 10 and 20 ^{o}C were comparable, with greater tolerance exhibited at 10 ^{o}C. This differential temperature effect is consistent with the findings of Saito (1962) who reported greater survival of *Undaria* at lower temperatures in low salinity water. One aquaculture company in New Zealand has now started using freshwater immersion to kill bio-fouling organisms associated with stripped rope. Preliminary assessment under field conditions indicates that decomposition of the residual fouling

biomass leads to the development of anoxic conditions (reduced dissolved oxygen, and elevated levels of un-ionised ammonia and sulphide) within the treatment water, which would be expected to greatly accelerate the effects of freshwater alone (authors, unpubl. data).

Freshwater trials with mussels suggested that immersion could be undertaken for a sufficient period (1–2 d depending on temperature) to achieve 100% mortality of *Undaria* without adversely affecting mussel health. While the hermetic response of mussels and other bivalves to salinity extremes is well documented (e.g, Davenport 1979; Berger and Kharazova 1997), the level of freshwater tolerance exhibited by the New Zealand Greenshell™ mussel is surprisingly high given that this species is more typically associated with marine waters of 30–35 psu (Jeffs et al. 1999). This level of tolerance suggests that seed-mussels could be treated in freshwater (e.g., immersed in bins) while being transported between the main aquaculture regions. The fact that low salinity conditions develop during mussel immersion is problematical, however, in that gametophyte survival is extended beyond 2 d. Because it is not practical to hold the mussels for longer than 2 d, the treatment water would need to be exchanged during transport, so that salinity was maintained at ≤ 1 psu to ensure *Undaria* mortality. This is possible, but would create logistic difficulties for implementation at a field scale.

6.4.3 Hot water

As was the case for the 10 and 20 °C freshwater treatments, *Undaria* gametophytes were considerably more tolerant of hot water than were plantlets. The exposure time required to achieve complete gametophyte mortality dramatically reduced with increasing temperature, consistent with the effects of heat on *Undaria* zoospore viability (Mountfort et al. 1999). Our results indicating 100% gametophyte mortality at 55 °C were consistent with the findings of Webb and Allen (2001) who recorded 100% mortality after exposure to 60 °C water for 5 sec.

In terms of developing a hot water-based treatment method that is effective against *Undaria*, it is encouraging that mussels were able to withstand hot water exposure at 55 °C for 5 sec in combination with a simulated inter-regional transport phase. In fact, subsequent work revealed that mussels can tolerate treatments of up to 60 °C combined with the transport phase. At 65 °C, however, mussels can tolerate the treatment or transport phases alone, but the two stressors in combination reduce long-term survival

to < 30% (authors, unpubl. data). This work suggests that the 55–60 ^{o}C temperature range is likely to be optimal for treatment.

Clearly, evaluation of heat treatment effects under field conditions would be an important further step. For *Undaria* this would include field assessment of heat effects on complex substrata (e.g., rope, shell), which may have different thermal properties to laboratory apparatus or provide refuges from short-term hot water immersion (Forrest and Blakemore 2003). In terms of effects on seed-mussels in a field situation, laboratory investigations for the present study indicated similar levels of survival irrespective of whether or not the mussels were cooled post-treatment. However, where seed-mussels are transported in 1 tonne bags the treatment would need to be applied in such a way that mussel health was not compromised by residual heat effects, for example because of retarded cooling in the middle of the bags.

6.5 CONCLUSIONS

The environmentally friendly methods described in this paper would all be applicable in the treatment of equipment such as mussel farm floats and rope, with the preferred method selected based on cost, practicality and other constraints. Natural air drying is particularly appealing, but for equipment such as rope stored in bags, may require exposure times of weeks to months to ensure effective treatment. Developing a method for eradication of *Undaria* from mussel seed-stock is more problematical because of the paramount importance of identifying treatment conditions that do not compromise mussel health. Freshwater immersion has the greatest potential as a simple, low-cost method that can be applied for this purpose, although the need to maintain negligible salinity levels in the treatment water introduces logistic constraints at a field scale. It may be feasible to develop methods for seed-stock based on heat treatment, but this is likely to involve greater costs than freshwater immersion, and requires field validation to confirm both the efficacy against *Undaria* and to identify a method of implementation that ensures mussel survival is not compromised. A useful direction for further research would be to also consider the efficacy of these methods against other bio-fouling pests. While *Undaria* is of direct interest as a pest organism, and provides a useful model to explore some of the issues that arise in the development of treatment methods, greater benefits in applying the various treatments will emerge from methods that are not species-specific, but aim to reduce vector risks overall.

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Chapter 7

Efficacy of Acetic Acid Treatments in the Management of Biofouling

PREFACE

This chapter describes an evaluation of dilute acetic acid as a treatment for marine biofouling. *Undaria* is used as a model organism that allowed rapid evaluation of a range of potential methodological approaches for treatment. The relative ease with which different life-stages of *Undaria* are amenable to experimental manipulation also provided the opportunity to assess acetic acid effects on microscopic as well as visible life forms. However, the Chapter also extends beyond *Undaria* to consider effects on a range of other common fouling organisms, and evaluates the application of effective treatments to reduction of biosecurity risks associated with seed-mussel transfer, hence builds on the work started in Chapter 6. This work has been published as follows:

Forrest BM, Hopkins GA, Dodgshun TJ, Gardner JPA. 2007. Efficacy of acetic acid treatments in the management of marine biofouling. Aquaculture 262: 319-332

The version presented here is the one resubmitted after response to journal page proof changes. This is my own work, with co-authors providing technical support or manuscript review.

Abstract

The expansion of artificial habitats and aquaculture activities in coastal environments has been accompanied by an increased demand for tools to mitigate the effects of biofouling pests. One approach is to manage anthropogenic pathways to prevent the spread of established pest organisms to uninfected localities that are beyond their natural dispersal capacity. This paper describes the efficacy of acetic acid treatments against a variety of cosmopolitan fouling taxa, and evaluates a potential application in the treatment of foulers transported with movements of shellfish seed-stock between mussel farming areas in New Zealand. Laboratory and field experiments demonstrated that immersion in 4% acetic acid (in seawater) for as little as 1 minute can eliminate many soft-bodied fouling organisms, with lower concentrations requiring longer immersion times. The effects of immersion treatment were enhanced when combined with a 24 h air exposure phase to simulate the inter-regional transport of mussel seed-stock. We demonstrate that it is possible to cost-effectively treat mussels to eliminate the majority of problematical foulers without resulting in significant adverse effects to the stock either by: (i) a 4% treatment followed by a rinse to remove the acetic acid residue before transport, or (ii) application of the 4% treatment at the end of the transport phase. A concentration of 4% is equivalent to the acetic acid content of domestic vinegar, hence does not represent a significant environmental or occupational risk provided appropriate measures are put in place for handling and waste disposal. Acetic acid concentrations remain stable over time in the presence of organic matter, but may change during repeated use of treatment solutions. To ensure treatment criteria are being achieved, field determination of acetic acid levels can be made using simple titration-based approaches. Because of an apparent buffering effect in the case of sequential shellfish seed-stock immersion, pH could not be used to estimate acetic acid concentrations in this instance, but may provide a simple and reliable field-based indicator for other fouling treatments. Further work to refine the treatment method should seek to maximise the 'window' between pest mortality and mussel survival, to provide assurance that high risk species can be eliminated with minimal risk of adverse effects on seed-stock. Where treatments that are completely effective against all pest organisms result in unavoidable mussel mortality, decisions about whether or not to apply them must balance treatment costs and benefits against the unmanaged risks and consequences of pest incursion.

7.1 INTRODUCTION

The development of coastal environments has resulted in the creation of extensive areas of artificial habitat, including rock walls, wharf pilings, marina pontoons, vessel moorings and aquaculture structures. The association of non-indigenous fouling species with such structures and their proliferation at high densities has been documented in a number of studies (e.g., Hay, 1990; Clapin and Evans, 1995; Floc'h et al., 1996; Hay and Villouta, 1993; Lambert and Lambert, 2003; Bulleri and Airoldi, 2005; Coutts and Forrest, 2007). Such infestations can lead to increased costs for management (e.g., defouling) and, in the case of shellfish aquaculture, economic losses resulting from over-settlement and smothering of the crop (Verlaque, 1994; Hecht and Heasman, 1999;

Carver et al., 2003; Lane and Willemsen, 2004). Furthermore, infested structures may function as reservoirs that facilitate the spread of pest species to areas where they previously did not occur, through natural dispersal or through the infection of vessels and other vectors (Airoldi et al., 2005; Floerl and Inglis, 2005).

New Zealand has a number of fouling pests whose adverse economic effects have been documented, or whose actual or potential impacts on natural ecosystems are also recognised. These include the Asian kelp *Undaria pinnatifida*, solitary ascidians such as *Ciona intestinalis* and *Styela clava*, and the colonial ascidian *Didemnum vexillum* (Forrest and Taylor, 2002; Carver et al., 2003; McDonald 2004; Coutts and Forrest, 2005, 2007; Le Blanc, *pers. comm.*). In the many situations where widespread eradication of such organisms is not feasible, the mitigation of adverse effects associated with their invasion can theoretically be achieved in two main ways: (i) managing anthropogenic vectors to prevent spread to uninfected localities that are beyond their natural dispersal capacity, or (ii) reducing pest density or biomass on infected structures to levels that avoid significant adverse effects (Forrest et al., 2006), for example by mitigating direct effects (e.g., smothering of aquaculture stock) or reducing the reservoir of propagules for secondary spread.

The development of tools for such purposes is at a relatively early stage in the marine environment, although some promising progress has been made (e.g., McEnnulty et al., 2001; Wotton et al., 2004). Among the various approaches evaluated, a number of low cost environmentally-friendly methods have been described, for example the use of polyethylene wrapping to contain and smother fouling biota on vessels, wharf pilings and marina pontoons (Coutts and Forrest, 2005, 2007), and the application of air drying, water blasting, fresh water and hot water to manage fouling on aquaculture vectors (Forrest and Blakemore, 2006). The feasibility of such approaches may be limited by their slow rate of treatment, in which case biocidal agents could potentially be used to accelerate treatment effects. In this regard, considerable research has been undertaken on the anti-fouling efficacy of a variety of chemicals such as chlorine, lime and brine solutions (e.g., McEnnulty et al., 2001; Carver et al., 2003; Rajagopal et al., 2005). The efficacy of continuous low dose chlorination, for example, has been demonstrated as a control agent for mussel fouling in cooling water systems (Rajagopal et al., 2003; Taylor, 2006). Recent research has also highlighted the efficacy of acetic acid (the

active ingredient in vinegar) for situations where a fast-acting single dose treatment is required (Carver et al., 2003).

This paper further evaluates the efficacy of acetic acid as a rapid treatment agent for fouling. We first describe effectiveness against a variety of cosmopolitan fouling organisms (including recognised pests) in relation to concentration and exposure time. This provides a knowledge base that is relevant to a wide variety of applications. Examples include the development of immersion treatments for infected moorings and other equipment (e.g., marine farm ropes), or acetic acid additions to marina pontoons or vessels that have been encapsulated using methods described by Coutts and Forrest (2005, 2007). In such instances, the application of the treatment is relatively straight-forward because the primary goal is to administer the chemical to target organisms at a sufficient concentration and duration to ensure mortality. In this paper we examine a less tractable situation arising in shellfish aquaculture where fouling pests may inadvertently be transferred with seed-stock movements between marine farming regions; a biosecurity risk that has been recognised internationally for many years (e.g., Wolff and Reise, 2002; Wonham and Carlton, 2005). The challenge in this case is to develop treatments that are effective against target pest species but do not result in significant adverse effects on the stock.

7.2 MATERIALS AND METHODS

7.2.1 General approach

Laboratory and field experiments were conducted with some common fouling taxa, including pest organisms known to be transferred via aquaculture and other vectors, to identify lethal acetic acid treatments for a range of concentrations and exposure times. The tolerance of shellfish seed-stock to effective acetic acid treatments was then determined using cultivated green-lipped mussels, *Perna canaliculus* (New Zealand's dominant aquaculture species). Finally, some practical considerations for implementation of acetic acid treatments at a field scale were evaluated. A concentration of 4% acetic acid was the maximum used in most of the trials on the basis that this was equivalent to the content of domestic vinegar, and hence would not represent a significant occupational or environmental risk provided appropriate measures were put in place for handling and waste disposal.

The key focus was to evaluate methods suitable for managing fouling associated with mussel seed-stock transfers, hence the design of our experiments needed to account for mussel industry handling practices. In this respect, seed-stock moved between main aquaculture regions (typically hundreds to thousands of kilometres apart) are transported in large (~1 m^3) bags after being stripped from the crop line and subjected to a vigorous declumping, washing and screening process to remove most of the visible fouling biomass (Forrest and Blakemore, 2006). Shorter distance transfers may be conducted with the mussels remaining intact on the crop line. In both cases the seed-stock can be out of the water for an extended period (from a few hours to > 1 d) during transfer. Hence, experiments with fouling organisms and mussel seed-stock were designed to evaluate the effects not only of the acetic acid treatment, but also the effects of treatment combined with air emersion during transport. Our goal was to identify treatment conditions that resulted in complete mortality of fouling species while ensuring an acceptable (≥ 90%) level of mussel survival.

7.2.2 Acetic acid effects on fouling

This component of the study focussed on a suite of fouling organisms that we used as indicators of treatment effects. These were *Undaria pinnatifida* and *Ciona intestinalis* which are recognised fouling pests, and nine other fouling taxa that are globally widespread (e.g., Furlani, 1996) and represent a range of morphologies (e.g., filamentous, soft-bodied, calcareous) that we regarded as useful surrogates for structurally and functionally similar pests. The latter were solitary tunicates (*Cnemidocarpa bicornuata*, *Corella eumyota*), colonial tunicates (*Botryllus schlosseri*, *Botrylloides leachi*), encrusting (*Watersipora subtorquata*) and erect (*Bugula neritina*) bryozoans, tube-dwelling serpulid (*Hydroides elegans*) and terebellid (Family Terebellidae) polychaetes, and a filamentous green macroalga (*Cladophora* sp.).

Except for *Undaria*, we were able to collect the indicator organisms from passive fouling on 1 m lengths of weighted rope suspended for 8 months from marina pontoons. For *Undaria* it was necessary to create artificial cultures to achieve a sufficient density of plants. The different life-stages of *Undaria* are relatively amenable to experimental manipulation, therefore working with this species also provided the ability to rapidly evaluate a range of potential methodological approaches prior to treatment of the fouled ropes. Furthermore, *Undaria* provided the opportunity to assess acetic acid effects on microscopic as well as visible life forms, recognising that while mussel seed-stock

declumping and washing greatly reduces visible fouling, fragments and microscopic life-stages can survive the process (Forrest and Blakemore, 2003, 2006).

Experiments with *Undaria*

Initial work with *Undaria* compared the effects of acetic acid on the survival of gametophytes and plantlets (sporophytes < 50 mm length), and on the viability of reproductive (sporophyll) tissue to account for instances where either mature sporophytes or fragments of sporophyll are transferred with seed-stock (Table 7.1). Acetic acid concentrations mixed in both seawater and fresh water were compared, to determine the most effective diluent for subsequent work. Gametophytes were cultured on sterile 24-well Falcon™ plates, whereas plantlets were cultivated on weathered rope (Forrest and Blakemore, 2006). Sporophylls were collected from a mature *Undaria* population, with the experimental units comprising a disc (10 mm diameter) of excised tissue held within each of the 24 wells on a Falcon™ plate. These three life-stages were exposed to acetic acid treatments as indicated in Table 7.1, with a post-treatment seawater rinse used to remove any residual chemical.

Gametophyte and plantlet mortality was assessed one week post-treatment using methods described by Forrest and Blakemore (2006). For the sporophyll tissue, the post-treatment procedure involved high humidity (> 95% RH) air exposure for 24 h at 17 ^{o}C to induce partial dehydration, and then rehydration in filtered (25 μm) UV-sterilised seawater at 17 ^{o}C. This partly mimicked conditions that *Undaria* would be exposed to during inter-regional seed-mussel transfer (i.e., high humidity emersion) but, more importantly, was expected to provide optimal conditions for spore release (Saito, 1975). After 1 h of rehydration, the sporophyll disc was removed from each well, and the seawater replaced with a nutrient-enriched growth medium. The effect of the acetic acid treatment on sporophyll tissue viability, defined here as its ability to release competent spores, was assessed as the density of gametophytes attached to the bottom of each well two weeks after treatment. Viability was assigned on a ranked scale (1 – 5) to reflect gametophyte densities in each well as follows: 0 = absent; 1 = 1 – 5; 2 = 6 – 20; 3 = 21 – 50; 4 = 51 – 100; 5 = > 100.

This initial work revealed that sporophyll tissue was more resilient to the effects of acetic acid than were gametophytes or plantlets. Furthermore, while fresh water dilutions were marginally more effective, the difference was not sufficient to offset the

Table 7.1 Summary of experiments conducted with *Undaria pinnatifida*, fouled ropes, and mussel seed-stock.

Experimental component	Acetic acid (%)	Exposure time (min)	Diluent	Experimental temperature (^{o}C)	End-point	n
Undaria gametophytes	0.1 – 2	1	Seawater & fresh water	Ambient	Mortality	5
Undaria plantlets	0.1 – 2	1	Seawater & fresh water	Ambient	Mortality	5
Undaria sporophyll	0.1 – 2	1	Seawater & fresh water	17 ^{o}C air for 24 h post-treatment	Viability [1]	5
Undaria sporophyll	2 and 4	1, 2, 3, 4	Seawater	17 ^{o}C air for 24 h post-treatment	Viability [1]	4
Fouled ropes (field) [2]	2 and 4	1, 2, 3, 4	Seawater	Ambient 6 – 17 ^{o}C air for 24 h, seawater 15 – 16 ^{o}C during ongrowing	Mortality	1–4
Seed mussels (laboratory) [2]	4 and 8	2	Seawater	10, 15 and 20 ^{o}C air for 24 h	Attachment	4 [3]
Seed mussels (field) [2]	4	1, 2, 4	Seawater	Ambient 11 – 18 ^{o}C air for 24 h, seawater 15 – 17 ^{o}C during ongrowing	Survival	3 [3]

Notes:

[1] See text for details of sporophyll viability method.

[2] Experiments with fouled ropes and mussels included evaluation of 'rinse' vs 'no rinse' post-treatment.

[3] Replicates used in the mussel experiments each comprised 20 mussels for the laboratory work and approximately 70 mussels for the field trial.

practical convenience of using seawater during routine field operations. Hence, further experiments assessed the effects of seawater dilutions of acetic acid on sporophyll tissue using the method described above, at the concentrations and immersion times indicated in Table 7.1. These experimental conditions were chosen on the basis of pilot work indicating that effective treatments at concentrations of 1% or less would require exposure times > 10 minutes, which would not be practical within the context of many aquaculture operations. The design included a comparison of the effects of a post-

treatment seawater 'rinse' vs 'no rinse', with the latter involving leaving the residual acetic acid on the sporophyll discs prior to their rehydration. The purpose of this work was primarily to identify treatments that were completely lethal to *Undaria* (i.e., only major changes among the different treatments were of interest), therefore statistical analyses were not conducted.

Experiments with fouled ropes

The 1 m lengths of fouled rope were treated using the same concentrations and exposure times described above for *Undaria* (Table 7.1), and similarly included a comparison of rinse vs no rinse post-treatment. To understand the limiting processes operating during mussel seed-stock movement, we compared the effects of treatment and transport in isolation, and the combined effects resulting from treatment followed by transport, and vice versa. Transport was simulated by holding treated ropes in plastic bins (covered to simulate high humidity during transport) at ambient temperatures for 24 h (Table 7.1). As a result of the substantial biomass present on the ropes (hundreds of kilograms in total), replicate ropes were included for 'control' and 'transport' only, with single ropes used for other treatments. While this did not provide a measure of treatment variability, we were nonetheless able to examine consistency in patterns of efficacy with increasing exposure time. After four weeks of on-growing from marina pontoons post-treatment, the wet weight of fouling biomass was measured on each rope, and the fouling indicator species surviving the various treatments were described.

7.2.3 Effect of acetic acid on mussel seed-stock

Short-term treatment effects on seed-stock were measured as the percentage of mussels that reattached via their byssus 24 h post-treatment, according to methods described by Forrest and Blakemore (2006). This provided a fast screening method for evaluating relative effects on mussels from a range of preliminary experiments. However, the subsequent survival of treated mussels was up to 12% less than their initial 24 h attachment, hence survival after one month of on-growing was used as the assessment end-point in trials where the long-term effects of treatment were of interest. Based on work with blue mussels, *Mytilus edulis*, we assumed that treated *Perna canaliculus* surviving acetic acid treatments for at least 1 month would not suffer longer-term effects on growth or condition (Le Blanc, *pers. comm.*). Trials with seed mussels (26 – 56 mm shell length) were designed to mimic industry handling practices and used the

same approach as described above for the fouled rope treatments. For all investigations, mussels were briefly shaken prior to treatment to induce valve closure (as occurs after industry scale declumping) to avoid the significant mortality that can occur if the mussels are gaping before immersion.

Pilot work indicated mussel attachment of > 88% (survival > 83%) across a range of concentrations (1 – 8% acetic acid) and exposure times (1 – 20 minutes). Based on this, and on results from the *Undaria* and fouled rope work, a more rigorous laboratory experiment compared mussel attachment under the conditions outlined in Table 7.1. While it was intended that field methods would be developed using 4% acetic acid as a maximum, the 8% treatment was included to gauge the potential implications of over-dosing the mussels, in the event that this inadvertently occurred during industry operations. The effects of treatment alone were compared to controls for each concentration using two-factor ANOVA (Statistica 7, StatSoft Inc.), following an arcsine square-root transformation of the raw data. A separate three-factor ANOVA examined the effects of concentration, temperature and transport effects (i.e., transport only, and treatment/transport combinations), with pairwise contrasts examined using Tukey's HSD.

Effects on mussel survival were subsequently evaluated under field conditions using the same general design as described above, with experimental conditions outlined in Table 7.1. This work included a comparison of effects on declumped mussels vs those attached to crop line, to evaluate whether declumped mussels were more susceptible to acetic acid toxicity because of: (i) shell damage during the declumping operation, or (ii) increased valve gaping and foot activity in detached (by comparison with attached) mussels, as described for several other mussel species (Rajagopal et al., 2002, 2005). Mussel survival was assessed after on-growing for 1 month at the field site, and survival analysed using two- or three-factor ANOVA in the same general way as described above for mussel attachment.

7.2.4 Use of acetic acid in field operations

A key question for using acetic acid in pest management operations is how concentrations change in relation to repeated or extended use. Concentrations could conceivably decline via dilution (e.g., treated biomass is likely to release water) or consumption (because of the organic nature of the fouling) when multiple treatments are

undertaken sequentially. Similarly, it is useful to consider whether acetic acid concentrations degrade over time, to provide guidance on how often or for how long a treatment solution can be used. Furthermore, it is important that methods are available to measure acetic acid levels in treatment solutions at field sites, so that users can ensure they are maintaining target concentrations.

To evaluate options for users to accurately determine acetic acid concentrations, we established the relationship between dilutions of acetic acid (across the 0.1 – 5% range) with pH values measured *in situ* (Inlab®413 pH electrode), using three different batches of seawater collected from the same location but under different environmental conditions. To examine the effects of repeated use of treatment solutions, ten batches (each ~ 1 kg) of seed-mussels were sequentially immersed (each for 2 minutes) in duplicate bins of 4% acetic acid in seawater. After each immersion, measurements were made of pH, and samples taken for analysis of acetic acid via titration (AOAC 18th Edn 940.15). As a comparison, the concentration of acetic acid was also estimated from the pH vs seawater relationship. To examine stability over time, a post-treatment organic-rich seawater was simulated by mixing seawater and fouling detritus to achieve a turbid solution. This solution was diluted in seawater to make up duplicate solutions of 4% acetic acid (mean total suspended solids concentration 830 g m^{-3}; APHA 20th Edn 2540C). The change in pH and acetic acid concentration of duplicate samples was measured at regular intervals over 20 days, using the methods described above.

7.3 RESULTS

7.3.1 Acetic acid effects on fouling

***Undaria* life-stages**

Concentrations of < 1% acetic acid were effective against gametophyte and plantlet life-stages of *Undaria* after a 1 minute exposure, and slightly more effective in fresh water than seawater (Table 7.2). By contrast, reproductive sporophyll tissue was relatively resilient, with a concentration of 2% insufficient to prevent the release of viable spores. However, considerable variability in sporophyll resistance to treatment effects was evident. Some experiments used relatively young plants having thin flaccid sporophylls, and viability was reduced to a score of 1 or 2 after the 1 minute exposure. By contrast, identical treatments applied to older more thickened tissue resulted in viability scores (4 – 5) that were similar to controls.

Table 7.2 Lethal concentration thresholds of acetic acid, diluted in seawater or fresh water, to different life-stages of *Undaria* following a 1 minute immersion. Results are for two experiments conducted at acetic acid concentrations ranging from 0.1 to 2%.

Life-stage and assessment end-point	Lethal acetic acid concentration (%)	
	Seawater dilution	Fresh water dilution
Gametophyte mortality	< 1	< 0.5 and < 1
Plantlet mortality	< 0.5	< 0.1
Sporophyll viability	> 2	> 2

Subsequent experiments using relatively thick sporophyll tissue indicated that the efficacy of acetic acid at concentrations of 2 and 4% was a function of increasing concentration and exposure time, with effects markedly enhanced in all 'no rinse' treatments where the acid residue was left on the sporophyll tissue before rehydration (Figure 7.1). A 4% solution was completely effective after 1 minute in the no rinse treatment, with subsequent trials indicating complete effectiveness with 45 seconds. At 2%, a 4 minute exposure was required, although this treatment was characterised by a high level of variability; for example, viable sporophyll tissue after 3 minutes was present in only one of the four replicates (Figure 7.1).

Fouled ropes

The fouled ropes developed a relatively high biomass after 8 months (mean of 8.7 kg wet wt m^{-1}), largely resulting from a heavy oversettlement by *Ciona intestinalis*. Four weeks post-treatment the mean biomass on control ropes was 57% less than the baseline measurements (Figure 7.2). Furthermore, the transport phase alone resulted in a mean biomass reduction of 82%, which was 25% greater than in the controls, and largely attributable to a reduced biomass of *Ciona*. Among the acetic acid treatments, the level of biomass reduction ranged from 84 to 100% but there was little pattern in relation to concentration, exposure time, or post-treatment rinsing (Figure 7.2). Again, this largely reflected the considerable but variable reduction in *Ciona* biomass in all treatments which, associated with variation in water retention in *Ciona* (hence variability in wet

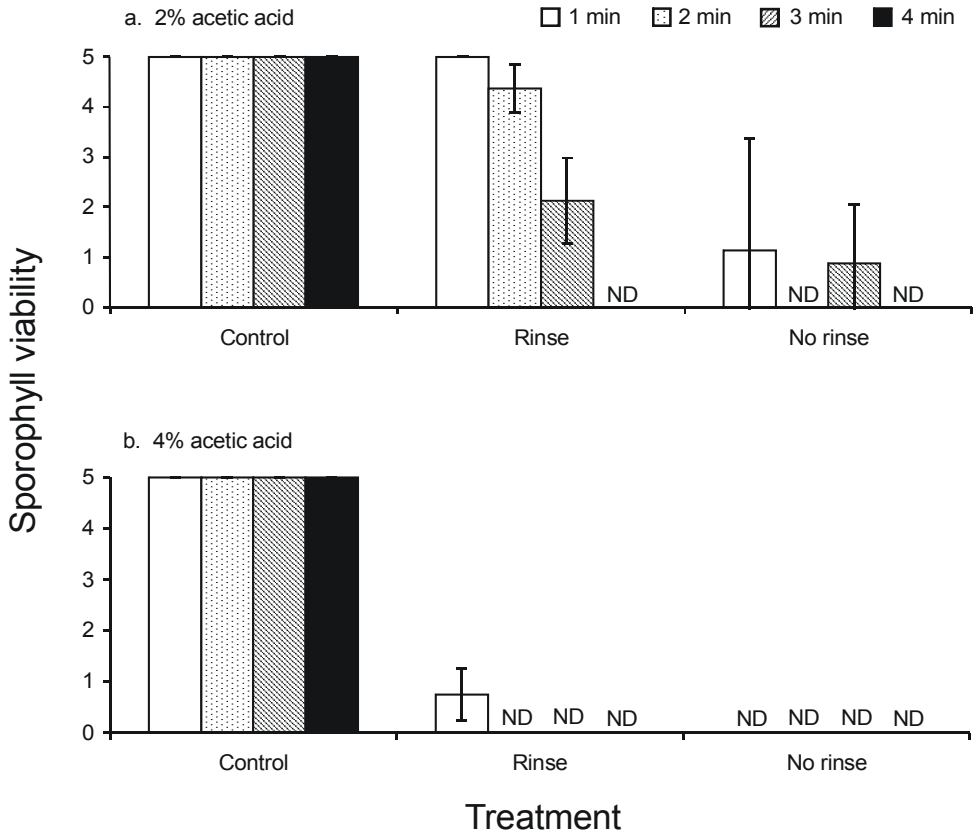

Figure 7.1 Mean sporophyll viability (± SD, n = 4) following immersion in 2 and 4% acetic acid for 1 – 4 minutes, with a comparison of effects from rinse vs no rinse. Viability was assessed semi-quantitatively according to gametophyte density two weeks post-treatment (see text section 2.2.1 for details). ND = none detected.

weight), masked any patterns in biomass decline among other taxa that might have otherwise been evident. In contrast with the biomass changes, acetic acid treatment effects were more clearly evident from changes in the indicator taxa. Results were consistent with the *Undaria* work in that they revealed an increasingly severe effect with increasing acetic acid concentration and exposure time, and as a result of not rinsing post-treatment (Table 7.3). All taxa survived the transport phase, and most survived the 2% treatment, although terebellid polychaetes and the green alga *Cladophora* sp. survived only the 1 and 2 minute immersions. In combination with transport and rinsing, 1 or 2 minute exposures to 2% acetic acid eliminated at least half of the taxa, with surviving species mainly consisting of ascidians, bryozoans and the

serpulid polychaete *Hydroides elegans* (Table 7.3). The latter species was the only one to survive the 'no rinse' treatment at 2%.

A concentration of 4% acetic acid was considerably more effective at eliminating fouling species. *Hydroides elegans* was the only species to survive all combinations of 4% treatment and transport across all exposure times, with the exception of the most severe test conditions (4% acetic acid, 4 minute exposure, no rinse) (Table 7.3). A single saddle squirt (*Cnemidocarpa bicornuata*) survived 3 minutes at 4% when the treatment was applied after the transport phase. *Cnemidocarpa* and other solitary ascidians also survived the effects of a 4% treatment alone (i.e., without the transport phase). By contrast, relatively soft-bodied colonial ascidians (*Botryllus schlosseri* and *Botrylloides leachi*) were eliminated by all of the 4% treatments.

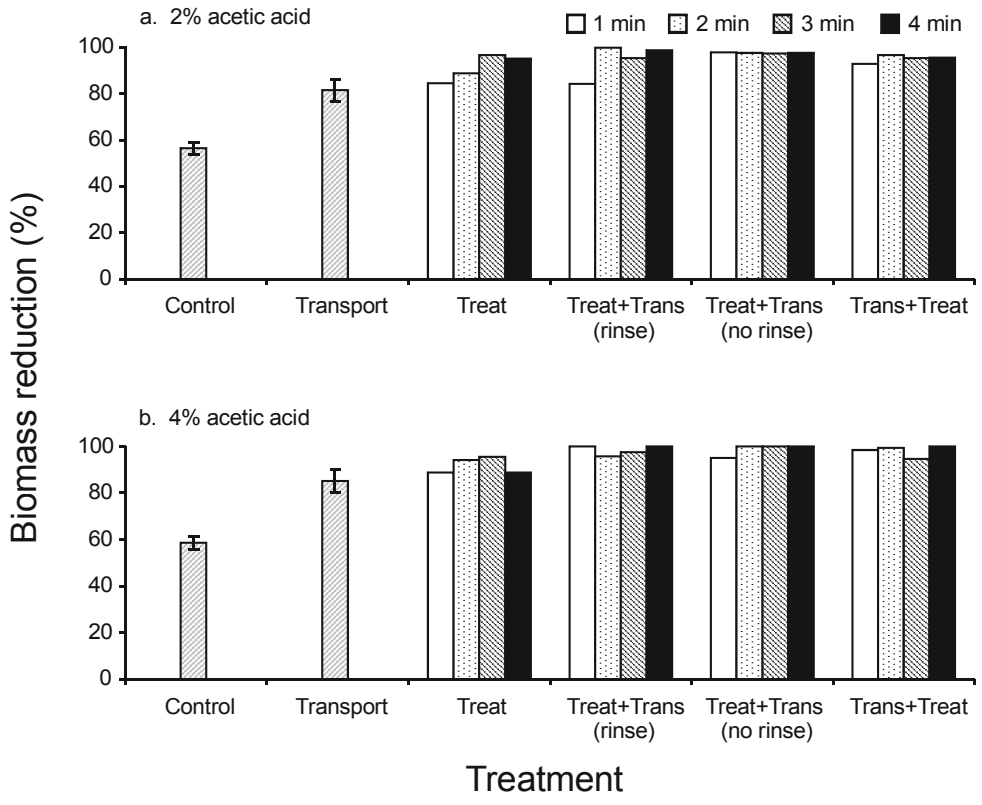

Figure 7.2 Percent biomass reduction in fouled ropes subject to various levels of acetic treatment and different types of pre- or post-treatment handling. See Section 7.2.3 for details.

Table 7.3 Survival of 10 indicator organisms (present on all ropes pre-treatment) after immersion in 2% or 4% acetic acid for 1 – 4 minutes. The effects of treatment (Treat) are shown, as well as treatment in combination with air exposure for 24 h to simulate inter-regional transport (Trans). Rinse and no rinse refer to post-treatment handling as described in Section 7.2.3. X = present on fouled ropes four weeks post-treatment, – = absent. All 10 taxa survived the effect of transport alone, hence this is not shown.

Indicator taxon	**Treat only**				**Treat+Trans (Rinse)**				**Treat+Trans (No rinse)**				**Trans+Treat**			
Time (mins)	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4
2% acetic acid																
Ciona intestinalis	X	X	X	X	-	-	-	-	-	-	-	-	-	-	-	-
Cnemidocarpa bicornuata	X	X	X	X	-	-	-	-	-	-	-	-	X	X	X	-
Corella eumyota	X	X	X	X	-	-	-	-	-	-	-	-	X	-	-	-
Botryllus schlosseri	X	X	X	X	X	X	-	-	-	-	-	-	X	-	-	-
Botrylloides leachi	X	X	X	X	X	X	-	-	-	-	-	-	X	-	-	-
Bugula neritina	X	X	X	X	X	X	X	-	-	-	-	-	X	X	-	-
Watersipora subtorquata	X	X	X	X	X	X	X	-	-	-	-	-	-	-	-	-
Hydroides elegans	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Terebellidae	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cladophora sp.	X	X	-	-	X	X	-	-	-	-	-	-	-	-	-	-
No surviving taxa	10	10	8	8	6	6	3	1	1	1	1	1	6	3	2	1
4% acetic acid																
Ciona intestinalis	X	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-
Cnemidocarpa bicornuata	X	X	X	X	-	-	-	-	-	-	-	-	X	X	X	-
Corella eumyota	X	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-
Botryllus schlosseri	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Botrylloides leachi	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bugula neritina	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Watersipora subtorquata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Hydroides elegans	X	X	X	X	X	X	X	X	X	X	X	-	X	X	X	X
Terebellidae	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cladophora sp.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
No. surviving taxa	5	4	4	2	1	1	1	1	1	1	1	0	2	2	2	1

7.3.2 Effect of treatments on mussel seed-stock

Mean mussel attachment (24 h post-treatment) in controls, after transport at 10 – 20 ^{o}C for 24 h, or after treatment for 2 minutes in 4% or 8% acetic acid, was consistently > 95% (Figure 7.3). However, a marked decline in attachment was evident in some treatment/transport combinations. In particular, the no rinse treatments reduced mean

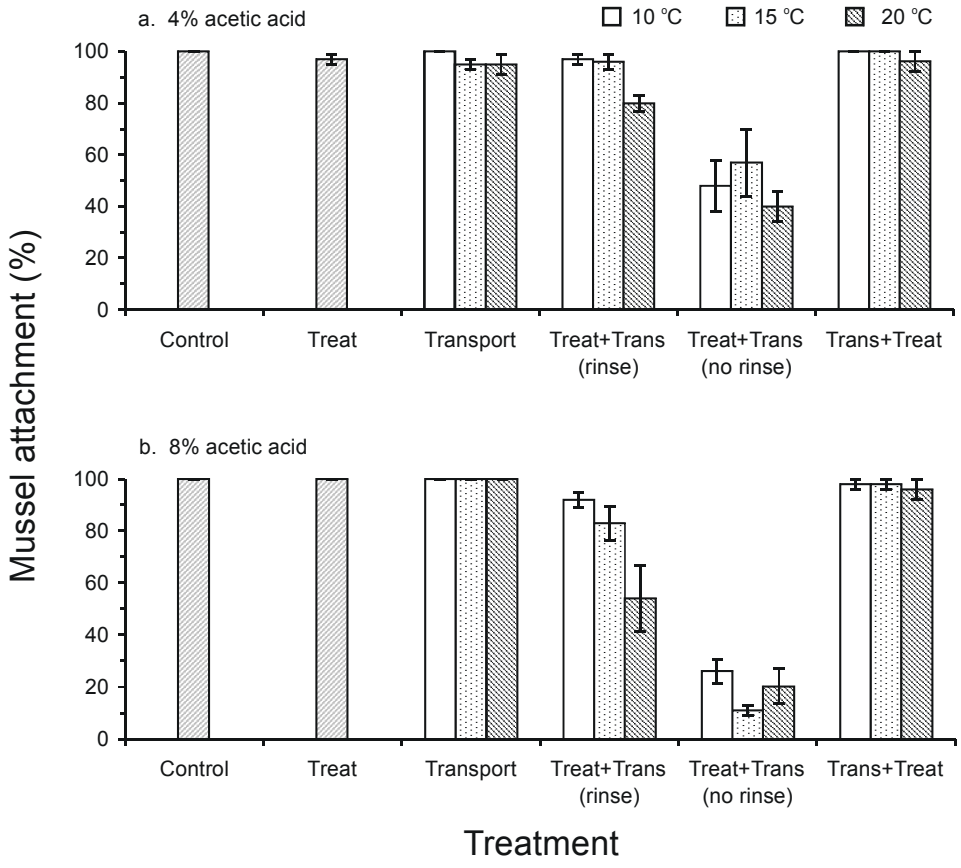

Figure 7.3 Mean mussel attachment (± SD, n = 4 batches of mussels) after immersion in acetic acid (4% or 8%) and air exposure for 24 h at 10 – 20 ^{o}C in temperature control cabinets to simulate transport.

attachment to < 57% and < 26% at 4% and 8% acetic acid respectively. This effect was to some extent mitigated by applying a post-treatment rinse, although there was a significant decrease in mussel attachment at 20 ^{o}C at both concentrations (Tukey's HSD, $p < 0.05$). Moreover, attachment at 20^{o}C was less than the 90% criterion for survival. At both acetic acid concentrations, combined treatment/transport effects were reduced, and survival maximised, by undertaking the treatment after the transport phase (Trans+Treat mean survival > 95%).

Mussel survival in field trials using 4% acetic acid showed patterns that were consistent with the laboratory attachment work. Survival following transport or treatment in isolation was > 91% at all exposure times, irrespective of whether the mussels were

declumped or attached to crop line (Figure 7.4). Not rinsing the mussels prior to transport had the most pronounced effect, reducing mean survival to < 67% in attached mussels and < 37% in declumped mussels. As was the case with the laboratory trials, this effect was mitigated by rinsing prior to transport, or by applying the treatment after the transport phase, which in most cases resulted in mussel survival of > 90%. Notably for all treatments, there was no evidence for a significant decline in mussel survival with increasing acetic acid exposure time (Tukey's HSD, $p > 0.85$), hence mortality in the no rinse treatments is primarily a function of processes occurring during transport.

7.3.3 Use of acetic acid in field operations

Values of pH were a consistently good predictor of acetic acid concentration (pH vs $\log_{10}$ acetic acid, Pearson $r > 0.99$), although increasing variance in the concentration estimate was evident with decreasing pH (i.e., increasing acetic acid), which was related to the pH characteristics of the seawater diluent (Figure 7.5). However, acetic acid concentrations could be estimated with a reasonable level of confidence (95% confidence intervals ± 0.5% of mean) under the treatment conditions of most interest (i.e., 4% or less). From an operational perspective, the error in estimation of acetic acid concentration could be further reduced by diluting a given treatment solution by about 10-fold before measuring pH. This would to shift the acetic acid concentration to < 0.5%, for which the variability in the concentration estimate is negligible (Figure 7.5).

Based on these findings, we anticipated that pH could be used as a simple and reliable field-based indicator of acetic acid concentration. However, results from sequentially immersing 10 batches of mussels in 4% acetic acid indicated otherwise, revealing a gradual (Pearson $r > 0.99$) increase in mean pH of 0.332 pH units from the baseline value. Based on Figure 7.5, this corresponded to an estimated acetic acid concentration decrease from approximately 4% to less than 2% (Figure 7.6). This was clearly not the case, however, because titremetric analyses revealed no appreciable or directional change in concentration from the baseline to batch 10 (Figure 7.6), which was consistent with observations that the volume of the treatment solutions did not appreciably change from the start to end of the two trials. By contrast with the sequential immersion results, the organic-rich seawater solutions mixed to 4% acetic acid remained stable over a 20 day period, with no appreciable or directional change in either pH (± 0.02 pH units) or acetic acid concentration (± 0.1%) from baseline conditions.

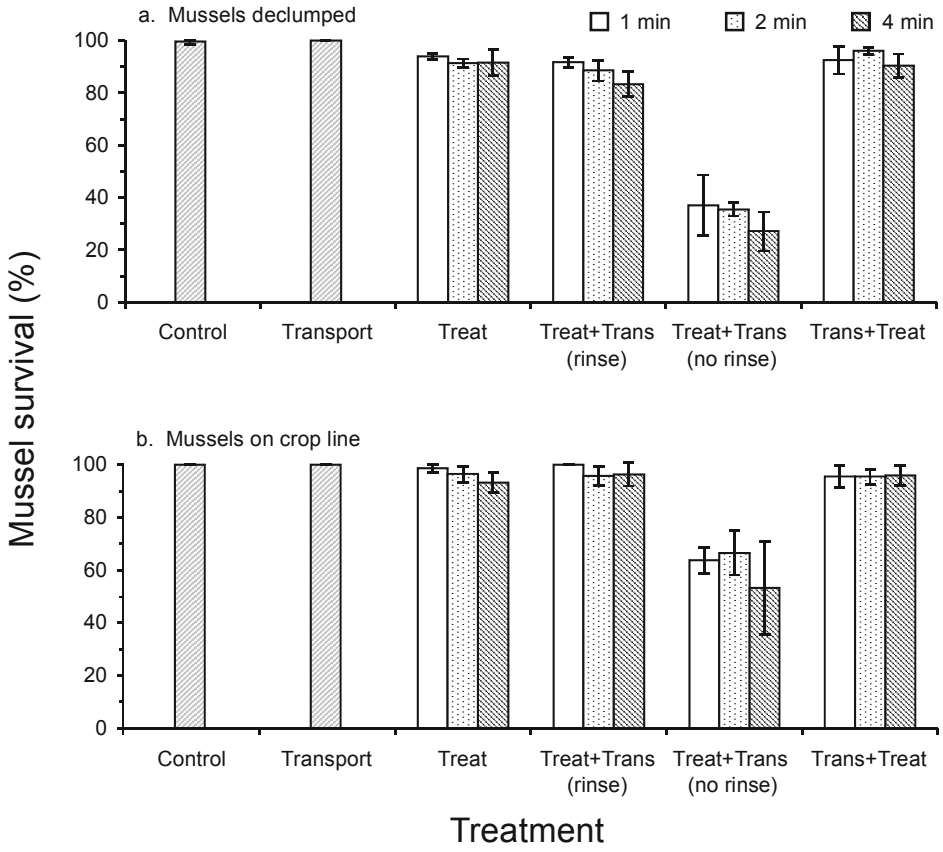

Figure 7.4 Mean mussel survival (± SD, n = 3 batches of mussels) after immersion in 4% acetic acid for 1 – 4 minutes and air exposure for 24 h to simulate transport. Declumped mussels are compared with mussels attached to crop line.

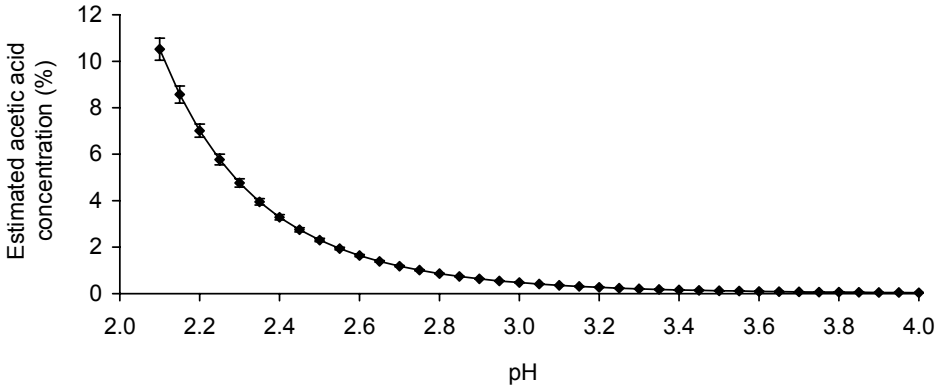

Figure 7.5 Estimated acetic acid concentrations (mean ± 95% CI) at different levels of pH. Values shown were derived from pH vs acetic acid relationships for three different seawater dilution series.

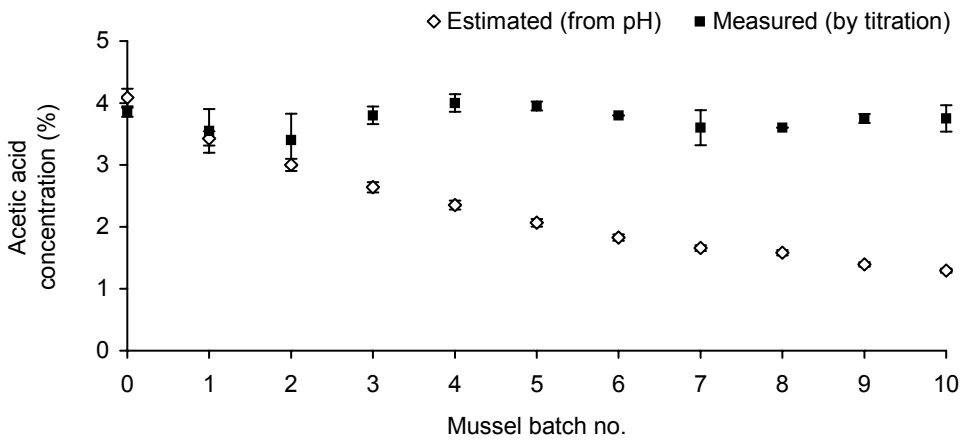

Figure 7.6 Comparison of acetic acid concentrations (n = 2, mean ± 95% CI) estimated from pH values (according to Figure 7.5) and measured by titration, following a sequential 2 minute immersion of 10 x 1 kg batches of mussels in the same 4% acetic acid solution. Mussel batch no. 0 represents the pre-immersion baseline value.

7.4 DISCUSSION

7.4.1 Acetic acid effects on fouling

All acetic acid treatments resulted in a considerable biomass reduction in the fouled rope immersion trials. However, the relative efficacy of different types of treatment was not clearly discernible, reflecting the fact that the solitary tunicate *Ciona intestinalis* was almost completely eliminated by all treatments, and was such a major component of the fouling that patterns among the other taxa were obscured. The pronounced reduction in *Ciona* biomass on control and transport ropes is in part likely to reflect physical damage from handling, because *Ciona* is relatively flaccid and fragile compared with the other indicator taxa used. While dessication and other processes operating during emersion (e.g., Lenz et al., 2004) may have also contributed to biomass reduction in the transported ropes, all 10 indicator organisms nonetheless survived this phase. This almost undoubtedly reflects the fact that transport was simulated in covered bins thus creating high humidity conditions that would facilitate survival (Sant et al., 1996; Schaffelke and Deane, 2005; Forrest and Blakemore, 2006). Ascidians and other soft-bodied fouling organisms would be less likely to survive if exposed to ambient air for this duration, as indicated by recent work with *Styela clava* and *Didemnum vexillum* (Coutts and Forrest, 2005, 2007). Nonetheless, in terms of

aquaculture seed-stock transfers, where high humidity conditions are invariably present, our results indicate that many fouling pests will survive routine handling practices. This highlights that effective management of biosecurity risks from aquaculture transfers almost certainly requires some form of additional treatment.

Immersion in acetic acid at concentrations of 2 and 4% for 1 – 4 minutes was highly effective against the variety of cosmopolitan fouling organisms tested. The effective concentrations and exposure times described here are comparable to other studies where acetic acid has been used. Carver et al. (2003) reported 100% mortality of *Ciona* after a 1 minute exposure to 5% acetic acid, with 30 seconds being 95% effective. Similarly, preliminary trials with *Styela clava* in New Zealand indicated complete mortality after a 1 minute immersion at 4% acetic acid, and after a 5 minute immersion at 2% (Coutts and Forrest, 2005).

Acetic acid efficacy in relation to treatment time was enhanced by the additional stress caused during 24 h transport. For example, a 4% treatment alone was insufficient to eliminate all fouling taxa after a 4 minute immersion period. However, when treatment was followed by transport, an immersion time of 1 minute was lethal to all taxa except the serpulid *Hydroides elegans*, irrespective of whether the fouled ropes were rinsed or not. Not rinsing the acetic acid residue from the fouling biomass prior to transport was nonetheless the most effective treatment. Of the range of organisms used in this study, *Hydroides elegans* was clearly the most resistant to the treatments, with complete mortality achieved only in the most severe combination investigated (4% treatment for 4 minutes, no rinse, 24 h transport). Presumably this reflects the morphology of this polychaete; its calcareous tube and operculum (Day, 1967) would enable it to prevent or reduce its exposure to acetic acid.

7.4.2 Effect of treatments on mussel seed-stock

Within the context of mussel industry operations, in which many tonnes of seed-mussels are routinely processed and transported, the relatively long immersion time required to eliminate resilient foulers like *Hydroides elegans* would often make the use of acetic acid impractical. More importantly, however, our results indicate that treatment conditions that are completely effective against such species would also be lethal to approximately half of the seed-stock. Assuming the tolerance of *Hydroides* is comparable to structurally and functionally similar taxa, for example other non-

indigenous serpulid pests like *Ficopomatus enigmaticus* (Read and Gordon, 1991; Probert, 1993), and various bivalve fouling pests (e.g., *Mytilus galloprovincialis* in New Zealand), then management of such species in relation to mussel seed-stock transfer would require an alternative approach.

Clearly, while the 4% no rinse option has considerable appeal for its simplicity and may have applications for a range of management scenarios (e.g., sterilising infected equipment), it is not feasible in the case of mussel seed-stock transfer. While mussels can survive the no rinse approach at lower concentrations (0.5 – 1% acetic acid), soft-bodied foulers can also survive the treatment. On the other hand, mussel survival generally met the 90% acceptance criterion in the 4% treatment provided the acetic acid residue was rinsed prior to transport, or the mussels were treated after the transport phase. In both cases, efficacy was similar and generally effective against soft-bodied organisms. Hence the application of such procedures, with an immersion phase consisting of exposure to 4% acetic acid for at least 1 minute, would eliminate many of the fouling organisms that are currently problematical to the mussel industry. The requirement to rinse to ensure an acceptable level of mussel survival would be reasonably straightforward for most operators, but would have other implications for field operations. For example, it would be important to ensure that this did not lead to re-inoculation of treated mussels by planktonic life-stages of pest organisms. The option to apply the treatment after the transport phase may be a simpler alternative, but would require biosecure management procedures to be adopted in the recipient region.

The level of mussel mortality was comparable to that described for cultured *Mytilus edulis* in eastern Canada subject to similar treatment conditions, although considerable mortality in *Mytilus* can occur if valve closure is not induced (e.g., by shaking) prior to immersion (Le Blanc, *pers. comm.*). Mortality in the present study was not strongly related to acetic acid immersion time or concentration (across the 1 – 4% range), but there was some evidence that warmer conditions during transport would lead to reduced survival, which would need to be accounted for during field operations (e.g., via temperature control or avoidance of excessively warm transport conditions). The reasons for this reduced survival were not explored, but could reflect direct heat stress on the mussels during emersion (Marsden and Weatherhead, 1998), or a more pronounced biocidal effect of acetic acid with increasing temperature (Breidt et al., 2004), which could be exacerbated by increased mussel gaping under such conditions.

In relation to current industry operating procedures for inter-farm seed-stock transfers, the similar tolerance of declumped vs attached mussels to the effects of the most feasible treatments (i.e., rinse, or transport then treat) is encouraging. It means that the treatment can be applied with minimal disruption to the industry, following the routine declumping and washing process. This procedure itself has the added benefits of mitigating biosecurity risk by removing most of the fouling biomass. Furthermore, the declumping process enables the entire mussel shell surface (and associated microscopic fouling) to be more easily exposed to the treatment than might otherwise be the case with heavily fouled seed-stock attached to crop line.

By contrast with rinsing, the reduced survival of declumped vs attached mussels in the no rinse treatment conceivably reflects greater valve gaping in declumped individuals during the transport phase, and hence exposure to a high humidity acetic acid environment. It has been demonstrated elsewhere that mortality to chemical toxicants in bivalves is greater in bioassays where detached vs attached animals are used (Rajagopal et al., 2002, 2005). This occurs because detached mussels show increased gaping, foot activity and byssus production, which exposes soft tissues to the toxicant.

7.4.3 Use of acetic acid in field operations

Our data indicate that 4% acetic acid solutions remain stable over time and in the presence of organic matter, hence do not appear to be consumed or complexed in the manner of commonly used chemicals such as chlorine (Taylor, 2006). While pH is a reliable predictor of acetic acid concentration in seawater dilutions, repeated mussel immersion at 4% resulted in an increase in pH without a corresponding change in acetic acid. The magnitude of change was such that the predicted acetic acid concentration was less than half of the actual value, which would have led to an erroneous concentration adjustment in a field situation. The increase in pH conceivably reflects dissolution of calcium carbonate from the mussel shell (which would have produced a buffering effect), although visual examination of treated shells did not reveal any effects on the shell surface as a consequence.

We do not anticipate that the increased pH would lead to a change in the effectiveness of treatment solutions, provided that acetic acid concentrations remain at the target concentration. Previous work has shown that the efficacy of acetic acid (and other weak organic acids) is primarily a function of the compound itself rather than altered pH

(Kimble, 1977; Spaulding et al., 1977; Verschueren, 1996; Breidt et al., 2004). Such findings are supported by work conducted as part of the present study, in which we observed that strongly-dissociating hydrochloric acid diluted to achieve the same pH as 1% acetic acid had a markedly lower biocidal effect on *Undaria*. By contrast, the efficacy of different types of domestic vinegar diluted to achieve a 1% acetic acid concentration was similar to a 1% concentration diluted from the glacial solution despite the pH of the former being 0.17 units less.

Clearly, while pH is not an appropriate field-based indicator of acetic acid concentration in the mussel industry application described here, its stability in the presence of organic matter suggests that it may nonetheless be useful in relation to acetic acid treatment of other types of fouling, but this would require further evaluation. In relation to shellfish seed-stock, an alternative approach for determination of acetic acid in treatment solutions would be a simplified titration-based method. This could be developed as a field 'kit' that aimed to detect known concentrations, based on additions of specified volumes of sodium hydroxide and treatment solution in the presence of a phenolphthalein colour indicator.

7.5 CONCLUSIONS

As part of an emerging suite of approaches to manage fouling organisms, this paper has revealed the efficacy of short duration acetic acid treatments against a variety of cosmopolitan taxa. The knowledge of fouler survival at different acetic acid concentrations and exposure times has application in a number of pest management scenarios, for example in the sterilisation of infested equipment or structures. In relation to management of mussel farming seed-stock transfers, this paper has also demonstrated that it is also possible to undertake treatments in ways that will eliminate many problematical foulers without significant adverse effects to the stock. This 'proof of concept' can be built on and refined in subsequent work. The ideal treatment will maximise the 'window' between pest mortality and mussel survival, thereby providing assurance that high risk species can be eliminated with minimal risk of adverse effects on seed-stock. In practice, however, treatments that are completely effective against pest organisms may result in some level of unavoidable mortality to mussels. In such instances, decisions about whether or not to apply the treatments must balance treatment costs and benefits against the unmanaged risks and consequences of pest incursion.

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Chapter 8

Setting Management Priorities Using a Risk Model

PREFACE

This chapter describes a risk-based decision support framework for setting priorities for the management of marine pest species. This Chapter is not about *Undaria per se*, but the logic behind the approach presented draws on experience with *Undaria*, and the seaweed is used to illustrate many of the points that are made. This is a novel contribution to marine biosecurity in that, for the first time, it provides a structured process for managing marine biosecurity risks. It goes beyond traditional risk assessment to provide a method for incorporating aspects of technical feasibility and cost/benefit into an overall risk management process. This chapter was initially drafted in 2003, and published as a book chapter in a form almost identical to that presented here. The citation for the publication is:

Forrest BM, Taylor MD, Sinner J. 2006. Setting priorities for the management of marine pests using a risk-based decision support framework. Chapter 25 In: Ecological Studies, Vol. 186, Biological Invasions in New Zealand, Allen RB, Lee WG (eds), Springer-Verlag Berlin Heidelberg

This work reflects my own writing, although initial discussions with co-author Dr Mike Taylor (Cawthron) were particularly instrumental in the early formulation of the ideas and the overall concept. Co-author Jim Sinner (Ecologic Foundation) played an invaluable role in refinement of the formulae that are presented with the framework, especially where cost-benefit elements have been incorporated.

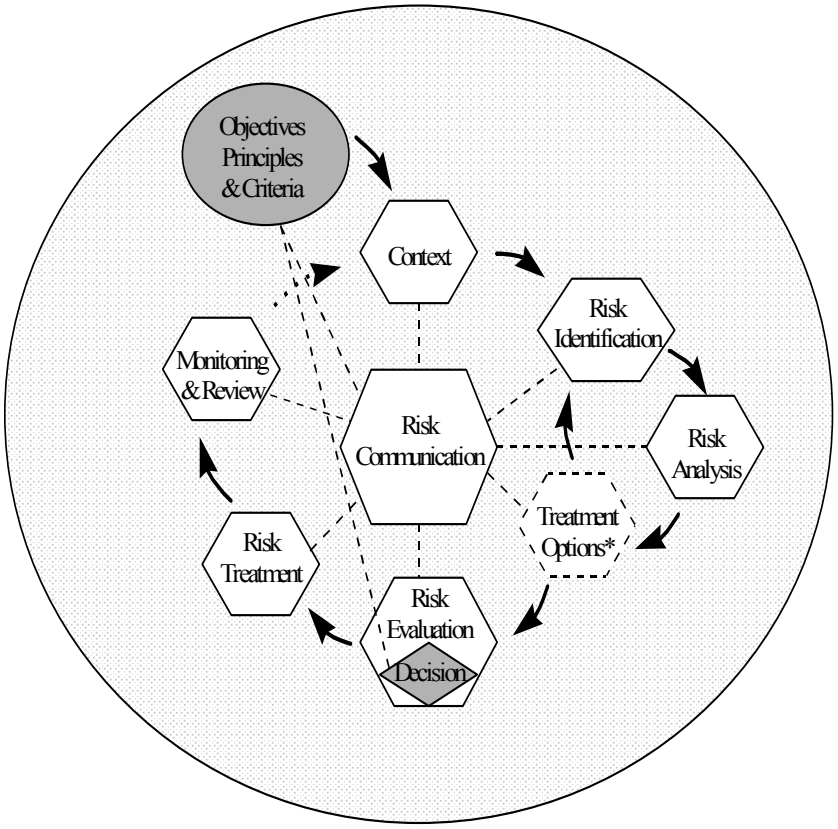
8.1 INTRODUCTION

At least 148 marine species have been accidentally introduced into New Zealand, with a further 4 deliberate introductions (Cranfield et al. 1998). A number of these threaten New Zealand's environmental, economic, social and cultural resources, with changes in patterns of trade meaning that further incursions of unwanted organisms are inevitable (Taylor et al. 1999). In recognition of such threats, the Biosecurity Strategy for New Zealand expands the traditional focus from terrestrial and freshwater issues to also emphasize management of risks from marine pest species. This chapter outlines the key elements of a decision support framework that will contribute to this goal by providing a systematic and transparent mechanism for identifying and analysing risks, and prioritising management objectives in the marine environment.

Our framework is based on the risk management process described by Sinner and Gibbs (1998), which involves four stages: risk identification, risk assessment[3], analysis of risk treatment options, and risk evaluation (Figure 8.1). In this chapter we provide a brief overview of key steps and considerations for the risk identification stage, and focus more on methodological approaches for the latter three stages. We build on lessons learned in developing a management strategy for the Asian kelp *Undaria pinnatifida* in New Zealand (Sinner et al. 2000), and reveal some of the peculiarities of bioinvasion and pest management in marine environments that contrast terrestrial and freshwater systems.

Our underlying premise is that a logical starting point in setting management priorities for marine pests is to consider the values we wish to protect from adverse impacts. This approach is particularly relevant to those having an interest in the protection of areas that are geographically defined at local and regional scales, such as aquaculture sites and Marine Protected Areas, but the same logic can also be applied at greater spatial scales. Hence in the sections below we describe a framework that allows: marine biosecurity risks to be identified in an explicit fashion; the probabilities that lead to a pest infestation estimated; the consequences of infestation at pest density assessed; and priorities to be established through comparison of the feasibility, benefits and costs of risk management.

[3] Sinner and Gibbs (1998), following the joint Australia/New Zealand Standard, use the term "risk analysis" for the step involving estimation of the likelihood of an event and its consequences. Terminology varies; here we use the term "risk assessment" for this step.

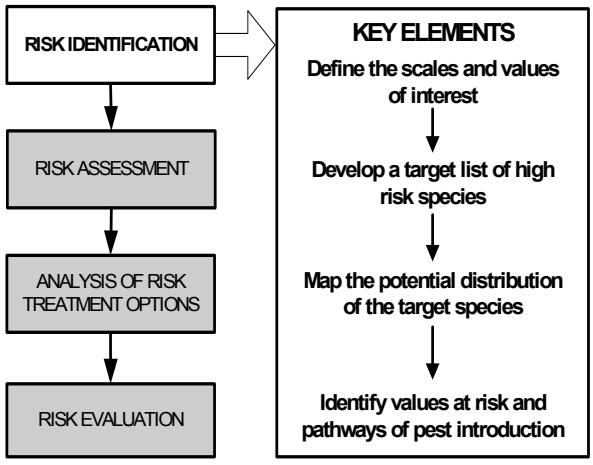


* Identification and analysis of Treatment Options is part of the Risk Treatment Step, but is shown separately here because, for some decisions, it needs to occur at a different point in the sequence than the implementation of selected treatments.

Figure 8.1 Risk management process described by Sinner and Gibbs (1998).

8.2 RISK IDENTIFICATION

The risk identification process is an information gathering phase that we have subdivided into four key steps, as shown in the adjacent diagram. Our framework first requires that values are identified and high value areas (HVAs) prioritised in a defensible way. For decisions post-border, we suggest that priorities should be determined based on comparisons that



are segregated according to the types of values being considered, such as distinguishing commercial aquaculture from marine conservation values. It would be more appropriate to assess the latter, for example, within the context of other conservation initiatives (including non-biosecurity initiatives), so that the maximum benefit for conservation is achieved within the available budget of the relevant organisation. However, we acknowledge that management interventions (especially at a national scale) may have benefits across the different environmental, economic, social and cultural value sets, in which case a process would be required to determine the measures having the greatest benefits overall.

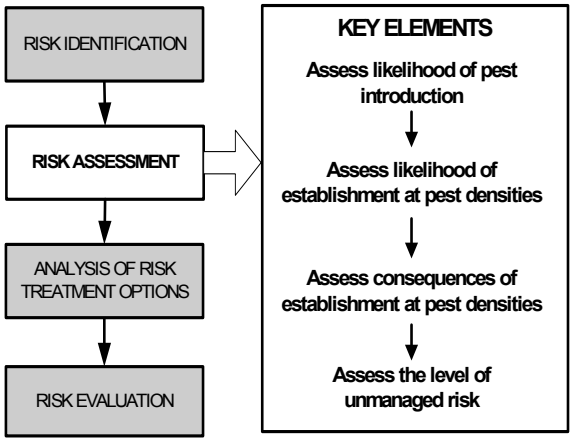
The second step, developing a target list of high risk pests, is a precursor to making predictions about their potential distribution, and thus the values that they threaten. A target list should be based on explicit selection criteria (e.g., Hayes et al. 2002; Hewitt and Hayes 2002; Hayes and Sliwa 2003). Screening for pests based on their common biological characteristics, which is an approach used in terrestrial and freshwater weed management (Groves et al. 2001), may have merit for some groups of marine species (e.g., Nyberg and Wallentinus 2005). However, this approach may not be generally feasible for marine environments where the more idiosyncratic features of particular species often facilitate their success as invaders (Forrest et al. 1997; Ruiz and Hewitt 2002). Until better screening tools are developed, identifying potentially high risk pests based on their invasiveness or impacts elsewhere is a useful starting point and one that can motivate stakeholder interest, even though this approach may not encompass the full suite of high risk species (McEnnulty et al. 2001; Simberloff 2003).

Given a target list, the potential distribution of each pest species in a recipient area assists in the identification of values at risk and the pathways to HVAs. A simple approach to estimate this distribution is to evaluate the 'match' between an organism's natural tolerances (e.g., temperature) and the environmental conditions in the recipient area (Smith et al. 1999). However, this type of assessment should be seen as conservative because experience in both terrestrial and marine systems has shown that it may underestimate actual pest distribution (e.g., Floc'h et al. 1996; Mack 1996).

The final step in the process is to identify the pathways by which target species might be introduced into areas considered high priority for protection. Natural dispersal via water currents is likely to be particularly important in the local or regional spread of pests, especially those with planktonic larval stages. At these scales and greater, the

importance of numerous human-mediated invasion pathways is also well recognised, with particular risks for the New Zealand marine environment posed by ballast water discharge (Hay et al. 1997; Inglis 2001), fouled hulls (e.g., Coutts and Taylor 2004), vessel sea chests (Coutts et al. 2003), and transfer of contaminated aquaculture equipment or shellfish seed-stock (Forrest and Blakemore 2002). It is important to note that nominally minor or unrecognised pathways can also pose significant risks in some circumstances (e.g., Hay and Dodgshun 1997; Coutts 2002) and need to be accounted for in the risk management process.

8.3 RISK ASSESSMENT



Following risk identification, key steps in the risk assessment stage shown in the adjacent diagram involve estimating the likelihood of each target species being introduced to each HVA and becoming established at pest density, and the associated consequences. For this purpose, we propose a standard 'chain of events' approach that combines these elements to determine the level of threat posed by pest species, with a process to rank risks according to the level of importance attached to each HVA. This essentially provides an index for each HVA whose score reflects the 'unmanaged risk', which can be represented in simplistic terms as:

$$R_{Uij} = P_{Iij} \times P_{PDij} \times V_j \times I_{ij} \qquad \text{where:} \qquad (1)$$

R_{Uij} = the unmanaged risk from species *i* in area *j*, which is the expected value of damage from the pest in the absence of measures to reduce the likelihood of introduction or to respond to an incursion;

P_{Iij} = the probability of introduction of species *i* to area *j*;

P_{PDij} = the probability that, once introduced, species *i* will reach pest density in area *j*;

V_j = the total value at risk in area *j*; and

I_{ij} = the consequences of establishment at pest density of species *i* in area *j*, in terms of the proportion of the values at risk that are lost due to the pest.

The highest values of R_{Uij} represent the greatest risks. The approach is hierarchical in that each main component can be broken into increasingly detailed parts given the availability of sufficient information. In the analysis of risk treatment options, these implied priorities are re-ranked taking account of the feasibility, efficacy, and costs of management. We recognize the importance of incorporating measures of uncertainty throughout this process but restrict our discussion here to the logic of our approach.

8.3.1 Likelihood of introduction: P_{Iij}

P_{Iij} represents the likelihood that a target species will be transported to an HVA during a given time frame, either by natural or by human-mediated pathways. With respect to human-mediated pathways this assessment can be a significant undertaking, as exemplified by risk assessment approaches for ballast water alone (e.g., Hayes and Hewitt 1998; Hayes 2002). For a broad decision-making tool, more simplistic approaches may be needed (e.g., Aurand et al. 2000; Hayes et al. 2002). We suggest that effort is made to at least separate the likelihood of target pest introduction (e.g., based on qualitative scores) into the key pathways, because management interventions would typically address specific pathways in order to reduce the probability. For example, in a situation where key pathways are identified as hull fouling (HF), ballast water (BW), aquaculture (AQ), and natural spread (NS), the probability of introduction can be expressed as:

$$P_{Iij} = f(P_{I/HFij}, P_{I/BWij}, P_{I/AQij}, P_{I/NSij}) \quad (2)$$

The nature of the function f for calculating the overall probability P_{Iij} of at least one introduction during a selected time period depends on the relationship between the individual probabilities. A probability $P_{I/UEij}$ can also be used to represent the possibility of introduction via some unexpected or unanticipated pathway. This identifies residual risk that is not being managed, even though it will not affect the relative management priorities that emerge from the analysis. In most cases it will be reasonable to assume that the probabilities are independent of each other but not mutually exclusive. In this case, the probability of at least one event is one minus the probability that none of them will occur (Snedecor and Cochran 1980). Thus, the expression is:

$$P_{Iij} = 1 - [(1 - P_{I/HFij}) \times (1 - P_{I/BWij}) \times (1 - P_{I/AQij}) \times (1 - P_{I/NSij}) \times (1 - P_{I/UEij})] \quad (3)$$

8.3.2 Likelihood of establishment at pest density: P_{PDij}

Environmental matching analyses made during the risk identification stage will provide rudimentary guidance on the likelihood of pest establishment in a recipient area, but prediction of infestation levels and hence potential impacts will be more difficult (Williamson 2001). An adequate knowledge of underlying invasion processes, likely infestation levels, and density-dependent effects is seldom available, with a general consensus that even with detailed study the prospect of making reliable predictions of invasion success is remote (e.g., Lawton and Brown 1986; Kareiva et al. 1996; Vermeij 1996; Forrest and Taylor 2002). Furthermore, knowledge of the general attributes of species and recipient communities that may influence the likelihood of success may not assist with prediction of whether a particular species will invade a particular locality and to what extent (Lawton and Brown 1986; Simberloff 1989; Lodge 1993).

Determination of the likelihood that an invader will reach pest density will therefore continue to rely on expert judgment. This can be formalized by providing categories for considering likely success based on factors such as: (1) invader attributes (e.g., extent of prior invasion success, reproductive potential and dispersal mode); (2) physical attributes of the recipient environment that may affect invasion success such as regimes of temperature, salinity, wave exposure, space availability, and substratum suitability; and (3) biotic attributes of the recipient environment that may affect invasion success such as the presence of grazers, predators, or competitors. Alternatively, in the absence of information, one can assign the same value to P_{PDij} for all species and sites, so that the evaluation of relative priorities is not influenced by this parameter, but this default approach may ignore potentially useful information.

8.3.3 Consequences of establishment: $V_j \times I_{ij}$

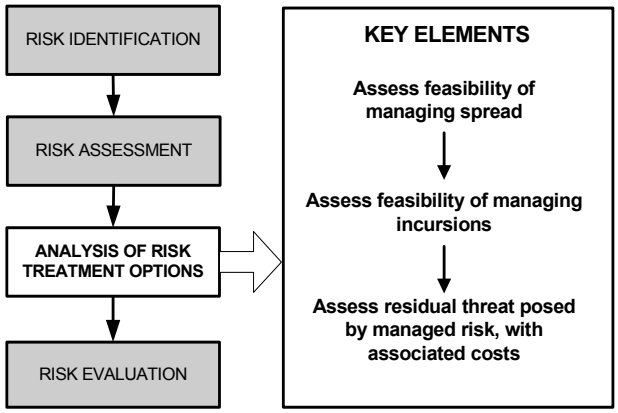
The third main component needed to determine unmanaged risk, R_{Uij}, is estimation of the severity of consequences of an introduced organism reaching pest density in a given HVA. This reflects not only the level of infestation, but also the type of values affected and the level of importance attached to an HVA. In the case of the former, for example, infestation by a conspicuous invader at a density resulting in only minor ecological effects could have impacts that are more than minor if the location were highly valued for its natural character (i.e., the pest density threshold depends on the type of values being considered). For current purposes we assume that different types of values will

be addressed separately. The model does, however, account for the fact that the consequences of a given pest density will be greater for HVAs of greater value.

The term V_j, the total value at risk in a given area, essentially applies a weighting factor to the unmanaged risk score, giving greater weight to HVAs of relatively high value. The term I_{ij}, the consequences of establishment at pest density, provides a further weighting according to the proportion of the values at risk that could be lost due to the pest. V_j could be expressed on any relevant scale, for example a dollar figure for commercial values or a 1-5 scale representing values of local through to international conservation significance, with qualitative scores assigned to I_{ij} to represent a scale from negligible to catastrophic consequences (e.g., Wotton and Hewitt 2004). Once V_j and I_{ij} have been determined, equation 1 can be calculated to estimate values for R_{Uij} to represent *relative* risk across species-site combinations.

8.4 ANALYSIS OF RISK TREATMENT OPTIONS

The key steps in the analysis of risk treatment options are shown in the adjacent diagram. Treatment options for invasive marine species are primarily: (1) management of spread to minimize the introduction of target species to HVAs, and (2) surveillance and response to new infestations. Clearly, 'no intervention' may also be valid in some circumstances, for example where the costs of intervention outweigh the benefits, where the risks are negligible, or where they are essentially unmanageable. A further option may be mitigation of adverse impacts, an example being closure of coastal shellfish resources for harvesting because of blooms of toxin-producing microalgae (Rhodes et al. 2001).



Development of effective measures to manage marine pests is at an early stage. Even for measures that are technically feasible, high costs and other constraints often preclude their implementation. Within our framework, consideration of the likely effectiveness of management leads to an assessment of the residual threat posed by managed risk, R_{Mij}. Costs of management, C_{Mij}, must also be estimated to enable

evaluation of which measures provide the most value for money. The level of managed risk and associated costs can be expressed as follows:

$$R_{Mij} = P'_{Iij} \times P_{PDij} \times V_j \times I_{ij} \times (1 - P_{SCij}) \quad (4)$$

$$C_{Mij} = C_{SMij} + [P'_{Iij} \times P_{PDij} \times C_{SCij}] \quad \text{where:} \quad (5)$$

R_{Mij} = the managed risk from species *i* in area *j*, which is the expected value of damage from the pest despite measures to reduce the likelihood of introduction and respond to any incursion (i.e., residual risk);

P'_{Iij} = the reduced probability of the introduction of species *i* in area *j*, after feasible measures to manage spread have been implemented;

P_{PDij}, V_j and I_{ij} are defined as per equation 1;

P_{SCij} = the probability of successful control of an incursion of species *i* in area *j*;

C_{Mij} = the expected cost of management measures to reduce the risk from species *i* in area *j*;

C_{SMij} = the cost of measures to manage spread that could be implemented to reduce the likelihood of introduction of species *i* to area *j*; and

C_{SCij} = the expected cost of incursion response to an introduction of species *i* to area *j*, i.e., the cost of incursion response discounted by the probability of an incursion.

Equation 4 is similar to equation 1, but requires consideration of the reduced risk of pest introduction to an HVA through management of spread (P'_{Iij}), and determination of the feasibility of management measures and the likelihood of successful control (P_{SCij}), so that the probability that control measures will fail ($1 - P_{SCij}$) can be incorporated into the expression of residual risk. The terms V_j and I_{ij} are independent of the other terms, and remain the same as in equation 1. For simplicity, we assume that the probability of establishment at pest density (P_{PDij}) also remains the same, even though the likelihood of pest introduction may have decreased (i.e., $P'_{Iij} \leq P_{Iij}$). This reflects the level of uncertainty (even for many well-studied pests) regarding the relationship between inoculum pressure and subsequent establishment.

The analyst has to exercise judgment about which management measures to include in the model. Situations will invariably arise where only one of P'_{Iij} or P_{SCij} will be relevant or meaningful. For example, for New Zealand's subantarctic islands (highly valued for conservation reasons) Sinner et al. (2000) demonstrated that managing pathways for *Undaria* would greatly reduce the risk of the seaweed's incursion, but that surveillance and incursion response were not feasible because of the isolated and rugged

nature of the islands, i.e., P_{SCij} was treated as zero and no incursion response was contemplated.

Similarly, there may be reasons to consider control of well-established pests to densities that avoid adverse effects, where management of spread is clearly pointless (i.e., $P'_{Iij} = P_{Iij}$). In effect, the analyst must determine which package of measures to evaluate for each species-site combination, based on what appears to be the most feasible. Alternatively, more than one package of measures may be compared for a given species-site (e.g., Sinner et al. 2000). These points are further considered below.

8.4.1 Reducing the risk of introduction through management of spread: P'_{Iij}

An assessment of the relative importance of natural vs human-mediated spread of a pest is central to decisions regarding the need for management of anthropogenic pathways, and Figure 8.2 outlines a screening tool that could be used for this purpose. Although the timescales in Figure 8.2 are arbitrary, they are included to highlight the principle that the more vulnerable a locality is to natural spread, the less likely that management of human-mediated pathways will be worthwhile. Clearly, however, the extent to which management measures are considered necessary or desirable, especially in the 'medium priority' categories shown in Figure 8.2, will depend on the values at stake. For example, a commercial aquaculture locality vulnerable to natural spread within a matter of a few years may be of such high value that it is worth evaluating the feasibility of managing human-mediated pathways to reduce the risk of pest introduction, perhaps to provide sufficient time for the industry to adapt or to enable development of effective incursion response measures.

Where analysis following Figure 8.2 suggests further evaluation of anthropogenic pathways is important, one then considers whether management is likely to be feasible, because effective management strategies may have major costs that limit their usefulness. In southern New Zealand, for example, marine farmers adopted a voluntary ban on movements of aquaculture equipment and shellfish seed-stock, the aim being to prevent the transfer of *Undaria* to a region where a management programme for the seaweed was in place, but they incurred costs from lost production when seed-stock from an alternative source was unavailable.

Often management will need to focus on measures to reduce rather than eliminate the spread of target species, for example by limiting contamination of transfer mechanisms

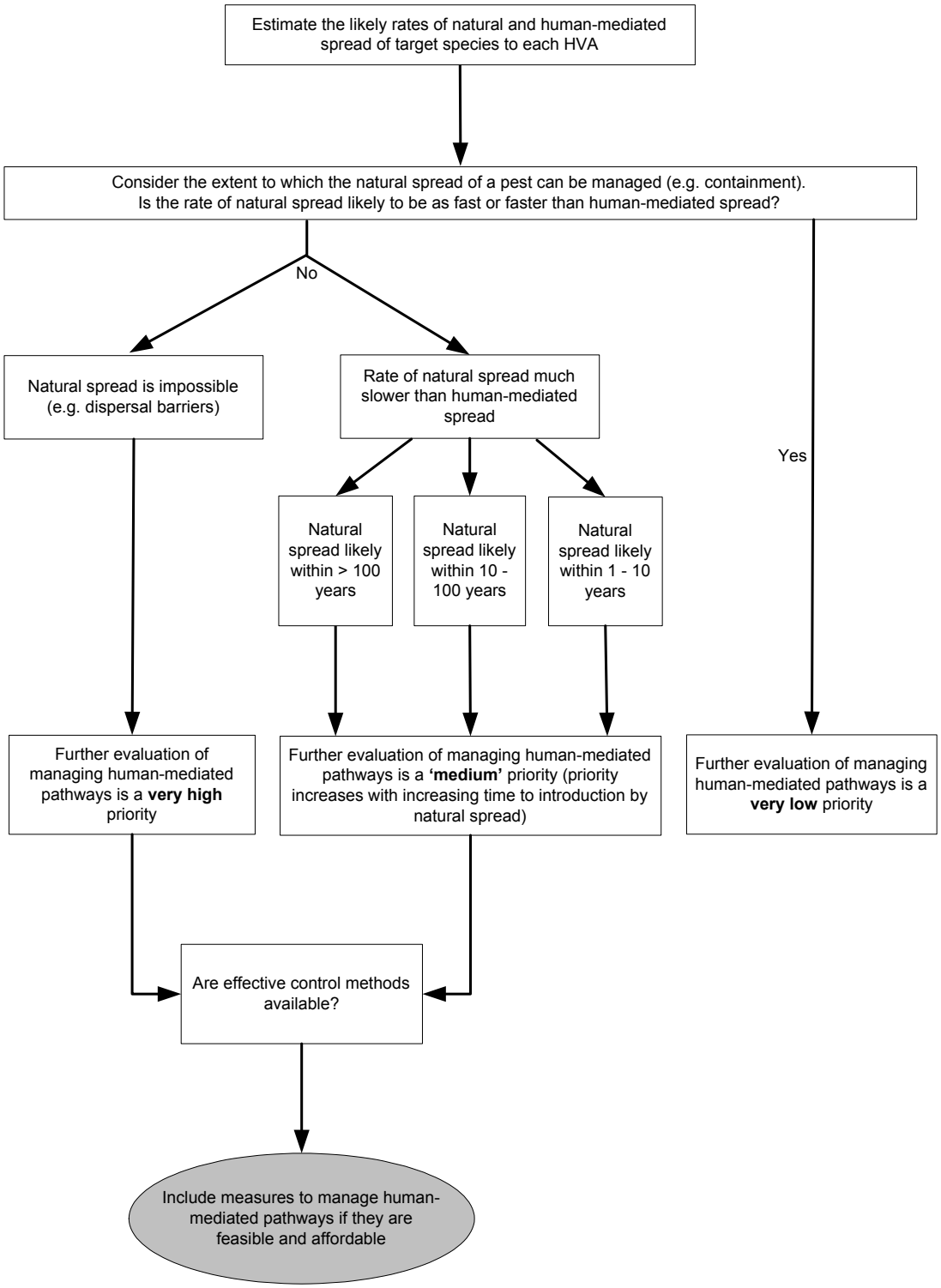


Figure 8.2 Decision tree for evaluating whether and to what extent management of human-mediated transfer mechanisms might be desirable for a given high-value area (HVA).

(e.g., through control of pest populations in source regions), pathway monitoring for target pests, and generic pathway management measures that may have added benefits beyond the target pest species (e.g., hull cleaning and anti-fouling, ballast water exchange). The ability to implement any or all of these will depend on factors such as the frequency and complexity of human-mediated pathways, the characteristics of the pest species or life-stage transported, the availability and cost of effective management measures, and the willingness of owners and operators of transfer mechanisms to partake in a management programme.

In terms of equation 4, P'_{Iij} is the residual risk that management measures will fail to prevent spread. Hence, the analyst will need to estimate the likely effectiveness of management for each pathway where feasible measures are available. This can be done by estimating each $P'_{I/Xij}$ directly (where X represents a pathway), or by estimating the proportion by which the measure would reduce risk of introduction via the pathway and multiplying this by $P_{I/Xij}$. For example, for a ballast water measure that reduced delivery of species *i* to site *j* by 50%, $P'_{I/BWij} = P_{I/BWij} \times 0.5$. Using the examples of human-mediated (ballast water, hull fouling, aquaculture) and natural spread pathways given in Section 24.3.1, this will allow determination of P'_{Iij} as follows (this can be calculated as in equation 3):

$$P'_{Iij} = f(P'_{I/HFij}, P'_{I/BWij}, P'_{I/AQij}, P'_{I/NSij}). \quad (6)$$

The analyst also needs to provide an estimate for the term C_{SMij}, which is the sum of costs for feasible measures to manage spread. These estimates do not need to be precise, but need to be reasonably accurate relative to the cost estimates for other measures.

8.4.2 Surveillance and incursion response: P_{SCij}

Table 8.1 highlights features of marine environments and marine pests that affect the feasibility of traditional approaches to surveillance and incursion response (i.e., eradication, containment, or control of pest populations). Key challenges in marine systems lie in the early detection of target pests, and in the development of practical and cost-effective incursion response tools that have minimal adverse side effects (Wotton and Hewitt 2004; Thresher and Kuris 2004). In these respects, classical biological control is considered high risk (e.g., Secord 2003), and the mechanical and chemical treatment approaches commonly used in terrestrial and freshwater environments are not

Table 8.1 Key features of marine environments and marine pests that affect the feasibility of surveillance and incursion response.

A. Receiving environment attributes

Relative ease of surveillance or response	Water clarity	Wave exposure	Bathymetric complexity	Biological complexity	Remoteness	Tidal state	Habitat availability
Easy	Clear	Sheltered	2D	Simple	Accessible	Intertidal	Limited
↓	↓	↓	↓	↓	↓	↓	↓
Difficult	Turbid	Exposed	3D	Complex	Remote	Deep subtidal	Unlimited

B. Invader attributes

Relative ease of surveillance or response	Invasiveness	Invader distribution	Invader conspicuousness	Habitat preferences	Propagule dispersal range
Easy	Low	Confined	Large or conspicuous	Specific	Short
↓	↓	↓	↓	↓	↓
Difficult	High	Widespread	Small or cryptic	Generalist	Distant

always applicable. Localized control of subtidal *Undaria* populations, for example, relies on diver detection and manual removal of the visible sporophyte stage of the seaweed, and is rarely successful (Hewitt et al. 2005). In the few situations where successful eradication of marine pests has been reported, there were usually particular (often unusual) circumstances that favoured a positive outcome, as revealed by examples with *Undaria* on a sunken vessel near New Zealand's Chatham Islands (Wotton et al. 2004) and the black-striped mussel *Mytilopsis sallei* in a Darwin marina (McEnnulty et al. 2001).

Given the poor record of post-invasion management success in marine systems, highest priority should ideally be given to preventing new incursions (McEnnulty et al. 2001; Eno and Hamer 2002), but the lack of completely effective management measures for this purpose means that unwanted introductions will continue. Hence in the context of our framework, initial judgment is required as to whether surveillance and incursion response is likely to be worthwhile, such that the P_{SCij} term is retained in equation 4 for more detailed evaluation. Figure 8.3 provides a structured approach to assist with this decision, leading through a series of questions that relate to the feasibility of

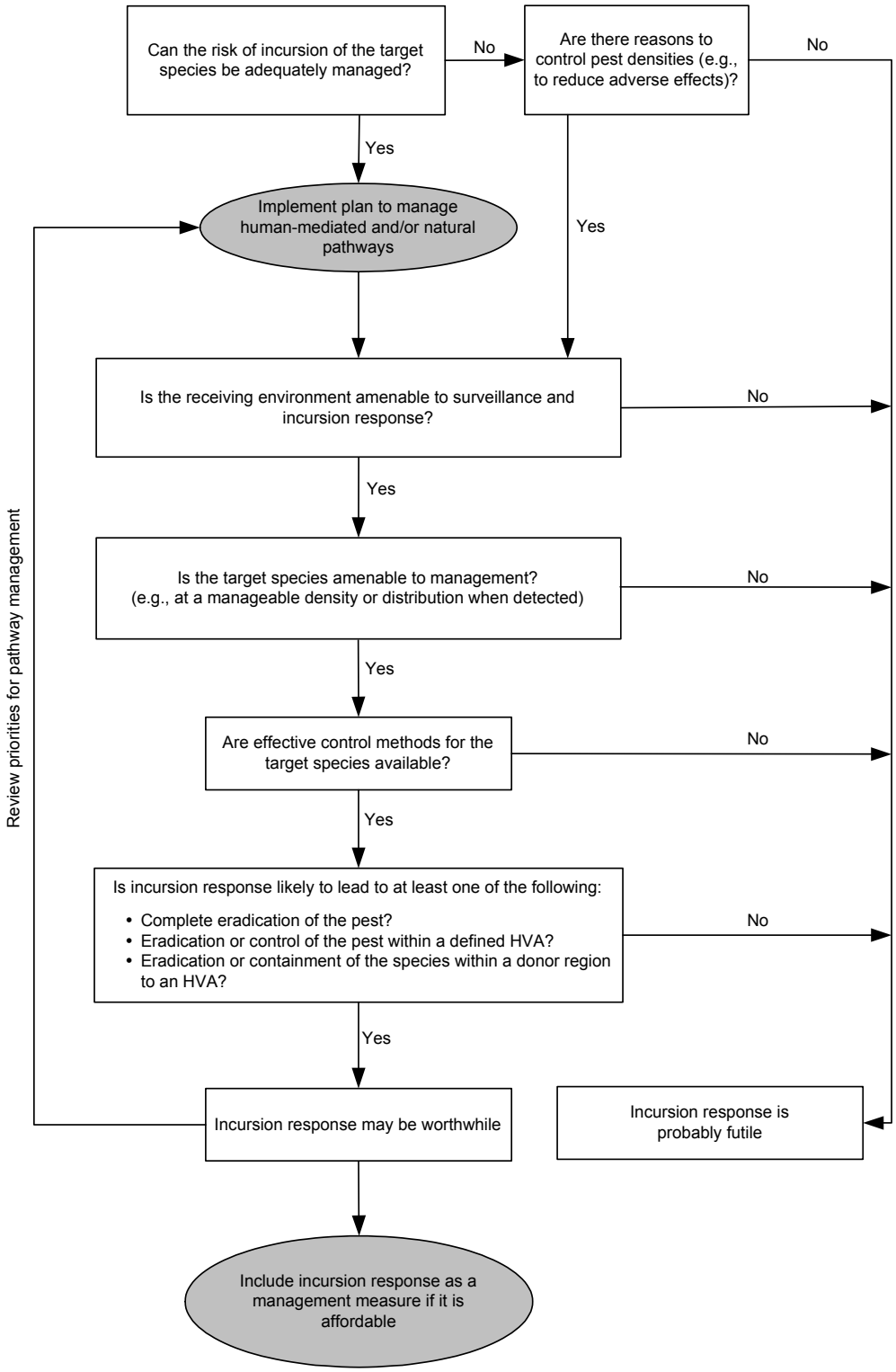


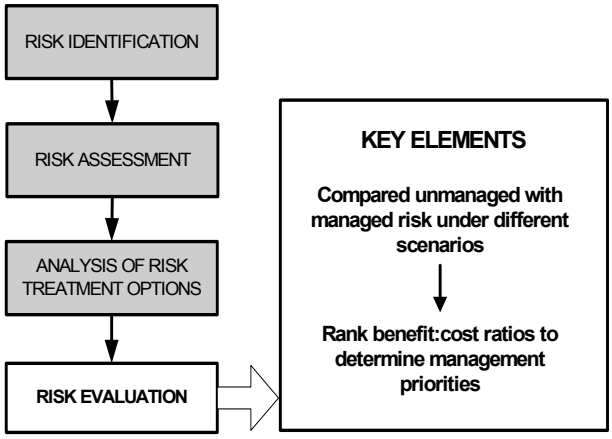
Figure 8.3 Decision tree for considering whether incursion response for existing and potential pests is likely to be worthwhile.

surveillance or incursion response based on the attributes of the pest and the receiving environment (e.g., Table 8.1), and on the availability of effective management options. A range of non-technical aspects also need to be considered as part of this process (McEnnulty et al. 2001; Wotton and Hewitt 2004).

The likelihood of successful incursion response, P_{SCij}, will clearly be species- and situation-specific, and driven to a large extent by the desired management endpoints (e.g., eradication vs control). In considering options for managing *Undaria*, for example, Sinner et al. (2000) described the cumulative probability in a given year that: (1) an infestation would occur; (2) the infestation would be detected while still at a 'Level 1' stage (i.e., no reproductive plants); (3) the response to Level 1 and 'Level 2' (i.e., reproductive plants present) infestations failed; and (4) control efforts failed, leading to an uncontrolled infestation. These same elements will not always be appropriate for different species or situations. For example, it may be desirable to manage *Undaria* and other biofouling pests on aquaculture structures to a level that avoids adverse effects, even when repeated incursions are inevitable and eradication is not feasible.

8.5 RISK EVALUATION: RANKING MANAGEMENT PRIORITIES

The key elements of risk evaluation are shown in the adjacent diagram. Risk evaluation involves comparing unmanaged risk with the risk after management, taking account of the costs of management. The most comprehensive analysis would be to determine priorities across all species and HVAs, and with respect to the full range of management measures. This involves estimating equations 1 and 4 for each *ij* combination and comparing the results. That is, for species *i* in area *j*:



$$R_{Uij} - R_{Mij} = \text{expected value of damage reduction.} \quad (7)$$

To indicate relative priorities for management, the expected value of damage reduction represents the benefits (B) of management and, from the resulting matrix, one can identify the species-site combinations whose management would provide the greatest returns. This can be compared to the costs (C) of management in the form of a benefit-cost (B:C) ratio, as follows:

$$B_{ij}:C_{ij} = [R_{Uij} - R_{Mij}] / [C_{SMij} + (P'_{Iij} \times P_{PDij} \times C_{SCij})] \quad (8)$$

The B:C ratios can be ranked from highest to lowest to determine relative priorities, within a species-site matrix. Where V_j has been expressed in monetary terms, B:C > 1 indicates a worthwhile expenditure, although when there are budget constraints only those actions with the highest returns would be implemented (see also Choquenot et al. 2004).

Another application of the framework would be to assess the relative return from management interventions that might be applied across all areas and species, e.g., a new hull fouling regulation. This requires estimating risk for the 'with' and 'without' management situations to calculate R_{Uij}, R_{Mij} and C_{ij}. The B:C ratio for measure X can be represented as follows:

$$B:C(X) = [SumR_{Uij} - SumR_{M(X)ij}] / SumC(X)_{ij}. \quad (9)$$

A similar ratio can be estimated for alternative management interventions (or combinations thereof) and the ratios compared to see which delivers the greatest benefits (i.e., damage avoided) per dollar spent. For exercises such as these, one could use representative species or taxa (i.e., representing key attributes of interest for risk species) rather than a comprehensive list of target species, in order to keep the evaluation process manageable.

In the case of *Undaria*, Sinner et al. (2000) applied a simplified version of this framework to selected HVAs by estimating the parameters $P'_I \times P_{PD}$ (as a single parameter) and P_{SC} in order to obtain cost estimates for ranking a range of management options. V and I were assessed qualitatively and used to describe the likely outcomes (i.e., benefits) of each option to inform decision makers in their selection of a preferred approach.

8.6 CONCLUSIONS

The aim of this framework is to provide an approach to setting priorities that caters for marine biosecurity threats (from existing or potential pests) to different types of coastal values or stakeholder sectors (e.g., aquaculture, conservation) at different scales of interest (e.g., national vs internal border control). It is a framework that promotes forward planning to avoid poorly informed ad hoc decision-making.

For full application, this approach would require a significant amount of data about particular pest species and the vulnerability of high value areas to those species. In many circumstances, this information will not be available or the analyst might consider that it is not possible to identify the species that pose the greatest risk (e.g., given uncertainty regarding how an organism will behave in a new environment). However, the framework can be simplified to accommodate these situations, e.g., by using representative species or taxa rather than a complete list of target species. Furthermore, at least in certain situations, some of the parameters or even dimensions of the framework can be condensed if there is insufficient information, or deleted if the management question does not require their consideration.

The data for implementation of this framework can be accumulated and refined over time, and there is clearly scope to automate the assessment process. Rudimentary first applications covering a range of scales and values, if properly documented, will provide a useful platform for further applications and, given that many policy decisions will require consideration of similar parameters, the tool will become progressively more sophisticated. In further development of the framework we emphasize the importance of information sharing among the various scientific disciplines and stakeholder groups involved in biosecurity both in New Zealand and overseas, since many of the issues and needs raised in relation to the marine environment are common to all.

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Chapter 9

General Discussion and Conclusions

9.1 POST-BORDER MARINE BIOSECURITY AND LESSONS LEARNED FROM *UNDARIA*

The purpose of this thesis is to evaluate the feasibility of post-border management of marine pests in New Zealand, based on research initiated at a time when there was a widely held view that the management of *Undaria* and other established pest organisms was likely to be futile. Since then, this mind-set has gradually changed and support for the concept of post-border management in marine systems has gained traction. Government moves to consider options for *Undaria*'s management in the late 1990s indicated some acknowledgment that post-border management may be feasible in some instances. Subsequently, in 2003 the Biosecurity Strategy for New Zealand identified the institutional arrangements necessary to ensure delivery of biosecurity outcomes in the marine environment, with post-border pest management now explicitly a function of the 'Post-Clearance' section of Biosecurity New Zealand. These broad shifts have been accompanied by: the development of regulatory frameworks for assessing marine pest risks (Hewitt et al. 2004); the development of post-border surveillance and incursion response systems for the marine environment (Wotton and Hewitt 2004); and the development of related operational and underpinning research programmes. Simultaneously, stakeholder groups such as aquaculture companies and their national agencies (e.g., the New Zealand Mussel Industry Council Ltd) have become increasingly active in the development of tools (e.g., codes of practice) to minimise the risk of inadvertent transfer of *Undaria* and other pest organisms with their activities.

The risk-based framework proposed in Chapter 8 was developed to support decision-making post-border, and provides a useful model for considering both the feasibility of management and related priorities. While the other chapters were not specifically conceived to support this framework (see preface to thesis), they nonetheless provide scientific and technical knowledge that is relevant to its application. Knowledge of

Undaria's commercial and ecological impacts (Chapters 2 and 3), natural dispersal potential (Chapter 4), and pathways of human-mediated spread (Chapters 2 and 5), are all critical to the assessment of *unmanaged* risk within the context of the model. Similarly, Chapters 6 and 7 provide examples of risk treatment approaches that would contribute within the Chapter 8 framework to the assessment of *managed* risk, and hence to the evaluation of risks in relation to the benefits and costs of management.

Despite the considerable effort and interest in managing *Undaria* at a national and regional level, the ultimate government decision was not to proceed with the seaweed's management except in a limited way, as described in Chapter 2. Clearly for *Undaria*, too little was done, and too late in the invasion process. Despite the national situation, however, there was some success at local scales. Furthermore, the lessons learned from *Undaria* and management approaches developed have contributed significantly to the knowledge base, capability-building, public awareness and institutional arrangements required for effective management of marine pests in New Zealand. The move away from a strong interest in *Undaria* in part reflects the fact that there have been incursions of other organisms whose potential impacts are regarded with greater concern within government agencies and amongst stakeholder groups, and which are considered relatively manageable owing in part to their confined distribution. Recent examples include the ascidians (also called tunicates or 'sea squirts') *Styela clava* and *Didemnum vexillum*, with the latter discussed in more detail below. For *Undaria* itself, management interest waned for reasons outlined in Chapter 2, namely:

1. Lack of evidence for significant impacts: this resulted in a lack of support from key marine users whose co-operation was essential to a comprehensive management programme. It also resulted in a lack of long-term central government commitment to *Undaria* management because of the view that limited funds should be spent on biosecurity issues of equal or greater importance.

2. A view that eradication of established populations or containment of *Undaria* was not feasible given the seaweed's widespread distribution: this meant that the focus of management was on a few geographically remote high value areas (HVAs), termed 'special HVAs' by Sinner et al. (2000). These were areas not considered to be susceptible to the natural spread of *Undaria* where the risk of the seaweed's incursion could be minimised through management of human transport vectors.

In the discussion below I consider in more detail these two themes, because many of the issues that emerge are applicable to marine pests generally. Building on the knowledge gained from *Undaria*, I then discuss key elements of a post-border management framework for marine pests that extends the concepts put forward in Chapter 8.

9.2 LACK OF EVIDENCE REGARDING IMPACTS

Chapter 3 described a study of *Undaria*'s ecological effects on low-shore communities based on work in Lyttelton Harbour, and also drew attention to some general considerations for assessing invasive species' impacts. Despite the intense interest in *Undaria* in New Zealand and globally, there has been little further advancement in knowledge regarding the effects of *Undaria* as it continues to spread. This largely parallels the situation for other marine pest species; despite widespread concerns regarding impacts there is still little known for many potential pest organisms (Garcia-Bethou et al. 2005). Furthermore, as was the case for the *Undaria* study in Chapter 3, knowledge is often derived from research that is limited in scope and has primarily site-specific relevance (e.g., Britton-Simmons 2004; Wikström and Kautsky 2004; Chapman et al. 2005; Neira et al. 2005; Sánchez and Fernández 2005).

Determining the consequences of invasion and the factors that lead to pest status is clearly a major challenge in invasion biology. As noted in Chapter 8, however, there is little optimism in the scientific community that invasion success or failure can be predicted with any certainty even for well-studied species or systems. In fact, Moyle and Light (1996) suggest that there is only one firm invasion rule, which they term the 'Frankenstein Effect': that new invasions are likely to have unexpected consequences. This is evident for *Undaria* and other species in New Zealand where knowledge of invasibility and impacts may not translate to adjacent areas, or hold for the same place over time, for example because invaders interact with different suites of indigenous species as they spread (Gust and Inglis 2006). In the case of *Undaria*, for example, spatial patterns and seasonal trends in sporophyte numbers can differ greatly between sites in close proximity, and densities can show a marked interannual variation at a particular site (Hay and Villouta 1993; Chapter 3). While a single *Undaria* sporophyte can in theory seed a new population, it is not a foregone conclusion that this will happen, or that conditions in the recipient habitat will favour the formation of significant infestations (i.e., high density canopy-forming stands). While the reasons

for such variation are unclear, a number of causal mechanisms have been invoked including physical disturbance, grazing, nutrient supply and a complexity of other factors, many of which may operate at local scales (Campbell 1999; Campbell et al. 1999; Valentine and Johnson 2003, 2004, 2005). While such spatio-temporal variation makes it difficult to predict invasibility and impacts, it also provides opportunities to explore the underlying causes. Nonetheless, it means that assessment of the impacts of invasive species in New Zealand will often default to expert opinion and overseas experience (Chapter 8), even though they both have clear limitations.

Where lack of hard evidence for impacts leads to regulatory agencies and stakeholders ignoring scientific predictions as to potential risks, management opportunities may unfortunately be lost. As was the case for *Undaria*, when the colonial ascidian *D. vexillum* was discovered in the heart of New Zealand's mussel growing region in December 2001, the response from stakeholders was minimal despite scientific advice regarding its potential as a significant fouling pest to aquaculture. Five years later after the organism infested mussel farms and decimated the crops, the aquaculture industry reacted by undertaking a self-funded full-scale eradication attempt that is still ongoing. Had a similar eradication effort been made at the early stages of invasion, *D. vexillum* would almost certainly have been eliminated for relatively little cost (Coutts and Forrest 2007), highlighting the benefits of applying the precautionary principle as proposed in Chapter 3.

9.3 ISSUES AROUND VECTOR MANAGEMENT AND INCURSION RESPONSE

9.3.1 Vector management

Chapters 2, 4 and 5 highlighted the range of human vectors that can be important in the spread of *Undaria*, with vessel movements and aquaculture pathways being particularly important domestically. With the acquisition of knowledge around pathways for equipment and seed-stock (Chapter 5), and development of tools for treatment of associated biofouling pests (Chapters 6 and 7), management of aquaculture vectors is becoming increasingly feasible. Unfortunately with regard to *Undaria*, the absence of this knowledge over the last two decades (and the lack of awareness and willingness to undertake management) has meant that the seaweed gradually spread to the main mussel farming regions in New Zealand. While management of pathways to these infested

areas is now clearly pointless for *Undaria*, future management needs may arise as new aquaculture regions develop and as patterns of *Undaria* distribution change, as discussed in Chapter 5. Simultaneously, the ability to manage all existing and potential aquaculture pathways remains highly relevant to future marine pest incursions. As such, the knowledge and tools developed in this thesis remain relevant, and form a foundation of information that can be built on in subsequent studies.

In the case of vessel traffic, while many transport mechanisms apply, hull fouling is implicated in the domestic spread of many unwanted marine organisms in New Zealand, and is a key pathway for the transfer of *Undaria* and biofouling species with the propensity to cause adverse economic impacts (Hay 1990; Coutts 2002; Floerl et al. 2004; Floerl and Inglis 2005; Coutts and Forrest 2007). For vessel traffic generally, and hull fouling in particular, the development of management solutions for non-indigenous species is arguably more difficult than in the aquaculture situation. On the one hand, management tools for hull fouling are readily available; for example treatments such as regular application of anti-fouling paints are highly effective in enhancing the resistance of vessels to colonization by hull fouling organisms (Coutts and Taylor 2004; Floerl and Inglis 2005; Floerl et al. 2005). Similarly, it is theoretically possible to identify and treat (e.g., by *in situ* cleaning) high risk vectors, or quarantine their movements. However, the implementation of such measures at a national scale poses difficulties that reflect: the greater scale of the problem (e.g., there are tens of thousands of vessels nationally); the diffuse and stochastic nature of vessel activity; and issues in gaining the co-operation and compliance of vessel operators (Dodgshun et al. 2004).

Current approaches to vessel management rely primarily on education of vessel operators to encourage behaviours (e.g., hull inspection and anti-fouling) that mitigate biosecurity risks (Hewitt et al. 2004). It is perhaps naïve to expect voluntary approaches to significantly reduce vessel risks when there is no strong incentive to take personal action; as a comparison, even when people's lives are at stake they still die in boating accidents through lack of appropriate safety equipment. Furthermore, even when awareness is raised, support for vector management measures is not necessarily assured; for example, where evidence of impacts is not available (see above), where management measures are perceived as inequitable (Sinner et al. 2000), or where there are high risk vectors that are essentially unmanageable (e.g., because their movements are unpredictable). Effective management of biosecurity risks associated with vessel

movements will conceivably require mandatory approaches, for example mandatory inspections of moored vessels and enforcement of regular anti-fouling regimes. While there would undoubtedly be strong objection to such approaches, they are little different from requirements that motor vehicles meet certain emission standards, pass a 6-monthly warrant of fitness and be annually registered for use on the road.

9.3.2 Incursion response

Objectives of incursion response

Complete eradication of a marine pest incursion early in the invasion process is clearly the most desirable management outcome, although this is often not achievable or may be undermined by re-invasion. A number of authors have highlighted the key elements, both technical and non-technical, needed for successful eradication of marine pests (Myers et al. 2000; Bax et al. 2001; Anderson 2005; Coutts and Forrest 2007). These include requirements such as: (i) the need for 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. Failure to achieve the latter has the potential to be a significant stumbling block for marine eradication programmes. The ability to detect pest organisms depends on a variety of attributes of both the pest and the receiving environment, as outlined in Chapter 8. With the considerable effort placed on pest surveillance and delimitation surveys in New Zealand and elsewhere, some sophisticated approaches have now been developed (Hayes et al. 2005; Gust and Inglis 2006). Nonetheless, they are still based on sampling and detection at defined levels of confidence and cannot guarantee finding all individuals. In fact, many first incursions to New Zealand or to new regions are found by accident or enquiry rather than active surveillance, as has been the case for a number of marine species including *Undaria*, *Styela clava* and *Didemnum vexillum* (e.g., Hay and Luckens 1987; Gust et al. 2005; Coutts and Forrest 2007; Chapter 2).

There is an additional argument that understanding the invasibility of different habitats could facilitate decisions around eradication. A pest organism will be more difficult to eradicate from an environment where it readily attains pest densities than one where it

struggles to establish. A corollary is that failure to eradicate a pest from one locality does not necessarily translate to failure elsewhere. In the case of the Big Glory Bay eradication programme, high water clarity and a high density of floating marine farm structures provided ideal habitat for prolific infestations of *Undaria* to develop, meaning that populations could quickly re-establish from a low density of sporophytes. By contrast, *Undaria* has failed to establish in natural seabed habitats in some parts of Nelson and Marlborough even where high densities exist on adjacent marine farms. For example, experimental work undertaken alongside the *Undaria* dispersal work in Chapter 4 included tagging sporophytes at a shallow subtidal population in outer Pelorus Sound, with the intention to document the subsequent pattern of spread. However, in the following season the *Undaria* population had disappeared from natural habitats in this locality (B. Forrest, unpubl. data). Interestingly, had we attempted to 'eradicate' the natural population in the first year, we would have claimed success in a situation where no intervention was necessary. Hence, natural environmental constraints clearly have the potential to complement human intervention and contribute to the success of eradication programmes. However, because eradication programmes are essentially uncontrolled experiments (Simberloff 2001), they never provide the ability to accurately gauge the relative importance of the two.

As discussed in Chapter 8, post-border management may involve not just eradication of existing or new incursions, but other measures such as containment to prevent spread or population control to manage pest densities to levels that avoid adverse effects. Clarity around management objectives is a critical element of post-border management, because decisions will affect both the scope of the programme and the time-frame over which it needs to be maintained. If the purpose is eradication for example, then effective pest surveillance and vector management are likely to be critical to success, but intensive management activities may only be a short-term requirement. On the other hand a population control programme (e.g., to manage densities to a level that avoids adverse effects) is likely to require a long-term commitment, but issues around pest detection and management of re-invasion may be less important (see Chapter 8).

Physical response methods

The example of *Undaria* highlights the problems in a marine environment where control relies on visual detection and hand removal (e.g., Hewitt et al. 2005). The success of hand removal depends on detecting and removing sporophytes before they

reach maturity and release spores, and failure to achieve this was one of the major downfalls of the southern New Zealand management programme (pers. obs.). Furthermore, ongoing and regular surveillance is needed to remove each new sporophyte that develops from the gametophyte 'seed bank', because this may persist for more than one year (Hay and Sanderson 1999; Hewitt et al. 2005). Because of the difficulties associated with hand removal, there has been a great deal of interest in complementary control methods that also target the microscopic life-stages of *Undaria*. As a result of research in relation to *Undaria* and other pest organisms, there is a toolbox of physical and chemical response methods that are relevant to this purpose (Creese et al. 2004; Wotton et al. 2004; Anderson 2005; Coutts and Forrest 2005, 2007; Coutts 2006). For example, Coutts and Forrest (2005, 2006) developed cost-effective approaches for sterilising wharf piles and marina pontoons by encapsulating them with polyethylene. As well as containing pest organisms, mortality can be achieved through addition of chemicals, or by allowing the passive development of anoxic conditions (e.g., Coutts and Forrest 2005). This type of approach is suited to relatively sedentary pest organisms and is highly labour intensive, hence likely to be applicable in only small-scale eradication programmes. However, the success of a small-scale programme (based on heat treatment) was demonstrated on the hull of the fishing vessel Seafresh 1, in an eradication campaign that almost undoubtedly prevented the establishment of *Undaria* at the Chatham Islands (Wotton et al. 2004).

Biological control

Classical biological control, involving the introduction of natural enemies (i.e., other non-indigenous species) to control target pests, does not appear to have been attempted as a response method for *Undaria*, nor any other marine pest organism. In terrestrial systems where such approaches have been widely applied (with varying degrees of success), biological control is usually regarded as a means of suppressing a pest to a level that avoids significant impacts, rather than as an eradication tool (Lafferty and Kuris 1996). Classical biological control is often considered high risk because of the potential for direct and indirect non-target effects on native ecosystems, for example when host-shifting occurs (Cory and Myers 2000; Pearson and Callaway 2003). For these and other reasons, this classical approach is not generally favoured in marine environments (Secord 2003), although a number of proposals in this regard have been put forward in recent years (e.g., for *Caulerpa taxifolia* by Meinesz 1999).

There are nonetheless a number of marine examples where alternative biologically-based control measures have been successful in particular circumstances (Minchin and Duggan 1989; Cigarría et al. 1998; Culver and Kuris 2000). Culver and Kuris (2000), for example, describe what appears to be a successful eradication of the epizoic sabellid polychaete *Terebrasabella heterouncinata*, a pest to the abalone industry in California. In that case the management approach involved reducing natural population densities of a preferred native host (an intertidal snail) below the threshold for sabellid transmission. Success was facilitated by the biological attributes of the invader, primarily that its capacity for natural dispersal was limited by its benthic larval stage. For *Undaria* there is also scope to consider augmentative biocontrol, where control is exerted by enhancing populations of natural predators (Secord 2003). *Undaria* is highly palatable to grazers because of its low phlorotannin concentrations relative to other native New Zealand brown algae, and benthic grazer control may be a key factor that explains the paucity of *Undaria* in natural habitats despite high densities on adjacent suspended structures (B. Forrest, unpubl. data).

Commercial harvest

Commercial harvest as a population control approach has a strong foundation in terrestrial pest management in New Zealand (Parkes 2006), and has recently received attention as a control measure for *Undaria* by Biosecurity New Zealand. This has primarily been in response to interest shown by various stakeholders who see a commercial opportunity. As well as a domestic market, overseas markets for *Undaria* exist by virtue of the fact that fresh or partially treated (e.g., blanched and salted) *Undaria* can be exported to Asian countries during their summer/autumn season when sporophytes are not present (Hay and Gibbs 1996). However, the efficacy of wild harvest as a control measure for *Undaria* may be limited for a number of reasons discussed by Sinner et al. (2000). For example, areas most accessible for harvest may not have economic densities, or may not have *Undaria* of suitable quality for human consumption, either because of poor water quality or poor product quality (e.g., the less desirable morphotypes of *Undaria*). Furthermore, the crop of *Undaria* that could be harvested from natural shores in New Zealand is probably minor compared to what could be harvested (with less effort) from fouled structures (e.g., marine farms), or grown by cultivation (Gibbs and Forrest 1999), and there is a recognised issue that permitting wild harvest could lead to incentives to deliberately spread *Undaria* for commercial gain. At this stage, it nonetheless appears that Biosecurity New Zealand

will release a limited number of permits for wild harvest to evaluate its efficacy as a population control measure.

9.4 BROADER CONSIDERATIONS FOR POST-BORDER MANAGEMENT

9.4.1 A post-border management framework

The post-border spread and establishment of non-indigenous marine species and the role played by human-activities in this process is increasingly recognised as a significant aspect of marine biosecurity that has previously been neglected because of greater interest in national border control (Wasson et al. 2001). Examples provided by *Undaria* and other species indicate that even when pest organisms become well-established, there may still be opportunities for management post-border; the benefits gained from even limited successes have the potential to greatly outweigh the costs (e.g., Sinner et al. 2000; Sinner and Coutts 2003; Coutts and Forrest 2007).

As demonstrated by *Undaria*, however, once a new marine organism becomes geographically dispersed, management options become increasingly limited, require long-term commitment, and will often be prohibitively expensive (Sinner et al. 2000). The unmanaged spread of a pest from its first point of introduction will in most instances lead to widespread infestation of vectors, and increased opportunities for HVAs or their donor regions to become infested. Hence, for a new pest incursion the first line of defence in a post-border management framework should clearly be an assessment of whether the organism can be completely eradicated and, if not, whether its spread can be contained. Containment could be regarded as either a long-term management approach or an interim solution to buy time to evaluate long-term options (Wotton and Hewitt 2004).

Eradication or containment at the point of incursion can be described as 'source-led' management approaches, because their purpose is to eliminate or control the entire pest population at source in order to generally protect national values that are threatened. If neither source-led management nor widespread containment is feasible, then prioritisation and protection of HVAs from the adverse effects of target pests becomes increasingly important. Unlike the source-led approach where the purpose is to protect values generally, the 'site-led' approach requires a spatially explicit assessment of HVAs and associated pathways, as described in Chapter 8. Clearly, post-border management such as described in Chapter 8 may contain elements of source-led and

site-led approaches, with the balance between the two determined according to where management effort needs to be placed in order to maximise benefits to biosecurity for the least cost.

In a geographic sense, the various points post-border (between the locality of a new incursion and the values at risk) where management intervention is possible can be considered as 'internal borders' for management. These are analogous to national borders for biosecurity, although management needs and opportunities will clearly differ. At a national scale, for example, there are a limited suite of human-mediated pathways to consider (mainly vessels), and control measures (e.g., ballast water exchange) tend to operate across all species irrespective of their pest status. In the post-border case, on the other hand, pathways can be highly diverse (Hewitt et al. 2004; Chapter 2) and management measures could consist of generic (e.g., hull cleaning regulations) as well as species-specific approaches. Furthermore, whereas national border control primarily focuses on the management of the human vectors of pest introduction, in the post-border case there is the opportunity to manage not just the pathway itself, but also the donor region (e.g., shipping port) where the human transport vector becomes infected.

The remaining discussion considers some of the theoretical and practical issues around the definition of internal borders for post-border management. I consider initially the definition of borders for different stages of the invasion process, but focus on defining internal borders for containing the spread of pest organisms in relation to their dispersal potential. I then discuss approaches to management of human-mediated pathways in relation to natural dispersal barriers, including the relevance of generic management approaches that encompass all species, versus approaches that target particular pest organisms. Finally, I discuss *Undaria* management in relation to these ideas, and present additional marine examples where the utility of managing internal borders has been demonstrated.

9.4.2 Approaches to defining internal borders for management

For eradication of benthic organisms at the border, the area in which related management activities (e.g., surveillance and incursion response) are undertaken is defined by characteristics of pest organisms and their environment (e.g., habitat, pest mobility, dispersal range of planktonic life-stages), and will almost invariably be pest-

and site-specific. Where eradication fails and containment at the point of incursion is not feasible, a significant challenge is to identify internal borders around which associated management approaches will be effective in preventing the widespread dispersal of marine pest organisms, hence reduce risk of infestation in HVAs. Although containment at source is preferred, it is likely that broader containment strategies will be needed. Two approaches for containing the spread of pest organisms in marine environments are direct management of vectors or control of pest population densities. As a management strategy, control of pest populations is of most relevance to relatively sedentary organisms having planktonic dispersal phases. For such organisms population management could seek to reduce pest densities to a level that minimised the risk of vector infection or led to a reduction in spread by natural dispersal. The success of such approaches is based on the premise that inoculation pressure, both in terms of the density of propagules (i.e., larvae or spores) or their frequency of release, is one of the primary correlates of invasion success (Ruiz et al. 2000; Allendorf and Lundquist 2003; Floerl and Inglis 2005; Lockwood et al. 2005; Verling et al. 2005).

The terrestrial analogue for such containment approaches is reflected in the development of 'barrier zones' for pest management. These are internal borders positioned at the spreading front of an invading population around which pest management activities taken place (e.g., Marsula and Wissel 1994). The utility of barrier zone management, involving surveillance for pests, and eradication or control to eliminate or contain populations, has been demonstrated in the case of gypsy moth spread in the United States (e.g., Tobin et al. 2004). A relevant concept developed from this work is that barrier zones may be shifted not only to follow the spreading front, but also backward in order to eventually eradicate the entire pest population. Such approaches can be effective and economically feasible in terrestrial systems, but may not be an optimal strategy unless natural barriers to population spread exist (Sharov and Liebhold 1998).

In marine environments, containing natural spread using approaches analogous to barrier zone management is unlikely to be feasible for most organisms given the relative difficulty of undertaking effective surveillance and population control over large spatial scales. Furthermore, rather than proceeding as an advancing wave, many marine invasions are characterised by sporadic leaps in distribution that reflect an association with human transport pathways. Similarly, controlling pest densities to decrease vector

infection may have merits in some circumstances, but direct management of human vectors to mitigate their risk of spreading pest organisms is likely to be a more feasible approach to containment. Although the terrestrial analogue is therefore not directly transferable, the basic concept of barrier zones can still be applied by defining internal borders around which vector management can be undertaken. As discussed in Chapter 8, the utility of vector management depends on the natural dispersal capacity of pest organisms, and may be pointless in a situation where a locality (e.g., an HVA) is vulnerable to natural spread. Hence a critical step in the definition of internal borders for vector management is evaluation of the natural spread potential of pest organisms; internal borders could in theory be established for 'hubs' of vector activity between which the natural spread of pests organism is prevented or restricted by dispersal barriers. Hence, in Section 9.4.3 I outline some considerations for defining natural dispersal barriers to spread, and show in Section 9.4.4 how such knowledge can assist with definition of internal borders for vector management, and related surveillance and response activities.

9.4.3 Defining natural dispersal barriers

Conceptual basis for defining dispersal barriers

A number of recognised marine pests have wholly planktonic existences (e.g., the Mediterranean comb jelly *Mnemiopsis leidyi*), although the majority are benthic organisms that have a planktonic dispersal phase. This mode of dispersal can facilitate natural spread across scales of metres to hundred of kilometres depending primarily on planktonic duration and hydrological conditions (Gaylord and Gaines 2000; Kinlan and Gaines 2003; Shanks et al. 2003). At broad spatial scales, natural dispersal barriers to planktonic organisms can be caused by oceanographic features such as zones of upwelling or current systems that lead to restricted exchange between water masses or the seaward advection of coastal propagules (Apte and Gardner 2002; Poulin et al. 2002; Waters and Roy 2004; Ayers and Waters 2005; Gibbs et al. 2006; Stephens et al. 2006; but see Viard et al. 2006). For benthic organisms, dispersal barriers may also exist in the form of habitat that is unsuitable for adult life-stages. For such organisms, barriers to dispersal can therefore arise as a function of the interaction between their planktonic dispersal characteristics and their environmental requirements.

Internal borders based on oceanographic and habitat barriers are depicted conceptually in Figure 9.1, where it is assumed that planktonic organisms or propagules travel uni-

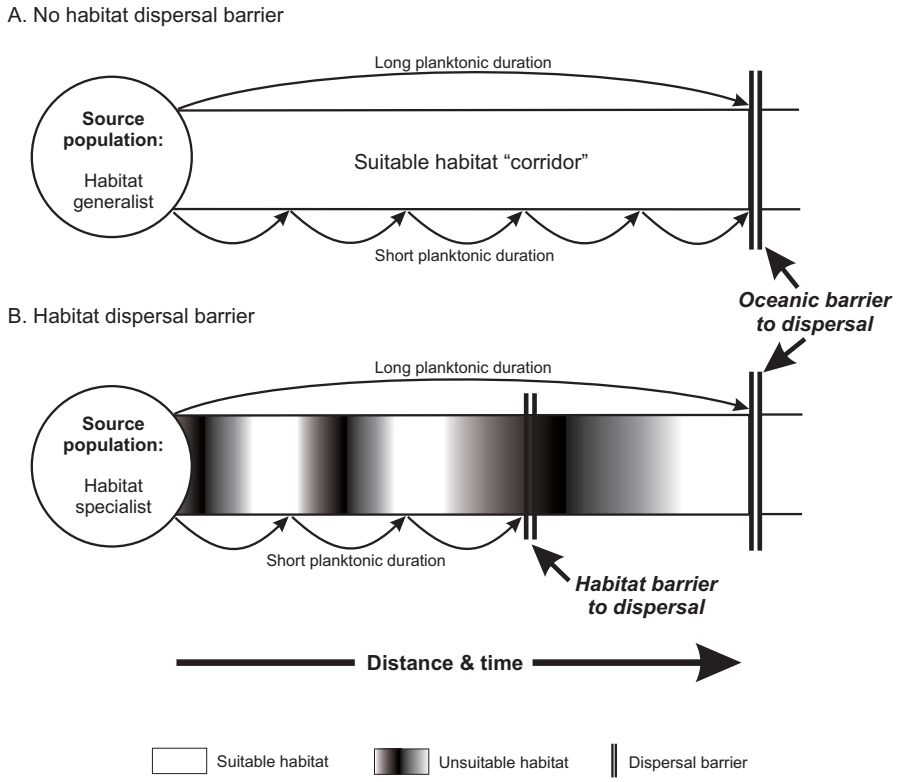

Figure 9.1 Representation of oceanographic and habitat barriers to dispersal in relation to planktonic duration and habitat suitability. The habitat dispersal barrier in B arises in an organism whose planktonic duration is too short to allow dispersal across areas of unsuitable habitat. The blurred boundaries are used to convey the idea that habitat suitability may be variable in space and time.

directionally in an environment within their thermal tolerance. In the case of a habitat generalist that is either wholly planktonic or has an extended planktotrophic dispersal phase, and hence is theoretically capable of relatively rapid long-distance dispersal, it is assumed that maximum dispersal range is limited primarily by oceanographic barriers (Figure 9.1A). The same oceanographic constraint would apply to organisms with more restricted dispersal phases (e.g., lecithotrophic larvae or macroalgal spores with a short planktonic duration), but such species may take considerably longer to spread across the same distance, for example via multiple generations of recruitment and subsequent release of planktonic propagules.

The contrast in dispersal scales in Figure 9.1A is simplistic, especially given the growing body of evidence that actual dispersal distances in organisms with a long planktonic dispersal phase are often less than theoretical maxima, for example because larval behaviour and other processes lead to retention of propagules (Todd 1998; McQuaid and Phillips 2000; Kinlan and Gaines 2003; Shanks et al. 2003; Drake and Lodge 2006; Levin 2006). Conversely, in benthic organisms typically regarded as having restricted planktonic phases (e.g., macroalgae), dispersal can be greater than predicted based on propagules, reflecting strategies (e.g., drifting, asexual reproduction) that lead to episodic leaps in distribution (Santelices 1990; Kinlan et al. 2005; Chapter 2).

The extent and rate of geographic spread in benthic organisms may be further constrained by unavailability of suitable habitat within their planktonic dispersal range (Figure 9.1B). Where habitat suitability is patchy, a benthic organism having a short planktonic duration is particularly susceptible to restriction in its spread because of dispersal barriers, as illustrated in Figure 9.1B. On the other hand, a long planktonic stage may be sufficient to disperse it across natural barriers to suitable habitats. Hence, its ultimate dispersal may be determined by oceanographic features as described above. Alternatively where habitat suitability is marginal Allee effects may arise (Keitt et al. 2001). For example, in the dispersal of a dioecious species like *Undaria*, settlement of conspecifics of the opposite gender may be too far apart for reproduction, or transient environmental conditions may reduce densities below the threshold required for reproduction to occur (Arrontes 2005; Lockwood et al. 2005).

Management limitations in definition of dispersal barriers

From a pest management perspective, determining oceanographic constraints on dispersal, or habitat barriers to establishment, poses numerous challenges. At a broad scale, oceanographic barriers could be determined from a knowledge of water currents, or discontinuities in populations or ecological communities. It has previously been recognised that biogeographic boundaries can be associated with oceanographic dispersal barriers, suggesting that it may be possible to infer the occurrence of such barriers according to biogeographic zones in existing species assemblages (e.g., Gaylord and Gaines 2000; Teske et al. 2006). For identification of internal borders for vector management, such approaches are potentially confounded, however, in that extant distributions may reflect factors other than natural propagule dispersal, such as

historical paleogeographic patterns, prior human-mediated spread, or environmental constraints on distribution. In the case of the latter, issues around climate change in relation to sea surface temperatures are of particular relevance (Stachowicz et al. 2002). A related consideration for oceanographic barriers is that they may lack permanence or exhibit 'leakiness' (Gaylord and Gaines 2000). For example, water current reversals or relaxation of upwelling events could break down broad-scale oceanographic barriers and lead to propagule dispersal counter to mean conditions (e.g., Byers and Pringle 2006). Hence, while oceanographic barriers may prevent propagule dispersal most of the time, there may clearly be times of greater connectivity between water masses.

In relation to habitat, gross differences in attributes such as substratum composition (e.g., rocky reef versus soft-sediment) or wave exposure (e.g., sheltered estuarine versus wave-exposed open coastal conditions) are relatively permanent features at ecological time scales. Such characteristics therefore provide a useful basis on which to define internal borders for managing the human-mediated spread of pest organisms with restricted habitat ranges. At smaller spatial scales within particular habitat types (e.g., rocky reef or soft-sediment), however, there is likely to be considerable spatio-temporal variation in habitat suitability to different pest organisms (Figure 9.1B), recognising that changes in propagule supply coupled with variation in benthic processes that facilitate or retard establishment will determine whether and to what extent a particular species invades a particular habitat at any given time (Pechenik 1999; Kolar and Lodge 2001; Grantham et al. 2003; Verling et al. 2005; Drake and Lodge 2006). As such, defining internal borders at such scales is probably unrealistic.

From Figure 9.1B, the extent to which habitat acts as a barrier to dispersal clearly depends on the particular requirements of pest organisms, and their planktonic dispersal capacity in relation to the spatial scales at which habitat dispersal barriers are distributed. Whereas the habitat requirements of pest organisms are often well understood, a major challenge lies in reliable estimation of their planktonic dispersal capacity. For this purpose a number of modelling approaches have been proposed that capture the bulk of propagule dispersal (e.g., Siegel et al. 2003), but may fail to account, for example, for the tails of the dispersal kernel that reflect episodic long distance transport (Kinlan et al. 2005; Levin 2006). An additional consideration is that the nature and spatial extent of a habitat barrier is likely to be highly specific to individual species, or groups of species with similar dispersal characteristics and environmental

requirements. In this respect oceanographic barriers differ in that they have the potential to limit the planktonic dispersal of all organisms to a similar extent (but see Gaylord and Gaines 2000), although generally they will operate at relatively broad spatial scales. In overview, therefore, internal borders for managing the human-mediated spread of marine pests could be defined at a broad scale (e.g., 100s to 1000s of km) according to oceanographic features that act as dispersal barriers to all species. In conjunction with oceanographic barriers, habitat barriers based on temporally persistent features could be defined at smaller spatial scales (e.g., a few kilometres or greater) for target organisms (or suites of similar organisms), especially those having both restricted habitat requirements and a limited planktonic duration

9.4.4 Management of human transport pathways in relation to dispersal barriers

Defining internal borders for vector management

The definition of natural dispersal barriers for marine pests provides the basis on which internal borders for vector management, and related activities (e.g., surveillance and incursion response) can be identified; management opportunities arise where human transport mechanisms provide the only link between a pest population and an HVA or its donor regions. To illustrate some relevant issues around definition of internal borders, a scenario in Figure 9.2 shows potential pathways for spread of a pest organism between its point of first incursion and an HVA. Spread proceeds either via natural dispersal where there are no barriers to this, or via vector activity between main hubs. A hub as depicted here is a centre of vector activity (e.g., a port environment) that may include multiple vector departure or arrival points (nodes); for example commercial docks, recreational boating marinas and so on.

Figure 9.2 starts with a source population of a pest with the potential to spread by natural dispersal or by human transport vectors. Vector risk, simplistically depicted by the weight of connecting lines between hubs, is related to: (i) attributes of the pest organism that influence vector infection such as density, fecundity, and planktonic duration of propagules in relation to hydrological conditions; and (ii) attributes of vectors that influence infection such as proximity to the pest population and effectiveness of management (e.g., anti-fouling in the case of a vessel hull); and additional attributes that influence risk such as the number of vectors, frequency of their movement, and residence time at destination (e.g., Floerl and Inglis 2005; Muirhead and MacIsaac 2005). In the scenario in Figure 9.2, the ultimate goal of protecting the HVA

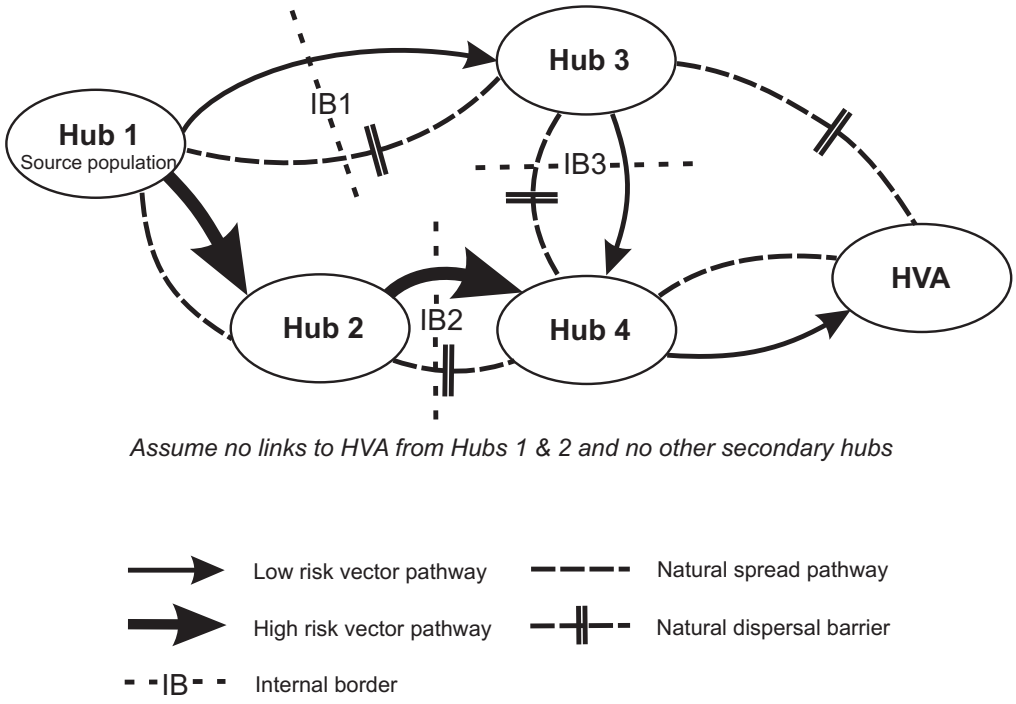

Figure 9.2 Conceptual representation of a simple network of vector hubs, illustrating how internal borders for vector management can be defined according to natural dispersal barriers for pest organisms.

relies on preventing the spread of the pest to Hub 4, because no natural dispersal barriers exist between the two. To prevent infestation of Hub 4, there are three key connections where natural dispersal barriers exist and hence where internal borders could be set up for vector management (i.e., IB1-IB3 in Figure 9.2).

To prevent widespread infestation of hubs (and hence increasing risk to the HVA), initial efforts should logically focus on the two internal borders most closely linked to the source population at Hub 1 (i.e., IB1 and IB2). There may be little point in managing transport pathways between Hub 1 and 2 because no natural dispersal barrier exists. However, this is a cost-benefit decision for managers that would need to consider the rate of natural spread of the pest organism. An organism with limited dispersal capacity may spread so slowly by natural mechanisms that management of human vectors is worthwhile (Chapter 8). Clearly, in a more realistic scenario where there are a web of connections between multiple vector hubs, the benefit of slowing

spread at the first internal border may be that it prevents the infection of other hubs that are more directly linked to HVAs.

Assuming for present purposes that no vector control is undertaken between Hubs 1 and 2, management measures would focus on IB1 and IB2 and comprise vector controls between Hubs 1 and 3 (IB1) and Hubs 2 and 4 (IB2), and development of pest surveillance and response programmes for Hub 3 and 4. As in the case of terrestrial barrier zones, in the event of an incursion leading to uncontrolled spread within Hub 3, IB1 between Hub 1 and 3 could be abandoned in favour of IB3 between Hubs 3 and 4, and a similar management approach adopted. Note that where vector management measures are developed around internal borders in the form of oceanographic barriers, the 'leakiness' of such barriers (see Section 9.4.3) suggests that biosecurity goals should be based around risk reduction rather than prevention of spread. A further point is that oceanographic dispersal barriers may operate in only one direction (Gaylord and Gaines 2000), meaning that situations could arise where management of vector activity between infested hubs has merits only for traffic moving in the direction where the barrier occurs.

Vector management approaches and allocation of effort in relation to internal borders

Where multiple hubs and pathways occur, a key consideration is the spatial allocation of vector management effort. One option, and arguably the most intuitive one, is to prioritise management according to pathway risk which, in the case of Figure 9.2, would mean a greater focus on IB2 than IB1. Alternatively one might apply equal management effort across all hubs and pathways, based on estimates for ballast water by Drake and Lodge (2004) that the most effective strategy to mitigate risk of introduction is to reduce risks across all vessels, rather than eliminating key hubs. The answers to whether one approach is better than another lie in part with consideration of the scale and complexity of the problem. In a complex network of nodes and pathways where multiple hubs became infested by a pest organism, a risk-based management approach would become increasingly less tractable and fraught with uncertainty. The uncertainty in pathway risk analysis is primarily due to stochastic pathways that may lead to the realisation of unrecognised or low probability events that have significant consequences (Chapter 8). Hence, a focus on known or quantifiable sources of risks may lead to important but less identifiable sources being overlooked.

The pest management context also becomes relevant in terms of the approach to vector control. Post-border management in marine environments to date has typically focused on single species (e.g., Sinner et al. 2000; Creese et al. 2004; Wotton et al. 2004; Anderson 2005; Coutts and Forrest 2007), but given that the spatial distribution of internal borders based on habitat barriers will differ among pest organisms, risk-based vector management in the context of multiple pests could be highly inefficient if targeted at single species and their particular vectors. Considered together, issues around uncertainty in pathway risk analysis and inefficiencies in single species management provide a strong argument that, even in situations where risk-based and species-specific approaches are clearly warranted, a greater benefit to biosecurity may arise if such programmes are underpinned by widespread implementation of vector management techniques. By focusing on pathways, such approaches are inclusive of multiple species, as advocated for the Great Lakes by Leung et al. (2006).

Hence, within a post-border management framework a blend of generic and species-specific approaches to vector management is likely to be more desirable than either approach in isolation. In the case of biofouling for example, regular anti-fouling of vessels is likely to have generic benefits in reducing the transfer of high risk fouling organisms (Coutts and Taylor 2004; Floerl and Inglis 2005). It would make sense, therefore, to apply such measures equally across all vessel pathways, especially in situations where implementation is voluntary, and encouraged through education and awareness campaigns. On the other hand, where specific pests are targeted for management and quarantine is critical to success, active intervention approaches may be necessary for specific vectors, such as sterilisation of infected vessel hulls or contaminated aquaculture equipment and seed-stock (e.g., Coutts and Forrest 2007; Chapter 5). However, the relatively high costs associated with the implementation of such tools may prohibit their general use, meaning that their application is limited to situations where risks are unacceptably high.

A notable benefit of generic vector management is that non-specific approaches would limit the human-mediated spread of indigenous biota as well as non-indigenous organisms. There are a number of examples in New Zealand where human transport has spread indigenous organisms beyond natural dispersal barriers, for example as a result of inadvertent transfers with vessel fouling or the deliberate movement of aquaculture seed-stock (e.g., Coutts 2003; Coutts and Forrest 2007). Coutts (2003), for

example, referred to the establishment of a northern New Zealand slipper limit (*Crepidula costata*) in a southern New Zealand harbour following the inter-regional movement of a barge. From a biodiversity perspective, the human-mediated transfer of indigenous organisms may be equally as significant as for non-indigenous species, especially in countries having high levels of regional endemism.

9.4.5 Feasibility of internal border management for different pest organisms

The extent to which different pest species are manageable depends on their natural dispersal capacity, habitat requirements, and a range of other attributes of both the pest organism and its receiving environment, as highlighted in Chapter 8. I use *Undaria* and other New Zealand examples below to highlight how such attributes, and particularly how the definition of internal borders based on oceanographic and habitat dispersal barriers, can be integral to the success of post-border management programmes for marine pests.

Management around oceanographic barriers

The planktonic dinoflagellate *Gymnodinium catenatum* produces biotoxins associated with paralytic shellfish poisoning (PSP), and has been responsible for closures of shellfish aquaculture areas worldwide (e.g., Rhodes et al. 2001). Experience with this organism in New Zealand indicates that definition of internal borders based on oceanographic barriers can have practical applications in marine pest management. In May 2000, a bloom of *G. catenatum* was detected off New Zealand's northwest coastline, in an area that encompassed the country's primary source of spat for mussel (*Perna canaliculus*) and Pacific oyster (*Crassostrea gigas*) farms in other parts of the country (MacKenzie and Beauchamp 2000).

Following the initial discovery, the subsequent detection of high densities of *G. catenatum* cysts in spat supplies led to a voluntary ban on seed-stock movements to all growing regions, and the development of treatment methods to eliminate or minimise cyst densities within infected material so that inter-regional spat transfers could continue (e.g., Taylor 2000; NZMIC 2002). The key pathway targeted was mussel spat transfer to the main mussel growing region in the north of the South Island. Based on knowledge of oceanographic conditions around New Zealand (e.g., Heath 1985; Carter et al. 1998) it was anticipated that these regions may not be vulnerable to the natural

spread of *G. catenatum*, such that management of risks associated with aquaculture pathways was worthwhile (Figure 9.3A, B).

Over the period the spat management programme was in place, the bloom tracked slowly southward to the bottom of the North Island and then moved rapidly northward along the east coast, reflecting a north-flowing coastal current system (Figure 9.3A, B). Although *G. catenatum* was detected at high densities in the Cook Strait region between New Zealand's North and South Islands, and at low densities in waters adjacent to mussel growing areas, PSP toxins were not detected in mussel stocks. Furthermore, the bloom did not progress further southward, conceivably because of the prevailing northerly current flows. Although oceanographic barriers to dispersal are not the only explanation for the failure of *G. catenatum* to bloom in South Island aquaculture regions (e.g., habitat conditions may also have been unsuitable for bloom formation), the apparent restricted dispersal between the North and South Island of New Zealand is consistent with genetic studies of mussels (Apte and Gardner 2002) and seastars (Waters and Roy 2004; Ayers and Waters 2005) indicating a marked north-south disjunction in this region.

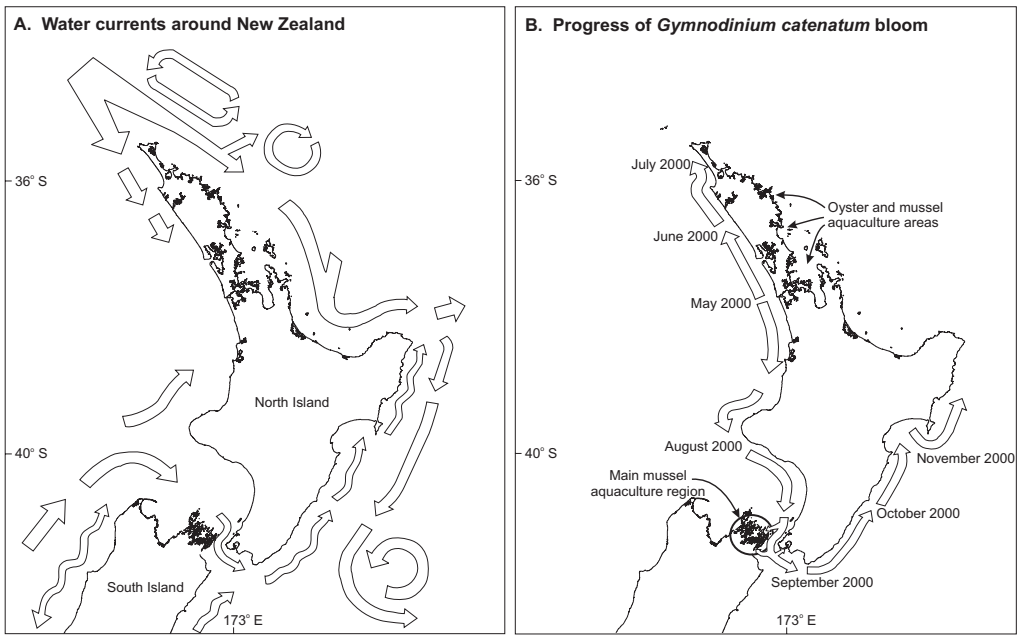

Figure 9.3 A. Water currents around New Zealand (modified from Carter et al. 1998); and B. The spread of the dinoflagellate *Gymnodinium catenatum* during a bloom in May-November 2000 (modified from Mackenzie and Beuzenberg, unpubl.)

Management around habitat barriers

Examples provided by *Undaria* and the biofouling ascidian *Didemnum vexillum* can be used to illustrate the importance of dispersal capacity and habitat barriers in the spread and management of benthic marine pests. Both of these species have a similar propagule dispersal capacity; typically in the order of hundreds of metres per year (Chapter 4; Coutts and Forrest 2007). However, *Undaria* has a number of features that make post-border management considerably more difficult than for *D. vexillum*. A key feature is that, although spore dispersal is limited, modes such as sporophyte drift can lead to episodic leaps in *Undaria* distribution across scales of kilometres or conceivably tens of kilometres (Chapter 4). Within the context of eradication efforts, these strategies make it difficult to define dispersal zones for surveillance, with the latter further constrained by *Undaria*'s temporally persistent microscopic gametophyte life-stage. The fact that *Undaria* can inhabit a range of artificial and natural substrata, and attain a high density population from a single reproductive sporophyll, ultimately mean that failure to detect all sporophylls is likely to lead to uncontrollable infestations where habitat is favourable (see Section 9.3.2).

When the above factors are considered, the reasons for the failure of the southern New Zealand management programme to eradicate or contain *Undaria* at a local scale (km to 10s of km) are understandable (see Chapter 2). Nonetheless, across greater spatial scales (e.g., 10s of km of greater) *Undaria*'s natural dispersal is likely to be prevented by dispersal barriers in the form of extensive tracts of deep water (e.g., between islands), soft-sediment or severe wave-exposure. Hence, where it is feasible to manage human transport vectors, these types of barriers identify internal borders around which containment of *Undaria* may be entirely realistic. This philosophy was the basis of recommendations made by Sinner et al. (2000) for national *Undaria* management, that priority be given to management of vectors to offshore islands of high conservation value that were beyond *Undaria*'s natural dispersal capacity (Chapter 2). Clearly, in the case of *Undaria*, prevention (i.e., containment of *Undaria*'s spread in relation to internal borders) will often be a more tractable management goal than cure, although the success of the Seafresh 1 eradication described by Wotton et al. (2004) is a reminder that the merits of management need to be considered on a case by case basis.

The colonial ascidian *D. vexillum* provides a useful contrast to *Undaria* in that, as well as having a planktonic dispersal capacity limited to hundreds of metres per year, this

organism has a highly restricted habitat distribution. Experience in New Zealand to date suggests that *D. vexillum* will almost exclusively establish on artificial structures (Coutts and Forrest 2007), with seabed populations only maintained where there is a substantial biomass immediately adjacent (pers. obs.). Consequently, the inter- and intra-regional spread of *D. vexillum* in New Zealand has been mediated by movements of vessels and aquaculture equipment, with spread at local scales (10s to 100s of metres) facilitated by artificial structures in close proximity. Such structures (e.g., marine farms, vessel moorings) thus act as stepping stones for the spread of this species in a manner that is conceptually identical to the model proposed for the spread of *Codium fragile* ssp. *tomentosoides* along the Adriatic coast of Italy (Bulleri and Airoldi 2005).

As a result of its restricted dispersal and habitat range, dispersal barriers, and hence opportunities to define internal borders for the management of *D. vexillum*, occur over small spatial scales (i.e., kilometres). By comparison with many other marine pests, these same characteristics make it relatively easy to define surveillance zones and to detect colonies when they are present. Furthermore, a number of incursion response tools are effective in eliminating *D. vexillum* from artificial structures, such that eradication of the organism is technically feasible given reasonable effort, commitment, and quality assurance (Coutts and Forrest 2007). For these reasons, and because the spread of the ascidian to mussel farms has recently resulted in significant fouling impacts, the aquaculture industry embarked on a programme to eradicate the species, as noted in Section 9.2. This is an ongoing programme (started in 2006) with an initial focus on eradication of priority outlying 'satellite' populations of *D. vexillum* that were discovered during regional surveillance of artificial structures. If these population are successfully eliminated, and further human-mediated spread contained, the ultimate goal is to eradicate the species from two remaining areas of significant infestation, in an approach that parallels terrestrial barrier zone management as described in Section 9.4.2.

Management of future pest incursions to New Zealand

There will be some instances (e.g., where risks are high as in the case of *D. vexillum*) where definition of internal borders for specific species at small spatial scales may be both justified and achievable, and other situations where such approaches pose difficulties. In terms of the development of relatively simple management methods that can be applied at local and regional scales, the *D. vexillum* example is probably an

exception rather than the norm. For many other pest organisms management may only be feasible at relatively broad spatial scales, or not be feasible at all. The example given above of *Gymnodinium catenatum* describes a situation where management of a key vector appeared to be worthwhile even for a planktonic organism, in this instance because there appeared to be oceanographic barriers to natural dispersal. This type of success suggests that further consideration of oceanic barriers in a New Zealand context would be worthwhile. Qualitative likelihood estimates of the connectivity between New Zealand ports have already been made by Stanton (1997). Such estimates could conceivably be enhanced through development of more sophisticated approaches, such as the web-based tool described by Condie et al. (2005) for evaluating the connectivity of water masses around the coast of Australia.

For new incursions of the more intractable pest species that will invariably arrive, it seems that novel solutions will be needed to manage at spatial scales within oceanographic dispersal barriers. For example, the northern Pacific seastar, *Asterias amurensis*, which is a voracious shellfish predator and globally notorious marine pest, is a mobile habitat generalist with an extended planktonic larval stage (Sutton and Bruce 1996; Ross et al. 2003). Given the proximity of donor regions at similar latitudes in Australia, and the association of this species with shipping vectors, it is more likely a matter of 'when' not 'if' this organism arrives in New Zealand. Effective post-border management tools for such species do not exist, and solutions may rest in part with development of novel methods such as molecular probes for detection of propagules (e.g., Deagle et al. 2003) or the development of semiochemical technologies for pest attraction (e.g., Ingvarsdóttir et al. 2002). In any new method, there is clearly a need to balance management efficacy against the risk of collateral impacts on the wider environment. In many instances this may mean that promising but high risk solutions are publicly and politically unacceptable (Thresher and Kuris 2004).

9.5 CONCLUSIONS AND FUTURE DIRECTIONS

New Zealand's 'leaky' borders mean that further incursions of non-indigenous marine species are inevitable, and with every new organism comes the increased likelihood that one will cause significant impacts on ecological and other values. Experience with *Undaria*, and more recently with biofouling pests like *Didemnum vexillum*, suggests that effective management of such organisms post-border is possible even when they

become relatively well established. It is also evident that the benefits gained from even limited successes have the potential to greatly outweigh the consequences of uncontrolled invasion. However, as unwanted pest species become increasingly widespread and established, management will need to become increasingly focused on the protection of specific values from adverse effects, but in many instances this will be prohibitively expensive or simply not feasible.

A comprehensive strategy should involve surveillance for new incursions and attempts at eradication of high risk species that are detected. Where this first line of defence fails, the next logical goal is containment of the organism to reduce the risk of spread generally (and to HVAs in particular), with surveillance and incursion response in HVAs where this is feasible. Containment is likely to be best achieved through the management of human transport vectors in relation to internal borders, and should consist of a blend of species-specific approaches, and generic measures that are applied nationally to minimise the human transport of all organisms. Achieving effective vector controls will be a major challenge in post-border management; tools are already available to manage many types of vectors, but the awareness or willingness to implement them are not always present, partly because of prohibitive costs. As such, many high risk pest organisms, even those that are relatively manageable, may eventually spread to HVAs despite best efforts.

The conceivable reality of post-border management in the future, therefore, is that there will be some successes and many failures. Given constraints on budgets there is a clear need, therefore, to determine post-border management priorities for New Zealand and focus on those high risk situations where management is most likely to be successful. One of the first steps in this process should be to identify potential high risk pest species so that incursion response plans can be developed prior to their arrival, and hence ad hoc decision-making avoided. Given a knowledge of potential and existing pests, priorities for post-border management can then be refined based on an understanding of the feasibility of surveillance, incursion response and management of spread in relation to the most important values at risk.

There is clearly a role for science in refining the knowledge and tools on which post-border management priorities and decisions are based. For example, there is a need for the development of novel management tools that can be applied across relatively large spatial scales, and which are publicly and politically acceptable. There is clearly also a

need for a better understanding of the consequences of invasion (Grosholz 2002; Simberloff 2003; Wotton and Hewitt 2004). The importance of this should not be under-estimated because it is a primary driver of stakeholder, regulatory and political interest; while the aquaculture industry interest in the *Undaria* issue was relatively low, for example, the will to manage *D. vexillum* emerged once a significant threat was recognised. Further scientific challenges lie in understanding the factors that lead to pest densities, including reasons for boom and bust cycles, time lags before population explosion, and positive interactions among invasive species that exacerbate their spread and proliferation in invaded habitats (Simberloff and Von Holle 1999; Floerl et al. 2004; Simberloff and Gibbons 2004; Diederich et al. 2005; Grosholz 2005).

In addition to the need for science-based solutions, there is a parallel need for resolution of a number of regulatory issues in marine biosecurity in New Zealand. One of these relates to clarification of agency roles at a regional level. For example there is a lack of clarity as to whether marine biosecurity issues should be addressed as part of regional council resource consent processes. This means that some regional councils have required Marine Biosecurity Management Plans for proposals where biosecurity risks are evident (Chapter 5), while for others the biosecurity issues have been side-lined. Another need relates to clarification around procedures for dealing with 'cryptogenic' species; those whose status as native vs introduced is unclear (Carlton 1996). Uncertainty regarding the status of *D. vexillum* in New Zealand in part contributed to the lack of central government response to an obvious fouling threat, resulting in a lost opportunity to eradicate this organism at a stage when such an outcome was clearly achievable (Coutts and Forrest 2007).

In conclusion, biosecurity is perceived as critical to New Zealand's viability as a nation, and there is an increasing public awareness and political support for marine biosecurity initiatives. We now have a single central government agency responsible for biosecurity working under one main piece of legislation, which is developing management systems spanning pre- to post-border. An effective biosecurity system will conceivably consist of vector management, surveillance, incursion response, and control measures that target particular pests or suites of functionally similar species (e.g., biofouling organisms), coupled with generic vector management approaches that aim to reduce human-mediated transport of all organisms at a national scale. New Zealand's geographic isolation and low population, hence relatively low level of vector activity,

makes the management of human-mediated pathways of spread entirely feasible in many circumstances. Hence, while there are clearly many challenges in the post-border management of marine pests, this is nonetheless a realistic goal, and probably moreso in New Zealand than in any other country in the world.

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