Chapter 5
Non-Native Aquaculture Species Releases: Implications for Aquatic Ecosystems

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Abstract Aquaculture is undergoing a rapid worldwide expansion. Of significant concern is the increasing use of non-native species, with subsequent escapes of these species and their associated pathogens and parasites posing a serious threat to native biodiversity, economic value and ecosystem function, particularly in regions rich in endemic species. The contribution of non-native species to the growth of the global aquaculture industry and the economic benefits that it has brought to many developing countries cannot be underestimated. However, minimizing the escapes of non-native aquaculture species must be a high priority for resource managers, conservationists and the aquaculture industry. This paper reviews intentional and unintentional non-native aquaculture introductions and the environmental consequences that escapes can have on the aquatic environment and presents a potential system of risk evaluation, management and funding mechanisms to assist in the long term sustainable development of the aquaculture industry.

Keywords Non-native species; aquaculture; introductions; aquatic ecosystems; biodiversity hotspots; risk evaluation and management

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5.1 Intentional Introduction of Non-Native Species for Aquaculture

Global aquaculture production has reached 59.4 million tonnes per year, worth US$70.3 billion accounting for almost 50% of world seafood production (FAO 2006a). It has experienced average annual growth rates of 8.8% from 1950 to 2004 (FAO 2006b) and it exceeded wild capture fisheries in Asia in 2002 (FAO 2006a) (Fig. 5.1). In many regions of the world, non-native species have been intentionally introduced for aquaculture purposes and have contributed significantly to the expansion of the industry (Welcomme 1992; Dextrase and Cocarelli 2000) (Fig. 5.2). These species provide considerable economic and social benefits, particularly in developing countries and are typically selected for production based upon: (a) the perceived poor performance of available native species relative to non-natives, including their slow growth rates, lower yield, reduced resistance to disease, tolerance to overcrowding and hardiness to environmental fluctuations; (b) proven production techniques that are readily transferred to new locations; and (c) new commercial opportunities, specifically in developing regions, utilising pre-established global markets (FAO 2006b).

The majority of intentional introductions have occurred in the last century for stocking and aquaculture purposes (Holick 1984; Welcomme 1991; Minchin and Rosenthal 2002; Goren and Galil 2005) and, with the current pace of technological development, it is highly likely that further non-native species and their hybrids will be trialled in countries outside their native range (Minchin and Rosenthal 2002). At present, four non-native species are the focus of intensive aquaculture efforts on

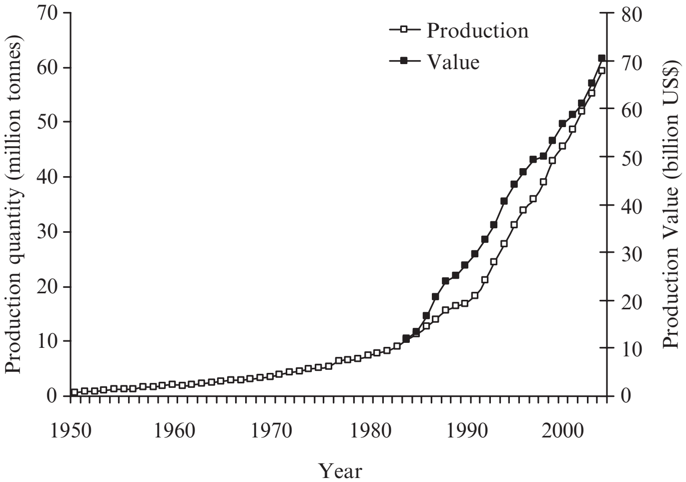

Fig. 5.1 World aquaculture production (including plants) (million tonnes) and value (billion US$) from 1950 to 2004 (FAO 2006a)

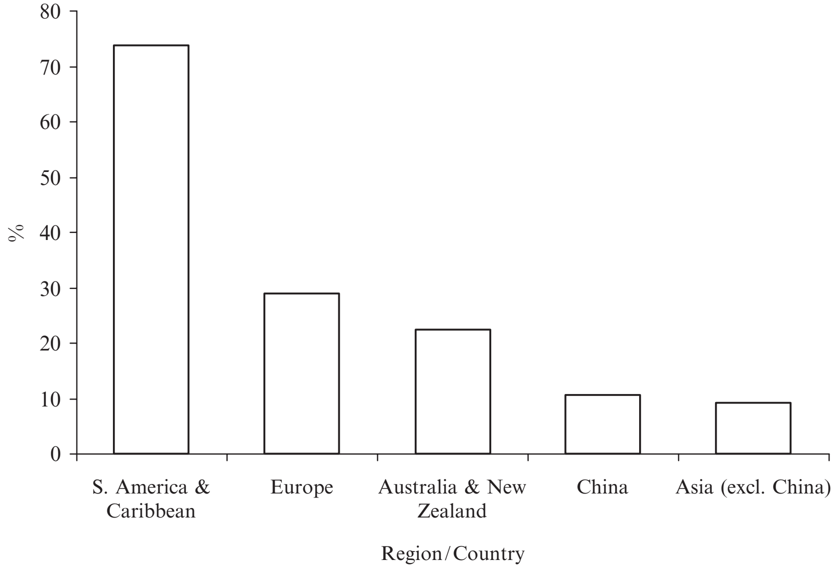

Fig. 5.2 Percentage of aquaculture production by weight (excluding plants) in 2005 based on non-native species in regions/countries with intensive aquaculture activity (FAO 2006a)

multiple continents; the Pacific white shrimp *Penaeus vannemei*, the Nile tilapia *Oreochromis niloticus*, the Atlantic salmon *Salmo salar*, and the Pacific cupped oyster *Crassostrea gigas* (FAO 2006b).

5.1.1 Non-Native Aquaculture Production in China

China has by far the greatest aquaculture industry, producing over 41.3 million tonnes in 2004 and approximately 10% of the total production in 2005 consisted of non-native species (FAO 2006a; Fig. 5.2). The importance of non-native species to the rapid increase in China's aquaculture production in the latter half of the 20th century can not be underestimated. In 1959, China introduced rainbow trout *Oncorhynchus mykiss*, an indigenous species of America, from North Korea: the first of many aquatic species introductions. Over the next half a century, more than one hundred aquatic species were introduced to China and over 20% of them have been widely cultivated (Zhu 2000). These species include 83 finfish species, such as *Oreochromis niloticus*, *Scophthalmus maximus*, *Colossoma brachypomum* and *Micropterus salmoides*; six crustacean species such as *Litopenaeus vannamei* and *Cherax quadricarinatus*; fourteen mollusc species such as *Argopecten irradians* and *Ampullaria gigas*; and nine species of turtle and tortoise (Li 2005). Currently, over 10,000 t are produced annually for each of thirteen introduced aquaculture species in China (Li 2005).

Tilapia was first introduced into China, either from Vietnam in the 1950s (Liu et al. 2000) or Africa in the 1970s (Zhu 2000), dependent on the source; this freshwater finfish is now cultivated in all 29 provinces in China with an annual production of 805,000 t (2003), which accounts for 60% of the world's total production (Li 2005). The bay scallop *Argopecten irradians* introduced from the United States in 1982 has increased in production from less than 100,000 t per year to an annual production of more than 600,000 t (Liu and Zhu 2006). The Yesso scallop *Patinopecten* (*Mizuhopecten*) *yessoensis,* an indigenous species of Japan, Korea and Pacific Russia was introduced to China in 1981 and in 15 years has become a major off-bottom mariculture species at a shell-on production of 910,000 t in 2004 (FAO 2006c). This production is three times the amount produced by Japan, the only other major producer of this species (FAO 2006c) and generates a total value of more than US$1 billion per year. The Zhangzidao Fishery Cooperation Group in Dalian is the largest producer and supplies 90% of total Yesso scallop products in China. About 800 million spat of Yesso scallop are produced annually by the hatcheries of the company, which seed the company's 400 km^2 seabed culture area.

Introductions of non-native species have also helped the industry in the face of serious problems. The outbreak of virus disease in Chinese shrimp *Penaeus (Fennerpopenaeus) orientalis* affected mariculture shrimp production in the early 1990s; however, the expanded farming of the Pacific white shrimp *Penaeus vannamei,* the Japanese shrimp *Penaeus japonicus* and the Giant tiger shrimp *Penaeus monodon* rapidly reversed the decline in shrimp production. The first commercial shipment of disease resistant *P. vannamei* broodstock from the Americas to Asia was from Hawaii to Taiwan Province of China in 1996, and from Hawaii to mainland China in 1998 (Wyban 2002). In 2004, over 735,000 t of *P. vannamei* were produced in China, more than the rest of the world combined (Chen 2006; FAO 2006a).

5.1.2 Non-Native Aquaculture Production in Europe

Several non-native species have been in various forms of culture for over 2,000 years in Europe. Perhaps the earliest species cultured was the Common carp *Cyprinus carpio* in ponds in Eastern Europe, which originated from the Manchurian region of China. However, it has only been since Victorian times that aquaculture in Europe evolved and this was mainly out of concern over the depletion of existing fisheries (Wilkins 1989). Early experiments on rearing native oysters *Ostrea edulis* to produce settlements in ponds during the first few decades of the 1900s were occasionally successful. However, it was stock movements of the native oyster from continental Europe that were used to increase production. These were supplemented with imports of the American oyster *Crassostrea virginica* to both Britain and Ireland. This became a regular trade over about 40 years from the ~1880s. Such long distance movements became possible with reduced journey times owing to the development of steam transport (Minchin 2006). Intercontinental trade soon led to the movement of other species including fertilised salmonid eggs, easily transported and managed in hatchery flow trays from the 1880s. Several species later became exchanged or spread to different world regions which led to introductions of the Rainbow trout *Oncorhynchus mykiss,* subsequently cultured in freshwater as well as in sea cages, and Brook trout *Salvelinus fontinalis* and Lake trout *S. namaycush* for stocking mountain lakes in Europe.

With increases in international trade, improved biological knowledge, production of food for young stages, and increased technological developments, cultivation became practical. The hatchery techniques for bivalves developed by Loosanoff and Davis (1963) in North America soon were utilised in Britain and France from the 1960s and 1970s making it possible to raise several species. Not all of the species imported and used in experimental trials were considered useful (Utting and Spencer 1992). It was the Pacific oyster *Crassostrea gigas* and the Manila clam *Venerupis philippinarum* that became widely used in culture throughout much of northern Europe. Total production of *C. gigas* reached 122,000 t in 2004 and 29% of all aquaculture production consisted of non-native species in Europe by 2005 (FAO 2006a; Fig. 5.2). Other species that were intentionally introduced but are in cultivation at comparatively small levels of production, are, for example, the Japanese abalone *Haliotus discus hannai* in Ireland and the Japanese shrimp *Penaeus japonicus* in Spain.

Some species arrived in Europe accidentally and have subsequently been utilised. One of these, the red alga *Asparagopsis armata,* arrived in ~1940 and is now cultivated for the production of cosmetic products (Kraan and Barrington 2005).

5.1.3 Non-Native Species Production in Latin America and the Caribbean

Countries in Latin America and the Caribbean have exhibited the greatest expansion in their aquaculture industries compared to other regions, experiencing a 21.3% annual growth rate since the 1950s, when aquaculture production was minimal (<7,000 t) (FAO 2006b). Substantial growth in aquaculture production began in the late 1970s, primarily supported by shrimp and salmon production in three countries in South America: Ecuador, Brazil and Chile. The development of the world shrimp market in the 1970s and 80s saw considerable investment in these countries, particularly in Ecuador, which concentrated on the native shrimp species *Penaeus vannamei*.

Brazil also concentrated its production efforts on shrimp and imported the non-natives *P. monodon* and *P. japonicus* in the 1970s (FAO 2006a). The culture of the non-native *P. vannemei* began to increase substantially in the early 1990s in Brazil and this species is now the dominant shrimp species grown in the country with the production of 76,000 t in 2004. The non-native Common carp, *Cyprinus carpio* and the various tilapia species, including the blue *Oreochromis aureus*, Mozambique *O. mossambicus*, Nile *O. niloticus* and Wami *O. urolepis* imported to Brazil in the 1960s and 70s also comprise a large proportion of Brazil's aquaculture production with 114,248 t produced in 2004 (FAO 2006a).

In the late 1980s, Chile began to develop their salmon industry based on the Atlantic salmon *Salmo salar*, which is native to the north-east Atlantic and had been introduced to Chile in 1935 (FAO 2006a). Since 1990, this industry has exhibited one of the highest average annual growth rates (31.4%) compared to other countries' aquaculture activities. The production of non-native salmonids had reached over 550,000 t by 2004 (Buschmann et al. 2006; FAO 2006a) and the Chilean government plans to double the production output of this species by 2013 (Ridler et al. 2006).

In the Caribbean, the four main aquaculture producers are Belize, Costa Rica, Cuba and Honduras. In Belize and Honduras, the non-native shrimp, *P. vannemei* is the dominant aquaculture species, comprising 97% and 80% respectively of the total aquaculture production in 2004 (FAO 2006a). In Costa Rica, 18,000 t of the non-native Nile tilapia were produced in 2004, comprising 73% of the total aquaculture production for the country. In Cuba, the main aquaculture species is the non-native Silver carp *Hypophthalmichthys molitrix,* which accounts for 54% of the aquaculture production (FAO 2006a).

In 2005, over 74% of the annual production in Latin America and the Caribbean was attributed to non-native species (FAO 2006a; Fig. 5.2), with an economic value of US$3.9 billion in 2004, representing 75% of the total value of aquaculture production in the region. This production is now concentrated on non-native Pacific Whiteleg shrimp *P. vannemei* (in non-Pacific countries), Atlantic *S. salar* and Coho salmon *Oncorhynchus kisutch*, Rainbow trout *O. mykiss*, Nile tilapia *Oreochromis niloticus* and various carp species (Fig. 5.3).

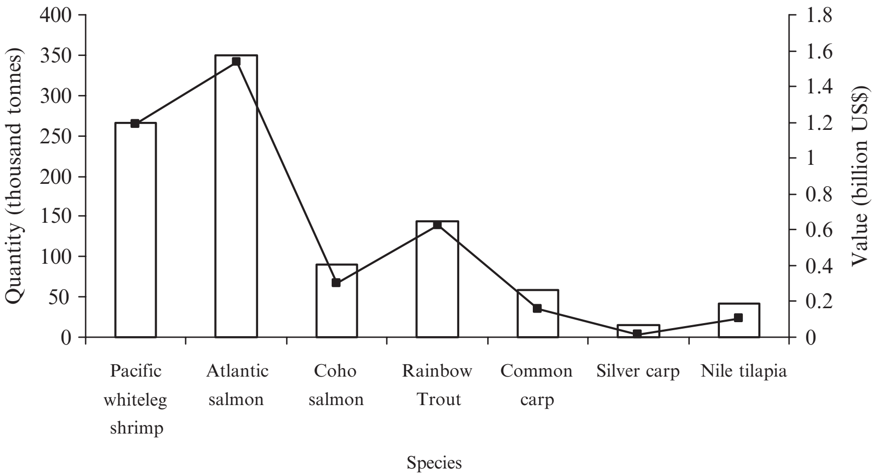

Fig. 5.3 Aquaculture production of non-native species (thousand tonnes) and value (billion US$) in Latin America and the Caribbean in 2004 (FAO 2006a)

5.1.4 New Zealand and Australia

Both New Zealand and Australia have significant and growing aquaculture industries that rely on their "clean and green" image – many of the common Northern Hemisphere diseases and parasites are absent from the aquaculture facilities in these two countries, yet almost a quarter (22%) of their aquaculture production by weight was based on non-native species in 2005 (FAO 2006a; Fig. 5.2).

5.1.4.1 New Zealand

New Zealand produces ~97,700 t of aquaculture product per year worth ~US$217 million (~NZ$315 million), equating to approximately 20% of total NZ fisheries production (NZAC 2006). Over 98% of New Zealand's aquaculture industry is based on three species: the endemic Greenshell mussel *Perna canaliculus*, and two non-native species, the Pacific oyster *Crassostrea gigas*, and the King (or Quinnat) salmon *Oncorhynchus tshawytscha*. Non-native species however, represent 33.3% of New Zealand aquaculture product by value (Table 5.1).

There are three species of salmon in New Zealand, all of which are non-native: King or Chinook salmon *O. tschawytscha* introduced from the United States in 1907, Sockeye salmon *O. nerka* introduced from Canada in 1902, and Atlantic salmon *Salmo salar* introduced in the 1960s (FAO 2006a). Only the King or Chinook salmon (also known as "Quinnat") are successfully farmed on a significant scale in New Zealand. This is in contrast to the rest of the world where salmon aquaculture is focused on the Atlantic salmon, except for some Chinook salmon in Canada and Coho salmon in Chile.

King salmon are grown in sea cages in the marine environment and in freshwater raceways throughout the South Island. There are about 29 salmon farms in New Zealand covering a total of around 128 hectares (as of December 2005) producing around 7,000 metric tonnes per annum. These 29 farms account for roughly half of the worldwide farmed king salmon production.

The main Pacific oyster farming areas are located in sheltered bays and harbours around the North Island. The farming method for Pacific oysters consists of wooden racks to which the oysters are attached. The racks are anchored in the lower intertidal region. There are about 236 Pacific oyster farms in New Zealand covering a total of ~928 hectares (as of December 2005) and producing over 2,000 t in 2004 (FAO 2006a).

5.1.4.2 Australia

Australia produces ~47.1 million tonnes of aquaculture product per year, worth ~US$480 million (~AU$610 million), equivalent to 30% of Australia's total fisheries

Table 5.1 Aquaculture production in 2004–05 for New Zealand (after NZAC 2006) and Australia (by state after (ABARE 2006). Bold names and figures represent non-native species information; figures in parentheses represent non-native species subtotals)

	New Zealand	Australia	NSW	Vic	Qld	WA	SA	Tas	NT
Value	$'000	$'000	$'000	$'000	$'000	$'000	$'000	$'000	$'000
FISH									
Salmon [a]	**81,000**	**112,357**	**0**	**0**	**0**	**0**	**0**	**112,357**	**NR**
Trout [b]	–	**12,862**	**1,784**	**10**	**533**	**0**	**545**	**0**	**NR**
Tuna	–	139,955	0	0	0	0	139,955	0	NR
Silver perch	–	2,821	2,431	0	390	0	0	0	NR
Barramundi	–	14,625	1,360	0	11,000	0	2,265	0	NR
Other	29,000	9,434	1,201	6,065	280	1,888	0	0	NR
CRUSTACEANS									
Prawns	–	49,864	4,464	0	45,400	0	0	0	NR
Yabbies [c]	–	1,866	362	78	0	**1,120**	306	0	NR
		(1, 120)							
Marron [d]	–	2,072	0	0	0	**1,485**	587	0	NR
		(1,485)							
Redclaw	–	1,402	2	0	1,400	0	0	0	NR
MOLLUSCS									
Edible oysters[e]	**26,000**	73,287	35,788[f]	0	700	0	**19,995**	**16,804**	NR
		(36,799)							
Pearl oysters	–	122,550	0	0	550	122,000	0	0	NR
Mussels	181,000	7,571	215	2,816	0	1,515	657	2,369	NR
Other	4,000	9,772	0	4,454	0	0	5,318	0	NR
OTHER (non food)		50,666	0	0	2,000	467	17,015	3,384	27,800
Total value	**321,000**	**611,103**	**47,605**	**23,946**	**61,720**	**128,475**	**186,643**	**134,914**	**27,800**
	(107,000)	**(264,623)**	**(1,784)**	**(10)**	**(533)**	**(2,605)**	**(20,540)**	**(129,161)**	**(?)**

[a] *Salmo salar, S. trutta.*
[b] New Zealand: *Oncorhynchus tshawytschca*; Australia: *Oncorhynchus mykiss.*
[c] *Cherax albidus* (introduced to WA from Victoria; (Morrissey and Cassells 1992), *C. crassimanus, C. destructor, C. glaber, C. plebijanus, C. quinquecarinatus.*
[d] *Cherax quadricarinatus* (introduced from Qld and NSW to WA), *C. rotundus.*
[e] New Zealand: *Crassostrea gigas;* Australia: *Crassostrea gigas,* and the natives: *Ostrea angasii, Sacostrea glomerata, S. cucullata.*
[f] some component of the New South Wales production is *Crassostrea gigas*

production. Approximately 60 species are under aquaculture production, of which several are introduced from other regions of the world or from other regions of Australia (see Table 5.1). While many of these species are for human consumption (e.g., salmonids, oysters and prawns), aquaculture in Australia includes a variety of products for other purposes.

Non-native species represent 43.3% of Australian aquaculture production by value (in 2004–05). These non-native species include introduced salmonids (Atlantic salmon, *Salmo salar*; Brown trout, *Salmo trutta*; Brook trout, *Salvenlinus fontinalis* and Rainbow trout, *O. mykiss*), Pacific oysters, and freshwater crayfish (yabbies: *Cherax albidus*; *C. quadricarinatus*) translocated from one Australian state to another.

Atlantic salmon and rainbow, brown and brook trout are cultured commercially in Australia. Tasmania is the major force in Australian production. Atlantic salmon and ocean trout (rainbow trout) are grown in sea cages, trout are also grown in freshwater dams and raceways where large supplies of cold, flowing water are readily available. Sea cage culture contributes more than 60% of the total salmon and trout production in Australia.

The Pacific oyster is extensively farmed in Tasmania and South Australia, and comprises a minor component of the industry in New South Wales where several native species are grown (*Ostrea angasii*, *Sacostrea cucullata*, and *S. glomerata*). The farming method is similar to New Zealand where both wooden racks and stakes are used. The blue mussel, *Mytilus edulis*, is commercially farmed in Victoria, Tasmania, New South Wales, and Western Australia.

5.2 Unintentional Aquaculture Related Introductions of Non-Native Species

Despite the apparent success in increased aquaculture production through the use of non-native species, current practices can pose significant risks of unintentional introductions from net pens or pond systems into freshwater and marine systems. These introductions have been widely reported (Naylor et al. 2001; Nico et al. 2001; SAMS 2002) and are often associated with weather events (e.g., flooding or hurricanes) or accidents of operation. It is estimated that; up to 2 million farmed Atlantic salmon escape into the North Atlantic each year (McGinnity et al. 2003), over 500,000 Atlantic salmon escaped from cages between 1987 and 1997 on the west coast of North America (McKinnell and Thomson 1997), up to 80% of adult salmon entering rivers in Norway were escapees (Fiske and Lund 1999) and that the introduced Rainbow trout *Oncorhynchus mykiss* now occupies over 51% of Slovenian territory (Povz and Sumer 2005). Mass escapes of the Pacific white shrimp *Penaeus vannemei* have also occurred in both the United States (Balboa et al. 1991; Wenner and Knott 1992; Howells 2001) and Thailand (Barnette et al. 2006).

5.3 Ecological Consequences of Intentional and Unintentional Introduction of Non-Native Species

The intentional and unintentional introduction of non-native cultured species represents a "biological introduction", which are human mediated movements of organisms to regions where they did not evolve. Biological introductions are widely recognised as a major threat to species diversity (CBD 1992; Worm et al. 2006) arising from habitat modification, changes in ecosystem functioning, extinction of native fauna and flora, disease transfer and genetic effects such as hybridisation with native congeners (Lovei 1997; Ruiz et al. 1997; D'Antonio et al. 2001; Jonsson and Jonsson 2006). Regions supporting high levels of endemic species are particularly vulnerable to these introductions.

5.3.1 Habitat Modification

The accidental or intended introductions of exotic species can cause significant changes to ecosystems (Ruesink et al. 2006). However, the response of natural communities to the introduction of a non-native species is complex, and impacts can have positive, negative or negligible, depending on the species, location, age, or type of habitat considered (Neira et al. 2005; Gribben and Wright 2006). To highlight the potential effects of introduced species on habitat structure, two case-studies will be considered: the first in Willapa Bay, Washington USA (Ruesink et al. 2006) and the second on the South African coast (Robinson et al. 2005).

In Willapa Bay, USA the Pacific oyster *Crassostrea gigas* was introduced in 1928 for aquaculture purposes due to the overexploitation of the native oyster *Ostreola conchaphila* (Ruesink et al. 2006) and is now the main oyster species cultivated. The Pacific oyster naturally recruits to uncultivated regions of the bay and forms dense intertidal hummocks of shell and live oysters (Ruesink et al. 2006). This recruitment has also been observed in a number of other countries where *C. gigas* has been introduced (Orensanz et al. 2002; Nehring 2003; Diederich et al. 2005). The importation of *C. gigas* and the development of the oyster industry also unintentionally brought the invasive smooth cordgrass *Spartina alterniflora* to Willapa Bay in the form of packaging material for transplanted *Crassostrea virginica* in 1890 (Townsend 1893, 1896; Feist and Simenstad 2000).

Species that can change habitat structure and modify the local environment are known as ecosystem engineers (Crooks 2002). *C. gigas* and *S. alterniflora* can substantially re-engineer a habitat to provide biogenic structures which provide substrate for fish, invertebrate and macroalgal recruitment and sediment accumulation (Ruesink et al. 2006). The expansion of culture sites and biogenic reefs formed by oysters can also cause significant changes in sediment porosity, bioturbation activity and have an effect on biogeochemical cycling (Ruesink et al. 2006).

In South Africa, the Mediterranean mussel *Mytilus galloprovincialis* was accidentally introduced in ~1979 and is now grown commercially (Robinson et al. 2005) with over 6,100 t produced in 2005 (FAO 2006a). *M. galloprovincialis* has become the dominant intertidal mussel along the west coast, where it has considerably modified the natural community composition by dominating rock surfaces (Robinson et al. 2005). *M. galloprovincialis* forms dense, multi-layered structures and supports a higher biomass per m^2 than the single layered beds of the indigenous mussels *Choromytilus meridionalis* and *Aulacomya ater* (Robinson et al. 2005). The increased vertical range of *M. galloprovincialis*, due to a greater dessication tolerance, higher fecundity and faster growth rates than the native species (Van Erkom Schurink and Griffiths 1990, 1991; Hockey and van Erkom Schurink 1992; Van Erkom Schurink and Griffiths 1992), has led to a massive increase in non-native mussel biomass along the South African west coast (Griffiths et al. 1992).

The introduction of certain non-native species to a region can considerably modify the system, as shown by the introduction of *C. gigas*, *S. alterniflora* and *M. galloprovincialis*. Predicting the impact that non-native species will have on habitat structure and, as a consequence, existing food webs and community composition is inherently difficult. A greater understanding of how these non-native "ecosystem engineers" alter energy flow, ecological processes, biogeochemical cycles and ecosystem function is critical in determining the impact that these species will have on the ecosystem as a whole.

5.3.2 Changes in Ecosystem Functioning

Ecosystem services are a set of ecosystem functions that are useful to humans and many are critical to our survival (climate regulation, air purification, pollination, nutrient recycling) while others enhance it (aesthetics) (Kremen 2005). Ecosystem functioning is intrinsically linked to biodiversity and changes in biodiversity and community structure can cause drastic changes in ecosystem function and hence in the provision of ecosystem services.

The majority of studies of ecosystem function have concentrated on biodiversity loss due to extinctions; however, many biological invasions have resulted in a net gain at the local or regional level (Sax and Gaines 2003). This causes a net increase in diversity at the ecosystem level and an important consideration is how these species additions affect ecosystem functioning (Stachowicz and Tilman 2005). Few studies have been undertaken to specifically address this question, although it is clear that invasive species can affect ecosystem structure and function (Stachowicz and Tilman 2005). For example, Levin et al. (2006) showed that invasion by a *Spartina* hybrid in San Francisco Bay (USA) shifted the system from an algae based to a primarily detrital-based system. Furthermore, the *Spartina* hybrid canopy changed the hydrodynamic regime causing drastic and multiple changes in the physical, chemical and biological properties in the benthic system (Neira et al. 2006). These changes caused a reduction in survivorship of key taxa that supported higher trophic levels, such as migratory shorebirds (Neira et al. 2006).

Tilapia has been used worldwide as an aquaculture species and has escaped in many regions where they are cultured (Peterson et al. 2005). Tilapia can significantly alter the ecosystem they invade, yet the impact is often hard to predict (Figueredo and Giani 2005). For example, the Redbelly tilapia (*Tilapia zilli*) was accidentally introduced into a power plant reservoir in North Carolina, where it reduced all aquatic macrophytes through grazing, which coincided with a dramatic decline in native fishes (Crutchfield 1995). The Common carp (*Cyprinus carpio*) has been widely translocated around the world for aquaculture purposes and through unintentional introductions is now a successful invader in parts of Europe, Asia, Africa, North, Central and South America, Australia and Oceania (Lever 1996; FAO 2002). Carp can reach high densities (1000 individuals ha^{-1}) and biomass (3144 kg ha^{-1}) (Harris and Gehrke 1997) and this can result in reduced photosynthetic production and visibility for visually feeding fish (Koehn 2004) through increasing the water turbidity whilst feeding (Fletcher et al. 1985; King et al. 1997), a decline in the abundance of aquatic plants (Fletcher et al. 1985; Roberts et al. 1995) and finally, cause trophic cascades in shallow lakes (Khan et al. 2003).

5.3.3 Extinction of Native Flora and Fauna

There is no doubt that biological invasions are causing dramatic widespread changes to communities and altering many ecological systems (Parker et al. 1999; Ruiz et al. 1999; Levi and Francour 2004; Neira et al. 2005; Gribben and Wright 2006; Ruesink et al. 2006). However, many extinctions have been attributed to biological invasions when there have been many other environmental factors (eutrophication, habitat loss, land use changes, over grazing) which could have played a key role in causing the decline of the native species (Gurevitch and Padilla 2004). Of the 762 species globally documented to have become extinct as a result of human activities in the past few hundred years, < 2% list non-native species as a cause (Gurevitch and Padilla 2004).

In many cases, species do not go "extinct", but are lost from a large part of their former range which greatly reduces and/or fragments the populations (Hobbs and Mooney 1997). Non-native species have been identified as part of the problem and, in combination with habitat loss, modification and degradation of the environment, have lead to the loss of species in a particular region. For example, Fellers and Drost (1993) resurveyed 16 historic sites and 34 other sites for the Cascade frog *Rana cascadae* and only found two frogs at one site. The population extinction was attributed to several factors, principally to the introduction of non-native predatory fish, drought and habitat loss due to management activities (Hobbs and Mooney 1997). The introduction of the Grass carp *Catenopharyngodon idella*, the Bighead carp *Aristichthys nobilis* and the Taihu Lake noodlefish *Neosalanx taihuensis* during the 1970s and 1980s to the southern provinces of Guangdong, Guangxi and Yunan from the Yangtze River system, has severely affected and has contributed to the extinction of some local finfish species (Li and Xie 2002). The Nile perch *Lates niloticus* was introduced into Lake Victoria in the 1960s apparently causing the extinction of many cichlids species – viewed as the biggest vertebrate extinction of the 20th century (Witte et al. 2000). However, Gurevitch and Padilla (2004) suggest that development of the railroad in the 1920s caused erosion and shoreline destruction (Verschuren et al. 2002) and urbanization during the 1970s increased eutrophication and decreased lake transparency from 8 to 1.5 m (Verschuren et al. 2002; Aloo 2003). Increased nutrient loading and anoxic events resulting in fish kills are now common. The increase in nutrient loads, however, has favoured the non-native water hyacinth *Eichhornia crassipes*, which alters nursery areas for juvenile fish (Witte et al. 2000).

Species in the marine environment are typically considered to have a lower risk of extinction because of the large continuous habitats they occupy and the life history characteristics of many species that results in extensive dispersal potential enabling the recolonisation and repopulation of impoverished areas (Gurevitch and Padilla 2004). Caution should be taken, however, as this perception was derived from experiences when marine populations were much larger than they are today (Dulvy et al. 2003) and when the current rate of exploitation of marine species and the level of associated by-catch of non-target species was significantly lower (Worm et al. 2006). Unintentional introductions of non-native aquaculture species are likely to increase with the rapid expansion of the aquaculture industry on a global scale and there is an urgent need for more research into the role of non-native species in pushing native species towards extinction and to evaluate their impact relative to that of other factors (Gurevitch and Padilla 2004).

5.3.4 Disease Transfer

To be economically viable, cultivation of a species must normally take place at a high density either within contained units (on account of the capital costs of the equipment) or as bottom culture on shores (where space may be limited). Under these conditions introduced pests, parasites and diseases are provided with increased opportunities to thrive (Minchin and Rosenthal 2002).

There are many cases of stock movements introducing unwanted pests, parasites and diseases and some have had serious economic impacts on aquaculture production, for example, oysters (Heral 1990), shrimp (Kinne 1984; Sindermann 1993) and fishes (Kinne 1984). For example, the trematode *Gyrodactylus salaris* was carried with Atlantic salmon *Salmo salar* from Swedish hatcheries to Norway (Johnsen and Jensen 1991) and resulted in serious salmon mortalities in the recipient region.

Movements of harmful biota over larger distances, however, are more common. For example, consignments of half-grown Pacific oysters have resulted in a large suite of invertebrates being spread throughout the world (Gruet et al. 1976), with their uneven shells providing a large surface area for the attachment of cryptic species. Biota may also reside in the mantle cavity, the gut or in various tissues. During the early large-scale movements of oysters these associated species were tolerated as a nuisance. However, with present knowledge and management such releases are unlikely to be repeated on account of the wide range of microbiota and syndromes that have been associated with such movements (Cheyney et al. 2000).

The movement of stock in seemingly small quantities can also have serious consequences for native species. For example, the importation of Japanese eels *Anguilla japonica* for cultivation trials in Europe released a rotund nematode that in its final stage lodges in the visceral cavity near the air bladder and has caused significant internal damage in other eel species such as the native freshwater eel *Anguilla anguilla* (Kennedy and Fitch 1990). This nematode is easily dispersed by copepods, and a wide range of paratenic hosts that include other fishes and insects. The species has now become widely spread in Europe and the consequences for the stock of the North Atlantic eel, already in decline, are unknown.

The spread of viral diseases through stock movements has been particularly prevalent in Penaeid shrimp and has caused significant declines in production (Subasinghe et al. 2000). Viruses may also be spread *via* other crustaceans, and barnacles may even be capable of transmitting these to different countries as hull fouling on ships. Pathogenic species may also be carried in the water and sediments in the ballast tanks of ships and many species in commercial culture have been found associated with hull fouling (Minchin and Gollasch 2002). No studies have been undertaken on the potentially harmful biota carried on ships' hulls although it is suspected that the oyster disease *Bonamia osteae* was carried to different bays on the hull of a barge (Howard 1994).

5.3.5 Genetic Impacts

Marine aquaculture species are increasingly being selected or modified with respect to genetic traits linked to performance. Cross (2000) described the genetic improvement of aquaculture species as an economic imperative and without it, the industry would find it impossible to compete. For example, Coho salmon *Oncorhynchus kisutch* with introduced growth hormone genes from Chinook salmon *Oncorhynchus tshawytscha*, demonstrated much faster growth compared to the control group (Devlin et al. 1994). Hybridization between the Yesso scallop *Patinopecten* (*Mizuhopecten*) *yessoensis* and a local species *Chlamys farreri* have also been undertaken to improve growth performance (Yang et al. 2004; Yu et al. 2006). In addition, Chinese researchers have recently introduced a new batch of Yesso scallop broodstock from Russia (Meng 2006) in an effort to reconstruct their genetic diversity (Li and Xue 2005). These experiments have produced new strains of scallops and some individuals have already been put out to sea for a pilot grow-out.

As a result, a substantial fraction of genetic variation in aquaculture species resides at a higher organisational level (among populations) than in natural populations where all variation resides below the family level (Youngson et al. 2001). Within the population, genetic complexes will develop, often relating to the environment in which the population has developed, constituting spatial, behavioural or temporal isolating mechanisms. Aquaculture practices of both inbreeding and selection of individuals for specific traits magnifies the development of genetic complexes in a population.

When aquaculture escapes breed with natural populations, hybridisation and subsequent introgression can lead to a breakdown of the genetic complexes which have developed, forcing a reduced fitness in the hybrid individuals (Skaala et al. 2006). This can lead to a decline in fitness and increased threat of extinction in the now hybridised natural population (Mooney and Cleland 2001). Outbred largemouth bass *Micropterus salmoides* crossed from two distinct populations suffered a reduction in fitness of approximately 14% relative to parental stocks (Goldberg et al. 2005). F2 generation hybrids suffered higher mortality rates and increased susceptibility to infectious disease. Collection and translocation between previously isolated stocks can have similar effects, which have been shown in stocks of black-lipped pearl oyster, *Pinctada margaritifera cumingii*, in French Polynesia (Arnaud-Haond et al. 2004). When large populations of invading species are introduced, the threat to native species is unavoidable, however evidence suggests that even when small populations of an invader are introduced (for example, escaping aquaculture individuals) the native population is still threatened (Mooney and Cleland 2001).

Hybridisation can be either inter- or intraspecific. Hybridisation between native brown trout, *Salmo trutta* and Atlantic salmon, *Salmo salar*, in Europe is an example of the former and female salmon escapees have been shown to hybridize relatively freely with the brown trout (Youngson et al. 1993). Intraspecific hybridisation would involve escape of a different strain of the species into a native population (Cross 2000). This is likely, due to the modification of aquaculture species with traits chosen for performance. Spawning success is lower for cultured salmon than for wild fish (Fleming et al. 1996, 2000), even when released to the wild as smolts (Jonsson et al. 1990). In Spain, where rivers have been highly stocked with non-native trout *S. trutta*, 25% of native populations had evidence of introgression by genes of hatchery origin (Almodóvar et al. 2001). Evidence has also been found for the introgression of the Mediterranean mussel, *Mytilus galloprovincialis* genes into native Australian populations (Sanjuan et al. 1997).

5.3.6 Trans-Boundary Effects

Biological invasions, whether intentional or accidental, are by their very nature not limited by geo-political boundaries. This is even more the case for marine bioinvasions where oceanic currents and natural dispersal mechanisms can lead to significant range expansions, following initial establishment, that transcend state and national boundaries. Examples include the escape and spread of the macroalga *Undaria pinnatifida* from aquaculture facilities in Brittany, Atlantic France (Pérez et al. 1984) across the English Channel to southern England and along the coasts northwards to the Netherlands and southwards to Spain (Fletcher and Manfredi 1995; Wallentinus 1999). Similarly, the expansion of the European green crab, *Carcinus maenas*, along the West Coast of North America following its initial accidental establishment in San Francisco Bay resulted in an expansion from the state of California, to Oregon and Washington (Grosholz and Ruiz 1995).

The Conference of Parties of the Convention on Biological Diversity (CBD 1992) in Decision VII/5, identified the need for regional and international collaboration to address trans-boundary impacts of mariculture on biodiversity, such as spread of disease and invasive alien species (paragraph 51), particularly where non-native species are grown for mariculture purposes. Similarly, the FAO through the Code of Conduct for Responsible Fishing (CCRF) (FAO 1995) and Technical Guideline Number 5 (FAO 1997) has explicitly addressed aquaculture development in relation to trans-boundary obligations. Article 9.1.2 of the CCRF identifies the potential genetic impacts of introduced (alien) species through introgression and competition with native stocks and Article 9.2.3 explicitly discusses the need for consultation with neighbouring states when considering the introduction of alien species into a trans-boundary system.

From an aquaculture perspective, trans-boundary effects include both the intentional release of a species that has the ability to disperse across geo-political boundaries and cause harm to a neighbouring coastal state, as well as, the operational or regulatory management failure to prevent or mitigate non-native species escapes that may cause harm to a neighbouring coastal state.

5.3.7 Implications for Biodiversity Hotspots

Biodiversity hotspots are defined as those areas where "exceptional concentrations of endemic species are experiencing exceptional loss of habitat" (Myers et al. 2000; Orme et al. 2005). Of the top five regions identified as major global hotspots for marine biodiversity (Roberts et al. 2002), two regions are major aquaculture producers; the Philippines and Indonesia with an annual production of over 1.4 million tonnes. The Caribbean is ranked ninth (Roberts et al. 2002) and this region has experienced an annual growth rate in aquaculture production of 21.3%, almost three times higher than the global production average of 8.8% since the 1950s. Over 65% of the production in the Caribbean is due to introduced species (FAO 2006b). Chile is identified in the top 25 terrestrial hotspots and has experienced an annual increase in aquaculture production of 40.0% from 1980 to 2004 (FAO 2006a). From a conservation perspective, it could be argued that concerns about aquaculture effects related to non-native species need to be primarily focused on these areas.

5.4 Future Directions

As the landings from capture fisheries stagnate (SOFIA 2004; Hilborn 2007), aquaculture is critical to the provision of global resources. The industry provides full time employment for over 3.3 million people in China alone (De Silva 2000) and many millions more could be employed either directly or indirectly in aquaculture worldwide – provided there is wise environmental management. A sustainable approach to coastal aquaculture is especially key given that 65% of humanity, 3.6 billion people, live within 150 km of the coast and are dependent on ecosystem based services (Cohen 1995; Sachs and Reid 2006) and that a number of major aquaculture regions support biodiversity hotspots. Much of the future aquatic production will be dependant on good water quality and how developments evolve that might otherwise conflict with the space required for cultivation. The present ease of transportation will allow for the movement of aquaculture species over large distances rapidly enabling a wide range of species to become transferred. Legislation and risk management in the movement of species is becoming recognised as an important area in order to prevent undesired impacts, as a result of an intended introduction.

5.4.1 Legislation for the Introduction of Non-Native Species for Aquaculture Purposes and Management Strategies

The International Council for the Exploration of the Sea (ICES) Code of Practice on Introductions and Transfers of Marine Organisms (ICES 2005a) and the European Inland Fisheries Advisory Commission (EIFAC) Code of Practice for Consideration of Introductions and Transfers of Marine and Freshwater Organisms (Turner 1988), provided guidelines for the intentional introduction of non-native species for aquaculture purposes. Furthermore, it has been recommended that the new IUCN code of practice should be incorporated into national development strategies (Hewitt et al. 2006). These codes aim to minimise negative impacts of non-natives used in aquaculture on the recipient environments. Australia and New Zealand are well advanced in the development of their national strategies; however, it is recognised that these procedures take time to implement and there are circumstances where there is an urgent requirement to provide food for the vast population, as in China or where there has been serious environmental degradation, as in the case of deforestation in the Indo-Pacific (Coates 1995).

5.4.1.1 Australia and New Zealand

Australia has experienced a number of high profile invasions from a variety of sectors resulting in serious environmental and economic impacts (Hewitt et al. in press). As a consequence of these invasions, Australia identified the need for a coordinated approach across national and state agencies through the development of a National System for the Prevention and Management of Marine Pest Incursions (National System) to address all potential marine pest vectors underpinned by a risk assessment framework and to specifically establish arrangements for prevention, emergency preparedness and response, and ongoing management and control (Hewitt et al. in press).

The National System is coordinated by the Department of Agriculture, Fisheries and Forestry with all Australian States and the Northern Territory, marine industries (shipping, ports, fishing, aquaculture), conservation groups and researchers. Australia's biosecurity system is largely managed under the Quarantine Act (1903). At present, biosecurity management of aquaculture is partitioned into: quarantine activities associated with import standards, established by Biosecurity Australia and implemented by the Australian Quarantine Inspection Service (AQIS); and operational management at State and Territory levels. The importation of a new species for use as an aquaculture product must be assessed and approved by Biosecurity Australia, with appropriate approvals by AQIS. Once these approvals are in place, importation can proceed once approvals from the State or Territory are provided. Under the current National System, it would be unlikely that approvals for a new importation of a species for open water culture would proceed due to the obligations to prevent and minimise impacts of non-native species in the marine environment. If approvals were given, the operator would be required to submit and have approved an Emergency Marine Pest Plan that outlines options for action in the event of escape or other problems such as a disease outbreak. Similarly, it is likely that ongoing monitoring would be required with mandatory reporting to State and Territory authorities.

For the purposes of New Zealand's regulatory requirements, non-native fish, aquatic life or seaweeds approved for use in New Zealand must be in the exclusive and continuous possession or control of the person undertaking the activity AND must be able to be distinguished or kept separate from naturally occurring fish, aquatic life or seaweeds.

Importation of plants and animals, including aquatic organisms for aquaculture, is rigorously controlled by the New Zealand Environmental Risk Management Authority (ERMA). Biosecurity arrangements restricting the importation and quarantine of new species (that is species not occurring in the wild prior to 1996) involves a thorough investigation of the potential risk of introducing this species into New Zealand including the disease risk it presents. The ERMA makes decisions on applications to introduce hazardous substances or new organisms, including genetically modified organisms.

5.4.1.2 China

In China, before the issue of the Quarantine Act of Import and Export Organisms (2004, People's Congress) and the Aquaculture Seedling Management Procedures

(2005, Chinese Ministry of Agriculture, MOA), the lack of management had resulted in a somewhat chaotic situation in non-native species introductions, and some species were introduced repeatedly. The Aquaculture Seedling Management Procedures was enacted to deal with this situation and, for the first time the introduction of broodstock, juveniles, larvae and fertilized eggs for aquaculture (research or production) purposes is under government control. All aquaculture seedlings are categorized by the MOA in collaboration with relevant branches of the State Council, as (i) whose import and export are forbidden; (ii) whose import and export rely on the approval of MOA; or (iii) whose import and export rely on the approval of Provincial Fisheries Administrations. Among other requirements, all applications for the import of aquaculture seedlings should contain a Safety Impact Report (including environmental and biological impact and possible disease transfer) and a Certificate of Origin. These measures are inadequate, in that there is still no integrated risk assessment system in China to prevent aquatic bio-invasion, no legislation governing the early-warning, removal and control of introduced species, and no ecological remediation and compensation liability measures to combat bio-invasion.

In recent years, however, Chinese central government has strengthened legislative and administrative measures supervising aquatic species introduction, and encouraging research efforts on risk assessment and the control of bio-invasions. Guided by the above mentioned acts and a number of regulations, the National Biosafety Office, affiliated with the State Environmental Protection Agency (SEPA), the MOA along with its provincial level agencies and the General Administration of Quality Supervision, Inspection and Quarantine of the People's Republic of China (AQSIQ) undertake the management of species introduction and the inspection of pests, parasites and diseases carried by any imported organisms. Strict inspection and risk assessment procedures have also been implemented on import and export of genetically modified organisms (GMO).

5.4.2 Risk Evaluation and Management

Risk evaluation has become a useful management tool to assess the biological and ecological aspects of ecosystems when using limited available data (i.e., managing under uncertainty). For example, ecologically sustainable development seeks a balance between the benefits and the costs (environment, economic, social) of an activity. In many instances, the information necessary to determine benefits and costs will be unknown and risk evaluation can aid the decision-making process. In simple terms, risk analysis is used to determine how often an event may occur (frequency) and what the consequences of such an event would be. Risk evaluations can inform decisions before allowing the import of a new species (pre-border) or before allowing release of a new species into the environment (post-border).

A standardised risk management process can be summarized in four steps: (1) establishing the context; (2) identifying the risk; (3) assessing the risks (risk analysis and risk evaluation); and (4) and treating the risks (e.g., Australian and New Zealand Standard Risk Management AS/NZ4360:2004). This is readily applicable to assessing pre-border biosecurity risk (e.g., microalgae import decision-tree; Campbell 2006b) and post-border biosecurity risk in the form of organism impact assessments (Campbell 2005, 2006a).

In an aquaculture context, risk evaluation must assess: (1) the introduced species being imported for commercial purposes (e.g., use of abalone, *Haliotis rufescens* and *H. discus hannai*, in Chile); (2) the mechanism of transfer to determine hitch-hikers including pathogens and parasites; and (3) the feed species (e.g., *Thalassiosira wiessfloggi* is fed to rotifers that are then used as aquaculture feed) imported to sustain both native and introduced aquaculture species.

Management of imported introduced species is typically controlled with the aid of Import Health Standards (IHS), that operate as codified rule structures that identify how, when and where a specific "risk good" can be imported, and adhere to the World Trade Organizations (WTO) related standards (Hewitt and Campbell 2007). IHS seeks to minimise the risk and identify appropriate management options (Orensanz et al. 2002; DAFF 2003; Pheloung 2003).

IHS's are often underpinned by species specific risk analyses. A decision tree model is one example of risk analysis where a series of simple yes/no questions progresses the assessment through the process, indicating where importation should be rejected, approved with or without stipulations (Fig. 5.4). The model can be qualitative, semi-quantitative or quantitative (data input dependent). Each step is assessed against a risk mitigation context (such as a management procedure) with the endpoint derived by the questions asked at each step in the process. The decision-tree applies the same set of criteria to all species, ensuring a consistent, objective and verifiable manner to assess all import requests and invariably considers specific national and international obligations.

Countries can also apply the risk evaluation embedded in the ICES Code of Practice (ICES 2005b), or develop more individualised importation processes (e.g., Hewitt et al. 2006). For example, a generic importation model for aquaculture species identifies risk as an integral component, followed by an economic assessment of cost: benefit (Fig. 5.5). The model is initiated when a request to import a non-indigenous species or non-indigenous genome occurs. Decision makers undertake a risk evaluation that defines: unacceptable impacts, risk methods used and *a-priori* states the acceptable level of risk. The process is supervised by a scientific review committee and produces contingency and action plans or guidelines that deal with the accidental release of a non-indigenous species.

5.5 Concluding Remarks

Rapid increases in the production of non-native species and the associated risks of unintentional introductions and pathogen and parasite transfers to native populations underscores the urgent need for concerted global action in advancing

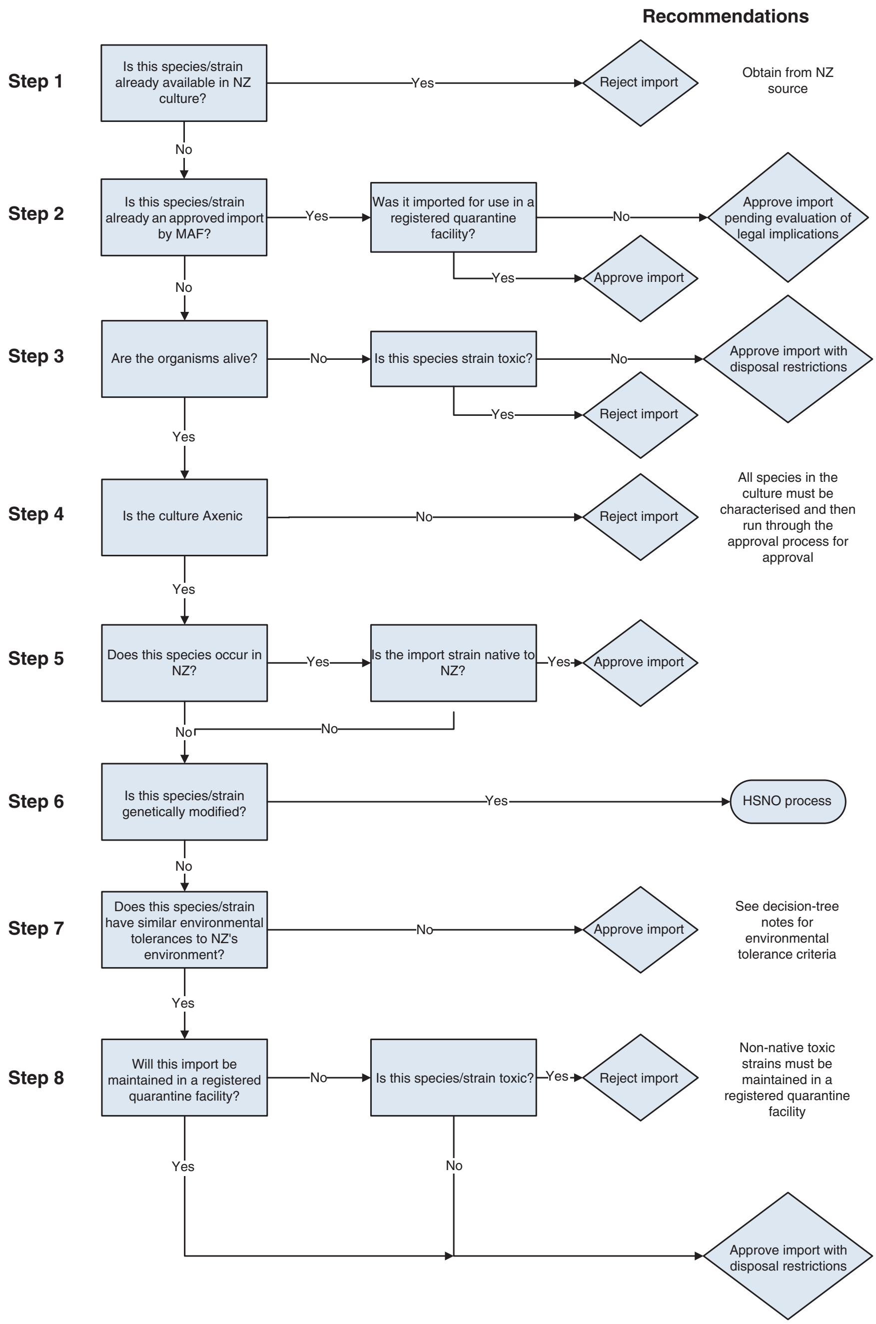

Fig. 5.4 Microalgae decision-tree developed for assessing the risk of importation of microalgae to New Zealand

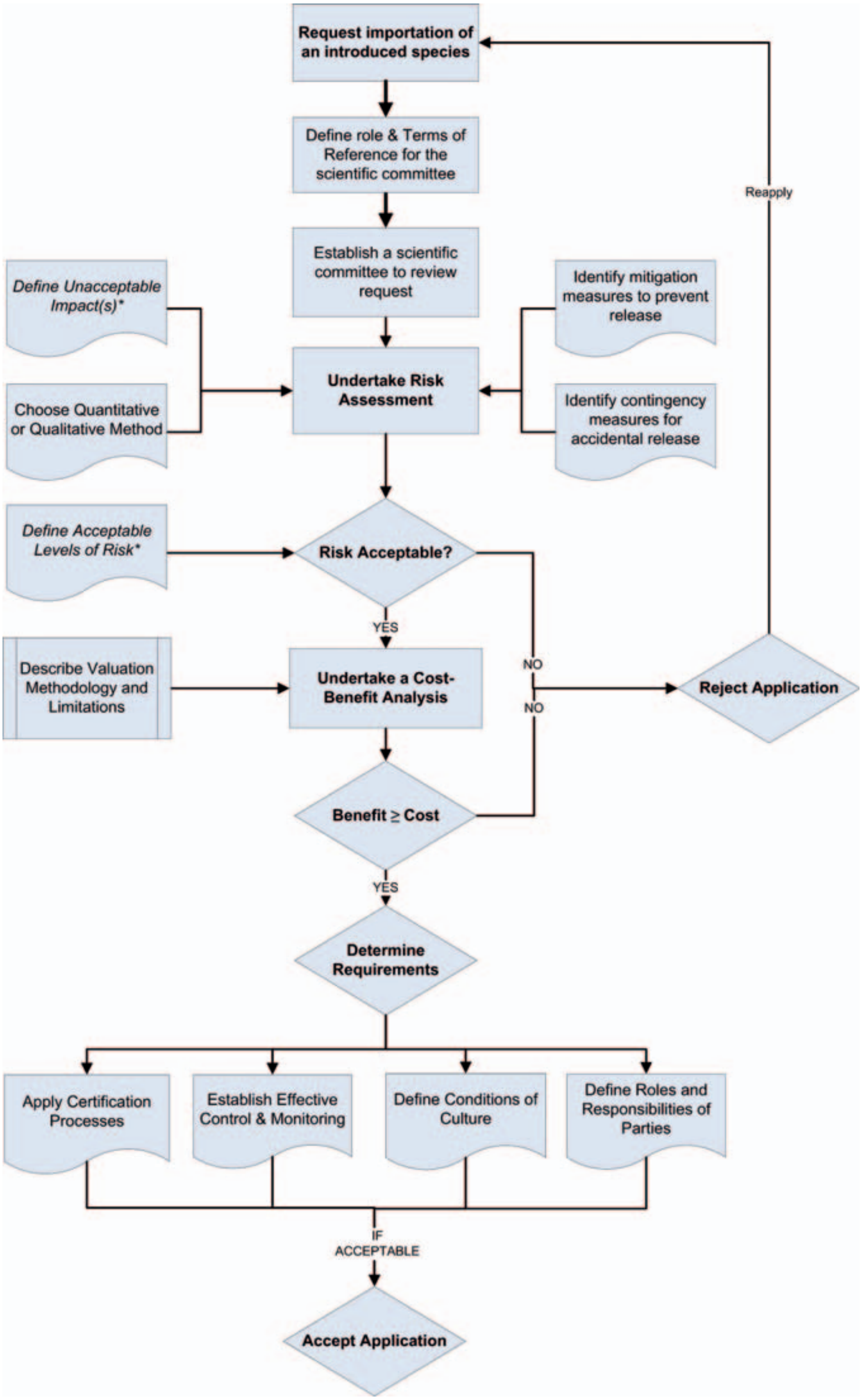

Fig. 5.5 A conceptual risk framework used for the importation of non-indigenous species for aquaculture purposes

environmentally sound aquaculture practices. If aquaculture is to be sustainable and the ecosystem safeguarded, particularly in regions of high endemism (i.e., biodiversity hotspots), effective controls on the introduction of non-native species associated with production are needed. Species diversity has been linked to increased robustness of systems to exploitation (Worm et al. 2006), making protection of biodiversity hotspots a clear priority (Webster et al. 2005). Existing codes of practice and risk evaluation models serve as important guidelines and should be carefully considered and actively promoted in planning non-native aquaculture. The 10-year Global Conservation Fund or the 5-year Critical Ecosystem Partnership Fund, which are aimed exclusively at hotspots (Brooks et al. 2006) should be used to assist the regions of highest risk to adopt international regulations and risk assessments for the introduction of non-native species for aquaculture purposes.

It is evident that a multi-disciplinary approach is needed to draw together experts, particularly from aquaculture, invasion biology, sociology and economics, which till now have had relatively limited interaction. In addition, efforts should be directed towards joint partnerships between countries and experts that have pioneered aquaculture research and those which possess the greatest biodiversity to improve growth rates, immunological resistance, product quality and market availability for native cultured species and/ or to design more robust aquaculture systems. Such action would either reduce the need to introduce non-native species for aquaculture purposes or minimise the risk of escape.

Finally, efforts should be advanced to increase the profile of concerns surrounding non-natives, in order to educate and involve a broad cross-section (scientists, industry, managers and the public) and promote sustainable aquaculture practices. This should include an international forum of experts and countries prepared to aid development in developing countries, symposia and workshops that engage a diverse community, ready access to the above codes of practice and related information, and explore market identity for environmentally sound products (Bartley and Minchin 1996).

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