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Research article

Rapid response to non-indigenous species. 1. Goals and history of rapid response in the marine environment

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Abstract

This paper is the first in a three-part series that addresses rapid response as a management approach to the recurrent problem of colonization by non-indigenous species, invasive tunicates in particular. "Rapid response" refers to the steps taken, starting before detection of the invasion of a non-indigenous species, through a decision process that may culminate in an attempt to eradicate the species before it becomes established in the new habitat. Rapid response is the second line of defense against non-indigenous species, when prevention measures have failed. We review the goals of rapid response, and its history in the marine environment, to place rapid response into context for the subsequent papers which will review the history of non-indigenous tunicate management in Prince Edward Island, Canada, and propose a framework for rapid response.

Key words: rapid response, tunicates, invasive species management

Introduction

The increasingly rapid speed of marine transportation has had the unintended side-effect of reducing transit time for propagules of aquatic organisms from weeks or months to as little as hours or days. This has greatly increased the pool of species transported to new areas; some of these introduced non-indigenous species will successfully establish; and a subset of these will become invasive pests. The end result is the homogenization of the species composition of widely separated aquatic communities and the destabilization, at least temporarily, of ecosystem structure that may result in the loss of valuable resources or functions. Preventing introductions between locations is obviously the most effective means of avoiding ecosystem disruption but there are no fool-proof strategies to accomplish this. For example, Carlton and Geller (1993) estimated that several thousand species are typically transported in ships' ballast water on any given day. Sea chests may be as important as ballast water in transporting non-indigenous species – recent work in Australia and New Zealand detected at least 150 different organisms in sea chests, including known invasive species such as green crab (*Carcinus maenus* (Linnaeus, 1758)), clubbed tunicate (*Styela clava* Herdman, 1881), and vase tunicate (*Ciona intestinalis* (Linnaeus, 1767)) (Coutts et al. 2003; Coutts and Dodgshun 2007). The number of species being transported by other vectors in the marine environment has not been evaluated; however, co-transfer of organisms during importation of molluscs for stocking or aquaculture purposes is generally accepted to be a major vector of inoculation of non-indigenous invasive species. Regardless of the vector, even with the best prevention strategies, some incursions are inevitable (USGS 2000; McEnnulty et al. 2001; Wotton and Hewitt 2004); consequently, managing new introductions becomes *de facto* the best possible practice. Critical to this practice is a structured or formal procedure that first allows for early detection of incursions and then determines the best response.

The capacity to quickly respond to an invasion in cases where prevention fails is known as "rapid response" and is an essential element of the marine invasive species management programs being developed in many countries, e.g., Australia, New Zealand, and the USA (McEnnulty et al. 2001; NEANS 2003; NEANS 2006; NISC 2003; WANS 2003; Wotton and Hewitt 2004). Canada currently lacks a formal rapid response procedure for non-indigenous marine or freshwater species; however, our aim is to use worldwide examples as guides to develop a Canadian rapid response planning protocol. This paper is the first in a three-part series on rapid response. Here, we review the scope and goals of rapid response, and present international examples of rapid response in the marine environment. In the second paper, we present case studies of non-indigenous tunicate invasions and subsequent management actions that have occurred in the estuaries of Prince Edward Island (PEI) within the past decade. The final paper will propose a framework for the planning and execution of rapid responses in Canadian waters and relate these steps to the process used to manage tunicate invasions in PEI.

The scope and goals of rapid response

Rapid response is considered to be the "second line" of defence against non-indigenous species, where prevention is the "first line". Invasions are more likely to be addressed successfully in their early stages. Populations that are still localized and have low abundance are more likely to be contained and eradicated than well-established populations (NISC 2003). In later stages of an invasion, with a more widely established population, partial mitigation of negative impacts still might be possible. However, in most cases, an effective rapid response leading to eradication will be less expensive (both economically and ecologically) than a long-term nuisance species management program (NISC 2003).

Most proponents of rapid response strongly recommend that the goal should be eradication of the target species (Crooks and Soulé 1999; McEnnulty et al. 2001; WANS 2003). Myers et al. (2000) define eradication as "the removal of every potentially reproducing individual of a species or the reduction of their population density below sustainable levels". Anything less than eradication means that the species and any attendant problems are here to stay. McEnnulty et al. (2001) consider rapid response in the context of Integrated Pest Management, in which eradication is an explicit goal only in extremely limited circumstances where the target pest is confined to a very restricted range. Permanent establishment is, however, the most likely outcome of species colonization in aquatic systems because detection and control are difficult and organisms often disperse rapidly. Rapid response in these instances involves assessing which species-management goals are attainable. The goal of the final response may be: eradication, containing the problem to a given area, suppressing population abundance to slow its spread, suppressing population abundance below an economic or ecological threshold, or learning to live with the problems caused by the species (Myers et al. 2000). Sharov and Liebhold (1998) developed a model to analyse the net benefits of management strategies for non-indigenous pests. The optimal strategy shifted from eradication to slowing the spread to finally doing nothing as the area occupied by the pest increased, the negative impact of the pest per unit area decreased, decisions were based on short-term economics, or the risks associated with uncertainty of predictions were smaller.

Wotton and Hewitt (2004) identified three main components of an effective rapid response system:
(1) Processes and plans to guide response actions
(2) Tools with which to respond
(3) The capability and resources to carry out the response.

The next section illustrates how these components, or the lack of them, have contributed to the success or failure of rapid responses world-wide.

The history of rapid response in the marine environment: International examples

Rapid response efforts are not new; documented examples of attempts to eradicate the early stages of marine invasions go back more than half a century. We present here a chronological overview of the few well-documented international case studies to illustrate the elements of successful and failed attempts at control of a marine invader, including one case involving an invasive tunicate.

Example (1): *Urosalpinx cinerea* (Say, 1822), Atlantic oyster drill or American whelk tingle – England, 1920. The Atlantic oyster drill is a predatory gastropod thought to have been introduced into England from eastern North

America with eastern oysters *Crassostrea virginica* (Gmelin, 1791) around 1880. Identification was problematic - the oyster drill was not positively identified as non-indigenous to England until 1928 (Orton and Winckworth 1928). Lacking a pelagic stage, dispersal was limited and slow. Indeed, the species has been largely limited to areas where eastern oysters were stocked or held for market (Orton and Winckworth 1928; Hancock 1959; Spencer 1992).

Oyster drill infestations caused very high mortalities of juvenile European flat oyster *Ostrea edulis* Linnaeus, 1758 (Hancock 1959; Spencer 1992). By 1954, large-scale removal and destruction of oyster drills were being advocated to improve survival of oysters (Hancock 1959) and a variety of mechanical, chemical, and biological controls were investigated. Mechanical or suction dredging methods proved ineffective and led to unacceptable losses of oysters (Hancock 1959). Handpicking was deemed too labour-intensive, and experimental traps were impractical in the large area that needed to be covered (Spencer 1992). Over 1000 chemical compounds were screened in the laboratory between the 1940's and 1970's but none were found that also did not kill the oysters or have unacceptable environ-mental costs (Loosanoff et al. 1960; Spencer 1992).

Abandoning eradication as a solution, the bivalve industry altered its husbandry practices. Commercial hatcheries reared juvenile oysters in trays or bags, behind fenced barriers, or under plastic netting, to a size relatively safe from predators (Spencer 1992). In addition, the Molluscan Shellfish (Control of Deposit) Order enacted in 1974 prohibited transfer of mollusks between designated areas, except under very restrictive license conditions, which effectively controlled the further spread of oyster drill and several other non-indigenous pests (Spencer 1992).

<u>Example (2): *Sargassum muticum* (Yendo, 1907), Sargasso weed – England, 1973.</u> Attached *Sargassum muticum* was discovered in the Isle of Wight in the early 1970s, and then at additional sites around the Solent (Critchley et al. 1986). The species had already established a reputation for invasiveness and causing problems for physical structures, aquaculture, recreational activities, and biodiversity following its arrival in western North America. The likely vector to both Europe and North America was the importation of Japanese oyster *Crassostrea gigas* (Thunberg, 1793) for aquaculture purposes (Critchley et al. 1986).

The management of the *S. muticum* problem in England was formalized when an *ad hoc* working group met in May 1973 and decided to attempt eradication. Mechanical methods and herbicides were considered inappropriate due to the ecological sensitivity of the affected areas. Hand-picking was considered a promising method because the entire plant and holdfast had to be removed to prevent regeneration. In addition, the public was enlisted to report sightings of the pest to Portsmouth Polytechnic Institute. A *Sargassum* "Wanted" poster was produced and widely distributed along the south coast of England. This public awareness campaign provided important information on the distribution of *S. muticum*.

Unfortunately, eradication was less successful than the public awareness campaign. Eradication attempts began in May 1973, and continued through 1974 and 1975, but the plant had already reproduced and rapidly recolonized the cleared areas. By September 1976, it was obvious that manual removal was ineffective (Critchley et al. 1986). Biological control was investigated but all herbivores tested preferred to graze on native alga, and some actually aided dispersal of *S. muticum* by causing fertile lateral branches to break off and float away.

The end result was that no effective control methods were developed and *S. muticum* continued to disperse. A large standing stock of alga and its associated epibionts developed in areas where few macrophytes were found previously. Major changes to habitats and community structure were observed, and the alga restricted recreational usage of marine waters in several areas of southern Britain.

This was the first attempt at extensive control of a 'marine weed'. Several other non-indigenous algae, also introduced with Japanese oysters, became established in the same area but have remained relatively confined and apparently have not threatened the local ecology. Partially due to these potential algal threats, the Wildlife and Countryside Act (1981) of Great Britain was enacted to control the importation of all non-indigenous marine species (Critchley et al. 1986).

<u>Example (3): *Caulerpa taxifolia* (Vahl, 1802), "Killer" alga – Mediterranean, 1984; Australia, 2000; California, 2000.</u> A strain of the tropical alga *Caulerpa taxifolia* has invaded several temperate regions of the world (Meinesz 2002).

This strain produces only male gametes but a fragment from any part of the alga can produce a new plant. To date, all specimens examined from the Mediterranean Sea and California, and from European aquaria, are genetically identical.

(a) Mediterranean Sea, 1984

In the early 1980s, the curator of the tropical aquarium at Stuttgart, Germany, noticed a strain of *Caulerpa taxifolia* that grew vigorously, even in cool water, and served as a food source for herbivorous tropical fishes. The Oceanographic Museum of Monaco acquired cuttings in 1982 (Meinesz 1999). In 1984, a small patch was noticed in the Mediterranean Sea near the museum outfall but it was assumed that it could not survive winter. A survey in spring 1989 confirmed that the initial patch had not only survived several winters but was spreading. There was disagreement as to whether this was a problem because some scientists viewed this alga as a substitute for the native *Caulerpa prolifera* (Forsskål, 1775) that had died back during harsh winters. Eradication by hand picking was considered in autumn 1991 but *C. taxifolia* had already spread along 3 km of Monaco's coastline. The issue of control was to be discussed at a conference in Nice, France, in December 1991, but it was cancelled by the host university because the topic was considered too controversial.

In hindsight, it was already too late to control dispersal of the species by 1991. One uncontrolled vector was commercial aquarists who had been collecting *C. taxifolia* in Monaco's waters and selling it to aquarists throughout the Mediterranean where it had already escaped into the wild. In 1991, small patches were discovered in locations about 350 km away from Monaco (Saint-Cyprien and Agay, France). The first directed government response occurred in 1992 when the Institut Français de Recherche pour l'Exploitation de la Mer began investigations aimed at eliminating the alga; however, none of the numerous physical, chemical, or biological techniques investigated were effective. Meanwhile, in 1992, new colonies of *C. taxifolia* were discovered along the Majorca coast of Spain and the coast of Italy. By 1994, it reached the Croatian coast of the Adriatic Sea. In each jurisdiction, exhaustive efforts were made to eliminate the cultivar from initial sites of infestation (usually manual removal by divers, sometimes aided by pumps) but all were unsuccessful, presumably due to the plant's ability to regenerate from tiny fragments (Meinesz 2002; Zuljevik and Antolic 2002). Legislation to prohibit the maintenance of all exotic species of *Caulerpa* in aquaria was enacted in Spain in 1996 but, even then, authorities recognized that spread as a fouling species on boats was uncontrollable. The only response left was to adapt to its existence and possibly to focus efforts on control techniques to slow down the rate of spread. Eradication attempts were limited to areas of particular concern, such as marine protected areas.

(b) New South Wales, Australia, 2000

The response to *C. taxifolia* in Australia was complicated by the fact that the species is native to some parts of the country (Millar 2002). The first population in New South Wales (NSW) was found on 18 March 2000 during a routine survey by fisheries officers in Port Hacking, 25 km south of Sydney Harbour. Following confirmation of the species identity by Royal Botanic Gardens (Sydney), additional surveys discovered numerous populations near and up to 200 km south of Sydney. The species was absent from these areas based on surveys conducted during the early 1980s; thus, the introductions are thought to have occurred between 1985 and 1989.

The primary response in Australia was legislative. *C. taxifolia* was listed as a Noxious Species by the NSW parliament on 1 October 2000; however, its noxious rating was "low". This meant that *C. taxifolia* could no longer be bought, sold or traded within NSW, but existing specimens in aquaria did not have to be destroyed. The legislation did not apply to other Australian states. The federal government already had legislation in place regarding the introduction of marine pests, and the Consultative Committee for Introduced Marine Pest Emergencies (CCIMPE) had the Mediterranean strain of *C. taxifolia* identified as a trigger species. However, population genetics studies showed the NSW populations to be more closely related to native populations from Queensland, Australia, than to the Mediterranean strain; therefore, the CCIMPE considered *C. taxifolia* in NSW to be a range extension, not warranting federal intervention unless it caused losses of biodiversity, fish stocks, or seagrass beds.

(c) California, 2000

The response of the State of California (CA) to the arrival of *C. taxifolia* benefited from the publicity surrounding the invasion in the Mediterranean Sea. In 1998 the Aquatic Nuisance Species Task Force reviewed known environmental effects of *C. taxifolia* and, in 1999, the invasive clone was added to the US Noxious Weed list, i.e., its sale and transport were banned (Kaiser 2000; Anderson 2005; Williams 2002). The concern was that if *C. taxifolia* became established in CA, "the whole rocky reef plant and animal assemblage off our coast would be dramatically transformed" (Kaiser 2000). This action was fortuitous because on 12 June 2000, a consultant diving in Aqua Hedionda Lagoon (Carlsbad, CA) collected specimens of an unusual seaweed, which she sent to a specialist who confirmed on 15 June that it was morphologically identical to the invasive form of *C. taxifolia* (WANS 2003). The consulting firm then contacted a variety of agencies concerned with non-indigenous species, water, and wildlife issues (WANS 2003). The legislative authority to begin immediate management of the *C. taxifolia* invasion in California fell to agricultural agencies due to the listing of the seaweed as a Noxious Weed, and their response was immediate (Williams 2002).

Several groups began investigating control possibilities by 22 June 2000 (WANS 2003). The lagoon was owned by the power company, which provided $123,000 to the project and hired the consulting firm to determine the extent of infestation. On 28 June, representatives from federal, state and local agencies, private entities, and stakeholders met, and agreed to cooperatively develop a response. Specimens were sent for genetics testing, which confirmed the identity of the invasive clone (Jousson et al. 2000). Meanwhile, biologists from the consulting firm began containment and initial treatments on 29 June. The plan was to cover the algal patches with PVC tarpaulins and to inject a chlorine solution or chlorine tablets underneath (Anderson 2005). Because there were no federally registered products for containment of marine algae except antifouling products for boat bottoms, a Research Authorization was obtained from the California Environmental Protection Agency for the use of chlorine-based products. An action plan was released on 12 July as the Southern California Caulerpa Action Team (SCCAT) Rapid Response Program. By then, the infested area within the lagoon had been cordoned off with enforcement by local police and game wardens. The local Regional Water Quality Control Board declared *C. taxifolia* to be a pollutant, which allowed access to the Pollutant Spill Emergency Fund, normally earmarked for "clean up and abatement". The designation as a "clean up and abatement action" also bypassed potential legal delays (Anderson 2005). Intensive public education efforts were initiated (WANS 2003) and a "Steering Committee" was formed to oversee activities. By September 2000, all known patches in the lagoon were treated.

In early August 2000, another small infestation of *C. taxifolia* was discovered in two artificial ponds in Huntington Harbor, near Los Angeles (Williams and Grosholz 2002). The local Regional Water Quality Control Board also obtained money from its emergency spill clean-up fund to treat that population. SCCAT then approached the California Legislature and obtained additional funds for continuing research on control methods, education, and detection beyond the known infested areas. Limited surveys and monitoring have so far not revealed any open coast populations, so the eradication appears to have been successful (Anderson 2005). Another outcome was that in September 2001 the State legislated a ban on possession of all forms of *C. taxifolia* and of eight other species with similar morphology (Anderson 2005).

Despite the generally positive reviews of this rapid response, there was some dissension and confusion at the time. Some academic biologists criticized the absence of a comprehensive research program, which they wanted to see implemented and managed by an independent scientific panel (Dalton 2000). Indeed, the only research on the invader was a small study to estimate biomass, size structure and density of the colonies (Williams and Grosholz 2002). Another problem was that the first specimen sent to Switzerland for molecular analysis was improperly preserved, causing a delay in confirmation of identification (Dalton 2000), and illustrating the need for "first responders" to suspected invasive species to be informed on collection and preservation protocols. Notwithstanding, treatment had not waited for taxonomic confirmation, or for biological research. As of November 2004, no plants had been seen for two years, and the projected time to declare complete eradication was under discussion, pending the outcome of quality assurance experiments using plastic

Caulerpa fronds to ensure that zero-detection surveys were not simply a case of "missed" plants in a low-density population (Anderson 2005).

Example (4): *Terebrasabella heterouncinata* Fitzhugh and Rouse, 1999, parasitic polychaete – California, 1996. *Terebrasabella heterouncinata,* a previously undescribed sabellid polychaete native to southern Africa, arrived as a parasite of South African abalone *Haliotis midae* Linnaeus, 1758 imported to California for aquaculture research in the early 1980's (Culver and Kuris 2000). It is unclear whether the abalone was imported with the knowledge of California agencies (Cohen 2002).

The effect of the non-indigenous parasite was serious as *T. heterouncinata* spread to the California red abalone *Haliotis rufescens* Swainson, 1822. By the mid-1990's, all abalone mariculture facilities in California were affected (Culver and Kuris 2000). Abalone shells could become infested with thousands of worm tubes, weakening and deforming the shell, and slowing the growth of soft tissue. The damage caused by infestations was sufficient to bankrupt some growers (Cohen 2002). In 1996, the parasite was discovered to have escaped into a rocky intertidal habitat outside of one mariculture facility. Fortunately, the limited dispersal ability of the benthic larval stage restricted distribution to a small area, the infestation was detected at an early stage, and eradication from the intertidal zone was begun within weeks of detection (Culver and Kuris 2000).

Laboratory experiments indicated that several native gastropods, especially the black turban snail *Tegula funebralis* (Adams, 1855), were highly susceptible to infection by *T. heterouncinata*. Thus, the eradication effort focused on breaking the transmission cycle in the wild by removing both the source of contamination and the native host. A screen was installed at the abalone mariculture facility to eliminate the release of additional infested material; the intertidal zone near the discharge area was cleared of all animal debris associated with the facility; and large numbers of black turban snail were removed by hand from the infested area. Monitoring over the following two years did not detect any parasites in the natural habitat. Environmental costs were apparently minimal, as the native black turban snail successfully recolonized the site (Culver and Kuris 2000).

A variety of factors contributed to the success of this program. The limited dispersal ability of the benthic larval stage meant that only a small area was infested by the parasite. Also, the dependency of the worm on a living gastropod for establishment enabled eradication. Moreover, the escaped population was detected at an early stage and removal of shells and living specimens of the host species from the intertidal site occurred within weeks of detection. Physical screening of the effluent from the aquaculture facility was in place within two months. In addition, the response was carried out with cooperation between private, public, regulatory and scientific communities, who acted in a timely manner (Culver and Kuris 2000).

Example (5): *Mytilopsis sallei* (Recluz, 1849), Black-striped mussel – Australia, 1999. The black-striped mussel is a dreissenid related to the zebra mussel *Dreissena polymorpha* (Pallas, 1771), and has a long history as an invasive pest. In the USA and India, the encrusting colonies have destroyed pearl farms, choked drainage and sewage systems, increased the cost of using and maintaining marine equipment, and reduced biodiversity (Field 1999). Black-striped mussels were initially discovered during a routine inspection of the Cullen Bay marina, Darwin, Australia. The mussels are thought to have arrived either on the hull of one of three yachts arriving the previous year from the Panama Canal, or on an Indonesian fishing vessel (Hutchings et al. 2002). Because of its history in other countries, the species was considered a serious threat to the northern Australian pearl farming industry, which had an annual value of AUS$200 million (Field 1999).

Marinas in Darwin Harbour, where the tidal range exceeds 8 m, had a double lock-gate system that allowed isolation of the infested water bodies. On April 1 1999, the Cullen Bay marina was quarantined by the Northern Territory Fisheries Division (Field 1999). Water enclosed between the lock gates was chlorinated, followed by chlorination of the entire marina and stormwater drains in the area. Divers systematically searched nearby marinas, discovering colonies of black-lipped mussels at several sites. These too were quarantined and treated. In addition, all 743 vessels that had left the marinas during the presumed period of infestation were located and inspected. Infested boats were hauled out of the water for >1 week to kill all encrusting organisms (McEnnulty et al. 2001;

Hutchings et al. 2002). Finally, sampling panels were deployed in all previously infested areas and were to be inspected regularly for at least one year (Field 1999). Offshore inspection of boats continued to find potential colonists; e.g., in September 2000 the same species was discovered on the hulls of two Indonesian fishing vessels, which were then denied entry into State waters (Hutchings et al. 2002).

McEnnulty et al. (2001) attributed the success of the eradication of black-striped mussels to: support at all levels of government and the community; early detection; rapid action; the legal capacity to enter, seize or destroy private property; the ability to isolate the infested water bodies from the local marine environment; the ability to track exposed vessels; and pre-existing information on chemical treatments for related taxa. The collateral damage associated with this eradication was high (everything in the water was killed), but was restricted to three locked marinas with environments that were already substantially modified and polluted. This localized cost was accepted because the alternative risked the much greater damage to marine infrastructure, fisheries, and the environment that could have resulted if the black-striped mussel had escaped to the open ocean coastline (McEnnulty et al. 2001).

Example (6): *Undaria pinnatifida* (Harvey, 1860), Brown alga – New Zealand, 2000. *Undaria pinnatifida* is a brown seaweed native to Japan, China, and Korea that has been introduced to New Zealand, Australia, England, France, Belgium, Italy, Argentina, and California (Wotton et al. 2004). The New Zealand (NZ) introduction was discovered in 1987, and the species has since spread to numerous coastal locations in the country via natural dispersal and translocation on vessel hulls and aquaculture equipment. In NZ, it outcompetes most native seaweeds because growth is rapid and reproduction occurs year-round.

On 17 March 2000, a 40 m steel trawler sank in 20 m of water about 2.2 km off Chatham Island, 700 km east of the mainland of New Zealand. This location had a unique community structure, several endemic seaweed species, and valuable fisheries. *U. pinnatifida* had not been present during surveys conducted in 1999. NZ Department of Conservation divers inspected the wreck six days after the sinking and found two *U. pinnatifida* sporophytes on the hull. The Ministry of Fisheries (MFish), lead agency for marine biosecurity in NZ, undertook a quailtative risk assessment that resulted in *U. pinnatifida* being listed as an "Unwanted Organism" under the Biosecurity Act 1993. The owners of the vessel were ordered either to move it to an area where *U. pinnatifida* was present, or to sink it in deep water away from the Chatham Islands. Pending the move, the owners of the vessel were ordered to inspect it by divers every 30 d and manually remove any sporophytes. Every second month, the ship, the nearest wharf, moored vessels, and the nearest rocky coastline were surveyed independently.

Salvage attempts during 2000 were unsuccessful. The situation was reviewed in February 2001. MFish determined that heat treatment was a feasible option for eradication which was undertaken during June 2001. Monthly monitoring continued until December 2003 and no sporophytes were detected on the wreck or nearby shoreline, wharf structures, or resident moored vessels. Private insurers paid the cost: NZ $2.5 million for salvage attempts, $380,000 for eradication, and $43,500 for monthly inspections. This was perhaps the first instance where public liability insurance paid the costs of managing a non-indigenous species; however, the cost of insurance for NZ shipping has risen as a direct consequence.

Eradication required a long term commitment – monitoring was continued for the maximum known period of gametophyte viability (3 yr). To minimize future marine invasion in the Chatham Islands, an inspection program was developed for vessels with visits longer than 12 hr. In addition, a community-led surveillance program was established for detection and reporting of marine pests.

Example (7): *Didemnum vexillum* Kott, 2002, tunicate – New Zealand, 2001. In October 2001, the harbourmaster at Whangamata, NZ, reported a mysterious growth dominating wharf piles, moorings and vessel hulls (Coutts and Forrest 2007). The organism was initially determined to be an unidentified species of *Didemnum*, possibly endemic to NZ. Because the tunicate was presumed native, it could not be listed as an Unwanted Organism. Instead, an information sheet was released to provide the public with guidance on restricting dispersal, and to alert stakeholders such as the aquaculture industry of a potential pest. The species has since been confirmed as *Didemnum vexillum,* a non-indigenous tunicate also found in the northeastern USA, western USA and Canada, and Europe, and most likely native to Japan (Lambert 2009; Stefaniak et al. 2009).

A routine survey by divers of a large barge anchored in Picton, Marlborough Sounds, NZ, detected large masses of *D. vexillum* on the hull and seabed below the barge on 18 December 2001. The barge had been anchored since 23 April. Of particular concern was the observation that green-lipped mussels *Perna canaliculus* (Gmelin, 1791) (the aquacultured mussel in NZ), were being smothered and killed. Even so, because the tunicate was not an Unwanted Organisms, the local district council had no grounds to refuse permission to continue the mooring of the vessel until 1 December 2002.

The New Zealand Mussel Industry Council (NZMIC) was concerned by the tunicate's smothering capabilities, rapid spread, and preference for artificial structures. In conjunction with various stakeholders, a management plan and monitoring program were developed. Activities in 2002 included surveys for existing colonies of *D. vexillum*, trials of tunicate removal devices, relocation of the infested barge following a week of exposure to freshwater, monitoring of *D. vexillum* as part of the NZMIC anti-fouling program, and development of additional educational material for public distribution. However, by 13 August 2002, the species had infested 39% of the wharf pilings in Port Shakespeare. Surveys in February 2003 revealed that newly established colonies were present on a recently refurbished salmon cage in Shakespeare Bay. The salmon cage was cleaned on 10 March but not the sea bottom under the cage. Quantitative surveys revealed that the colonies of *D. vexillum* that remained after cleaning on the barge moored in Picton had increased 16-fold. A 15 July 2003 survey found many *D. vexillum* colonies on the barge and seabed below. In fact, *D. vexillum* had spread throughout Shakespeare Bay, infecting mooring lines, buoys, and four of the 14 recreational vessels present. Moreover, 99% of the wharf pilings were infested and *D. vexillum* was established on the rocky seabed under the wharf. The barge was later deemed unseaworthy and ordered removed from the area by the Environment Court for Enforcement Orders (Resource Management Act 1991). Once cleaned of *D. vexillum* by hand-picking, the barge was sunk in >1,300 m of water on 31 August 2003. By then, the salmon cage was reinfested and had been relocated 200 m from mussel farms. PMNZL commissioned a cost-benefit analysis for Shakespeare Bay and subsequently decided to attempt eradication.

During September 2003, commercial divers wrapped all 178 wharf pilings in Port Shakespeare with black polyethylene plastic and PVC tape, which killed most fouling species within four days. At the same time, the infested mooring lines and buoys were removed from the bay, cleaned, and returned. Four vessels that had left the Bay were traced and inspected – three that were infested were cleaned by divers using putty scrapers and catch bags. Twenty more vessels were brought on land and treated with antifouling paint. Between October and November 2003, an attempt was made to smother *D. vexillum* colonies on the seabed underneath the wharf by covering them with filter fabric. Divers also identified five more infested barges in Shakespeare and nearby Ahitarakihi Bay. Three of these were beached on the mudflats at high tide in an attempt to desiccate *D. vexillum,* but the tunicate survived on the submerged side of the barges. The remaining two barges were structurally unsound so they were treated *in situ* by wrapping them with black polyethylene and treating with granulized chlorine mixed with freshwater and neutralized after 48 h.

A survey in July 2004 revealed that eradication efforts had failed: the plastic wrapping killed *D. vexillum* on the wharf pilings but 87% of the plastic wrappings themselves were infested; the tunicate had survived between gaps in the fabric strips that had been placed on the botttom; seven of the 22 moorings in Shakespeare Bay were reinfested; all five previously-treated barges and two additional barges were infested; all artificial structures (buoys, wire cables) surrounding the infected barges at Ahitarikihi Bay had become infested; and *D. vexillum* had spread throughout the salmon cages in East Bay. Within six months of removing the plastic wrapping, 80% of the wharf pilings were re-infested.. The stakeholders met in late July 2004 and decided they could only monitor the spread of *D. vexillum* and gather more information on potential effects and control options. This case study demonstrates that once a reproducing population of an invasive species is established, successful eradication is unlikely.

<u>Example (8): *Ascophyllum nodosum* (Linnaeus, 1753), Brown alga – California, 2002.</u> The North Atlantic fucoid *Ascophyllum nodosum* was encountered by scientists on 6 September 2002 during a survey of 126 intertidal locations in San

Francisco Bay (Miller et al. 2004). Specimens were identified by algal experts as *A. nodosum* ecad *mackayi*, a distinctive free-living form of the species, and voucher specimens were deposited in the collection of the University of California. While the presence of floating *A. nodosum* had been previously noted in San Francisco Bay, colonization had not been observed and there were no records of the *mackayi* ecad from the bay. The alga appeared healthy (no discolouration), had receptacles with developing oogonia, could potentially reproduce asexually, and showed no signs of deterioration 2.5 months after its initial detection. It was judged that the species would be able to withstand environmental conditions both inside San Francisco Bay and on the outer Pacific coast. The alga was apparently confined to a single site, near Redwood City, where it inhabited a patch ~300 m^2 in area.

Based on the limited patch size, the threat of dispersal by rafting, and the broad environmental tolerances, Miller et al. (2004) decided to manually remove the alga. Permission was granted by the California Department of Fish and Game and the Port of Redwood City, which owned and leased the shoreline where the *A. nodosum* was found. Manual removal was undertaken from 19 to 21 November 2002. An area of intertidal habitat was divided into 10 adjacent 1 m x 30 m transects, which were systematically searched. Approximately 174 individual plants and fragments, comprising 37.6 kg or 0.2 m^3 were removed from the site. The algae were examined, photographed, double-bagged in plastic, and disposed of in a landfill. Follow-up monitoring was conducted at 2-3 month intervals and all remaining *A. nodosum* were removed. *A. nodosum* was detected at the site until December 2002 (Miller et al. 2004). Subsequently, monitoring frequency was reduced to once or twice per year. The total cost of the eradication, including on-site monitoring through mid-2004, was US $4680, including 20 person-hours for planning and logistics, 82 person-hours for removal and monitoring, and the cost of travel and supplies (Miller et al. 2004).

After five years without *A. nodosum*, a few fragments were detected in 2007; these are thought to represent a new incursion rather than remnants of the original one (A.W. Miller, Smithsonian Environmental Research Center, pers. comm.).

<u>Lessons learned from these examples</u>

Eradication of a marine invader is seldom successful; this review of published studies of rapid response attempts in marine habitats is biased toward positive outcomes because failed eradication efforts are rarely reported in the scientific literature. Unfortunately this publication bias provides no opportunity to learn from the failures. Nevertheless, we can draw the following conclusions:

(1) Initial spatial distribution and the potential for dispersal are critical factors in determining the feasibility of eradication. Control is more likely in a closed system or one that can be quarantined; for example, *Caulerpa taxifolia* was successfully contained and eradicated in California lagoons, but not in the large open ecosystem of the Mediterranean Sea. Likewise, eradication of *Mytilopsis sallei* in Darwin, Australia, was feasible because the tide gates of the marinas could be closed. Control is also feasible when the number of vectors for secondary spread is limited, and especially when vectors are ineffective in dispersal (e.g., *Terebrasabella heterouncinata*).

(2) In all successful eradications, the identity of the species was immediately confirmed by expert taxonomists. Delays in positive identification often translated into opportunities for the species to disperse beyond the point at which eradication would have been feasible, e.g., *Urosalpinx cinerea*, *Didemnum vexillum*.

(3) Delays resulting in extensive population growth and secondary dispersal also resulted from the incorrect assumption that a species could not survive the local environmental conditions, as well as lack of consensus as to the desirability of eradicating the species, both of which occurred in the case of *Caulerpa taxifolia* invasion of the Mediterranean Sea.

(4) Communication with the public provided valuable information to researchers as well as logistical assistance with the rapid response efforts. Reports from the public contributed to knowledge of the distribution of species (e.g., *Sargassum muticum*) and also assisted in certain kinds of control activities such as diving (*Caulerpa taxifolia*) or shore-based (*Sargassum muticum*) hand-picking. The mixed messages from experts that were broadcast during the invasion of the Mediterranean by *Caulerpa taxifolia* may have contributed to its spread. If aquarists had better understood the potential for negative ecological effects they might have been less likely to release the species into the wild.

(5) Prohibition of transfer can be a successful management strategy for pests that hitch-hike on aquacultured species (e.g., *Urosalpinx cinerea* in England), or for aquarium species (e.g., *Caulerpa taxifolia* in Australia), as long as there is effective enforcement and few or no other vectors.
(6) Species with a history of causing serious problems elsewhere, such as *Caulerpa taxifolia* (during its California incursion) and *Mytilopsis sallei*, elicited stronger and more timely responses. In other words, there was less uncertainty in the risk assessment; therefore, local authorities were more willing to act immediately. In the case of *Sargassum muticum*, eradication failed despite knowing the species' history elsewhere; whether this was due to the alga's dispersal capabilities or the eradication effort being initiated too late is unclear.
(7) Advance preparation contributed to the success of the response to *Caulerpa taxifolia* in California. There were no major administrative hurdles or delays, such as those which prevented a response to *Didemnum vexillum* in New Zealand. In California, the appropriate legal instruments were in place before the arrival of the species, and the species was already on a "watch list" for immediate action. This level of awareness of *C. taxifolia* meant that its suspected presence was immediately brought to the attention of agencies that could put a response in motion, and no doubt contributed to the consensus of all relevant parties (i.e., private landowner, government) to proceed quickly.
(8) A successful eradication may require a substantial investment of time and effort. In all successful eradications, there was sufficient funding for the eradication and follow-up monitoring or repeated removals through at least one life cycle of the species. Eradication of a very small patch may require fewer resources, as evidenced by the example of *Ascophyllum nodosum*, but even this smaller-scale and relatively inexpensive project required six years of follow-up study.
(9) Adaptation or management of invasive non-indigenous species is often the only feasible response. In England, the oyster industry developed anti-predator management strategies to protect vulnerable juveniles from *Urosalpinx cinerea*, and recreational water users had to adapt to the presence of *Sargassum muticum*.

(10) Eradication is of limited value unless re-inoculation can be controlled. The successful removals of *Undaria pinnatifida* and *Mytilopsis sallei* have been threatened by the subsequent arrival of these species on boats travelling from infested or native locales.

References

Anderson LW (2005) California's reaction to *Caulerpa taxifolia*: A model for invasive species rapid response. *Biological Invasions* 7: 1003-1016, doi:10.1007/s10530-004-3123-z

Carlton JT, Geller JB (1993) Ecological roulette: the global transport of nonindigenous marine organisms. *Science* 261: 78-82, doi:10.1126/science.261.5117.78

Cohen AN (2002) The release of pest species by marine aquaculture: Lessons from a South African parasite introduced into California waters. In: Cohen AN, Webb SK (eds) Biological invasions in aquatic ecosystems: Impacts on restoration and potential for control, Proceedings of a workshop, April 25 1998, Sacramento, CA. San Francisco Estuary Institute, Oakland, CA, USA, pp 9-13

Coutts ADM, Moore KM, Hewitt CL (2003) Ships' sea-chests: an overlooked transfer mechanism for non-indigenous marine species? *Marine Pollution Bulletin* 46: 1504-1515, doi:10.1016/S0025-326X(03)00292-3

Coutts ADM, Dodgshun TJ (2007) The nature and extent of organisms in vessel sea-chests: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin* 54: 875-886, doi:10.1016/j.marpolbul.2007.03.011

Coutts ADM, Forrest BM (2007) Development and application of tools for incursion response: lessons learned from the management of the fouling pest *Didemnum vexillum*. *Journal of Experimental Marine Biology and Ecology* 342: 154-162, doi:10.1016/j.jembe.2006.10.042

Critchley AT, Farnham WF, Morrell SL (1986) An account of the attempted control of an introduced marine alga, *Sargassum muticum*, in southern England. *Biological Conservation* 35: 313-332, doi:10.1016/0006-3207(86)90092-3

Crooks JA, Soulé ME (1999) Lag times in population explosions of invasive species: causes and implications. In: Sandland OT, Schei PJ, Viken A (eds) Invasive species and biodiversity management. Population and Community Biology Series, Volume 24. Kluwer Academic Publishers, pp 103-126

Culver CS, Kuris AM (2000) The apparent eradication of a locally established introduced marine pest. *Biological Invasions* 2: 245-253, doi:10.1023/A:1010082407254

Dalton R (2000) Researchers criticize response to killer algae. *Nature* 406: 447, doi:10.1038/35020226

Field D (1999) Disaster averted? - black striped mussel outbreak in northern Australia. Fish Farming International Aug. 1999: 30-31

Hancock DA (1959) The biology and control of the American whelk tingle *Urosalpinx cinerea* (Say) on English oyster beds. *Fisheries Investigations* (Ser. 2) 22(10): 1-66

Hutchings PA, Hilliard RW, Coles SL (2002) Species introductions and potential for marine pest invasions into tropical marine communities, with special reference to the Indo-Pacific. *Pacific Science* 56: 223-233, doi:10.1353/psc.2002.0017

Jousson O, Pawlowski J, Zaninetti L, Zechman FW, Dini F, Di Guiseppe G, Woodfield R, Millar A, Meinesz A (2000) Invasive alga reaches California. *Nature* 408: 157-158, doi:10.1038/35041623

Kaiser J (2000) California algae may be feared European species. *Science* 289: 222-223, doi:10.1126/science.289.5477.222b
Lambert G (2009) Adventures of a sea squirt sleuth: unraveling the identity of *Didemnum vexillum*, a global invader. *Aquatic Invasions* 4: 5-28, doi:10.3391/ai.2009.4.1.2
Loosanoff VL, MacKenzie CL, Shearer LW (1960) Use of chemicals to control shellfish predators. *Science* 131: 1522-1523, doi:10.1126/science.131.3412.1522
McEnnulty FR, Bax NJ, Schaffelke B, Campbell ML (2001) A review of rapid response options for the control of ABWMAC listed introduced marine pest species and related taxa in Australian waters. Centre for Research on Introduced Marine Pests Tech. Rep. 23. CSIRO Marine Research, Hobart, Australia
Meinesz A (1999) Killer alga: The true tale of a biological invasion. University of Chicago Press
Meinesz A (2002) Summary of *Caulerpa taxifolia* invasion and management in the Mediterranean. In: Williams E, Grosholz E. (eds) International *Caulerpa taxifolia* conference proceedings. California Sea Grant College Program Pub. No. T-047, 6 pp
Millar A (2002) The introduction of *Caulerpa taxifolia* in New South Wales, Australia. In: Williams E, Grosholz E. (eds) International *Caulerpa taxifolia* conference proceedings. California Sea Grant College Program Pub. No. T-047, 9 pp
Miller AW, Chang AL, Cosentino-Manning N, Ruiz GM (2004) A new record and eradication of the North Atlantic alga *Ascophyllum nodosum* (Phaeophyceae) from San Francisco Bay, California, USA. *Journal of Phycology* 40: 1028-1031, doi:10.1111/j.1529-8817.2004.04081.x
Myers JH, Simberloff D, Kuris AM, Carey JR (2000) Eradication revisited: dealing with exotic species. *Trends in Ecology and Evolution* 15: 316-320, doi:10.1016/S0169-5347(00)01914-5
Northeastern Aquatic Nuisance Species Panel (NEANS) (2003) Rapid response to aquatic nuisance species in the Northeast: Developing an early detection and eradication protocol. Workshop Proceedings. Bar Harbor, ME. May 20-21 2003
Northeastern Aquatic Nuisance Species Panel (NEANS) (2006) Implementing rapid response to aquatic nuisance species in the northeast: Key components of a successful program. Proceedings of a workshop, Portsmouth NH, May 3 2005
Orton JH, Winckworth R (1928) The occurrence of the American oyster pest *Urosalpinx cinerea* (Say) on English oyster beds. *Nature* 122: 241, doi:10.1038/122241a0
Sharov AA, Liebhold AM (1998) Bioeconomics of managing the spread of exotic pest species with barrier zones. *Ecological Applications* 8: 833-845
Spencer BE (1992) Predators and methods of control in molluscan shellfish cultivation in north European waters. In: De Pauw N, Joyce J (eds) Aquaculture and the environment. European Aquaculture Society Spec. Publ. 16. Ghent, Belgium, pp 309-337
Stefaniak L, Lambert G, Gittenberger A, Zhang H, Lin S, Whitlatch RB (2009) Genetic conspecificity of the worldwide populations of *Didemnum vexillum* Kott, 2002. *Aquatic Invasions* 4: 29-44, doi:10.3391/ai.2009.4.1.3
United States Geological Survey (USGS) (2000) Early detection, reporting, identification, rapid assessment, and rapid response to invasive plants in the United States. Proceedings of a workshop on setting up a National Early Warning and Rapid Response System for Invasive Plants in the U.S. Fort Collins, CO, June 21-23, 2000. http://biology.usgs.gov/invasive/ publications/earlydetection2000
United States National Invasive Species Council (NISC) (2003) General guidelines for the establishment and evaluation of invasive species early detection and rapid response systems. Version 1. June 2003, 16 pp
Western Aquatic Nuisance Species Panel (WANS) (2003) Model rapid response plan for aquatic nuisance species. Prepared for the Western Regional Panel on Aquatic Nuisance Species by California Department of Food and Agriculture. http://www.fws.gov/answest/documents.htm
Williams SL (2002) The role of science in management of *Caulerpa taxifolia* in the United States. In: Williams E, Grosholz E. (eds) International *Caulerpa taxifolia* conference proceedings. California Sea Grant College Program Pub. No. T-047, 12 pp
Williams SL, Grosholz ED (2002) Preliminary reports for the *Caulerpa taxifolia* invasion in Southern California. *Marine Ecology Progress Series* 233: 307-310, doi:10.3354/meps233307
Wotton DM, Hewitt CL (2004) Marine biosecurity post-border management: developing incursion response systems for New Zealand. *New Zealand Journal of Marine and Freshwater Research* 38: 553-559
Wotton DM, O'Brien C, Stuart MD, Fergus DJ (2004) Eradication success down under: heat treatment of a sunken trawler to kill the invasive seaweed *Undaria pinnatifida*. *Marine Pollution Bulletin* 49: 844-849, doi:10.1016/j.marpolbul.2004.05.001
Zuljevik A, Antolic B (2002) Appearance and eradication of *Caulerpa taxifolia* in Croatia. In: Williams E, Grosholz E (eds) International *Caulerpa taxifolia* conference proceedings. California Sea Grant College Program Publication Number T-047, 10 pp