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CLUSTER REPORT ON RESOURCE REQUIREMENTS FOR THE AICHI BIODIVERSITY TARGETS

TARGET 9: INVASIVE ALIEN SPECIES

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Anticipated costs of meeting the Aichi Biodiversity Targets for 2020: Target 9 - Invasive Alien Species

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1 Introduction

The Convention on Biological Diversity (CBD) has 193 signatory countries all of which are legally bound to abide by the legislation regarding biodiversity. Article 8(h) of the CBD states that "*each Contracting Party shall, as far as possible and as appropriate, prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species*". In October 2010 in Nagoya, Aichi Prefecture, Japan, the CBD met at the tenth Conference of Parties (COP). They adopted a revised Strategic Plan for Biodiversity for 2011-2020. Twenty Aichi Biodiversity Targets were set under strategic goals. Strategic Goal B aims to "*reduce the direct pressures on biodiversity and promote sustainable use*". Target 9 (which is listed under Goal B) deals specifically with invasive alien species:

"By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment."

Invasive alien species (IAS) are classed as one of the biggest threats to global biodiversity, along with habitat destruction and climate change (Vitsouek *et al.* 1997). Although biological invasions have been occurring for millennia, the rate of new introductions (either intentional or unintentional) continues to increase and the numbers of documented IAS are increasing exponentially (McGeoch *et al.* 2010), maybe in part due to increased research efforts. It is thought that globalization, and in particular, income growth and transport efficiency play a major role (Hulme 2009). With goods traded in greater volumes than ever before, between a wider network of countries and with quicker transit times, IAS are being unintentionally introduced to the rest of the world.

As per the CBD definition, an alien species is *"a species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce*". Accordingly, an IAS is "*an alien species whose introduction and/or spread threaten biological diversity*". The International Plant Protection Convention (IPCC) defines a pathway as *"any means that allows the entry or spread of a pest"*.

There are both negative and positive impacts associated with IAS (Bax *et al.* 2003). The negative impacts include impacts on the environment, impacts on human health and economic losses from marine activities, such as fisheries, aquaculture and tourism, leading to loss of income and wellbeing. Dealing with the problem incurs opportunity costs associated with the financial resources allocated to personnel, scientific and technical resources. Positive impacts include direct and indirect benefits from the use of the IAS. The relative magnitudes of the advantages and disadvantages are likely to vary considerably between different IAS in their various contexts. Most economic studies on IAS have dealt with the damage costs caused by the invading species (e.g. Perrings 2002). The cost of IAS to ecosystem services can be vast. For example, in the 1980s, the introduction and subsequent population explosion of the comb jelly *Mnemiopsis leidyi* into the Black Sea caused dramatic reductions to the pelagic plankton communities which led to the collapse of several commercial fisheries (e.g. anchovy) which only recovered years later (Shiganova 1998). While estimates of the damage costs incurred by IAS or the benefits derived from their control are few,

there have been even fewer analyses of the costs of research on and prevention and control of IAS, and these are likely to be influenced by a wide range of variables (Perrings *et al.* 2000).

The UK Department of Environment, Food and Rural Affairs (DEFRA), the United Nations Environmental Programme – World Conservation Monitoring Centre (UNEP – WCMC) and ICF GHK aim to develop a plausible range in which the total global cost of meeting the Aichi Targets is likely to fall. The overall study will include a broad analysis of type and scale of actions necessary for achieving the targets, and an initial aggregated assessment of resource requirements, and proposals on future actions required.

This study provides a rough order-of-magnitude estimate of the costs of meeting Target 9. All investment and expenditure needs are expressed in US$ at 2012 prices (unless otherwise specified). Note that there are potential overlaps and synergies between Target 9 and some of the other Aichi targets. These include Target 5 (habitat loss), Target 7 (aquaculture), Target 12 (threatened species), and Target 14 (ecosystem restoration). Dealing with IAS forms a necessary component of these other targets. We have not made any adjustments in this study, but rather assume that the estimates for the other assessments are net of the cost of dealing with IAS. In addition, while IAS removal is a form of ecosystem restoration (Target 14), further ecosystem restoration after the removal of IAS is often an aspect of IAS management, and we have not included these costs in this study.

2 Types of IAS and pathways of invasion

Of the species that find their way to localities outside of their natural range, only about 1% are thought to establish in the host country and then become invasive (Williamson 1998). These species are generally those that find themselves in favourable habitats but without the forces such as predation, disease and competition, that keep their populations in check within their natural range. Invasive alien species thus represent all types of biota, but tend to be pioneer-type species that are tolerant of a wide range of conditions. Although IAS also extend to disease and pathogens affecting humans, livestock, forestry and crops, in this study we have concentrated on plant, invertebrate and vertebrate invaders of freshwater, coastal and terrestrial habitats only.

Introductions of IAS are facilitated by human activities such as trade and travel, which in turn are increased through economic growth and globalization. The propensity for introduced species to proliferate is also facilitated by human impacts on the environment, including climate change. Introductions can be intentional or unintentional. The main means of introductions and spread of IAS are as follows:

- **Aquaculture, mariculture and horticulture**, which can lead to the escape and establishment of IAS.
- **Agriculture and forestry** results in the introduction of many species which can become invasive and spread into other areas when not properly managed.
- **Transport systems** result in the unintentional transport of organisms. Ship ballast water, which is carried from one port and discharged in another area, is one of the most significant pathways for marine bio-invasions. Fouling on the hulls of ships and other boats is also a

means of transport of organisms. Other such pathways include organisms transported with raw goods such as timber, or that end up on the vehicles themselves.

- **Tourists**, who number nearly a billion international arrivals annually, and can transport species intentionally (e.g. as souvenirs) or unintentionally (e.g. in the soil on their shoes).
- **Trade in pets, aquarium species and ornamental plants** is responsible for legal and illegal movements of species across borders, resulting in introductions of escapees into new areas.

3 The extent of the IAS problem

Invasive species pervade nearly every type of ecosystem and pose one of the biggest threats to biodiversity worldwide. Impacts of IAS on biodiversity can be at the ecosystem, species or genetic level. They alter natural habitats, modify ecosystem processes such as hydrology, fire regimes and nutrient cycling, and they prey upon and/or compete with indigenous species for resources such as space and food. These impacts lead to significant changes in ecosystems and result in the loss of diversity and populations. Their impacts on ecosystem services can be felt at regional and national scales, such as the impacts of IAS on water supply in South Africa (Higgins *et al.* 1997, Turpie 2004). Invasive species are a significant factor in the plight of endangered birds, amphibians and mammals (McGeoch *et al.* 2010). Some of the most prolific offenders have established virtually worldwide distributions outside of their native ranges. Invasive species lead to the homogenisation of biodiversity, through the direct loss of genes or gene complexes, and hybridisation of IAS with indigenous species (CBD 2001).

Islands deserve particular attention in terms of the threat that IAS pose on species. Islands represent less than 5% of the earth's land area yet harbor 80% of known species extinctions since 1600 and 40% of today's IUCN Critically Endangered species (IUCN 2011, Ricketts *et al.* 2005). Invasive alien vertebrates (IAV) are implicated in the majority of insular extinctions and remain a key risk to today's threatened species. (Alcover *et al.* 1998, Atkinson 1989, Blackburn *et al.* 2004, Clavero & Garcia Berthou 2005). Eradication of IAV from islands has proven an effective conservation tool, resulting in remarkable recoveries of endangered species and threatened island ecosystems (Aguirre-Muñoz *et al.* 2008, Campbell & Donlan 2005, Howald *et al.* 2007, Nogales *et al.* 2004, Veitch & Clout 2002). Over 1,100 successful IAV eradications have been implemented on islands worldwide; with practitioners undertaking removals from increasingly larger, more remote, and more technically challenging islands each year (Howald *et al.* 2010, Keitt *et al.* 2011, Phillips 2010). Based on this history of successful eradications, it is widely accepted that funding and social issues (acceptance by island residents and users of eradication efforts), rather than island size, are now the main limitations to where invasive alien vertebrates can be successfully eradicated (Campbell & Donlan 2005, Howald *et al.* 2007, Oppel *et al.* 2011).

The effects on biodiversity over time are unpredictable, but it is predicted that global climate change will further impact on biological invasions. Five IAS related potential consequences of climate change include altered transport and introduction mechanisms, establishment of new IAS, altered impact and/or distribution of existing IAS, and altered effectiveness of control strategies (Hellman *et al.* 2008). These impacts could potentially benefit indigenous species too, however it seems certain traits will allow IAS to better capitalise on the resulting changes to the environment (Dukes & Mooney 1999).

Several organisations and networks collate country-level information on IAS at a regional or global scale (Table 1). However, most datasets suffer from some degree of data limitation (both in quality and global representation). According to the Group on Earth Observations Biodiversity Observation Network (GEO BON) current global observation systems in monitoring IAS information are limited, but with moderate effort, additional datasets required to cover all aspects of IAS management can be constructed by 2020 (GEO BON 2011). Currently, there are geographic and taxonomic gaps in IAS knowledge (Pfenninger & Schwenk 2007). Knowledge of IAS impacts in mountain areas, tropical coastal and marine areas, forests and continental islands is particularly sparse. Most countries, especially low to middle income countries, need to establish their current baselines on status, impacts and trends of invasive alien species.

In addition to the above, the University of California Santa Cruz (UCSC) in collaboration with Island Conservation are in the process of developing a database to keep track of trends in control or eradication programmes on islands. From a total of 700 global island eradications, only 50 or so contain data on costs of eradication. The focus is on single species mammal eradication programmes on islands which are mostly less than 200 km^2 (E. McCreless, UCSC, pers. comm.). The **Threatened Island Biodiversity (TIB)** database is the most comprehensive hub of information describing IUCN threatened species on islands at risk from invasive vertebrates, and to date a total of 1347 islands have been identified for 1118 Critically Endangered (CR) or Endangered (EN) species (largest island is New Guinea). The TIB database provides key guidance by highlighting where eradications can be employed on a global scale to prevent extinctions. TIB partners include Island Conservation, the UCSC Coastal Conservation Action Laboratory, Birdlife International, and the IUCN Invasive Species Specialist Group.

Table 1. Detailed list of currently available observational datasets used to predict trends in IAS and their influence on the ecosystem and human responses. Modified from GEO BON 2011.

Observation dataset	Source	Year initiated	Update frequency	Geographical coverage
Pressure				
Extent of IAS	GISIN[1];	2004	Regular	Global
	SAHFOS[2]	1931	Regular	N. Atlantic & N. Sea
Trends in IAS	DAISIE[3]	1970	Regular	27 EU & 10 non-EU states
State				
Trends in IAS-driven species extinction	IUCN[4] Red List, RLI[5] dataset	1980	4-10 yearly	Global
Response				
IAS international policy	CIB[6]	1950	Annual	International
Ballast water treatment	IMO[7]	2012	In port	Global

[1] GISIN – Global Invasive Species Information Network; [2] SAHFOS – Sir Alister Hardy Foundation for Ocean Science; [3] DAISIE – Delivering Alien and Invasive Species Information for Europe; [4] IUCN – International Union for Conservation of Nature; [5] RLI – Red List Index; [6] CIB – Centre for Invasion Biology; [7] IMO – International Maritime Organisation.

A list of the "**world's 100 worst invasive alien species"** has been compiled by the Global Invasive Species Database and La Fondation TOTAL d' Entreprise in collaboration with the IUCN's Invasive Species Specialist Group (ISSG). Due to the complexity of interactions between a species and its environment, certain assumptions were made in creating this list. Species were selected based on the severity of their impact on biodiversity and/or human activities, as well as "their illustration of important issues surrounding biological invasion." Only one species from each genus was included in this list. Nevertheless it is interesting to note that there is a fairly good relationship between the number of these "worst 100 IAS" recorded per country and the total number of IAS recorded per country (Figure 1).

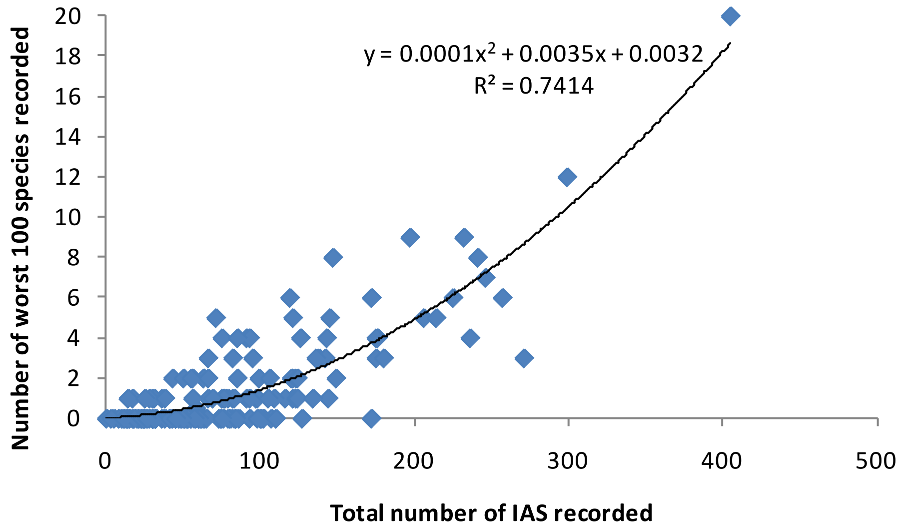

Figure 1. Relationship between the number of the 'world's worst 100' IAS recorded and the total number of IAS recorded per country.

In addition, a few published studies have collated and analysed data on IAS at a global level (or for a large sample of countries). These include the recent study by McGeoch *et al.* (2010), in which a pressure-state-response framework was used to analyse progress in meeting the CBD's 2010 Biodiversity Targets relevant to IAS using a set of three indicators: i) The number of documented IAS per country (pressure); ii) the impact of IAS on Red List species (state); and iii) legislation regarding IAS (response). Ideally, such an analysis needs to be expanded to include the risk of further invasions, analyse the impacts on biodiversity and ecosystem services, and human-driven control and pathway management (GEO BON 2011).

4 Actions required to meet Target 9

Three main types of action are required to meet Target 9:

(i) research and prioritisation of IAS and pathways to be targeted;
(ii) control and eradication measures to reduce existing IAS; and
(iii) preventative measures (including quarantine, surveillance, awareness raising, policy and legislation) to avoid new introductions.

4.1 Research and prioritisation

Not every IAS is a priority, and in order for expenditure to be efficient, it is important to prioritise management efforts on the basis of potential damage by different types of invaders in different areas to biodiversity and society, as well as the costs and effectiveness of the potential actions that can be taken. This requires the completion of an initial biological baseline survey, from which a regular monitoring programme (collecting data on the extent and magnitude of invasions) can be established. Considerable research has already taken place on the nature and invasion pathways of IAS, risk analysis, impact assessment, cost-benefit analysis, and the relative effectiveness of different actions (e.g. Jones *et al.*2008, Campbell & Donlan 2005, Campbell *et al.*2011). Nevertheless ongoing research will be imperative in order to fill gaps in our understanding of current invasions as well as with regard to new invasions.

4.2 Control and eradication

Four main strategies for dealing with established IAS are mitigation, containment, control and eradication (CBD 2009):

- **Mitigation** deals with attempting to minimise the damage caused by an IAS, e.g. by relocating an endangered species.
- **Containment** aims to restrict the spread of IAS and involves detecting and eradicating any new infestations outside of the containment area.
- **Control** aims to achieve a reduction in the density and abundance of IAS to below an acceptable threshold, such as a level that allows native species to recover and regain ground.
- **Eradication** is the elimination of the entire population of an IAS from a managed area.

These measures are part of a gradient of interventions (Gherardi & Angiolini 2004). The relative impacts of these different measures are shown in Figure 2. In some cases, mitigation measures and/or adaptation can be more sensible (McNeely 2001). For priority species, control or eradication is desirable, as opposed to mitigation or containment. While control is usually more manageable, particularly on mainland areas, in many situations, particularly for the most damaging invasive alien vertebrates on small and medium sized islands, eradication has proven feasible and economically desirable compared with protracted control measures. Furthermore, complete eradication of an IAS tends to result in greater conservation gain for native species than sustained control where reduced numbers of the invasive still impact native species. In addition to these measures, it is often necessary to include rehabilitation of damaged ecosystems or populations, however these actions have not been considered in this study.

The methods used are common across most types of IAS, and include mechanical removal, biological control, chemical treatment, habitat management and construction of barriers (based on Gherardi & Angiolini 2004). Eradication, involving the removal of every potentially reproducing individual from an area or reducing its population density below sustainable levels, is optimal in that it removes the need for further control and associated costs. Eradication of IAS on continents and very large islands is generally only successful where the problem is contained within a manageable area, such as in the early stages of an invasion. Successful eradication of the most damaging invasive vertebrates (rodents, cats, ungulates) on small to midsize islands (less than 400 km sq) is now commonplace (Keitt *et al.* 2011).

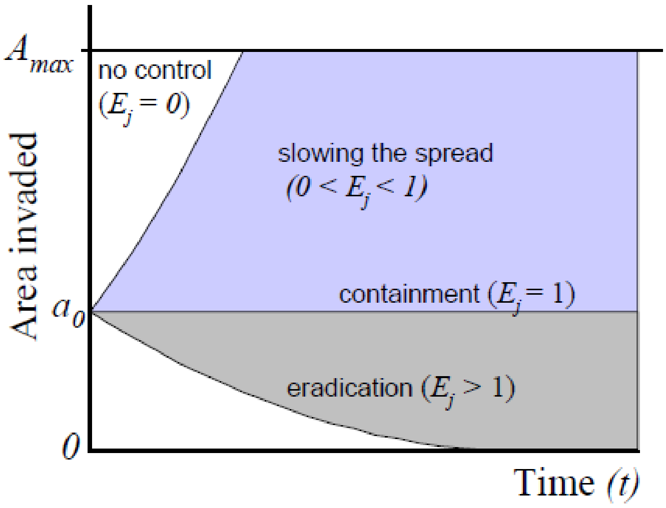

Figure 2. Impacts of different control measures on established IAPs (source: Wise *et al.* 2007).

4.3 Prevention of new introductions

Prevention has been shown to be a cost-effective means of dealing with IAS prior to introduction (Leung *et al.* 2002). Once high-risk species and pathways have been identified, measures need to be set in place to prevent the transport and establishment of species preferably before they leave the source area (pre-border), or once they arrive at the border.

In order to prevent intentional introductions, pathways need to be monitored and regulated (and strict authorisation, quarantine measures and standards have to be put in place). Examples of intentional introductions include the trading of goods (from foodstuffs to timber), the horticultural industry and the exotic pet trade. In these cases, the costs should be borne primarily by those who are responsible for the introduction and who will stand to benefit economically as a result (McNeely 2001).

Measures for preventing unintentional introductions include monitoring of pathways (e.g. maritime ports, recreational boating marinas and airports) and vectors (e.g. ships or airplanes), quarantine and border control (biosecurity), treatment or vector control, raising public awareness and developing appropriate legislation to support enforcement.

In order to prevent IAS from entering the country, the Risk Assessment process requires the development of 'white', 'black' and 'grey' lists (McNeely 2001). The placing of a species on one of these lists, demonstrates the level of threat it represents. The 'white' list contains species which are considered safe, although some monitoring is required (as the behavior of a species in a new environment cannot always be predicted). The 'black' list contains species which are likely to become invasive (often based on their invasiveness in other countries) and introduction of these is strictly prohibited. 'Grey' lists contain species which have unknown consequences most often due to a lack of biological data or resources. In all cases, the precautionary principle should be applied wherever possible (Fairbrother & Bennet 1999).

The CBD developed a set of interim guiding principles for the prevention and mitigation of impacts of alien species. These have been modified by McNeely (2001) to include the following strategic responses:

1. Build capacity to address IAS problems;
2. Build research capacity;
3. Develop economic policies and tools for addressing IAS issues ;
4. Strengthen national, regional and international legal and institutional frameworks to address IAS;
5. Institute a system of environmental impact and risk assessment for IAS;
6. Build public awareness of the problem of IAS;
7. Promote sharing of information about IAS;
8. Build responses to IAS into other relevant sectors;
9. Build IAS issues into global change programmes; and
10. Promote international cooperation to deal with problems of IAS.

Several countries have now developed 'rapid response' measures for those scenarios where prevention measures are not sufficient and the invasive species is only identified once introduced or established (McEnnulty *et al.* 2001, NISC 2003, NEANS 2006, Locke & Hanson 2009). Most rapid response systems ultimately aim for complete eradication, that is, the removal of all reproductive potential from the population and there are several successful cases of this (Locke & Hanson 2009).

In order to be effective, both pre- and post-invasion measures must be reflected and enforced through the adoption of appropriate national and international policies. However, in order to tackle this, countries require expertise in IAS ecology and management, a sufficient degree of development and long-term financial support for projects (GEO BON 2011). The number of agreements and the number of countries party to these have been increasing exponentially since 1951 (McGeoch *et al.* 2010). Just over half of all countries signatory to the CBD have national legislation which covers IAS, and there are ten international agreements (currently in force) which specifically address the control/eradication and prevention of IAS.

5 Overall approach for estimating costs

This study was based on existing information in the published and unpublished literature sourced from internet searches, correspondence and interviews with key informants and organisations. Due to time constraints, the amount of effort that could be put into this task was limited, and the search could not be exhaustive.

Available data on IAS status and costs and a range of country statistics were collected from websites, published papers and unpublished reports. All costs were first converted to 2012 values using CPI index values published by the World Bank, then converted to US dollars. Cost data were summarised into a spreadsheet database. Data were separated into (a) research and prioritisation, (b) control costs of IAS on mainland areas, (c) eradication costs of IAS on small islands and (d) prevention measures, and were recorded by type of IAS and by country. Data on (a) and (d) were comparatively sparse. While most attention has been paid to estimating the costs incurred by IAS and the costs of their control or eradication, very little has been written on the costs of preventative measures, or on the research, risk assessment and prioritization measures that have to be undertaken to inform strategies for dealing with IAS.

A major limitation in the above data was that, in most cases, information on costs reflected actual expenditure rather than the estimated expenditure required to meet targets such as the Aichi 9 target. In South Africa, estimates of both were available for the control of the most costly group of IAS, and the required expenditure was estimated to be roughly ten times the actual expenditure over the past 15 years. Thus to get around this, as well as the shortage of data on control costs generally, we made use of damage cost data to produce estimates of the required expenditure on control. Information on damage costs were thus also collated by country and were used to estimate control expenditures required using an estimated average benefit-cost ratio of 10:1, based on benefit-cost ratios that have been estimated for a variety of control programmes (Table 2). Thus even where control costs were available (e.g. Pimentel *et al.* 2005), we adjusted estimates of control costs where large damage costs were reported and control costs were zero or substantially less than our estimates based on damage costs. Where studies only reported current control costs, these were not adjusted, and may be underestimates of the actual expenditure required to meet the Aichi 9 target.

Another limitation was that many studies reported costs at a site level or limited spatial scale, such as at a provincial level. Where possible, information in these studies was used to provide an estimate scaled up to the national level, based on information that could be found on the full extent of the problem. If this could not be done then the data were omitted from the study.

For each country, control cost data obtained for each IAS were summarised into fourteen groups of IAS. Estimates of different types of costs pertaining to single or multiple groups of IAS were obtained for a total of 33 different countries on all five continents. One study (van Wilgen *et al.* 2012) was used to produce estimates for woody IAS for all African countries on the basis of area of different biomes, bringing the total number of countries for which we had at least partial estimates to 55. We were able to compile a relatively "complete" sets of estimates (in which it was felt that most or all IAS groups were represented) from existing information for twelve countries, most of which were high income countries.

Table 2. Cost-benefit ratios reported for the control or eradication of a range of IAS.

Species	B-C ratio	Source
Water hyacinth	5:1 – 67:1	Wise *et al.* 2007
Water hyacinth	13.6:1	OTA 1993
Red water fern	1,130:1	Emerton & Howard 2008
Purple loosestrife	26.5:1	OTA 1993
Parthenium weed	0.2:1 – 5.4:1	Wise *et al.* 2007
Triffid weed	1.7:1 – 3.1:1	Wise *et al.* 2007
Diffuse knapweed	17:1	Frid *et al.* 2009
Hawkweed species	185:1	Frid *et al.* 2009
Melaleuca	11.4:1	OTA 1993
Sea lamprey	30:1	OTA 1993
Larger grain borer	3.6:1 - 15.6:1	Wise *et al.* 2007
Alfalfa blotch leafminer	8.5:1	OTA 1993
Mediterranean fruitfly	19.6:1	OTA 1993
Feral swine	134:1 – 1,562:1	Engeman *et al.* 2004

Data were extrapolated to all countries based on national level indicators and statistics, or in the case of small islands, based on data from the TIB database. These extrapolations were based on statistical relationships as far as possible. In the absence of any robust relationship, geometric mean of the costs per unit area of each group was used to produce estimates of missing values. In order to account for differences in costs in different countries, cost data were analysed using the 'international dollar', on the basis of purchasing power parity (PPP), and extrapolated estimates were converted back to actual dollar costs. Detailed assumptions are given in the sections that follow.

The cost estimate for eradication of IAS from small islands included islands from the TIB database where IUCN CR or EN birds, mammals, amphibians, reptiles were confirmed as breeding or probably breeding in the last 20 years, where the presence of rodents was confirmed, suspected or unknown, which had fewer than 1000 people, and which were smaller than 40,000 hectares for rodent and predator projects, and smaller than 55,000 hectares for ungulate projects. This was considered achievable by 2020. Impacts and eradication costs of IAS on small islands were analysed in relation to island size and extrapolated based on known infestations on islands.

In general, our study aimed to produce order-of-magnitude estimates and does not pretend to be accurate. Estimates were broken down into investment and recurrent expenditure. Investment was assumed to occur during 2012-2014, and includes (a) research and prioritisation, (b) expenditure on infrastructure, awareness raising, policy and legislation to put prevention measures in place and (c) the eradication of IAS from priority small islands. Recurrent expenditure included expenditure on control, starting from 2012, as well as ongoing expenditure on prevention measures such as surveillance, quarantine, and the maintenance of equipment such as ballast water treatment systems. There was too little information on the breakdown of costs into government versus private expenditure to produce a general estimate of this breakdown, but this was described where known. For the same reason, estimates of the breakdown of the costs into labour, materials, *etc.*, could not be attempted.

6 Results

The following sections provide a review and summary of the available information on each of the different types of actions required to meet Aichi Target 9, as well as the assumptions used in the extrapolation of data to generate global estimates.

6.1 Costs of research and prioritisation

Considerable progress has already been made in terms of the accumulation of baseline information at a global level. The UNEP-WCMC 2010 BIP IAS Indicator project, which was run through the Global Invasive Species Programme, received US$0.149 million in funds over 2009 and 2010 (initial costs of setting up the project). This project aimed to determine how successful the CBD parties had been in meeting the 2010 Biodiversity Target. However, the current observation networks described in Table 2.1 require expansion so that they have greater global coverage. GEO BON (2011) estimated that in order to meet Target 9, €110,000 per annum (US$0.141 million in 2012 values) would be required to construct the additional observation databases.

In the 1990s the Office of Technology Assessment (OTA) of the **United States** undertook a study to determine the economic and environmental impacts of all IAS in the country. Several years later, a 400-page report was published with details of the results. The estimated costs of completing this study were just over US$1.1 million (Wittenberg & Cock 2001).

Of the few countries for which information on research expenditure was found, the expenditure was by far the highest in the **United States**, where US$257.8 million was allocated to IAS research in 2012 (NISC 2012). Williams *et al.* (2010) estimated that research on IAS in **Great Britain** amounts to some GBP17.4 million (US$29.7 million) per annum. In **South Africa**, the costs of researching biological control of seven invasive alien weed species from 1932 – 2000 was estimated to be ZAR41.1 million (US$1.6 million per annum in 2012 values). Of this research budget, 42% was spent on one species - *Lantana* (van Wilgen *et al.* 2004). McLeod (2004) estimated that research costs for nine species of vertebrates in **Australia** (including feral cats, rabbits, feral pigs, wild dogs, mice, feral goats, cane toads, wild horses and carp) amounted to AU$16 million (US$21.0 million) per annum. Currently, the majority of research funding seems to be directed towards IAS that have an impact on economic activities such as agriculture, rather than on biodiversity *per se*.

Based on the above, we estimated that research costs are typically about 2 – 15% of control costs, and using the overall cost estimates given below, this would suggest that research costs should be in the order of $3-5 billion annually, with higher investment initially. Given that past research has concentrated on industrial pests, we estimated that at least half of this should be spent on IAS threats to biodiversity.

6.2 Costs of control on mainland areas

No-one has attempted to produce a comprehensive estimate of the costs of IAS at a global scale, either in terms of their impacts or the costs of prevention and control. Based on their work in the United States, Pimentel *et al.* (2002, in Williams *et al.* 2010) suggest that the total loss to the world economy could be in the order of 5% of annual production. Based on estimates put forward by a group of experts, the GEF-6 Needs Assessment suggested that the costs required for the 155 CBD countries to deal with IAS for the period 2014-2018, might be in the order of US$67.5 – 103.2 million.

A handful of studies have been carried out at a national or multi-national scale on the costs incurred by multiple IAS. In 1993, the Office of Technology Assessment (OTA) of the **United States** Congress compiled the first national estimate of costs associated with IAS, with 79 species causing an estimated US$97 billion in damages from 1906-1991 (approximately US$1.1 billion per annum). Pimentel *et al.* (2000) calculated costs of IAS on the **United States, United Kingdom, South Africa, Brazil, India and Australia** which amounted to US$314 billion annually. The annual costs for the six countries to control the problem were estimated to be US$30 billion. The estimate for damage costs incurred to the **United States** alone, originally published by Pimentel *et al.* (2001), were updated to US$120 billion per year (Pimentel *et al.* 2005). This estimate was calculated by considering impacts resulting from the following groups: plants, mammals (rats and other), birds, reptiles, fishes, arthropods, molluscs, livestock diseases and human diseases. This includes combined damage and control costs (direct costs), however, it does not adequately reflect the full value of losses of biodiversity and ecosystem services, as some of the values associated with these are challenging to express in monetary terms.

In **Canada**, economic losses to fisheries, agriculture and forestry production as a result of ten IAS were estimated using available literature on indirect and direct costs (including costs of control, reduced yield, reduced land value, trade bans on exported goods, compensation paid to farmers, health care costs, and reduced tourism and tourism-related revenues). Where this data was unavailable for certain regions, but the distribution of the species was known, an extrapolation was completed to estimate national costs. The value of the resource or industry was also considered in the calculations. Characterised costs amounted to CDN$187 million per annum (US$213.5 million) (including disease costs). In addition to this, due to a lack of comprehensive, nationwide data, a simple empirical model was constructed to estimate the effects of sixteen IAS on annual production, excluding non-market values. The estimated losses in fisheries, agriculture and forestry production as a result of IAS were estimated to be CDN$13.3 – 34.5 billion per year (US$15,186.6 – 39,393.8 million; Colautti *et al.* 2006).

Williams *et al.* (2010) conducted a comprehensive analysis of the economic cost of IAS to **Great Britain** based on a detailed questionnaire sent to key organisations as well as existing published, unpublished and internet-based data. They estimated a total annual direct economic cost of GBP1,291 million, GBP245 million and GBP125 million to England, Scotland and Wales respectively, with an overall total direct cost of approximately GBP1.7 billion (US$2,900.7 million). Their estimates explicitly included some estimates on control costs.

Other national scale studies include those on China, Germany and Sweden. The 283 invasive alien species in **China** were estimated to incur direct costs totalling US$3,231.4 million (in 2012

values). The total economic impact, including indirect losses, was estimated to be US$19,477.2 million (2012 value), some five times the direct loss, and 1.9% of GDP (Xu *et al.* 2006). Their study aggregated impacts at an industry level rather than at an IAS level (or IAS group) making their estimates difficult to compare with others. Nevertheless they noted that, while the costs to China were only a ninth of those in the United States, they were a similar proportion of GDP. Gren *et al.* (2009) estimated the total costs (impacts and control) of 13 different IAS in **Sweden** to be between SEK1,620 and 5,081 million(US$259.7 – 814.5 million), commenting that the most reliable estimates related to human and animal health impacts, with the costs of impacts on biodiversity being least reliable. Rheinhardt *et al.* (2009) estimated the costs of controlling a variety of types of IAS in **Germany** to be in the order of EUR109-263 million in 2009 (US$141.7 – 342.0 million in 2012 values).

Most other studies have been more limited in scope (types of IAS or economic sectors considered) or geographic scale (sub-national or project-level studies). For example, Leavold *et al.* (2007a,b) estimated the costs of two species of IAS in multiple African countries. The majority of studies describe the impacts of a particular species in a particular area (e.g. de Groot *et al.* 2003). In addition to studies on the impacts of IAS, numerous studies, mostly in the grey literature, describe management of particular species in particular areas, and discuss the costs of management, with a view to achieving efficiency in control methods. There have also been a number of studies incorporating estimates of both impacts and control costs to inform cost-benefit analyses of particular control programmes, as well as retrospective analyses to inform future strategy (e.g. Wise *et al.* 2007, van Wilgen *et al.* 2012).

The estimates collated in this study yielded estimates of control costs for various types of IAS in 55 countries, of which 15 countries had estimates that were considered reasonably complete in terms of the range of IAS covered. For the countries for which sufficient data were available, the costs of controlling IAS other than agriculture/forestry pests were about 33% of the total.

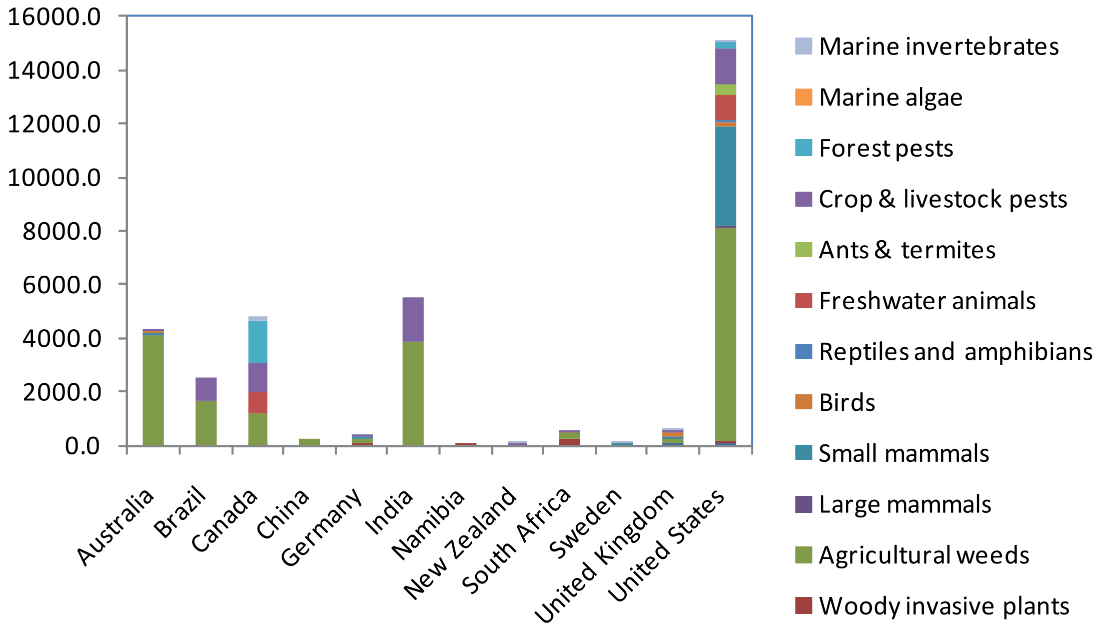

Figure 3. Estimated control costs for different groups of IAS in eleven countries

Estimates of total control costs were extrapolated from the available data in two ways: (1) based on average control costs per IAS, and (2) based on total control costs per country. The average control costs per unit area (at a national scale) could be estimated for each type of IAS. Costs were highest for woody invasive plants, agricultural weeds and agricultural insect pests (Table 3). These were also the groups for which most data were available, probably as a result of the high stakes and costs involved as well as the extensiveness of these problems. Extrapolating these average costs to the countries of the world, and adjusting for PPP yield an estimate of **total control costs of $37 387 million** per annum.

Table 3. Minimum, maximum and mean control costs for each type of IAS, at a national scale, standardised to International $ per km^2.

Type of IAS	n	Min	Max	Geometric Mean
Aquatic weeds	9	0.3	223.1	9.1
Woody invasive plants	37	0.2	2 762.9	139.6
Agricultural weeds	20	1.4	1 339.8	42.6
Large mammals	3	1.3	21.3	6.2
Small mammals	9	2.7	327.3	17.8
Birds	4	0.6	47.2	3.4
Reptiles and amphibians	4	0.3	2.2	0.8
Freshwater animals	7	0.2	187.2	9.8
Ants & termites	1	43.7	43.7	43.7
Crop & livestock insect and snail pests	13	1.4	1 386.5	46.8
Forest insect pests	5	2.3	143.0	21.6
Marine algae	1	19.2	19.2	19.2
Marine invertebrates	6	0.2	9.8	3.3

Total costs for 15 countries were analysed in relation to a number of country statistics and indicators. The best relationship that we could find was with a combination of country size and the number of alien species recorded as per the GISD (in terms of international dollars):

$$\ln \text{Cost}_{PPP} = -3.842 + 0.0064 * \text{IAS} + 0.627 * \ln(\text{Area km}^2) \quad (n = 15,\ R^2 = 0.78,\ P < 0.0001)$$

This relationship was used to estimate the costs per country in international dollars, which were then converted back to actual estimated costs. In other words the discrepancies in the costs of inputs in different countries were taken into account. This yielded an estimate of total costs in the order of **$41 679 million per annum**, not dissimilar from the estimate obtained above. The mean of these two estimates was $39 533 million.

7 Costs of eradication of invasive alien vertebrates on islands

An Invasive Vertebrate Eradication Costing Model was developed using known costs associated with the planning and implementation of 37 projects that successfully eradicated vertebrates on islands ranging from six hectares to over 400,000 hectares. This model was applied to estimate costs of removing invasive alien vertebrates from islands that support IUCN listed CR and EN species. The islands were identified using the **Threatened Island Biodiversity (TIB)** database; the most comprehensive source of information describing IUCN threatened species on islands at risk from invasive vertebrates. The TIB has identified to date a total of 1347 islands that support 1118 Critically Endangered (CR) or Endangered (EN) species (largest island is New Guinea). The TIB database provides key guidance by highlighting where the eradication of IAV's can be employed on a global scale to prevent extinctions. Major TIB contributors include Island Conservation, the UCSC Coastal Conservation Action Laboratory, Birdlife International, and the IUCN Invasive Species Specialist Group.

It is important to note that the costing model was not developed to provide specific costs for an individual restoration project on an island. Each project to remove an invasive vertebrate from an island is unique and requires detailed planning. Thus, costs will vary depending on the associated complexities. The model has been developed with a goal of providing accurate cost estimates for suites of projects, such as a total cost for removing invasive vertebrates from all islands in a country or in an archipelago.

In the costing model, projects were grouped as rodent projects of 50 ha or less, rodent projects of greater than 50 ha, ungulate projects and predator projects. Rodent projects of 50 ha or less were assigned a flat cost assuming implementation would be undertaken using the application of rodent bait via bait stations or hand broadcast. For rodent projects above 50 ha costs were based on estimates from aerial broadcasts of bait. Costs were separated into five different categories: implementation, planning, non-target mitigation, additional costs associated with inhabited islands-hereafter social costs, and isolation. We divided project costs into these categories so that we could assess them separately, but also because many projects do not report the full costs of all of these project components. In doing so, we were able to more accurately reflect actual implementation costs and maximize the sample size.

Costs for implementation of all projects other than rodents 50 ha or less were based on a cost per hectare. Costs for rodent project 50 ha or less were assigned a flat cost of $50,000. All other costs (planning, non-target mitigation, social) were calculated as a percentage of implementation cost (Table 4). The higher cost of planning for rodent projects reflects the increased up-front planning required. Ungulate and predator projects allow for adaptive management during implementation and do not require as much up front planning (Campbell & Donlan 2005, Campbell *et al.* 2011).

To address social costs, 100% of implementation costs for rodent projects and 10% of implementation costs for ungulate and predator projects were added to islands with known human populations of more than 10 people that were not part of a military camp or research station. The high cost for rodent projects reflects the increased effort associated with engaging island residents and addressing concerns related to the use of a rodenticide on their island. Isolation costs (distance

of island to nearest port) are known to be a factor in budgeting IAV eradication projects. The costing model suggested that islands with an isolation of more than 1,000 km from the nearest large port incur a cost of about 35% of the implementation costs. However, isolation information has not been calculated for the islands in the TIB database and therefore could not be added to individual islands that meet this characteristic. Therefore, 11.6% (1/3 of 35%) was added to all implementation costs. This accounts for one in every three islands incurring isolation costs. Similarly, non-target mitigation costs for eradication projects are known to be a factor in budgeting IAV eradication costs. However, it was not possible to identify what islands would require non-target mitigation because the TIB does not provide detailed information on all native species on each island. Therefore, the cost model assigned a non-target mitigation cost to all projects under the assumption that, on average, the amount budgeted would cover mitigation costs for those projects where it is required. The method in which both isolation and mitigation costs were assessed reinforces that the costing model is designed to provide average cost estimates across multiple islands, rather than accurate costs for each individual island.

Table 4. Costing model parameters.

Project Type	**Costs (% of implementation cost)**			
	Planning	**Isolation**	**Non-target**	**Social**
Rodent projects <50 ha	100%	No costs added	15%	100%, $4M cap applied
Rodent projects >50 ha<10,000ha	50%			
Rodent projects >10,000ha	Planning cost cap of $2M applied			
Ungulate projects	10%		5%	100%
Predator projects	10%		5%	10%

The TIB database summarizes information on all IUCN CR and EN vertebrates that occur on islands. This information was filtered to provide a list of islands where IUCN CR and EN species are threatened by IAV and that are appropriate candidates for the eradication of those IAV. This included all islands less than 50,000 ha and with fewer than 1,000 inhabitants. Eradications on islands larger than 50,000 ha and with more than 1,000 inhabitants is definitely possible. However, identifying for which islands this holds true is challenging without on the ground information. Thus, to be conservative in costing this approach, these larger island opportunities were left out of the costing effort. Using the filters of less than 50,000 ha and less than 1,000 inhabitants, the TIB identified 496 islands that support breeding habitat for 210 CR or EN species, representing 38% of all islands holding CR or EN species. The eradication of IAV's from these islands would provide protection for 19% of insular CR and EN species. These islands occurred within 68 ISO Country codes (including countries and external territories such as French Polynesia).

The total cost of removing invasive alien vertebrates from these islands was estimated to be **approximately $1,411 million**, which if spread over the period from now to 2020 would require an outlay of *at least* $176 million per annum (assuming delays will incur additional costs). A breakdown of the costs for the top 20 countries is shown in Table 5 If eradication expenditure is taken as an

upfront cost, it is assumed that future expenses will be limited to far less costly monitoring and preventative actions.

Table 5. A breakdown of total cost, number of threatened species, number of islands and total area for the top 20 countries with the highest number of threatened species. Values in US$ millions.

Country	Number of threatened species	Number of Islands	Total area of islands (km^2)	Estimated cost (US$)
Indonesia	15	24	1907.97	132.8
Australia	11	17	1192.66	113.2
French Polynesia	11	19	512.72	92.0
India	6	15	997.33	88.8
Phillipines	14	17	575.10	82.4
United States	9	18	948.24	80.4
New Zealand	12	9	1214.70	80.2
Bahamas	9	22	486.09	54.0
Mexico	13	12	692.68	52.6
Fiji	4	16	291.44	44.1
Japan	10	29	148.09	40.6
MP	8	9	176.85	29.4
Chile	5	5	163.64	27.6
Falkland Islands	1	6	220.30	26.7
South Africa	4	18	350.37	26.3
Greece	2	22	90.56	26.2
Ecuador	4	3	185.32	25.9
Saint Helena	6	2	180.51	23.6
Vanuatu	1	1	158.52	22.4
Virgin Islands (British)	3	10	69.48	21.3

The total amount of threatened biodiversity that would be protected by the eradication of damaging IAV on all 496 islands in this analysis is unknown because a complete list of the native species on these islands is not available. However, it is possible to review a few select islands in more detail. For example, Floreana Island, Galapagos, has 43 IUCN threatened species and the costing model predicts it will cost $22 million to remove damaging IAV. This provides a cost of roughly $0.5 Million per IUCN threatened species protected. The Juan Fernandez archipelago, Chile, supports 42 IUCN threatened species and the costing model predicts it will cost $16 Million to remove IAV from the archipelago. This equates to a cost of $0.4 Million per IUCN threatened species protected.

7.1 Costs of prevention/invasion pathway management

7.1.1 Government expenditure on biosecurity

Great Britain, Australia, New Zealand and the United States have completed nationwide studies addressing the biosecurity costs incurred in dealing with IAS and have published their biosecurity budgets. Other countries, such as South Africa, have budgeted for some biosecurity measures, with most of the current emphasis being on protection of agriculture, forestry and fisheries. These budgets give an indication of current funding and are unlikely to equate to required funding. Indeed, both Australia and New Zealand believe that they need to increase their national budgets by 9- to 25-fold in order to significantly reduce the threat (ISC 2012). As with research, biosecurity efforts and resources also tend to target industrial rather than environmental biosecurity.

In **Great Britain** (England, Scotland and Wales) the government spends some GBP17.8 million per annum (equivalent to US$30 million in 2012) on quarantine & surveillance (Williams *et al.* 2010). These costs are covered primarily by the Department of Forestry and Rural Affairs (DEFRA) (GBP15.8 million) but also the Forestry Commission (GBP1.9 million). These agencies focus on animal and plant pests. The costs associated with organisms imported for research purposes are excluded from this, as these are incurred privately, by the person or organization responsible.

The government of **Australia** has invested more than AU$1.6 billion since 2009 towards a reliable biosecurity system (Department of Agriculture, Forestry and Fisheries 2012). This allows for continued protection of human health and biodiversity, while minimizing trade restrictions on exports. In 2012, Australia committed AU$379.9 million (US$391.3m) over seven years to construct a new quarantine facility for high risk imports, AU$19.8 million (US$20.4m) over three years to support information and communication technology systems, and AU$124.5 million (US$128.2m) over four years towards biosecurity operation at airports and mail centres nationwide. From these data we have assumed an initial investment of US$412 million and annual operating costs of some US$95 million per year.

In **New Zealand**, the Ministry of Primary Industries is responsible for biosecurity, for which a budget of NZ$136.505 million (US$108 million) was allocated for 2012-2013. This included border biosecurity monitoring and clearance (i.e. managing the biosecurity risk associated with international trade and travel), border biosecurity systems development and maintenance (the development and maintenance of standards and systems that manage biosecurity risk associated with imports and exports), and implementation and development of biosecurity policy advice and domestic biosecurity surveillance (NZ MPI 2012).

In **South Africa**, the South African National Biodiversity Institute (SANBI) runs government-funded programmes into IAS prevention (including surveillance, the development of a DNA Bar-coding system to identify and verify the presence of IAS and an advocacy and awareness programme). The annual budget allocated for prevention in 2012-2013 was estimated at ZAR17.4 million (US$2 million), but is probably less than a fifth of what it should be.

In the **United States**, the National Invasive Species Council summarized the funding made available from several government departments (including the Department of the Interior,

Department of Agriculture, Department of Commerce, Department of State Department of Defense, Department of Homeland Security, Department of Transportation and the U.S. Agency for International Development) for the fiscal year of 2012, a total of US$338 million on IAS prevention measures, of which US$95 million was for education and public awareness and US$62 million was for leadership and international co-operation.

There were too few data to find any robust statistical trends in the national-level expenditure on biosecurity. One could expect the required expenditure on biosecurity to be a function of the numbers of points of entry, the volumes or values of freight or the numbers of passengers passing though these points. The value of imports is closely correlated to the value of GDP, and in fact the above figures correspond more closely to GDP than other statistics, mainly due to the high expenditure by the US. However as a percentage of GDP, expenditure by the US and UK is relatively modest (0.001-2% of GDP) compared with that of Australia (0.007%) and particularly New Zealand (0.076%). Working with PPP-adjusted estimates, the best relationship obtained from this very small sample was with the total number of IAS recorded ($\ln Cost_{PPP} = 0.0085xIAS + 1.3437$, $R^2 = 0.799$, $P<0.05$). This is reasonable if current IAS numbers can be an assumed correlate of future threat. Based on this relationship, some **US$2,065 million should be spent annually** on prevention measures globally. Based on information from Australia, the initial set up costs (constructing facilities, setting up databases and technology systems etc) could amount to four times the operating costs, or about US$8,260 million. While the emphasis to date has been on the prevention of IAS that threaten industry, it would be reasonable to assume that the measures that need to be taken to prevent IAS that may threaten biodiversity would amount to at least two thirds of the total, due to the overlap in interests.

7.1.2 Private sector costs

Prevention costs are transferred to the private sector by means of legislation concerning imports and safety precautions that have to be taken in transportation, and these costs can be substantial. In **Great Britain**, commercial shipping vessels have various legal requirements they must comply with regarding hull maintenance. Ferries are required to dry dock annually for a hull inspection and clean while tankers dry dock every three to five years. These cleans are required for safety reasons and also to rid the vessels of biofouling, a known vector for marine IAS (Eno 1996, Gollasch 2002, Drake & Lodge 2007). These activities cost British ships some GBP31.4 million per annum (Williams *et al.* 2010). However we have not included this cost in our analysis since antifouling measures are incurred as a necessary form of ship maintenance already.

International legislation will soon require a substantial increase in the private expenditure required to combat the introduction of IAS in ballast water discharged from ocean-going commercial vessels. In 2004, the International Maritime Organisation (IMO) adopted the **International Convention for the Control and Management of Ships Ballast Water & Sediments (BWM Convention).** The BWM Convention is not yet in force. The Convention, will enter into force 12 months after ratification by 30 States, representing 35% of world merchant shipping tonnage. As of the 31st May 2012, there were 35 countries bound by the treaty, but these represent only 28% of the worlds shipping tonnage. Therefore, this convention will not be enforced until the support is more representative of the major powers in the shipping industry. When the BWM Convention comes

into force, it will require that on-board ballast water treatment systems are fitted to vessels. Any ocean-going cargo vessel built before 2009 must be retrofitted with a ballast water treatment system by 2014/2016 (depending on vessel size). In 2011, the world's cargo carrying fleet was estimated at 55,138 ships of 991 billion Gross Tonnage and 1,483 billion Dry Weight (Maritime Knowledge Centre 2012), and some 3,445 new merchant ships are built per year (Williams *et al.* 2010).

Ballast water treatment systems cost from US$380,000 for a 200 m^3/hr plant to US$875,000 for a 2000 m^3/hr plant in 2007 (Williams *et al.* 2010), with annual operating costs ranging from US$0.009 – 0.296 million per vessel (King *et al.* 2010). Continued technological development and the "learning curve ratio" (the price reduction per doubling of capacity), as well as economies of scale that will come into play once the installment of these systems becomes mandatory, can be expected to lead to a drop in price. For example, as wind energy expands in the UK in accordance with national policy, prices have been projected to drop by 55-70% over the period 2002-20 (Milborrow 2002, Dale *et al.*2004). Given the shorter period and the smaller market, we estimated a drop in the above prices of 45% for the period considered. Thus the total cost for the existing global fleet to be fitted with systems could be in the range of US$11,524 to US$26,535 million (average US$19,030 million). The annual maintenance of these systems, plus the annual installation of new systems on newly built ships would amount to US$993 to US$10,634 million per annum (average US$5814 million per annum).

The above costs all pertain to the marine transport system. It is reasonable to expect that depending on the legislation existing in different countries, some regulations will also affect road and air transport systems. There is currently no information available on the extent of this (and the associated costs).

7.1.3 Building capacity and legal frameworks

There have been several internationally-funded efforts recently to help developing countries to build their capacity in order to carry out effective biosecurity programmes. For example, the Global Invasive Species Programme (GISP) spent an average of US$50 000 on legislative projects in each of **Kenya, Senegal, Uganda, Mozambique**. These were funded primarily by the World Bank BNPP, but also UNEP-Regional Seas. The projects entailed the building of legal and institutional capacity, the development of legal and institutional frameworks for IAS (with workshops related to this in developing countries) and the development of marine biofouling guidelines. Several GEF-funded projects have been carried out in developing countries which have concentrated on strengthening capacity to control the introduction and spread of alien invasive species. Average expenditure in **Sri Lanka, Cameroon**, **Cuba, Argentina, Mexico, Chile, Indonesia, Vietnam, Philippines** and **Cambodia** was $10.9 million per country.

Similarly, the Global Ballast Water Management Programme (GloBallast) funded by GEF/UNDP/IMO is assisting developing countries to reduce the transfer of harmful aquatic organisms and pathogens in ships' ballast water, implement the IMO ballast water guidelines and prepare for the new BWM convention. Six developing countries (**Brazil, China, India, Iran, South Africa** and **Ukraine**) were selected to pilot this project which was initiated in 2000 and the final

phase is expected to be completed in 2014 with a projected total cost of US$44.3 million (in 2012 values) for the six countries, equating to US$7.4 million per country on average.

Working from the higher estimate for the broader programmes described above, this could be extrapolated to a global cost of US$1,564 million for all low to middle income countries.

7.2 Summary of global estimates

A summary of the above estimates is presented in Table 6. An estimated total initial investment of some US$44,000 million is required for enabling effective prevention systems and eradicating IAS in situations where this is possible (i.e. mainly from islands). Almost half of this cost is to be borne by the private sector (principally the commercial shipping industry with the installation of ballast water treatment systems). Recurrent expenditures from 2012 to 2020 amount to approximately $380 000 million, with control costs being counted from 2012, others counted from 2015 after initial investments are complete. Control costs account for the bulk of recurrent expenditure, amounting to almost $40,000 per year. In contrast, implementing eradications of the most damaging invasive alien vertebrates on small and medium sized islands is estimated at approximately $1,411 million as a once-off investment. Thus the total costs of reaching the Aichi Target 9 to achieve a measurable reduction in the prevalence of and damage caused by IAS by 2020 amounts to just over $423,500 million. Excluding the costs of dealing with industrial pests, and focusing on IAS that threaten biodiversity only, the corresponding costs would probably amount to some $186,200 million. However, the separation of these activities is difficult and is based on some crude assumptions.

Table 6. Summary of estimated costs of control, eradication and prevention measures. Costs are estimated (a)without (Sc1) and (b) with the costs of addressing IAS that affect agriculture and forestry (Sc2).

Measures	Initial Investment 2012-14 (US$ millions)		Recurrent expenditures (US$ millions per year)		Recurrent expenditures (US$ millions total for period)	
	Sc1	Sc2	Sc1	Sc2	Sc1	Sc2
Research and prioritisation						
Baseline surveys, research and prioritisation	7,500	15,000	1,000	3,000	6,000	18,000
Control and eradication						
Control of mainland IAS	-	-	13,046	39,533	104,366	316,262
Eradication of priority IAS on islands	1,411	1,411	-	-	-	-
Prevention						
Developing capacity and legal frameworks	1,564	1,564	-	-	-	-
Biosecurity measures	4,581	6,940	1,145	1,735	6,871	10,411
Ballast water treatment (private)	19,030	19,030	5,814	5,814	34,884	34,884
TOTAL	**34,086**	**43,945**	**21,005**	**50,082**	**152,121**	**379,557**

8 Discussion

Along with the growing interest and awareness in the economic value of biodiversity and ecosystem services during the last two decades, there been a growing interest in the costs of environmental degradation as a result of a range of threats, one of which is IAS. Thus much of the initial focus on economic aspects of IAS has concentrated on valuing the losses in production as a result of IAS. In particular, interest has centred on the IAS that impact on production in the primary sectors - agriculture, forestry and fisheries, species that affect human health, and those that incur significant management costs, such as the clearing of industrial water intake pipes fouled by invasive mussels. Those whose main impact is on biodiversity have received less attention, with the exception of IAS on islands. Different studies estimating the economic impacts of IAS use a variety of approaches, and whether a study includes non-market impacts has a significant bearing on value estimates (Williams *et al.* 2010). Compounding the matter is the fact that not all studies provide descriptions of what types of costs were considered or how the estimates were derived, and there is a need for standardisation of these estimates (Simberloff 2004). Nevertheless, since non-market costs are difficult to estimate, most estimates tend not to include these. Similarly, studies vary in whether they describe direct impacts on economic output or total economic impacts which include multiplier effects. Obtaining data on government expenditure on IAS management (especially budgets allocated to prevention and research) was one of the more challenging aspects of this study. CBD National Biodiversity Strategy and Action Plans should strive to be more informative regarding the biodiversity issues which are addressed, including information on funding allocations. Thus achieving an estimate of global damages is fraught with difficulty, and estimation of prevention and control costs is equally tricky.

We have presented a rapid estimate of the global cost of achieving a significant target in the reduction of IAS, based on limited data and using statistical analysis as far as possible, but relying on a number of assumptions. In an analysis of the IAS management costs from 16 countries (many of these partial estimates), Williams *et al.* (2010) failed to find any consistent trends in terms of factors such as country size and GDP influencing overall costs. The estimates used in the study were more comparable in terms of the types of IAS considered and the level of coverage for each country, but the relationships were still based on few data. An exception was that in the case of estimating the cost of eradication of IAS on islands, data have been collected in a more standardised way, so that a more robust model could be developed from which to estimate future costs.

Our estimates are incomplete in that not all costs have been fully taken into account, and our assumptions are conservative. However, much of this cost is borne by high income countries such as the United States, which are already incurring a substantial proportion of the public expenditures required. Our estimates should not be projected beyond 2020, since the efforts to make a substantial reductions in existing IAS populations, reduced rates of introductions and increased vigilance during this period should, together with technological improvements, lead to lower costs of control in future years.

Indeed, estimating the cost of dealing with IAS is also a dynamic problem, in that actions in any given year will affect costs in subsequent years. These dynamics are driven by the biology of the IAS themselves, which vary considerably taxonomically and geographically. Without incorporating such dynamics into the analysis, our estimates remain crude, but are likely to be in the right order of

magnitude for the period under consideration. Our estimate is a preliminary first estimate, and much more could be done with existing information given sufficient time, with the combination of geographic information systems and bio-economic modelling. The quality of information produced from several years of research on IAS in Great Britain (Williams *et al.* 2010), as well as the recent UK National Ecosystem Assessment (2012), is testament to the value of investing in this kind of research. Future research should concentrate on producing more detailed estimates of damage costs and the research, control and prevention costs required for a larger sample of countries.

Taking indirect impacts (multiplier effects) into account, the total economic cost of damage caused by invasive species has been estimated as being about 5% of global GDP (Pimentel *et al.* 2001), which equates to some US$6.5 trillion per annum. A more conservative estimate of damages of about 2% of GDP (e.g. based on Xu *et al.* 2006 - China, MAF 2009 – New Zealand) would equate to about US$2.6 trillion. The recurrent expenditure of $50 billion estimated in this study is equivalent to less than 2% of the more conservative estimate of damage costs.

Is this level of investment justified? Certainly in the case of islands, up-front investment in the form of eradication is justified in that it obviates future costs and timely and complete action will make a critical difference to endangered species. For mainland infestations, control will only be achieved if enough resources are devoted to the problem. Thus it is necessary to focus on priority species and priority areas in order to be more efficient (van Wilgen *et al.* 2012). Inefficient, unfocussed expenditure will be money wasted. There is considerable evidence from both models and empirical data of the cost savings achieved by early action. An analysis of five IAS in Great Britain (Asian long-horned beetle, carpet sea squirt, water primrose, grey squirrel and coypu) showed that costs of control increase exponentially as an invasion progresses, demonstrating that early intervention is worthwhile (Williams *et al.* 2010). Thus, as with any form of restoration or mitigation expenditure, the expectation is that heavy investment in this period will pay off in future periods in terms of damage costs avoided. From an economic perspective, the level of investment required in each period is an optimisation problem which should take the rate of time preference into consideration through discounting costs and benefits in future time periods (Turpie *et al.* 2004). If studies continue to focus on the impacts of IAS on economic sectors such as agriculture, then such an analysis might not necessarily yield the level of control of all groups of IAS required by the Aichi Targets by 2020. However, better understanding of the losses of ecosystem services and their intangible and non-market values is likely to justify greater investment in some types of IAS that receive less attention. For example, Nunes *et al.* (2004) used stated preference methods to elicit public willingness to pay for a hypothetical marine protection programme on the North Holland coast which is subject to harmful algal blooms. These IAS affect marine health and recreational activities on beaches, to an extent that beaches are often closed to members of the public. The public's willingness to pay justified an expenditure of EUR225-326 million (US$318-460 million) on the installation of a local ballast water disposal facility.

Even with sound economic justification, meeting this target will require significant financial resources that might be difficult to access in some situations. While IAS externalities associated with the transport sector can be internalized to some degree, such that the private sector carries much of these costs, for the most part, funding the management of IAS is a public sector problem, especially when it comes to the control or eradication of existing IAS, including where they occur on private lands (Turpie & Heydenrych 2000). However, through presentation of IAS damage costs and

calculation of cost-benefit ratios, this expenditure can be justified, especially given that the problem will only escalate without management. Governments must be encouraged to allocate a proportion of their national budgets towards biosecurity (pertaining to industrial and environmental threats). This may be a daunting task in low to middle income countries, however IAS management can also lead to poverty alleviation and job creation. South Africa's Working for Water programme is an excellent example of this, creating 371,587 jobs in previously disadvantaged communities (G. Preston, DEA Working for Water programme, pers. comm.), focussing on the employment of women in particular.

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10 Bibliography

Aguirre-Munuoz, A., Croll, D.A., Donlan, C.J., Henry III, R.W., Hermosillo, M.A., Howald, G.R., Keitt, B.S., Luna-Mendoza, L., Rodreguez-Malago, M., Salas-Flores, L.M., Samaniego-Herrera, A., Sanchez-Pacheco, J.A., Sheppard, J.B., Tershy, R., Toro-Benito, J., Wolf, S. & Wood, B. 2007. High-impact conservation action: a case study from the islands of western Mexico. Ambio **37**: 101-107.

Albuquerque, F.S., Peso-Aguiar, M.C. & Assunção-Albuquerque, M.J.T. 2008. Distribution, feeding behavior and control strategies of the exotic land snail *Achatina fulica* (Gastropoda: Pulmonata) in the northeast of Brazil. Brazilian Journal of Biology **68**: 837-842.

Alcover, J.A., Campillo, X., Marcias, M. & Sans, A. 1998. Mammal species of the world: additional data on insular mammals. American Museum Novitates **3248:** 1–29.

Atkinson, I.A.E. 1989. Introduced animals and extinctions. Pages 54–75 in Western, D. & Pearl, M.C. (eds) Conservation for the twenty first century. Oxford University Press, New York.

Bax, N., Williamson, A., Aguero, M., Gonzalez, E. & Geeves, W. 2003. Marine invasive alien species: a threat to global biodiversity. Marine Policy **27**: 313–323.

Beale, R., Fairbrother, J., Inglis, A. & Trebeck, D. 2008. One Biosecurity: A working partnership. The independent Review of Australia's Quarantine and Biosecurity Arrangements, Report to the Australian Government.

Blackburn, T.M., Cassey, P., Duncan, R.P., Evans, K.L. & Gaston, K.J. 2004. Avian extinction and mammalian introductions on oceanic islands. Science **305**: 1955–1958.

Britton, J.R. & Brazier, M. 2006. Eradicating the invasive topmouth gudgeon, *Pseudorasbora parva*, from a recreational fishery in northern England. Fisheries Management and Ecology **13**: 329–335.

Butchart, S.H.M., Resit Akcakaya, H.R., Chanson, J., Baillie, J.E.M., Collen, B., Quader, S., Turner, W.R., Amin, R., Stuart, S.N. & Hilton-Taylor, C. 2007. Improvements to the Red List Index. PLoS ONE 2(1): e140. doi:10.1371/journal.pone.0000140

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E.C., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Hernández Morcillo, M., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J. & Watson, R.. 2010. Global biodiversity: indicators of recent declines. Science **328**: 1164–1168.

CAB International 2004. Prevention and management of alien invasive species: forging cooperation throughout West Africa. Proceedings of a workshop held in Accra, Ghana, 9-11 March, 2004. CAB International, Nairobi, Kenya.

Campbell, K. & Donlan C.J. 2005. Feral goat eradications on islands. Conservation Biology **19:** 1362–1374.

Campbell, K. J. Harper, G. Algar, D. Hanson, C. C. Keitt B. S. & Robinson S. 2011. Review of feral cat eradications on islands. Pages 37-46 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). Island invasives: eradication and management. IUCN, Gland, Switzerland.

Chenje, M. & Mohamed-Katerere, J. 2006. Invasive alien species. Pages 331-347 in Africa Environment Outlook 2. United Nations Environment Program.

City of Portland. 2008. City of Portland invasive plant strategy in response to resolution 36360. http://www.portlandonline.com/bes/index.cfm?c=47815

Clavero, M. & Garcı´a-Berthou, E. 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology and Evolution **20**: 110.

Cofrancesco, A.F.Jr., Reaves, D.R. & Averett, D.E. 2007.transfer of invasive species associated with the movement of military equipment and personnel. Report prepared by the US Army Corps of Engineers® Engineer Research and Development Centre.

Colautti, R.I., Bailey, S.A., van Overdijk, C.D.A., Amundsen, K. & MacIsaac, H.J., 2006. Characterised and projected costs of nonindigenous species in Canada. Biological Invasions **8**: 45–59.

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

Dale, L., Milborrow, D., Slark, R. & Strback, G. 2004. Total cost estimates for large-scale wind scenarios in UK. Energy Policy 32: 1949–1956.

Department of Agriculture, Forestry and Fisheries, Australian Government Biosecurity budget announcements 8 May 2012. http://www.daff.gov.au/bsg/biosecurity-reform/budget-announcements/2012-13-budget

DEFRA 2008. The invasive non-native species framework strategy for Great Britain. Department for Environment, Food and Rural Affairs, London, UK.

De Groote, H., Ajuonu, O., Attignon, S., Djessou, R. & Neuenschwander, P. 2003. Economic impact of biological control of water hyacinth in Southern Benin. Ecological Economics **45**: 105-117.

Ding, J., Mack, R.N., Lu, P., Ren, M. & Huang, H. 2008, China's booming economy is sparking and accelerating biological Invasions. BioScience **58**: 317-324.

Doeleman, J.A. 1989. Biological control of *Sylvinia molesta* in Sri Lanka: An assessment of cost and benefits. Australian Centre for International Agriculture Research: Canberra.

Dukes, J.S. & Mooney, H.A. 1999. Does global change increase the success of biological invaders? Trends in Ecology & Evolution **14**: 135-139.

Emerton, L. & Howard, G. 2008, A toolkit for the economic analysis of invasive species.Global Invasive Species Programme, Nairobi.

Engeman, R.M., Smith, H.T., Severson, R., Severson, M.A., Woolard, J., Shwiff, S., Constantin, B. & Griffin, D. 2004. Damage reduction estimates and benefit-cost ratios for feral swine control from the last remnant of a basin marsh system in Florida. USDA National Wildlife Research Center - Staff Publications. Paper 334.

Eplee, R. E. 1992. Witchweed (*Striga asiatica*): an overview of management strategies in the USA. Crop Protection **11:** 3-7.

Essl, F. & Rabitsch, W. 2002. Neobiota in Österreich. Umweltbundesamt, 432 pp.

Essl, F. & Rabitsch, W. 2004. Austrian Action Plan on Invasive Alien Species. Produced for the Federal Ministry of Agriculture, Forestry, Environment and Water Management. Stubenbastei 5, 1010 Vienna, Austria.

Fairbrother, A. & Bennet, R.S. 1999. Ecological risk assessment and the precautionary principle. Human and Ecological Risk Assessment **5**: 943-949.

Fish Management Chemicals Subcommittee 2010. Maintaining North America's healthy native aquatic ecosystems: Rotenones role in eradicating invasive fishes, parasites and diseases.

Frid, L., Knowler, D., Murray, C., Myers, J. & Scott, L. 2009. Economic Impacts of Invasive Plants in BC. Prepared for the Invasive Plant Council of British Columbia by ESSA Technologies Ltd., Vancouver, BC. 105 pp.

Frost & Sullivan. 2010. Global Ballast Water Treatment Systems Markets. Report M494-15 (March 2010).

Gherardi, F. & Angiolini, C. 2004. Eradication and control of invasive species, in Biodiversity conservation and habitat management, [Eds. Gherardi, F., Gualtieri, M. & Corti, C.], in Encyclopedia of Life Support Systems (EOLSS), Developed under the Auspices of the UNESCO, Eolss Publishers, Oxford ,UK [http://www.eolss.net].

GEF 2010. Strengthening capacity to control the introduction and spread of alien invasive species in Sri Lanka. (GEFSEC project ID: 2472)

GEF 2010. Development and institution of a national monitoring and control system (Framework) for living modified organisms and Invasive Alien Species (IAS). (GEFSEC Project ID: 3561)

GEF 2011. Enhancing the prevention, control and management of invasive alien species in vulnerable ecosystems in Cuba. (GEF project ID: 3955)

GEO BON. 2011. Target 9 – Control of invasive alien species. In: Adequacy of Biodiversity Observation Systems to support the CBD 2020 Targets. Report prepared by the *Group on Earth Observation Biodiversity Observation Network (GEO BON), the IUCN and the World Conservation Monitoring Centre, for the Convention on Biological Diversity*, Pretoria, South Africa, pp. 33-35.

GISP 2005. Matthews S. & Brandt K. (Eds) South America Invaded: the growing danger of invasive alien species. Global Invasive Species Programme.

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

Global Invasive Species Programme (GISP) 2007. Invasive species and poverty: Exploring the links.

Keller, R.P. & Perrings, C. 2010. Ecosystem services economics international policy options to reduce the harmful impacts of alien invasive species. UNEP.

Goswami, S.L. 2009. Biosecurity Economics: Conflicting results in evaluation criteria. Contributed paper presented at the 53rd Annual Conference of the Australian Agricultural and Resource Economics Society, Cairns, February 11th – 13th, 2009.

Gren, I.M., Isaacs, L. & Carlsson, M. 2009. Costs of alien invasive species in Sweden. Ambio **38**:135–140.

Howald, G., Donlan, C. J., Galván, J. P., Russell, J. C., and Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal,M., Saunders, A. & Tershy, B. 2007. Invasive rodent eradication on islands. Conservation Biology **21**: 1258–1268.

Hulme, P.E. 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. Journal of Applied Ecology **46**: 10–18.

Jones, H. P., Tershy, B. R., Zavaleta, E. S., Croll, D. A., Keitt, B. S., Finkelstein, M. E. & Howald, G. R. 2008. Severity of the effects of invasive rats on seabirds: a global review. Conservation Biology 22: 16–26.

Karatayev A.Y., Mastitsky S.E., Burlakova L.E. & Olenin, S. 2008. Past, current and future of the central European corridor for aquatic invasions in Belarus. Biological Invasions **10**: 215-232.

Keitt, B.S., Campbell, K.J., Saunders, A., Clout, M., Wang, Y., Heinz, R., Newton, K. & Tershy, B.R. 2011. The Global Islands Invasive Vertebrate Eradication Database: A tool to improve and facilitate restoration of island ecosystems. In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). *Island invasives: eradication and management*, pp. 74-77. IUCN, Gland, Switzerland.

Keller, R.P., Drake, J.M., Drew, M.B. & Lodge, D.M. 2011. Linking environmental conditions and ship movements to estimate invasive species transport across the global shipping network. Diversity and Distributions **17**: 93–102.

King, D.M., Riggio, M. & Hagan, P.T. 2010. Preliminary overview of global ballast water treatment markets. Maritime Environmental Resource Centre (MERC) Ballast water Economics Discussion Paper No. 2.

Knowler, D. & Barbier, E. 2000. The economics of an invading species: a theoretical model and case study application. pp 70-93. In: Perrings, C., M.Williamson, M. & S. Dalmazzone (eds). The Economics of Biological Invasions. Edward Elgar, Cheltenham, U.K.

Knowler, D., 2005. Re-assessing the costs of biological invasion: *Mnemiopsis leidyi* in the Black Sea. Ecological Economics **52**: 187-199.

Kroeger, T. 2007. Economic impacts of live wild animal imports in the United States. Prepared for Defenders of Wildlife.

Leavold, V. Lloyd, L. & Lepetit, J. 2007a. Development of case studies on the economic impacts of invasive species in Africa - *Mimosa pigra*. Global Invasive Species Program February 2007.

Leavold, V. Lloyd, L. & Lepetit, J. 2007b. Development of case studies on the economic impacts of invasive species in Africa – *Salvinia molesta* in Senegal. Global Invasive Species Program February 2007.

Leistritz, F.L.,Thompson, F. & Leitch, J.A. 1992. Economic impact of leafy spurge (*Euphorbia esula*) in North Dakota. Weed Science **40**: 275-280.

Leung, B., Lodge, D.M., Finnoff, D., Shogren, J.F., Lewis, M.A. & Lamberti, G. 2002. An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. Proceedings of the Royal Society of London B **269**: 2407-2413.

Locke, A. & Hanson, J.M. 2009. Rapid response to non-indigenous species. 1. Goals and history of rapid response in the marine environment. Aquatic Invasions **4**: 237-247.

Lupi, F. Hoehn, J & Christie, G. 1999. Valuing non indigenous species control and native species restoration in Lake Huron. Benefits and costs in natural resource planning. (W. Douglas Shaw ed.) Western Regional Research Publication.

Lupi, F., Hoehn, J. & Christie, G. 2003. Using an economic model of recreational fishing to evaluate the benefits of sea lamprey control on the St. Mary's river. Journal of Great Lakes Research **29**: 742-754.

Maritime Knowledge Centre. 2012. International shipping facts and figures – information resources on trade, safety, security, environment. Report prepared for the International Maritime Organisation (IMO) March 2012.

Martins, T.L.F., Brooke, M.D.L., Hilton, G.M., Farnsworth, S., Gould J., & Pain, D.J. 2006. Costing eradication of alien mammals from islands. Animal Conservation **9:** 439–444.

McEnnulty, F.R., Bax, N.J., Schaffelke, B. & Campbell, M.L. 2001. A review of rapid response options for the control of ABWMAC listed introduced marine pest species and related taxa in Australian waters. Centre for Research on Introduced Marine Pests Technical Report 23. CSIRO Marine Research, Hobart, Australia.

Macdonald, I.A.W., Reaser, J.K., Bright, C., Neville, L.E., Howard, G.W., Murphy, S.J. & Preston, G. (eds.). 2003. Invasive alien species in southern Africa: national reports & directory of resources. Global Invasive Species Programme, Cape Town, South Africa.

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A. Chanson, J. & Hoffmann, M. 2010. Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. Diversity and Distributions **16**: 95–108.

McGeoch, M.A., Chown, S.L. & Kalwij, J.M. 2006. A global indicator for biological invasion. Conservation Biology **20**: 1635–1646.

McGeoch, M.A., Spear, D., Kleynhans, E.J. & Marais, E. 2012. Uncertainty in invasive alien species listing. Ecological Applications **22**: 959–971.

McLeod, R. 2004. Counting the Cost: Impact of Invasive Animals in Australia, 2004. Cooperative Research Centre for Pest Animal Control. Canberra.

McNeely, J. 2001 a. Global strategy for addressing the problem of invasive alien species. A result of the Global Invasive Species Programme (GISP). 63 pp.

McNeely, J. 2001 b. Invasive species: a costly catastrophe for native biodiversity Land Use and Water Resources Research **1**: 1–10.

Milborrow, D.J., 2002. Will downward trends in wind prices continue? Windstats Newsletter 15 (3), 1–3.

Mwebaze, P., MacLeod A., Tomlinson, D., Barois, H. & Rijpma, J. 2010. Economic valuation of the influence of invasive alien species on the economy of the Seychelles islands. Ecological Economics **69**: 2614–2623.

Myers, J.H., Simberloff, D., Kuris A.M. & Carey, J.R. 2000. Eradication revisited: dealing with exotic species. Trends in Ecology and Evolution **15**: 316-320.

National Invasive Species Council (NISC). 2003. General guidelines for the establishment and evaluation of invasive species Early Detection and Rapid Response Systems. Version 1. 16 pp.

Nimmo-Bell. 2009. Economic costs of pests to New Zealand. A Report prepared by Nimmo-Bell for MAF Biosecurity New Zealand. MAF Biosecurity New Zealand Technical Paper No: 2009/31.

NISC 2005. Focus group conference report and pathways ranking guide, June 21-22, 2005. Washington, DC.

NISC 2012. National Invasive Species Council, Invasive Species Interagency Crosscut Budget. March 14, 2012. U.S. Department of the Interior.

Nogales, M., Martin, A., Tershy, B.R., Donlan, C.J., Witch, D., Puerta, N.,Wood, B. & Alonso, J. 2004. A review of feral cat eradication on islands. Conservation Biology **18**: 310–319.

NEANS Northeastern Aquatic Nuisance Species Panel. 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. 24 pp.

Nunes, P.A.L.D. & van den Bergh, J.C.J.M. 2004. Can people value protection against invasive marine species? evidence from a joint TC–CV survey in the Netherlands. Environmental & Resource Economics **28**: 517–532.

NZ MPI 2012. New Zealand Ministry of Primary Industries Estimate of Appropriations 2012/2013.

OECD Organisation for Economic Co-operation and Development. 1993. OECD core set of indicators for environmental performance reviews. OECD Environment Monographs No. 83. OECD. Paris.

Office of Technology Assessment. U.S. Congress (OTA). 1993. Harmful non-indigenous species in the United States. OTA Publication OTA-F-565. US Government Printing Office, Washington D.C.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. & Bodey, T.W. 2011. Eradication of invasive mammals on islands inhabited by humans and domestic animals. Conservation Biology, 25: 232–240

Panzacchi, M., Bertolino, S., Cocchi, R. & Genovesi, P. 2007: Population control of coypu *Myocastor coypus* in Italy compared to eradication in UK: a cost-benefit analysis. - Wildlife Biology **13**: 159-171.

Parks and Wildlife Service. 2007. Plan for the eradication of rabbits and rodents on Subantarctic Macquarie Island.

Perrings, C., Williamson, M. & Dalmazzone, S. (eds). 2000. The economics of biological invasions. Edward Elgar, Cheltenham, Gloscester, UK.

Perrings, C., Williamson, M., Barbier, E.B., Delfino, D., Dalmazzone, S., Shogren, J., Simmons, P. & Watkinson, A., 2002. Biological invasion risks and the public good: an economic perspective. Conservation Ecology **6**, 1. <http://www.consecol.org/vol6/iss1/art1>.

Perrings, C. 2005. The socioeconomic links between invasive alien species and poverty. Report to the Global Invasive Species Program.

Pfenninger, M. & Schwenk, K. 2007. Cryptic animal species are homogeneously distributed among taxa and biogeographical regions. BMC Evolutionary Biology **7**: 6 pp.

Phillips, R.A. 2010. Eradications of invasive mammals from islands: why, where, how and what next? Conservation Biology **110**: 1-8.

Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'Connell, C., Wong, E., Russel, L., Zern, J., Aquino, T. & Tsomondo, T. 2000. Economic and environmental threats of alien plant, animal, and microbe invasions. Agriculture, Ecosystems and Environment **84**: 1-20.

Pimentel, D. 2002. Biological invasions: economic and environmental cost of alien plant, animal and microbe species. CRC Press LLC, USA.

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

Radtke, H. & Davis, S. 2000. Economic analysis of containment programs, damages, and production losses from noxious weeds in Oregon. Report prepared for Oregon Department of Agriculture, plant division, noxious weed control program.

Reinhardt, F., Herle, M., Bastiansen, F. & Streit, B. 2003. Economic impact of the spread of alien species in Germany. 248 pp. English version.

Ricketts, T.H., Dinerstein, E., Boucher, T., Brooks, T.M., Butchart, S.H.M., Hoffmann, M., Lamoreux, J.F., Morrison, J., Parr, M., Pilgrim, J.D., Rodrigues, A.S.L., Sechrest, W., Wallace, G.E., Berlin, K., Bielby, J., Burgess, N.D., Church, D.R., Cox, N., Knox, D., Loucks, C., Luck, G.W., Master, L.L., Moore, R., Naidoo, R., Ridgely, R., Schatz, G.E., Shire, G., Strand, H., Wettengel, W. & Wikramanayake, E. 2005. Pinpointing and preventing imminent extinction. Proceedings of the National Academy of Science United States of America **51**: 18497–18501.

Rockwell, H.W.Jr. 2003. Summary of a Survey of the Literature on the EconomicImpact of Aquatic Weeds. Report for the Aquatic Ecosystem Restoration Foundation - August. http://www.aquatics.org/pubs/economics.htm.

Sand, P.F., Eplee, R.E. & Westbrooks, R.G. 1990. Witchweed Research and Control in the United States, Monograph Series of the Weed Science Society of America **5**: 154 pp.

Shiganova T.A. 1998. Invasion of the Black Sea by the ctenophore *Mnemiopsis leidyi* and recent changes in pelagic community structure // Fisheries Oceanography – GLOBEC Special Issue. Ed. Steve Coombs. pp 305-310.

Shine, C., Reaser, J.K. &. Gutierrez, A.T. (eds.). 2003. Invasive alien species in the Austral Pacific Region: National Reports & Directory of Resources. Global Invasive Species Programme, Cape Town, South Africa.

Simberloff, D. 2004. Community ecology: is it time to move on? American Naturalist **163**:787–799.

Stumpf, E., 1998. Post-harvest loss due to pests in dried cassava chips and comparative methods for its assessment. A case study on small-scale farm households in Ghana. GTZ. 172 pp. http://www2.gtz.de/post_harvest/documents/new_else/x5426e00.htm.

Sturtevant, R. & Cangelosi, A. 2000. The Great Lakes at the Millennium: Priorities for Fiscal 2001. Prepared for the Northeast Midwest Institute, Washington, DC.

Sun, J.F. 1994. The Evaluation of Impacts of Colonization of Zebra Mussels on the Recreational Demand in Lake Erie. Fourth International Zebra Mussel Conference, Madison, Wisconsin. March. Availability: http://www.sgnis.org/publicat/108.htm

Turpie, J.K. & Heydenrych, B.H. 2000. Economic consequences of alien infestation of the Cape Floral Kingdom's Fynbos vegetation. pp 152-182 in Perrings, C., Williamson, M. & Dalmazzone, S. (eds) The economics of biological invasions. Cheltenham, U.K.: Edward Elgar. pp. 214-261.

Turpie, J.K., Winkler, H. & Midgley, G., 2004. Economic impacts of climate change in South Africa: a preliminary analysis of unmitigated damage costs. In: Blignaut, J., de Wit, M. (Eds.), Sustainable Options. UCT Press.

USDA, 2001. Agricultural Statistics. U.S. Department of Agriculture, Washington, DC

van Wilgen, B.W., Richardson, D.M., le Maitre, D.C., Marais, C. & Magadlela, D. 2001. The economic consequences of alien plant invasions: examples of impacts and approaches to sustainable management in South Africa. Environment, Development and Sustainability **3**: 145–168.

van Wilgen, B.W., De Wit, M.P., Anderson, H.J., Le Maitre, D.C., Kotze, I.M., Ndala, S., Brown, B. & Rapholo, M.B. 2004. Costs and benefits of biological control of invasive alien plants: case studies from South Africa. South African Journal of Science **100**: 113-122.

van Wilgen, B.W., Le Maitre, D.C., Wannenburgh, A., Kotze, I.M., van den Berg, L. & Henderson L. 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. Biological Conservation: **148**: 28–38.

van Wyk, E. & van Wilgen, B. W. 2002. The cost of water hyacinth control in South Africa: a case study of three options. African Journal of Aquatic Science **27**: 141-149.

Veitch, C.R. and Clout, M.N. 2002. Turning the tide: the eradication of invasive species. Auckland, Invasive Species Specialist Group of the World Conservation Union (International Union for Conservation of Nature and Natural Resources,IUCN) (in press).

Venette, R.C. & Larson, M. 2004. Mini Risk Assessment, Giant African Snail, *Achatina fulica* Bowdich (Gastropoda: Achatinidae). Department of Entomology, University of Minnesota, St Paul (US)

Vitousek, P.M., Mooney, H.A., Lubchenco, J. & Melillo, J.M. 1997. Human domination of Earth's ecosystems. Science **277**: 249–299.

Wallace, N.M., Leitch, J.A. & Leistritz, F.L. 1992. Economic impact of leafy spurge on North Dakota wildland. North Dakota Farm Research **49**: 9-13.

Walpole, M., Almond, R.E.A., Besançon, C. Butchart, S.H.M., Campbell-Lendrum, D., Carr, G.M. Collen, B., Collette, L., Davidson, N.C., Dulloo, E., Fazel, A.M., Galloway, J.N., Gill, M., Goverse, T., Hockings, M., Leaman, D.J., Morgan, D.H.W., Revenga, C., Rickwood, C.J., Schutyser, F., Simons, S., Stattersfield, A.J., Tyrrell, T.D., Vié, J-C. & Zimsky, M. 2009. Tracking Progress Toward the 2010 Biodiversity Target and Beyond. Science **325**: 1-3.

Williams, F., Eschen, R., Harris, A., Djeddour, D., Pratt, C., Shaw, R.S., Varia, S., Lamontagne-Godwin, J., Thomas, S.E. &Murphy S.T. 2010. The Economic Cost of Invasive Non-Native Species on Great Britain. CABI Report.

Williamson, M. 1998. Measuring the impact of plant invaders in Britain, in Starfinger, S., K.Edwards, I. Kowarik and M. Williamson (eds.). Plant Invasions. Ecological Mechanisms and Human Responses, Leiden, Backhuys: 57-70.

Williamson, M.H. 2002. Alien plants in the British Isles. In: Pimentel D (ed) Biological invasions: economic and environmental costs of alien plant, animal and microbe species. CRC Press, Boca Raton, FL

Wise, R.M., van Wilgen, B.W., Hill, M.P., Schulthess, F., Tweddle, D., Chabi-Olay, A. & Zimmerman, H.G. 2007. The economic impact and appropriate management of selected invasive alien species on the African content. Report prepared for the Global Invasive Species Programme.

Wittenberg, R. & Cock, M.J.W. (eds.) 2001. Invasive Alien Species: A Toolkit of Best Prevention and Management Practices. CAB International, Wallingford, Oxon, UK, xvii - 228.

Woodfield, R. & Merkel, K. 2006. Final Report on Eradication of the Invasive Seaweed *Caulerpa Taxifolia* from Agua Hedionda Lagoon and Huntington Harbour, California. Prepared for the Steering Committee of the Southern California Caulerpa Action Team.

Xu, H.G., Qiang, S., Han, Z.M., Guo, J.Y., Huang, Z.G., Sun, H.Y., He, S.P., Ding, H., Wu, H. & Wan, F.H. 2006. The status and causes of alien species invasion in China. Biodiversity Conservation **15**: 2893–2904.