ASSESSMENT AND CONTROL OF BIOLOGICAL INVASION RISKS

Edited by Fumito Koike, Mick N. Clout, Mieko Kawamichi, Maj De Poorter and Kunio Iwatsuki

YNU
Yokohama National University,
21st Century COE Program

BDNJ
Biodiversity Network JAPAN

IUCN
The World Conservation Union

Assessment and Control of Biological Invasion Risks

Compiled and Edited by Fumito Koike,
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With the assistance of Keiji Iwasaki, Nobuo Ishii, Nobuo Morimoto, Koichi Goka, Mitsuhiko Takahashi as reviewing committee, and Takeo Kawamichi and Carola Warner in editorial works.

The papers published in this book are the outcome of the International Conference on Assessment and Control of Biological Invasion Risks held at the Yokohama National University, 26 to 29 August 2004.

Publication of this book was aided by grants from the 21st century COE program of Japan Society for Promotion of Science, Keidanren Nature Conservation Fund, the Japan Fund for Global Environment of the Environmental Restoration and Conservation Agency, Expo'90 Foundation and the Fund in the Memory of Mr. Tomoyuki Kouhara.

Published by:
SHOUKADOH Book Sellers, Japan and the World Conservation Union (IUCN), Switzerland

Citation:
Koike, F., Clout, M. N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds). *Assessment and Control of Biological Invasion Risks.* Published by SHOUKADOH Book Sellers, Kyoto, Japan and the World Conservation Union (IUCN), Gland, Switzerland. 216pp.

ISBN: 978-4-87974-604-7

Printed by SHOUKADOH Book Sellers, Japan

Available from:
IUCN Publications Services
Rue Mauverney 28, 1196 Gland, Switzerland
Tel: + 41 22 999 0000, Fax: + 41 22 999 0002, E-mail: books@iucn.org
www.iucn.org/publications/
A catalogue of IUCN publications is also available.

SHOUKADOH Book Sellers
Shimotachiuri Ogawa Higashi, Kamigyo-ku, Kyoto 602-8048, Japan
Tel: +81-75-441-3155, Fax: +81-75-441-3159 E-mail: shoukadoh@nacos.com

Cover photos: Mieko Kawamichi, Fumito Koike and Akio Hosaka (*Leiothrix lutea*)

Raccoon (*Procyon lotor*) naturalised in Japan probably predates the endemic salamander (left, *Hynobius tokyoensis*). Photo: Masato Kaneda. (*See* p.196 Hayama *et al*.; p.148 Koike)

Raccoon dog (*Nyctereutes procyonoides*) is native in east Asia, and probably competing with feral raccoons. Photo: Go Abe. (*See* p.116 Abe *et al*.)

Coypus (*Myocastor coypus*) naturalised in wetlands. Photo: Mieko Kawamichi. (*See* p.142 Baker)

Mongoose (*Herpestes javanicus*) captured by the control operation. Photo: Ministry of the Environment, Japan. (*See* p.157 Abe *et al*.; p.165; Ishii *et al*; p.122 Watari *et al*.)

Black rat (*Rattus rattus*) predates fruits of the endemic pandan (*Pandanus boninensis*) in the oceanic islands of Ogasawara. Photo: Keita Fukasawa. (*See* p.127 Clout and Russell for eradication of rats from islands)

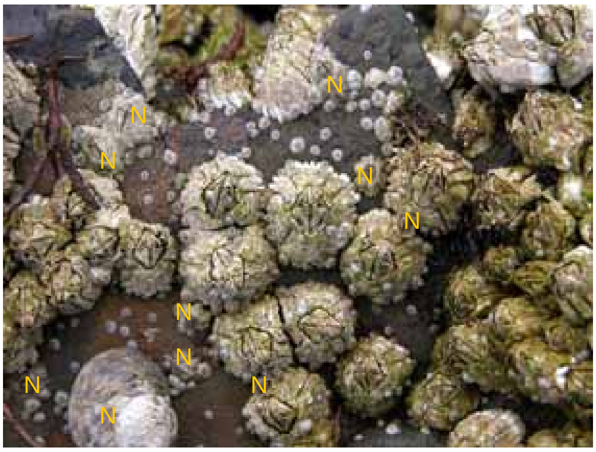
Formosan squirrel (*Callosciurus erythraeus taiwanensis*) and related taxa naturalised in Japan and southern Europe. Photo: Miho Sato. (*See* p.204 Hori *et al*.)

Plate 1

***Musculista senhousia* (mussel)** modifies soft-sedimented habitat to a dense mat of mussels. Photo: Masumi Yamamuro. (*See* p.113 Crooks)

Moon snail (*Euspira fortunei*) predates clam shell causing economic damage. Photo: Kenji Okoshi. (*See* p.104 Iwasaki)

Chinese mitten crab (*Eriocheir sinensis*) naturalised widely. The crabs accumulated at the migration barrier (a salmon support pathway). Photo: Stephan Gollasch. (*See* p.104 Iwasaki)

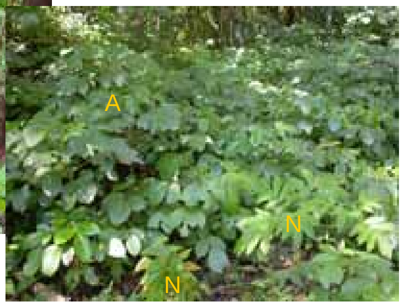


***Balanus glandula* (barnacle)** dominates this intertidal rocky shore (adults and new recruits). N: native species. Photo: Ryusuke Kado. (*See* p.104 Kado and Nanba; p.94 Otani; p.106 Iwasaki)

Mediterranean blue mussel (*Mytilus galloprovincialis*) dominates this intertidal shore. Photo: Keiji Iwasaki. (*See* p.95 Otani; p.104 Iwasaki)

Plate 2

Photo: Koichi Mikami.



***Bischofia javanica* (Euphorbiaceae)** is a secondary forest tree in the native range. The species dominates in climax forests in oceanic islands having dishermonic flora. A: *B. javanica*; N: native species. Photo: Keita Fukasawa. (*See* p.65 Kato et al.; p.73 Koike and Kato)

***Podocarpus nagi* (Podocarpaceae)** is a shade tolerant climax forest tree in the native range. The species was intentionally introduced 1200 years ago to the Kasugayama Shrine, and spread in the climax forest. Photo: Yuri Maesako. (*See* p.5 Koike)

Velvetleaf (*Abutilon theophrasti*, Malvaceae) has weedy (left) and cultivated non-weedy (right) strains. Photo: Shunji Kurokawa. (*See* p.84 Kurokawa *et al.*)

Plate 3

GLOSSARY

Terminology has developed independently in different sectors that address Invasive alien species (IAS), reflecting their different mandates. Managers, law makers and researchers sometimes have different requirements for terminology. Even within ecological sciences, botanists and zoologists may use terminology differently. This glossary explains the terminology as used in this volume. However, sometimes there is a need to use terminology differently, for example where a particular term has been defined in the national law of a country (e.g. "Invasive Alien Species" in Japanese law or "invasive species" in U.S. law). In such cases, this particular meaning, as used in that case, will be clarified in the text.

native species: a species, subspecies, or lower taxon, living within its natural range (past or present) including the area which it can reach and occupy using natural dispersal out of its natural range even if it is seldom found there.

endemic species: a species that is native to a particular area and ONLY in that area. (Example: the kiwi bird is native to New Zealand, and it is endemic there, while the white fronted tern is also native to New Zealand but is not endemic there because it also is native to Australia).

alien species: a species, subspecies, or lower taxon introduced outside its normal past or present distribution. Synonym in this volume: "introduced species". (Synonyms used elsewhere: non-native, non-indigenous, exotic).

introduction: the movement, by human agency, of a species, subspecies or lower taxon outside its natural range (past or present). This movement can be either within a country or between countries.

- *intentional introduction*: the purposeful movement by humans of a species outside its natural range and dispersal potential (for example: introduction of a species for aquaculture, for fishing, for aquarium use, for use as a crop or a garden plant, for biological control, etc....)
- *unintentional introduction:* an introduction of a species outside its natural range introduced “unwittingly” by humans or human delivery systems (for example: in ballast water, as contamination in soil or seeds, a contamination inside or on a container, as a parasite or pathogen of an intentionally introduced species, etc. It also includes the use, by a species, of tunnels and canals).

introduced species: same as *alien species*

invasive alien species: an alien species whose establishment and/or spread threaten ecosystems, habitats or species with economic or environmental harm. Note: national law and regulations in many countries often contain a <u>legal</u> definition of "invasive alien species" or "invasive species" which can be different from the ecological definition in this glossary. Where this is the case in this volume, the specific meaning will be clarified.

pathway: For the purpose of this manual, a pathway is broadly defined as the means (e.g. aircraft, ship or train), purpose or activity (e.g. agriculture, forestry, fisheries or horticulture), or commodity (e.g. timber, grain) by which an alien species may be transported to a new location, either intentionally or unintentionally.

biodiversity: the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems. Synonym: biological diversity.

ecosystem: a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.

habitat: means the place or type of site where an organism or population naturally occurs. The place where an organism is currently absent but potentially viable is included in the context of metapopulation ecology.

pest: This term is sometimes used as synonym to invasive alien species. (Note: many use it in an agricultural context only. Moreover, in the context of International Plant Protection Convention and some other international instruments, it has a well defined meaning, specific to that context).

weed: a plant growing where it is not wanted; Sometimes used as synonym for invasive alien species that is a plant. It includes woody species (e.g. shrubs, trees) and it includes natural and semi-natural environments (i.e. undisturbed environments).

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Various ecological problems and biological invasion

Kohei Urano
Yokohama National University
Leader of the 21st Century COE Programme of Japan, Ministry of ECSST
"Environmental Risk Management for Bio/Eco-Systems"

The Convention on Biological Diversity has been adopted by many countries, resulting in the development of national biodiversity strategies. This illustrates the international recognition of the importance of protecting ecosystems. However, ecosystems still face many threats, some of them growing and spreading so rapidly as to cause irreversible deterioration in many countries and areas.

The Biodiversity Synthesis Report of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) showed many undesirable phenomena such as loss of forest, conversion of terrestrial biomes, extinction of species and so on. This report also describes various ecosystem services, and explains how their changes have affected humans. Furthermore, it proposes international actions to achieve sustainability for future generations.

Ecosystem changes have principally been attributed to loss of habitat, over-exploitation of ecosystem resources, biological invasion, nutrient pollution, toxic chemical pollution, and global climate change.

Loss of habitat results from the expansion of agricultural land, urban sprawl and industrialisation in many countries. With this loss of habitat, many native plants and animals lose their living spaces.

Over exploitation of ecosystem resources is brought on by over-fishing, over-hunting and over-logging. This has led to the exhaustion of natural biological resources and the extinction of species.

Biological invasion is an issue of growing importance in many countries, due to the significant increase in international transportation and trade. Introduction of invasive alien species can disturb the balance of local ecosystems or even destroy it due to absence of competitors or natural enemies. As a result, native species can be threatened or even driven into extinction.

Nutrient pollution is caused by the release of nitrogen and phosphorus from domestic sewage, industrial wastewater, farming, etc. This nutrient pollution can cause an abnormal population growth of some species, leading to ecosystem disturbance and biodiversity reduction.

Toxic chemical pollution results from the spread of agricultural pesticides or the discharge of industrial wastewater, flue gas and solid wastes. This can result in reduced reproduction or increased mortality in sensitive species. Persistent organic pollutants (POPs) such as PCB, DDT, HCB and DXNs accumulate through the food chain, in the bodies of fish, birds and mammals and can then do harm to humans. .

Global climate change is brought by discharge of greenhouse gases such as carbon dioxide, methane, dinitrogen monoxide and fluorocarbons. Global change leads to change and loss of habitat of various species, it disturbs ecosystem balance and it reduces biodiversity.

Funded by the Ministry of Education, Culture, Sports, Science and Technology (ECSST), our COE programme was designed to collect and analyse environmental risk information in East and Southeast Asia including Japan. Its final aim is to build a centre of practical environmental science for risk management of bio/eco-systems. This programme involves studies on three of the main ecosystem stressors mentioned above, namely biological invasion, nutrient pollution, and toxic chemical pollution. In this volume, we focus on one of them: biological invasion. The contents of this volume are based on the "International Conference on Assessment and Control of Biological Invasion Risks" held 26-29 August, 2004 at Yokohama National University.

I sincerely hope that this book will be of worldwide benefit to people, for assessment and management of biological invasion risks.

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Alien species and wild flora

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INTRODUCTION

Excellent reviews have been published on the significant influence that alien species can exert over native wild flora and fauna (for instance Boufford 2001). I will give some actual examples of the influence of alien species on the native flora of Japan.

The second version of the National Biodiversity Strategy of Japan, edited by the Ministry of the Environment and endorsed by the government in 2002, recognises that there are three components to the biodiversity crisis in Japan. These are: (1) influence by human activities - shown clearly by the increasing number of endangered species, (2) influence on artificially modified areas abandoned by human activities, and (3) influence by more recent impacts such as artificially produced chemical materials, alien species and so on. The first two, as well as chemical materials under (3) were already under regulation by law in Japan, but alien species were not. Finally in 2004, however, we succeeded in establishing a law to control invasive alien species in Japan. However, it is not sufficient to have a law – it will, in addition, be necessary for all citizens in Japan to realise the threat posed by invasive alien species and to participate in activities to control them. These Proceedings are part of this.

Conservation International has designated the Japanese Archipelago as one of the hot spots for biodiversity on earth. This is the first example of such nomination from industrialised countries, and the rich flora of Japan is one of the reasons for this (Boufford *et al.* 2004).

ALIEN SPECIES WITH AND WITHOUT ADVERSE EFFECTS

In the Invasive Alien Species Act of Japan, endorsed in 2004, it was noted that the alien species designated as having adverse effects on ecosystems by the Japanese government should be controlled carefully. The definition of adverse effect on ecosystems in this law means adverse effects on 1) ecosystems, 2) human safety or 3) agriculture, forestry and fisheries (The Invasive Alien Species Act – Law No 78 (June 2, 2004) can be read at *www.env.go.jp/en/topic/as.html*.)

There are a vast number of alien species introduced into Japan (Shimizu 2003); the alien species introduced before the so-called Meiji revolution (before 1870s), are excluded from the Invasive Alien Species Act. Furthermore, the alien species in the Act are restricted to those introduced from abroad. We also have a number of species which were introduced into Japan before historical records started. We call them pre-historic introductions. Many of them were brought to Japan as supplementary food plants, such as *Monochoria vaginalis*, *Lycoris radiata*, *Stellaria neglecta*, *Amaranthus lividus*, *Capsella bursa-pastoris*, *Astragalus sinicus*, which have become very popular in developed areas of Japan. Many other species were introduced into Japan during the Yedo dynasty, (from the 17th century to the first half of 19th century) and are now integrated in the landscape (e.g. in paddy field areas), such as *Trifolium repens*, *T. paratense*, *Oenothera stricta*. Such species are now considered integrated elements of the flora.

One of the best Japanese novelists, Osamu Dazai, once wrote that *Oenothera tetraptera* looks very well in the landscape of Mt Fuji when he was in Yamanashi Prefecture; Mt Fuji is considered one of the best landscapes in Japan, even though *O. tetraptera* is an alien species. In other words, *O. tetraptera* is now well integrated and accepted as part of the landscape of Japan, and it is not considered as a problem.

I am enumerating such examples to show that some alien species have adjusted very well within the Japanese flora, especially where the environment was developed by human activities in various ways. There is no reason to be concerned about these alien species at the moment. However, there are several other alien species which are considered to be harmful.

Several native plant species of Japan have invaded other parts of the world where they were introduced and are considered pests there, for example *Lygodium japonicum*, *Pueraria lobata*, *Reynoutria japonica* and *Hedera rhombea*.

ALIEN PLANTS ALREADY NOMINATED AS INVASIVE

Based on the Invasive Alien Species Act, the following introduced plants have already been

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nominated as invasive alien species: *Alternanthera philoxeroides* (Berberidaceae), *Hydrocotyle ranunculoides* (Apiaceae), and *Gymonocoronis spilanthoides* (Asteraceae). These three species listed in the first round are widely distributed and vigorous and they are considered to have adverse effects. *Hydrocotyle bonariensis* and *H. umbellate* are also listed as allied species which are distributed widely as weeds.

The following nine species have been added in the second list: *Azolla cristata* (Azollaceae), *Myriophyllum aquaticum* (Haloragaceae), *Sicyos angulatus* (Cucurbitaceae), *Coreopsis lanceolata*, *Rudbeckia laciniata, Senecio madagascariensis, Veronica anagallis-aquatica* (Asteraceae), *Pistia stratiotes* (Araceae) and *Spartina anglica* (Poaceae). Among them, *Azolla cristata, Myriophyllum aquaticum*, and *Pistia stratiotes* are aquatic species that grow vigorously in their new habitat after invasion, strongly influencing native flora there. *Coreopsis lanceolata*, *Sicyos angulatus* and *Veronica anagallis-aquatica* are land plants growing also in wet areas such as river-sides where many native species are seriously threatened by their invasion.

Rudbeckia laciniata invades into conservation areas like National Parks and forms large and compact colonies damaging various native species. To protect natural flora and fauna in protected areas, this alien species must be managed urgently.

Senecio madagascariensis was introduced into Japan rather recently but it is rapidly invading and increasing its distribution areas; moreover, this species is a concern in various sites throughout the world.

Spartina anglica has not yet been introduced into Japan, but it has caused concern for native species in wetlands in various sites globally. Introduction of this species into Japan must be prevented.

ALIEN PLANTS CONSIDERED TO HAVE ADVERSE EFFECTS BUT UNABLE TO BE NOMINATED AS IAS AT THE MOMENT

There are other species which are considered to be in the category of IAS but who have not formally been designated as such. Some of them are species introduced for quick growth in developed areas. In the future, native species should be selected for this purpose, but there are few native Japanese species which can presently be used for this, and hence introductions from abroad continue. Some of those species in this category are establishing in the wild where they out-compete native vegetation or hybridise with closely related Japanese species Native flora is seriously affected by these introduced species, but the current demand for green cover of seriously developed areas requires the continued use of such species, even though it is recognised that this will be problematic for coming generations. It is very important to breed various cultivars from native species to use, even though this will take time.

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Assessment and control of biological invasion risks

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INTRODUCTION

Biological invasion is usually an irreversible process. Contaminated chemicals in the environment will be decomposed after several years. Invasive alien species, however, reproduce themselves and persist. New invading alien species will change the nature of forests, rivers and lakes in the future. Biological invasion by alien species has recently been recognised as an environmental issue, and standardised procedures (technical as well as legal) to assess and control invasion risks have not yet been established. New ideas and basic studies are urgently required to deal with this new environmental issue.

CURRENT STATUS OF INVASION

The Earth's history of continental drift has created the large scale distribution pattern of species (Frodin 1984, Cox and Moore 1993). Earthworms have very limited ability of dispersal, and the present distribution of earthworm families are partly determined by past plate tectonics (Blakemore *et al.* 2006 - this volume). In addition to such global pattern, recent speciation also occurred at smaller geographical scale. The geographical range of most species is significantly smaller than the circumference of the Earth. It is close to 1000km in woody plants and about 10,000km in ferns (Fig.1). This spatial scale is similar to the geographical range of animals (Shoener 1987, Gaston 1994). Human mediated transports of 100km can cause biological invasion by some species that have a small distribution range; and those of 1000 km may cause invasion by many woody species.

Although natural migration often occurs, the distance over which human mediated transport occurs is three or four orders of magnitude larger than natural dispersal ability (Fig. 2). Japan imports grains and hay cubes for livestock, mainly from North America, and these contain viable weed seeds. 21.6kg of corn imported from the USA contained seeds of 545 weed species and for 21 of those species there were more than 100 seeds (Kurokawa 2001). Such mass transfers beyond 10,000km occur every year.

Human activities, that transport alien species, mainly occur in towns and villages. Alien species that grow in disturbed areas will easily be naturalised, because many seeds of such species are transported unintentionally and many suitable habitats are spread around the place of arrival (Fig.3). However, alien

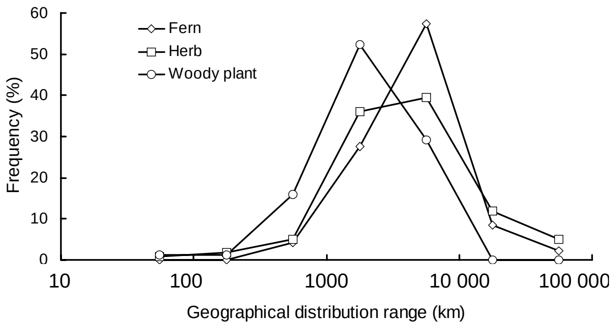

Figure 1 Distribution range (west-east width) of native plants in Kanagawa Prefecture, Japan. Species were randomly sampled at the probability of 20% from a flora list (Flora-Kanagawa Association 2001). Lower taxon such as subspecies was not considered. Geographical range was calculated from longitudinal distribution range and circumference at the latitude of 35º.

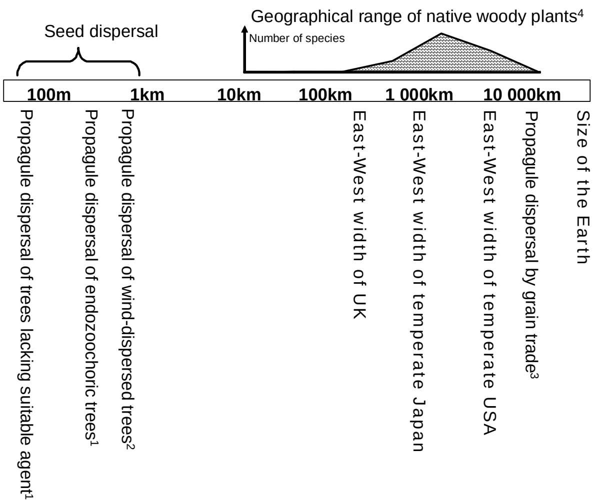

Figure 2 Spatial scale of propagule (seeds, fruits, detached bulbs, etc.) transfer by humans and natural agents. Dispersal distance: 1ha of area receives at least one seed at the probability of 0.5. [1]Komuro and Koike 2005, [2] Clark *et al.* 1999, [3] Kurokawa 2001, [4] East-west geographical range of woody plants native in Kanagawa Prefecture, Japan (Fig. 1)

Pages 4-12. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks.* Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

forest plants have difficulties in spreading in urban and rural landscapes with highly fragmented habitats (Fig.4). Such species may not spread continuously, but eruptive (spasmodic) spread may occur due to stochastic colonisation of the metapopulation (Fig.5). Forest species will quickly spread after the population arrived at continuous forest, as is the case with the alien *Impatience* in Europe (Kornaś 1990, Williamson 1996).

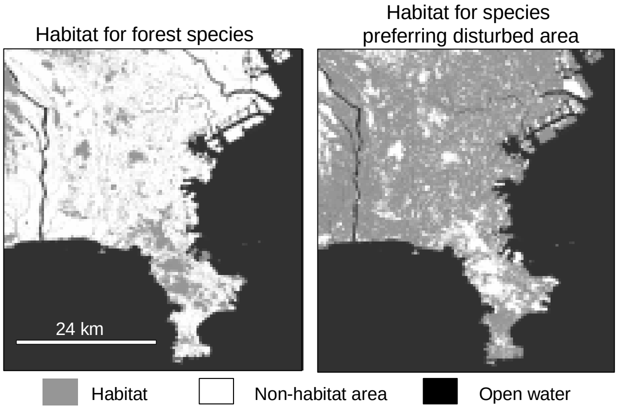

Figure 3 Habitats for disturbed site species and those for forest area in Kanagawa Prefecture (suburb).

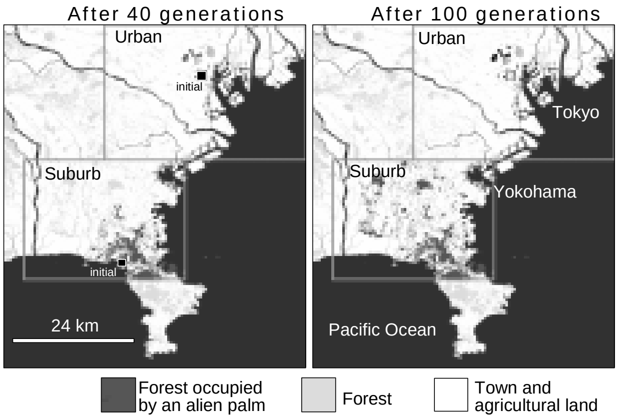

Figure 4 Hypothetical simulation of range expansion of an alien palm (*Trachycarpus fortunei*) in urban and suburban landscapes. The effect of landscape on range expansion was evaluated based on this hypothetical simulation, although the palm has already distributed everywhere.

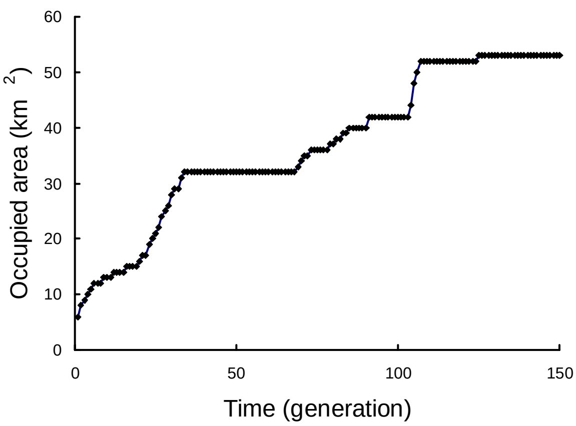

Figure 5 Simulated range expansion of an alien palm (*Trachycarpus fortunei*) in the urban landscape based on the same simulation as in Fig. 4.

This phenomenon might be one reason that spreading alien species are usually r-strategists (Ehrlich 1989, Rejmánek and Richardson 1996, Pheloung *et al.* 1999, Grtokopp *et al.* 2002, Crooks 2006 - this volume) and are adapted to disturbed habitats. Another reason for the dominance of r-strategists may be the lack of time to spread for K-strategists. Beech is sometimes invasive (Healey *et al.* 2002); however, the life span of Japanese beech (*Fagus crenata*) is about 200 years (Nakashizuka and Numata 1982). It is only 160 years ago that the first steamship crossed the ocean and this is less than one generation for long lived woody plants. A shade tolerant climax forest tree with limited dispersal ability, *Podocarpus nagi*, was introduced to the ancient capital of Japan 1200 years ago (Suganuma 1975), and the species spread gradually to an approximately 1 km by 1km area in the Kasugayama Forest Reserve, a World Heritage site of the ancient capital Nara (Maesako *et al.* 2003). Many species living in stable environment may not have had sufficient time yet to establish and/or spread in the wild. In the next century, however, the number of alien species that are established and/or spreading in stable environments may continue to increase and they might become an important issue.

A global list of all alien species does not exist, and the number of alien species worldwide is not clear, but numbers are known for several countries.

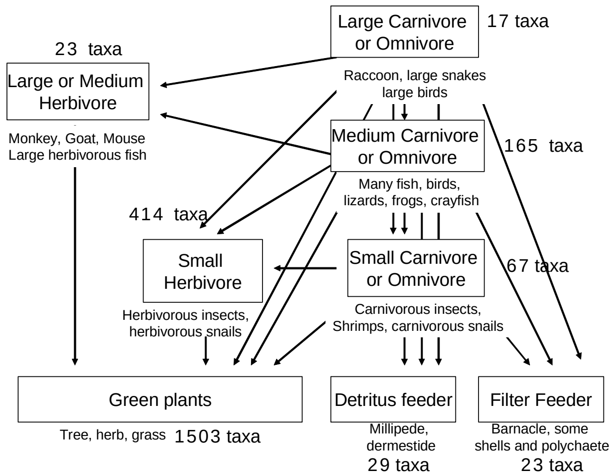

Figure 6 Number of taxa (species and sub-species) established in Japan after 1868, when Japan opened its ports to the World. Information was based on Ecological Society of Japan (2002). The number of taxa is tentative due to lack of information on food habit in some species.

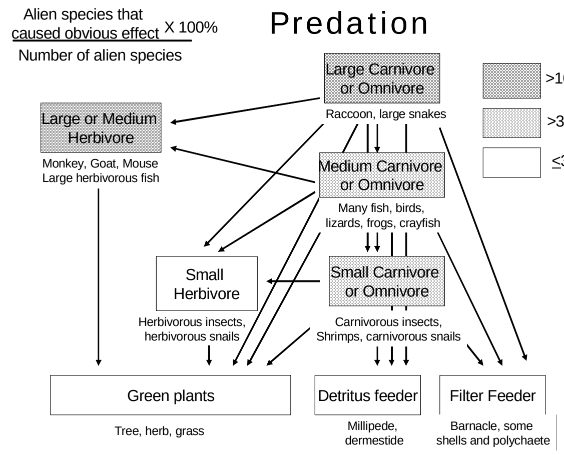

Figure 7 The fraction of species causing serious effects on native species through predation. Figures are tentative due to lack of information.

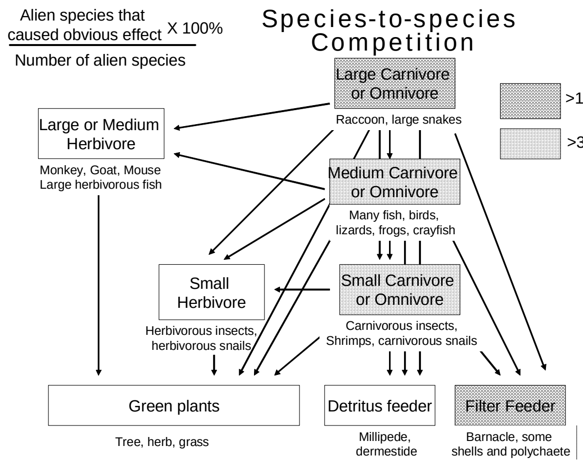

Figure 8 The fraction of species causing serious effect on native species through species-to-species competition. Weak competition working among various species in communities (diffused competition) was not included. Figures are tentative due to lack of information especially for plants and small animals.

Based on the data from Japan, the number of alien species greatly differs in the different trophic levels (Fig. 6). There were about 1500 taxa (species and lower taxon such as subspecies) of alien green plants established in Japan. Large carnivores or omnivores (raccoon, large snakes, etc.), large or medium size herbivores (goat, mouse, etc.), filter feeders (barnacle, shellfish, etc.), and detritus feeders (millipede, dermestide, etc.) had 2 order of magnitude fewer alien species than green plants. Small herbivores (herbivorous insects, herbivorous snails, etc.), medium size carnivores or omnivores (fish, birds, crayfish, etc.), and small size carnivores or omnivores (carnivorous insects, shrimps, carnivorous snails, etc.) had an intermediate number of alien taxa.

Some alien species reduce the abundance of native species through predation or parasitism. Large sized species tend to cause serious effects (Fig. 7).

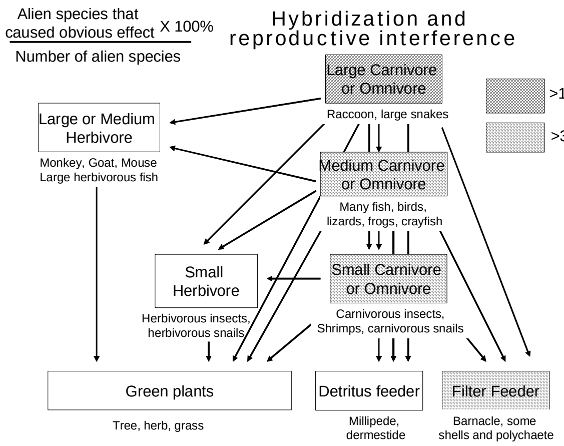

Figure 9 The fraction of species causing serious effect on native species through hybridization and reproductive interference. Figures are tentative due to lack of information especially for plants and small animals.

Feral goats (*Capra hircus*) wiped out the vegetation on small islands (Shimizu 1993) and ship rats (*Rattus rattus*) predate the seeds of endemic plants in Ogasawara (Bonin) Islands (Watanabe *et al.* 2002). Large carnivorous fish (*Micropterus salmoides*) feed on shrimp and native fish species (Yonekura *et al.* 2004), and the distribution range of the endemic Amami rabbit (*Pentalagus furnessi*) is decreasing due to predation by the Indian mongoose (*Herpestes javanicus*) (Sugimura 2002). Predation by alien species does not only reduce the prey species, but causes indirect increases in species on the next lower trophic level (Maezono and Miyashita 2003). The Indian mongoose (*Herpestes javanicus*) causes the increase of insects through the decrease of frogs (Watari *et al.* 2006 - this volume).

Another ecological mechanism to reduce native organisms is competition. The land use by native raccoon dog (*Nyctereutes procyonoides*) around farmland can be eliminated by the feral raccoon (*Procyon lotor*) (G. Abe *et al.* 2006 - this volume). The alien barnacle (*Balanus glandula*) may be a competitor to native barnacle on rocky shore (Kado and Nanba 2006 - this volume). Although the introduced canopy tree (*Bischofia javanica*) decreased the dominance of native plants significantly in Ogasawara Islands, reports on serious species-to-species competition between green plants, and between herbivorous animals are less frequent than in the case of large carnivorous animals (Fig. 8). As there are more than one thousand alien plant species, it is possible that their effect might not have been realised yet in many cases. It may also be difficult to determine significant effects on native flora caused by alien plants due to the diffused nature of competition (Hubble 2001). Such effects require further study.

"Ecosystem engineers" are species that change

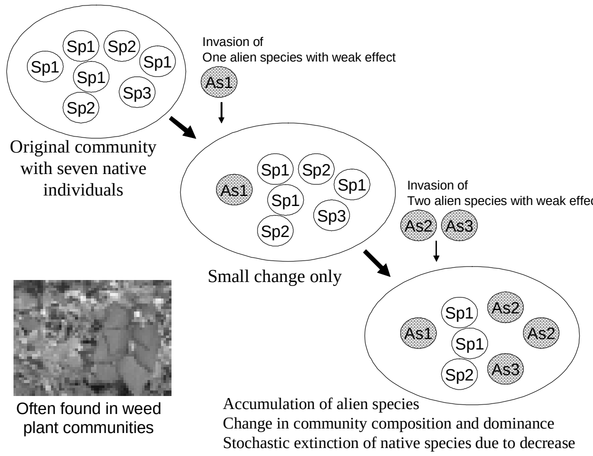

Figure 10 Hypothetical diagram of community change caused by alien species with weak competition effects. Sp_n represents individuals of *n*-th native species, and As_m of *m*-th alien species. Such phenomenon often occurs in weed communities.

physical and chemical environments. These species are similar to resource competitors except that their effects are not limited to native species of the same niche, but extend to a much wider range of species. Earthworms can change soil environments and have an impact on plants (Blakemore *et al.* 2006 - this volume); in the soft-sediment habitats of sea and brackish water a mussel (*Musculista senhousia*) turns the bottom into a dense mat of shells and this affects many types of organisms (clams, eelgrass, etc.) either negatively or positively (Crooks 2006 - this volume).

Records of hybridization and reproductive interference are also frequently found in large carnivorous or omnivorous animals (Fig. 9). Reproductive interference between native and alien salmonoid fish (native *Salvelinus leucomaenis* and alien *Salvelinus fontinalis*) through asymmetrical reproductive behaviour was reported (Kitano 2002). Such interference can usually only be determined after in depth research, and this has not been carried out for most species. This should be remedied, and such research should cover all alien species.

Due to the development of legislation to eliminate the import of invasive alien species and due to the publicity around alien species issues, new intentional releases of large alien animals are likely to be reduced in future. However, the introduction of alien species that have a weaker effect on native ecosystems (e.g. many alien plants) will continue, and the cumulative effect of such weaker effects of alien species will nevertheless cause a significant reduction of native species through neutral replacement (Hubble 2001, Fig. 10). Such alien species with weaker effects and alien species living in stable environments (as discussed before) will increase gradually in the future. As a result, biological invasion issues will increase even more in importance in the next century.

RISK MANAGEMENT

Most countries have a quarantine system to deal with alien species that are detrimental to humans or to economically important species (Tanaka and Larson 2006 -this volume). Some countries have extended their quarantine system to deal with alien species that are harmful to ecosystems. However, no ideal legal and social system has so far been put in place anywhere, as discussed in this volume (Ikeda 2006, Courtney 2006 and Takahashi 2006 -this volume).

The prohibition of introduction of invasive alien species is an important management approach, but it is not sufficient on its own. Due to the volume of imports quarantine systems can not check all the imported commodities and instead they usually check selected samples. Unfortunately, even if only a few individuals are introduced into a natural habitat this can result in their establishment, and severe damage can result.

As another approach to management, risk feedback to importers and carriers may be a possible (Fig. 11). In many cases, the people obtaining benefits are different from the people exposed to risks. For example: carriers, importers, and growers of new crops benefit from the introduction but other citizens, such as traditional farmers and fishermen, and the government carry the risk of naturalised alien species. Possible impacts from alien species include those to cultural values, economic damage to farmers and fishermen, and financial costs to the government who must pay for control and eradication programmes. Tort liability, a mechanism of risk feed back, could be an effective approach for invasion risk management (Courtney 2006 -this volume).

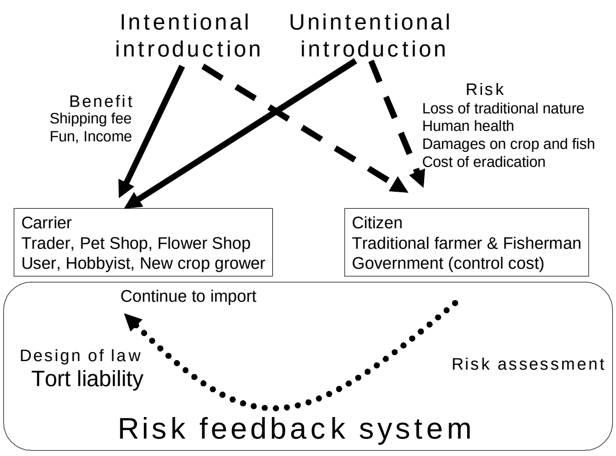

Figure 11 Risk feed back from the people exposed to risks to the people obtaining benefits from the introduction of alien species.

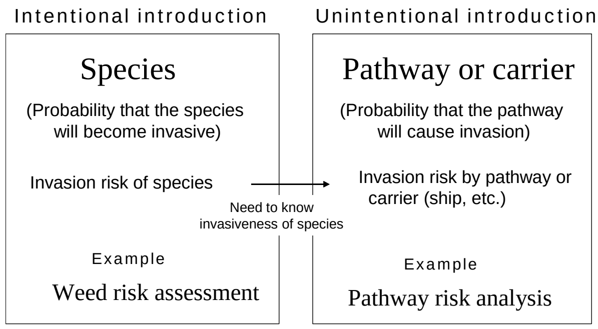

Figure 12 Two types of risk assessment for intentional and unintentional introduction.

In Japan, the illegal release of a carnivorous alien fish (large-mouth bass, *Micropterus salmoides*) for game fishing continues. It is quite difficult to identify who released the fish, because it is usually done secretly. The legal system on its own is not effective and other approaches, such as education and raising awareness need to be considered.

RISK ASSESSMENT

Risk management should be based on reliable risk assessment. For intentional introduction, risk assessment is to evaluate the probability that the species will become invasive (Fig. 12). Weed risk assessment (Pheloung *et al.* 1999, Kato *et al.* 2006 - this volume) is widely used for intentional introductions. For unintentional introduction, the aim of risk assessment is to evaluate the probability that the pathway (e.g. importing the specific commodities from the specific foreign country, ballast water from a specific region) will result in invasion.

Research on how to predict invasiveness of a species based on biological traits is in progress, but it has had limited success so far (Williamson 1996, Goodwin *et al.* 1999, Gollasch 2006 - this volume). Although weed risk assessment is successfully used in Australia, New Zealand and Hawaii (Pheloung *et al.* 1999, Daehler *et al.* 2004, etc.), its success is due to the use, as a variable, of history of invasion elsewhere in the world (Koike and Kato 2006 - this volume). Biological traits are also considered in weed risk assessment, but their effectiveness for prediction is low. Many biological traits are specific to a given taxonomical group and ecological guild. The traits for terrestrial plants are quite different from those of aquatic fish. However, history of invasion elsewhere in the world can be used for various species. A risk assessment system based solely on such historical records will be possible for all species, and applicable as the first stage of risk assessment.

In spite of the associated difficulties, biological traits must be used in risk assessment for species that are not known as invasive yet. The palm *Arenga engleri* is considered a major pest in Ogasawara Islands, but invasion in other regions has not been reported at this stage (Kato *et al.* 2006 - this volume). If we can predict species composition and dominance in a community based on biological traits, we can predict invasion by new species. There are two approaches to predict community composition and dominance based on traits of species (Fig. 13). One is a mechanistic approach, studying mechanisms in the community and using computer simulation for prediction; and another is through statistical data mining. Most invasion risk assessment is done using the latter because the simulation technique is not well enough developed yet.

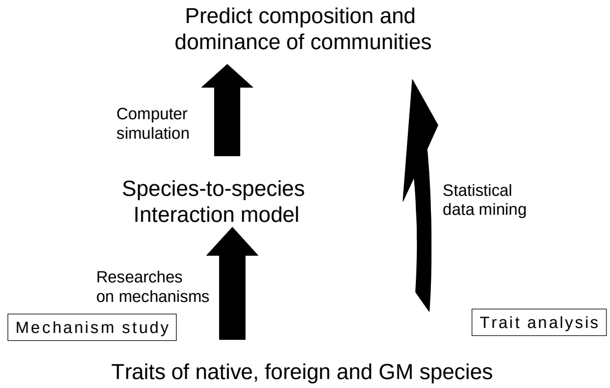

Figure 13 Two approaches to predict community species composition and dominance from biological traits of the species pool.

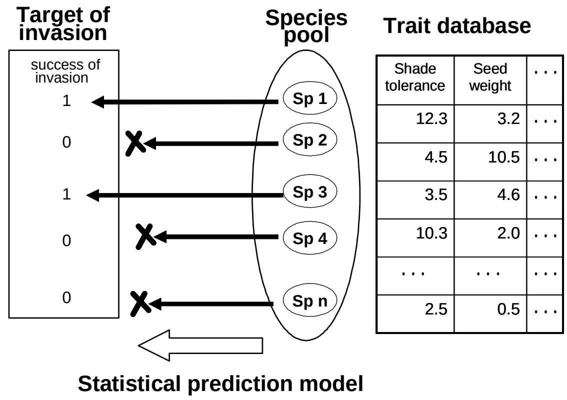

Figure 14 Hypothetical diagram of statistical modeling of invasion risk assessment. The target of invasion (country, island or biological community) is the area used to judge success or failure of invasion (1=success, 0=failure). The species pool is the set of species (sp 1 to sp n) having the chance to invade the target area. The statistical prediction model is obtained based on the trait database and the record of success or failure of invasion.

In statistical invasion risk assessments the target of invasion (country, island or biological community) and the species pool, which include the species that have the chance to invade to the target area, are defined first (Fig. 14). Success of invasion for each species is examined, and traits of the species are obtained from literatures and field research. The statistical prediction model is usually obtained assuming success of invasion as the dependent variable, and various traits as independent variables. Grotkopp *et al.* (2002) assumed their target area as whole Southern Hemisphere, and the species pool as introduced alien pine species (Table 1). In Koike (2001) the climax forest community was the target, and local flora was assumed as the species pool.

The selection of the target and the species pool will influence the success of the prediction model. If we consider a wide area with a mixture of various communities as the target, prediction using biological traits becomes difficult (Table 1). If we consider a given biological community as the target, prediction will become easier. Since the same trait in an alien species can influence invasion probability positively or negatively, depending on the target "receiving" community, risk assessment should be done separately for each target community. For instance, for climax forests key traits that were effective in prediction were shade tolerance and tall maximum tree height (Koike 2001, Fig. 15). Alien trees with such traits became extremely hazardous pest (such as *Bischofia javanica* in Ogasawara Islands). However, these same traits will prevent invasion into arable weed communities. This illustrates that if we specify the target community, prediction of invasion based on biological traits of an alien species will become possible.

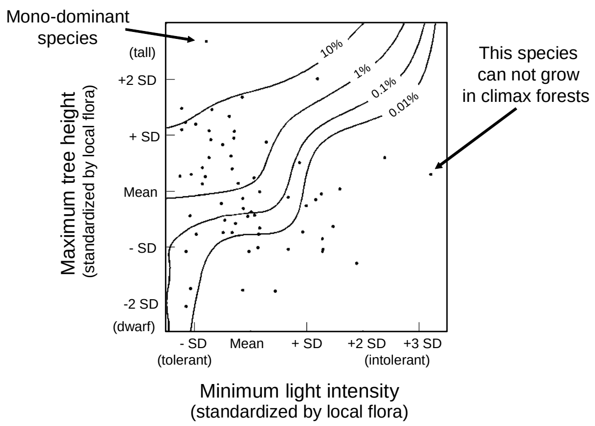

Figure 15 The dominance of woody species in climax forests can be predicted by shade tolerance and maximum height (redrawn from Koike 2001). Each plot represents a species. Data of four forests (boreal mixed, cool-temperate deciduous, warm-temperate evergreen broad-leaved, and subtropical evergreen broad-leaved forests) were plotted on the same axes. Contours show percentage dominance in basal area.

The science of community ecology has made rapid progress in recent years. Traditionally, ecologist had assumed that alien species compete with the native species in same niche, and that the alien species will invade if it wins the competition. However in plant communities, instead of such species-to-species competition, an alien species often causes weak effects on many surrounding species (Hubble 2001), decreasing the abundances of many native species, but only by a little bit. In the case of alien and native turtles in freshwater (Chen 2006 - this volume), species-to-species competition was also not obvious. We are not able to predict invasion by a simple competitive experiment between alien and native species of the same niche; instead estimating the range of ecological traits (envelope) which allow invasion into the target biological community may be a more suitable approach (Weiher and Keddy 1999, Koike 2001, Fig. 15).

Nevertheless, competitive exclusion between two species is often found in vertebrate communities that have few (or sometimes only one) species. Feral raccoons can exclude native raccoon dogs from the forest edge adjacent to farmland and residential areas in Japan (G. Abe *et al.* 2006 - this volume). In the case of vertebrates, behaviour excluding other individuals from the home range and feeding sites can cause competitive exclusion of native species. A clear hierarchy of winner and loser individuals will be established, and the winner will access resources. In plants, such asymmetrical resource competition typically occurs in fertile soil, where the closed canopy of taller species accesses most of the light energy (Tilman 1993, Wilson and Tilman 2002). The relation between animal behaviour excluding other individuals and the number of co-existing species in

Table 1 Target of invasion and species pool in invasion risk assessment models. The target of invasion is the area used to judge success or failure of invasion (country, island or biological community), and the species pool is the set of species considered that have the chance to invade the target area (Fig. 14). References are classified into four groups based on the target and species pool used.

Target	Species pool	
	Wide range of species	**Congener**
Community	Koike 2001	Tofts and Silvertown 2002*
Mixture of community	Biological part of WRA (Pheloung *et al.* 1999)	Goodwin *et al.* 1992 Grtokopp *et al.* 2002

*Performances in juvenile stage only

the community needs to be studied, in order to assist with the prediction of impacts on native species from competitive exclusion by alien species.

CONTROL AND ERADICATION

Management techniques are rapidly developing. Future range expansion is now not so difficult to predict (Koike 2006 - this volume). In an area of 10 x 10km eradication for large animals can be successful (Clout and Russell 2006 - this volume). There are three types of successful eradications: (1) in a very early stage of invasion with a very small distribution range, (2) in a very intensively managed environment such as a greenhouse or in human health (e.g. smallpox), and (3) where there is a geographical limit such as in islands or when the species has special habitat requirements. Control or eradication of actively spreading populations in natural habitats is very difficult, and techniques to halt spatial spread and to eradicate whole populations should be further developed.

Alien species often have a dense population at the centre of the distribution range, and low density in marginal areas. Such distribution can be caused by two completely different mechanisms (Fig. 16). One is a source-sink structure with high population growth rate at the central core habitat (source population) and the marginal population (sink population) supported by migrants from the core area. In this case there is no geographical range expansion. The other case is that of an actively spreading population, with high population growth rate at the marginal (low density) area. Density dependent population growth is commonly found in alien animals (S. Abe *et al.* 2006 - this volume). In this case the geographical distribution range is expanding.

In a population with source-sink structure, intensive catch at the central source population will be effective as a management programme (Baker 2006 - this volume); however, in an actively spreading population, removal of marginal isolated populations is important and effective to slow down range expansion (Koike 2006 - this volume).

DATABASE

Alien species databases are required for assessment and management of invasion risks. In risk assessment, information on native geographical range including climate information, current distribution outside its native range, and damage to the ecosystem and economy, is critical to assess the invasion risk for the focal species (Goodwin *et al.* 1999, Koike and Kato 2006 - this volume). However, there is no

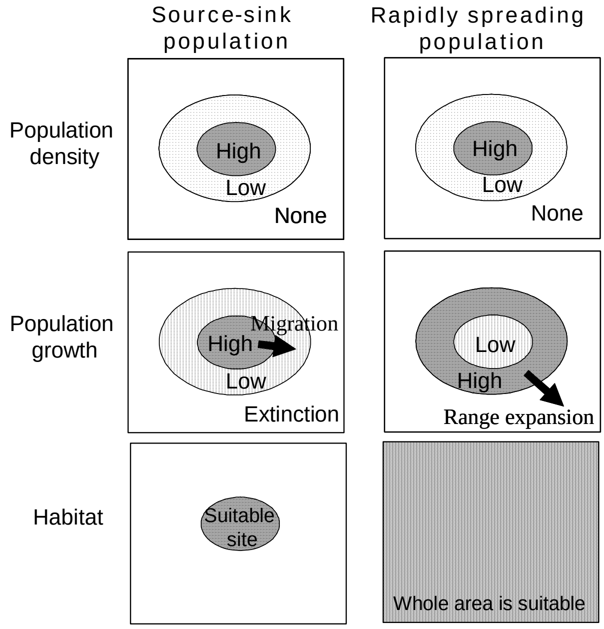

Figure 16 Spatial structure of populations with source-sink structure (left) and of rapidly spreading population (right).

standardised measure yet to evaluate the impact on native ecosystem. Key biological traits for use in invasion risk assessment are also not known except in the case of forest plants and herbaceous weeds in disturbed land (Pheloung *et al.* 1999, Koike 2001). Research on key traits in various habitats should be done immediately to determine what information should be included in databases. In addition, information on ecological traits of native species (at least randomly selected species from the local flora) is also necessary for more accurate prediction, because they also play a role in the success of alien species invasions (Koike 2001).

For eradication and other management, databases need to include effective management methods and techniques (trapping technique, herbicide information, etc.) for each species. Databases with reports of successful and failed eradication projects (Rejmánek and Pitcairn 2002, Clout and Russell 2006 - this volume) and their fiscal costs may be helpful to evaluate the feasibility of planned eradication programmes.

THIS VOLUME

This volume is the compilation of papers presented at the conference “Assessment and control of biological invasion risks” held in 2004 at Yokohama National University. Organization of this book is objective oriented. Papers were not classified by habitats or taxonomical criteria, but by objectives such as risk assessment, risk management and eradication. We tried to extract aspects common to various species instead of accumulating species specific information. This volume also includes contributions reporting on the current status of invasion and on properties of alien species in East Asia.

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Part 1

Risk Management

Risk analysis, the precautionary approach and stakeholder participation in decision making in the context of emerging risks from invasive alien species

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Abstract The recent development of “risk analysis” will be described focusing on stakeholder participation in decision making on addressing emerging risks, such as biodiversity loss or ecological destruction of habitat by invasive alien species. First, we discuss how such a type of “risk” is perceived and identified among the stakeholders involved in “risk analysis” which consists of "hazard identification", "risk assessment" and "risk management". The interdisciplinary nature of “risk” is a substantial factor in understanding why risk is differently perceived and managed by stakeholders involved at local, regional and global scales. Secondly, it will be discussed why the precautionary approach is one of the most fundamental approaches to decision making on this issue, in view of the scientific uncertainty and the complexity resulting from socio-cultural dynamics in our post-industrial society. Finally, the case of the “Ballast Water Convention” will be presented as an example of applying the precautionary approach as part of good risk governance in the case of invasive alien species.

Keywords: ballast water; risk analysis; risk governance, stakeholder participation; biosecurity; precautionary approach to invasive alien species

INTRODUCTION

Because of the drastic increase in both scale and variety of global trade, tourism and transport across national or regional boundaries, we have faced an increased risk to humans and to the environment due to biological invasion of alien species into the terrestrial, freshwater, and marine ecosystems since the 1990’s. Traded commodities have provided pathways and vectors for intentional as well as unintentional introductions of alien species. Maritime transport has been one of the biggest means of global transfer of marine organisms, either inside the vessels, or attached to them. A large volume of ballast water results in the transportation and discharge of a large number of marine organisms at locations where they are alien. Toxic algae, such as dinoflagellates that used to be dominant in one specific coastal zone, have suddenly occurred in other coastal zones, primarily due to the large volume of ballast water being transported across the oceans (GloBallast, 2000). Biological invasions have become relevant in many contexts of biology, ecology, marine and environmental sciences as well as environmental policy sciences, and concerns include risk to human health, loss of biological diversity, habitat impacts, and other socio-economical damage.

This paper focuses on “risk analysis”. We need to examine inter-disciplinary aspects of the hazards created by invasive alien species, including the difficulty of predicting effects of invasion, leading to high scientific uncertainty, and the added complexity due to different values among stakeholders involved. Risk analysis is considered as one of principal approaches to deal with scientific assessment and decision making in the context of uncertainty and complexity. For example, some of the recent international agreements or conventions such as the Agreement on Sanitary and Phytosanitary Measures (SPS Agreement, 1994) or the International Plant Protection Convention (IPPC, 1997) state the need for measures taken to be based on scientific assessment of risks to human, animal, or plant health. Risk analysis is expected to underpin decisions or negotiations either at bilateral or multilateral level.

In the first part of this paper, we will discuss how modern risk analysis has been developed as an interdisciplinary approach to deal with complex and uncertain processes in the management of emerging risks in our post-industrial society. The second part will discusses the “precautionary approach or precautionary principle” as a fundamental approach in decision making, relating to invasive species, due to the high scientific uncertainty that originates from the limitations of scientific knowledge on invasiveness and to complexity of risk events in both natural and

Pages 15-26. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

social systems. Finally, we will illustrate the need for the interdisciplinary concept of "risk analysis" and the precautionary approach in decision making locally as well as globally, using the case of the "Ballast Water Convention"

RISK ANALYSIS FOR THE EMERGING RISKS FROM INVASIVE ALIEN SPECIES

Interdisciplinary nature of risk analysis

It has been argued that many of the emerging biological hazards affecting biodiversity loss or resulting in the global expansion of new infectious diseases to humans, animals and plants, have been technologically and socio-culturally induced, because they are deeply associated with our modern ways of production and consumption in the post-industrial society. Moreover, these risks are often latent, invisible, or inter-generational and inter-specific and can usually be characterised as "of low frequency but with catastrophic consequence". Hence, these risks are often understood as "virtual-reality" in terms of the difference between the possibility and the reality (Renn and Klinke 2001)

To deal with those types of emerging risks, the modern concept of "risk" has evolved from the conventional view of "risk" as "expected value of the probability of a hazardous event occurring multiplied by the magnitude of the consequence of the hazard" into an ontological or sociological concept of "risk" that allows us to take into consideration a wide rage of socio-cultural characters of emerging risks from biological hazards. Among such attempts to define the interdisciplinary concept of risk we take one of the simplest ones. That is:

> "*The potential for the realisation of unwanted, adverse consequences to human life, health, property, or the environment*".

This simple definition (*http://www.sra.org/resource_glossary.php/*) originated from the extended discussions on defining an interdisciplinary concept of risk at the SRA (Society for Risk Analysis: a professional and academic association founded in 1982 with major memberships in the USA, Europe and Japan). In this context, when we specify "a potential" in the framework of risk analysis, it should be noted that a certain degree of value judgement on "adverse or unwanted" has been included. To address this issue of "value" inherent either explicitly or implicitly in any concepts of risk, Kaplan and Garrick of the SRA proposed the expression in terms of a "risk triplet" (Kaplan and Garrick 1981). The risk triplet consists of scenario S, likelihood P, and possible consequence D in relation to the following three basic questions of risk analysis:

1) What is the nature of the event that can happen?
2) How likely is it?
3) What are the consequences?

$$\text{Risk} = R\{< S_i, P_i, D_i >\}$$

S_i: a set of scenarios concerning the nature of the possible events

P_i: a set of likelihoods concerning the frequency, probability or ambiguity,

D_i: a set of consequences concerning the unwanted possible damages to humans, animals, plants, or the environment.

The relationship between the conventional concept of risk (P_i x D_i) and this interdisciplinary one R {< S_i, P_i, D_i>} is illustrated in Fig. 1 (Ikeda 2004). In addition to the scientific knowledge of Pi and Di, scenario Si can put questions and conditions on ontological and socio-cultural factors in relation to our anthropocentric activities in the complex and uncertain worlds. For example, in cases of risk analysis for invasive alien species, the scenarios will try to identify what are the most critical endpoints to be assessed in terms of possible impacts to human, animals or plants, to achieve better risk management. As generic indices of endpoints, we could, for instance, take the number of species endangered, declining population of native species, loss of habitats or landscapes, disease outbreaks in human and non-human environments, economic loss of agricultural or fishery products and so on, depending on our concerns in relation to the biohazard under consideration.

Once we have identified the adequate endpoints to be assessed, the next step is to specify the

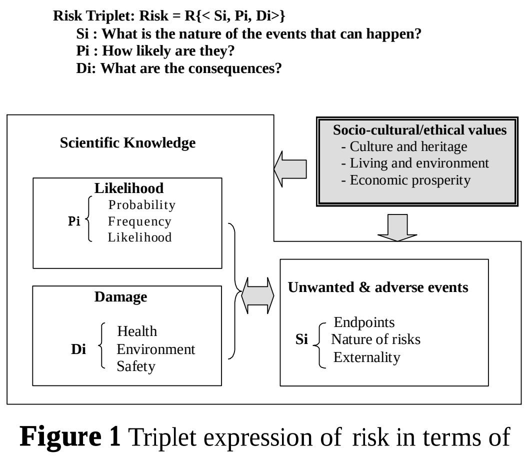


Figure 1 Triplet expression of risk in terms of interdisciplinary concept

particular types of risks that are associated with the endpoints, both in terms of likelihood and of damage. Then we also need to explore possible "external impacts" which might be accidentally or intentionally brought about, by posing the following questions:

1) Are the impacts or damages within our management limit or are they out of our control?
2) What scale or degree of irreversibility could these impacts have?
3) Are there likely indirect impacts on other genotypes or species, beyond the biological and ecological boundaries?
4) Do these impacts pose a moral or ethical problem with regards to nature or global ecosystems?

These are typical questions to arrive at scenarios (S_i) that should be clarified in the stage of problem formulation, when we begin the scientific or objective evaluation of P_i and D_i. It is the role of "risk analysis" to provide answers to these questions either in qualitative or quantitative ways in relation to the risk triplet R = {< S_i, P_i, D_i>}. Hence, the framework of risk analysis consists of identifying the possible biohazards, assessing the likelihoods associated with vectors and pathways, the extent of exposure to the endpoints, and their responses to the exposures.

Risk analysis framework for ecological and biological hazards

An example of "risk analysis" in the biological or ecological context is the "ecological risk assessment" (US EPA 1998) that was based on the classical framework of risk analysis proposed by US National Research Council (US NRC 1983). The NRC framework that was intended primarily for regulating hazardous chemicals to protect human health and protect against environmental degradation consists of the following three processes (see the upper part of Fig. 2 as modified for ecological risk analysis):

1) *Research Processes* (or problem formulation) on the possible risk events in laboratories, fields, communities, mostly based on both natural and social sciences.
2) *Risk Assessment* which provides an objective and integrated judgment in terms of scientific evaluations on hazards identification,

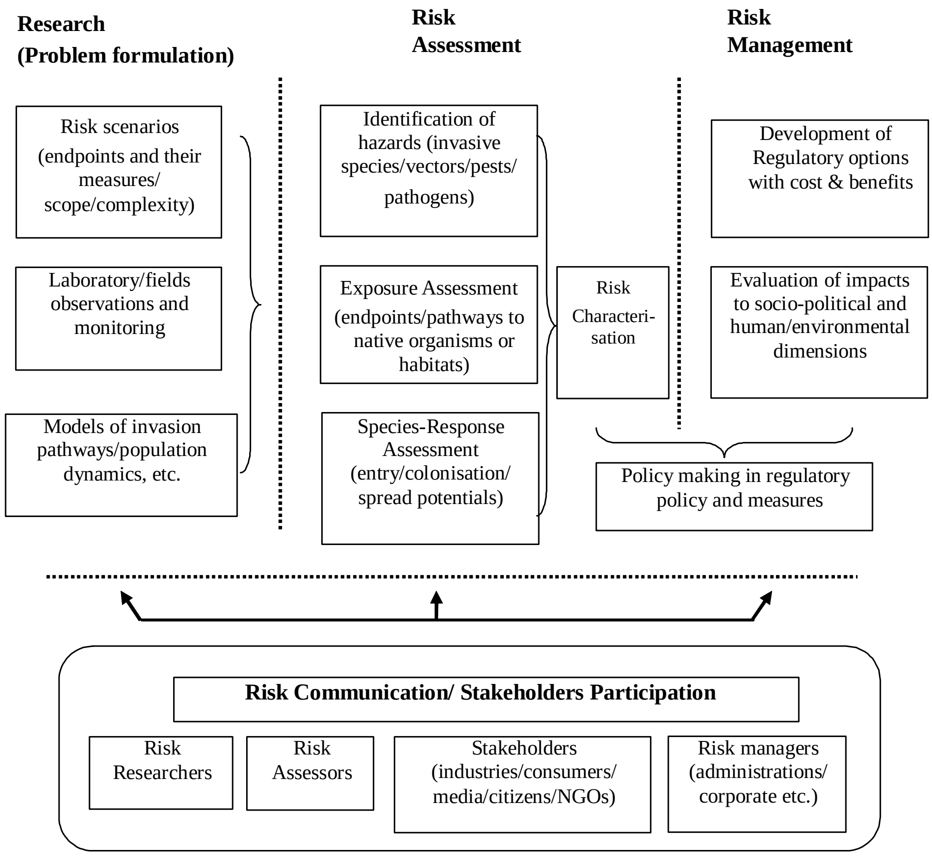


Figure 2 An example of a Risk analysis framework for ecological and biological hazards (Modified from the classical framework by NRC 1983)

dose-response, and exposure assessment as a form of risk characterisation,

3) *Risk Management* which is a subjective decision making process of selecting regulatory measures among alternative options, in conjunction with the outputs of scientific risk assessment and other socio-economic and cultural conditions.

One of the most important features of the NRC framework is to have made a clear conceptual or functional separation between "risk assessment" and "risk management". The explicit representation of the risk scenarios sits at the interface of the two processes for making better assessment and regulatory decisions, while dealing with high scientific uncertainty and complexity of values among researchers, assessors, managers and stakeholders. However, after several controversial cases failed to gain public acceptability or credibility, the critical role of *risk communication* and stakeholder involvement in risk analysis has been widely acknowledged as the fourth process in the risk analysis framework (shown in the lower part of Fig. 2).

Risk analysis for biological or ecological hazards could follow the same processes by modifying some of the elements in the general risk assessment procedures to be more focussed on biological invasion. For example "possible hazards or stressors" could be replaced by "invasive species" or "vectors", their "exposure to the ecological endpoints" by "pathways" to native indigenous species/ecosystems, and "dose-responses" by "species-responses" (as displayed in the upper part of Fig. 2). In addition, because it provides for participation of stakeholders throughout the entire cycle of risk analysis, the process of risk communication has been advocated, in particular, for cases where there is high scientific uncertainty and where the issue of values of stakeholders is complicated. Risk communication creates a social platform to share different risk perceptions, associated with the human dimension of invasive alien species (US NRC 1996).

Following the ecological risk assessment framework (US EPA 1998), several frameworks have been proposed for assessing risks, targeting invasive species associated with traded commodities or with transport, such as in the case of ships' ballast water. A typical framework is the "Generic Non-indigenous Aquatic Organisms Risk Analysis Review Process (abbreviation: Review Process)" developed by the U.S. Aquatic Nuisance Species Task Force. The Review Process consists of three major steps (Orr 2003):

1. *Initiation (problem formulation) step*: seeking the data for invasive species in terms of their possible origins, pathways and past and present mitigation actions. This allows the development of risk scenarios.
2. *Risk assessment step*: preparing a list of alien organisms of concern with their pathways, and then, conducting a species-based "organism risk assessment" in terms of probability of species establishment and consequences of the establishment (e.g. economic impacts). ,
3. *Risk management step*: developing mitigation measures and procedures primarily based on the output from the risk characterisation.

One of the interesting features of the Review Process is that it uses a simple and flexible set of codes indicating the degree of uncertainty that the risk assessors encountered, from a level of high certainty up to a level of high uncertainty. Uncertainty can originate from assessment methodology, assessor's bias or errors, or biological/ecological unknowns. The assessment also includes codes indicating the types of references used, e.g. literature, extrapolation from related species, etc. (Orr 2003). In their model, the "organism risk assessment" is expressed by the following forms of P and D in the risk triplet Risk = R {< S_i, P_i, D_i>}:

P: Probability of establishment:

$P = (X_a) \cdot (X_b) \cdot (X_c) \cdot (X_d)$

(X_a) = probability of being associated with possible pathways.

(X_b) = probability of entry potential surviving in transit.

(X_c) = probability of maintaining population in the habitat.

(X_d) = probability of spreading beyond the habitat area

D: Consequences of establishment:

D = Ya + Yb + Yc

Ya : Economic impact

Yb: Environmental and ecological impact

Yc: Socio-cultural impact (human-dimensional influence) (source: Orr 2003)

As far as a specific risk assessment framework for ballast water is concerned, a variety of models have been proposed, mostly qualitative, but some quantitative. For example, the Norwegian Maritime Directorate developed an integrated assessment model called EMBLA (Environmental Ballast Water Management Assessment). The model has three components: 1) Initial and detailed hazard screening, 2) Hazard Analysis, and 3) Impact/Consequence Assessment. It has basically the same structure as the US Review Process, but it is oriented more to practical operations to reduce invasion risks of alien species. It aims to combine realistic policy measures

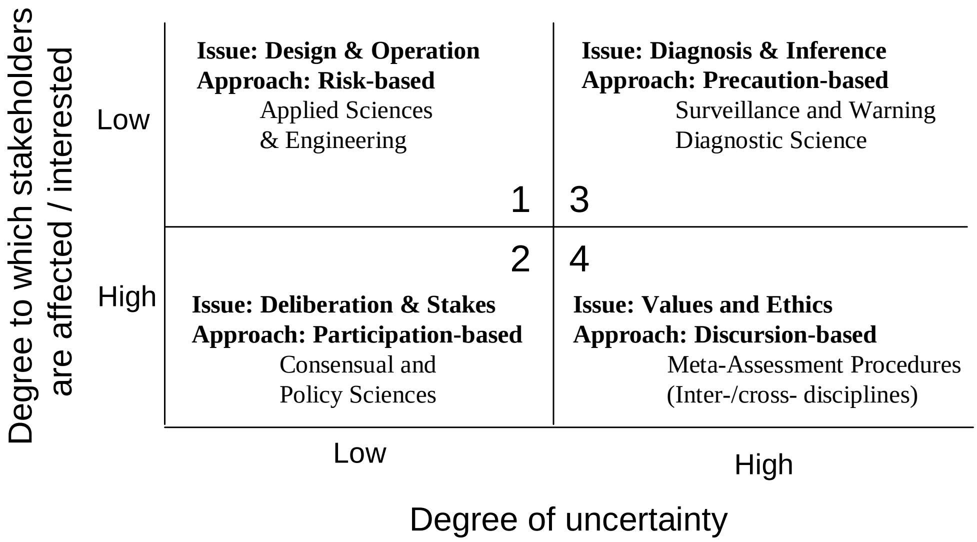


Figure 3 Classification of "nature of risk"

and risk assessment in order to prevent the transfer of harmful aquatic organisms (Norwegian Maritime Directorate 2005). The Australian Quarantine and Inspection Service (AQIS) developed a risk assessment model for ballast water, based on event-tree analysis, where probabilities are assigned in a series of possible events (Hayes 2003):

Risk of "ballast water" = p(a) p(b) p(c) p(d)

where

p(a) :contamination of donor port with the invasive species,
p(b) :infection of vessel with the species,
p(c): survival of the species during transport, and
p(d): survival of the species in recipient port.

It is, however, obvious that some of the most difficult issues involved in "risk analysis of the biological hazards" have not been fully addressed yet. In particular, it is essential to produce a reasonable set of interdisciplinary risk scenarios S_i so that they can work as a bridge between scientific assessment and subjective policy measures in order to achieve the management goals. The task is therefore to develop possible risk scenarios Si that focus on the specific nature of risks from biological invasion, while dealing with high uncertainty and complexity. The risk scenarios must be able to be used under the current regulatory institutions, monitoring or surveillance systems and resources limitation and must be able to address differences in risk perceptions and in social or ethical values among stakeholders in terms of biodiversity or biosecurity. In addition, local, national and global perspectives need to be accommodated as appropriate.

RISK ISSUES AND MANAGEMENT APPROACHES TO DEAL WITH UNCERTAINTY AND COMPLEXITY

We can classify the above described "nature of the risk" into the following four areas by dividing two axes into "high" and "low" respectively (Fig. 3). One axis is used to indicate the degree of uncertainty of our knowledge (a scientific evaluation axis) and the other axis to indicate the degree to which stakeholders are affected or interested in the risk events (a socio- economic/cultural/ethical evaluation axis) (Ikeda 2000).

Area 1: This is an area where scientific knowledge about risk events is fairly certain and there are few value-based stakeholder issues. An evaluation of the results of the scientific assessment is possibly using objective indices or standards. Here, we can rather logically take regulatory measures for risk reduction based on the objective risk assessment. In this area, the most important management issue is the objectiveness of risk information so as to ensure accountability for the regulatory decisions and choice of options to reduce the risks up to some acceptable levels. The management strategy is a **"**risk-based**"** regulatory approach under the existing legislative framework including setting an acceptable level of risks, monitoring and surveillance for invasive species, etc.

Area 2: This is an area where scientific knowledge about risk events is fairly certain, but there are rather significant issues among the stakeholders influencing their evaluation of the outcomes of the risk assessment. In this area, a "participation-based" consensus-building approach to risk management is required as a major management strategy. Here, it becomes critical to ensure not only the reliability but also the transparency of the procedure of risk assessment. This requires not only that there is stakeholder participation at all levels of the regulatory decision making, but also that some democratic institutional arrangements are in place to ensure such stakeholder collaboration.

Area 3: This is an area where the level of uncertainty in scientific knowledge is fairly high, but there are few value-based issues among the stakeholders when evaluating the results of the risk assessment, in spite of uncertainty. Examples are infectious diseases or epidemics. Here, it is inevitable that the risk assessment includes a high degree of qualitative scenario or subjective scores under high uncertainty. In this area, the main management strategy is to effectively promote "precaution-based" diagnosis and presumption under high uncertainty. Hence, it would be desirable to allocate significant resources, not only to monitoring or surveillance systems for early warning, but also to the development of risk communication for sharing the risk information with the experts and public.

Area 4: This is an area where uncertainty in scientific knowledge is high and there are significant issues among stakeholders, resulting in potential conflict when evaluating the assessment and/or in relation to the setting of acceptable levels of risk. Since existing approaches based on scientific and objective indices or standards can not be used in this area, it is necessary to develop a "discursion-based" meta-science which evaluates both scientific and socio-cultural factors (human dimension of invasive species problems) including anthropological, ethical and value judgment issues. At the same time, as is in "Area 3", we need to create a social platform of "risk communication" to allow discussion of the different risk perceptions among the stakeholders.

In Areas 1 and 2, we can make scientific assessments of possible events either qualitatively or quantitatively, with low uncertainty – for example, the probability of traffic accidents or odds ratio of excess deaths by SO_2 air pollution above some regulatory SO_2 standard. In such cases, the important management issues are 1) how to properly choose risk scenarios and how to decide on monitoring and regulating the related indices or measures for the risks on the endpoints, and 2) how to ensure stakeholder participation in the decision making and consensus building processes. Some of the specific risk management for invasive alien species in the quarantine systems under the SPS agreement falls in these domains, where we have reasonably good scientific data, allowing objective assessment.

However, since most of the emerging risks from invasive species are likely to be in situations of high uncertainty in our scientific knowledge, risk management issues relating to protecting native biodiversity/ecosystems from impacts by new genetically modified species or by alien, potentially, invasive species fall into the Areas 3 or 4. Whether the case will be in Area 3 or in Area 4 depends on the of socio-cultural issues among the interested stakeholder groups who have risk perceptions deeply rooted in their ontological or anthropological perspectives beyond the conventional science paradigms. Here, we need an approach of stakeholder participation in decision making that centres around a "precautionary framework or principle" with the support of a discursive "risk communication" among stakeholders, as is displayed in the lower part of Fig. 2 (Renn and Klinke 2001). We use "risk governance" as a specific type of risk management strategy in which interdependence or intercommunication among the stakeholders involving governmental or non-governmental actors is essential in the pursuit of collective decision making aimed at attaining policy goals.

PRECAUTIONARY APPROACH TO EMERGING RISKS, UNDER HIGH UNCERTAINTY AND COMPLEXITY

The term "precautionary approach" first appeared in the ministerial declaration of the second International Conference on the Protection of the North Sea, in London (1987), reflecting increased environmental concern about the marine and coastal pollution problems, such as oil spill accidents and the associated massive death of marine mammals in the 1980's. A typical example is the Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR Convention, 1992) which introduced a "precautionary regulation on any substances" discharged into the sea in spite of the limitations in scientific knowledge (and monitoring data) to anticipate potential threats (catastrophic events). This strict form of precautionary measure was, in fact, supported by most of the coastal countries who began to shift their regulatory approach from the traditional ones to the precautionary one (Horiguchi, 2000). Since then, the precautionary approach or precautionary principle (as it is called in some other environmental policy areas) has become one of the basic concepts to address the emerging environmental risks in the fields of marine pollution.

Because of the history of this concept, there is variation in the formulation of the precautionary approach or precautionary principle depending on the aims of the conventions or agreements. In addition, a number of different terms are used, such as "precautionary framework", "precautionary approach" or "precautionary principle". Here, we use "precautionary approach" which the international organisations of WHO and FAO are currently using as a term in the context of the problems of public health, food safety and animal or plant sanitary issues. The different versions of the concept, for instance, appeared in: 1) The declaration from the ministerial meeting of the North Sea marine pollution protection in 1990: 2); Rio Agenda 21 of the United Nations Environment Programme 1992; 3) Wingspread Declaration in 1998, and 4) EU Guidelines, 2000, and so on. The common features of the above examples can be formulated in terms of "*threat*", "*uncertain*", "*action*", and "*mandatory*" (Sandin 1999) as follows:

> If there is (1) a *threat*, which is (2) *uncertain*, then (3) some kind of *action* is (4) *mandatory*.

The former two terms are associated with the evaluation of the scientific knowledge about the nature of the risk event. The latter two terms are related with decision making about the management strategy or regulatory measures. Here we have again two substantial dimensions of uncertainty in scientific knowledge and complexity relating to socio-cultural issues. In more elaborated words, we could state that:

> -- If there exists a serious or irreversible *threat* (potentially dangerous) to human health and the environment, and also
> -- the causal association between the hazards and the endpoints is *uncertain* (not fully scientifically established),
> -- then, *action* (preventative or regulatory measures) is *mandatory* (with the status of the action subject to some conditions, e.g. balancing "cost-effectiveness" or "costs and benefits", depending on institutional or resource constraints).

In addition to the above conditions, the Wingspread Declaration 1998, which is one of the strongest assertions, added another condition, about the burden of proof being on the developers (promoters of the development). However, as far as the precautionary regulation or management issues are concerned, many problems remain ambiguous and require further discussion. These are, for example, 1) the degree of threat or irreversibility where precautionary measures should be taken without full scientific validation of the link between biological invasion and consequences, 2) the content of precautionary measures, and 3) the cost burden for malpractices, including the problem on what degree of the burden of proof should be required, etc. O'Riordan (1999) has elaborated six core elements of the precautionary approach which are relevant to the discussion of stakeholder participation in decision making in the context of biohazard risks from invasive alien species:

1) Action before scientific justification
2) Proportionality of response (cost effectiveness)
3) Ecological margin to allow for human ignorance
4) Shift of the onus of proof onto polluters
5) Concern for future generations
6) Ecological debts incurred by developed countries in the past

Among these elements, a couple of policy elements, such as "proportionality of response", "shift of the onus of proof onto polluters" and "ecological debt", are closely related to the issues of balancing costs and benefits of preventive actions or balancing ecological effectiveness and ethics of policy measures under high scientific uncertainty and complexity of stakeholder issues. The first of these elements is concerned primarily with the adequacy of precautionary actions in terms of the cost-effectiveness or opportunity cost during the whole management process. As for the second of these elements, for example, the ballast convention imposed responsibility to the administrators of the ship with some exceptional conditions of granting discharge of ballast water when ships operate only between specified ports. In general, it considers that a proof of safety is required beforehand and potentially affected states shall be consulted aiming at solving the concerns. The latter element of "ecological debts" is related to the sensitivity or vulnerability of global marine ecosystems and the responsibility of developed countries in terms of dumping industrial pollutants and over-utilisation of resources.

In the following section, we will discuss how these core elements of the precautionary approach are taken into considerations in the draft convention of ballast water as a major institutional vehicle towards decision making to address the emerging risks of biohazards from invasive marine organisms, incorporating the precautionary approach and stakeholder participation.

THE EXAMPLES OF BALLAST WATER AND RISK GOVERNANCE

Risk management scenarios in the context of the IMO ballast water convention

Ballast water, which is freshwater or seawater loaded at a cargo discharging port, is necessary for a ship's safe operations to maintain stability. When it is discharged at a cargo loading port, organisms in the ballast water create a set of potential risks not only to native marine species, but also to economic values and human health, e.g. through harmful algae blooms, infectious diseases, etc. Once an alien species has established within a new ecosystem it is usually impossible to eliminate and restore the ecosystem back to the original. The volume of ballast water transported worldwide is estimated at about 10 billion tons per year. It is estimated that Japan, exports 300 million tons and imports 1.7 million tons of ballast water per year (Kikuchi, 2001). In addition, there is a risk that biological invasions will become more prevalent due to global climate change.

The first official report of damage caused by ballast water was by the WHO in 1973, on a cholera epidemic in Latin America. Although the WHO concluded that ballast water accounted for the 1991 epidemic in Peru (WHO 1999), some experts insisted that there were many other possible vectors of *Vibrio cholerae*, and thus, ballast water should not be blamed on its own. (Seas, *et al.* 2000). During the1980s, a global scale of toxic phytoplankton blooms in most of the coastal zone of marginal seas strongly suggested a possible role for ballast water in the global transportation of invasive alien species (Carlton 1985). Nevertheless, scientific or biological mechanisms of the release of ballast water in the establishment of invasive alien species have not been well understood (Ruiz, *et al.* 2000). For example, in the case of the paralytic shellfish poisoning on the coast of Tasmania islands in 1986, either Japan or Spain was suspected as the origin of ballast water that acted as a vector for the toxic plankton sampled from the ballast tank of a log bulk carrier. However, toxicological research could not verify that species in the ballast water were definitely the sources of the shell fish epidemics (Oshima, 1992).

Because of the global and trans-boundary nature of invasion by marine alien species, individual countries and shipping societies have addressed concerns in relation to ballast water by preparing their own guidelines based on precautionary measures. Countries with significant concern about potential impacts of marine invasive alien species on their native marine resources and ecosystems worked towards the establishment of a global legal framework beyond territorial waters. Eventually, in 2004, the International Maritime Organisation (IMO) adopted the convention on the management of ships' ballast water at a diplomatic conference in February 2004. Ratification is anticipated by 2009 (GloBallast 2003).

According to the draft convention adopted by IMO in 2000, one of the most important issues of risk governance is how to introduce mandatory regulations to minimise the transfer of harmful aquatic organisms and pathogens from original habitats to other places while increasing the level of participation from port- and flag-states and non-governmental bodies, and other countries who are concerned about marine invasive alien species. Such mandatory regulations requiring a "precautionary approach" include measures to lessen the chance of taking on board harmful organisms along with ballast water, prevention of loading and discharging ballast water in shallow water, procedures for the exchange of ballast water at sea and for discharge to reception facilities at port-states.

As expected under regime theory, developed in the field of policy and political sciences (Young and Osherenko, 1993), the development process of the ballast water regime can be divided into the following three stages according to the degree of scientific knowledge and values and participation of the actors and stakeholders concerned - based on both scientific reports of ballast water prepared by the joint GEF/IMO/UNDP project (Gollasch 1997, IMO 1998, GloBallast 2002, 2003, 2004) and our interviews to the port regulatory authorities in Japan (Hijikata and Ikeda 2003):

1) guideline formation (before the adoption of the IMO guidelines in 1991),
2) guideline development and review (from 1992 to 1997),
3) convention formation (from 1998 to 2004).

In this section, we will examine how the framework of "risk analysis" relates to regime analysis of the ballast water convention. Then, based on examination of the development process of the draft ballast water convention, we also discuss what this show us about interdisciplinary or human dimensional factors that we need to incorporate in the framework of risk analysis for emerging risks from invasive alien species. In order to examine the core governance issues relating to the ballast convention, we need to specify key stakeholders and actors as well as the specific factors and issues of importance to stakeholders that participated in the regime formulation.

Actors in the ballast water convention

1) Port states (the US, Canada, Australia, Germany, Brazil, etc.):

They are the primary actors that insisted on imposing mandatory regulations. However, within those countries the issues of concern are not completely the same, for example, in the US and Canada the concern focused on economic costs incurred to control invasive species; in Brazil and Chile there was an interest in reducing the risks to human health and aquaculture contamination (e.g. concerns caused by pathogens such as *Vibrio cholera*), and in Australia and New Zealand there was a concern about degradation of tourism resources, as well as concern about red tides and shellfish poisoning.

2) Flag states (Panama, Liberia, Greece, etc.):

Since the number of flag-of-convenience ships is increasing, control of the flag states in regards to compliancy with international regulations has become "loosened". In addition, since flag states are often port states at the same time (such as Panama or Japan), there may be conflicting interests. As for Japan, it has insisted that regulations should not hamper trade.

3) Associations of ship owners (International Association of Independent Tanker Owners (INTERTANKO), The International Chamber of Shipping (ICS):

While they are observers at the Marine Environment Protection Committee (MEPC), at the Ballast Water Working Group (BWWG) and Correspondence Group, they directly contributed to the formation of the regime by stating opinions and submitting comments. They provided voluntary guidelines. They stressed not only the need for affordable treatment techniques, but also the intention to participate in the preparation of IMO's guidance, when the Convention was adopted.

4) Environmental NGOs (e.g. The World Conservation Union (IUCN), Friends of the Earth International (FOEI):

They mainly focus on conservation of biodiversity and have made a continued effort with campaigns to improve the awareness of marine invasive species problems.

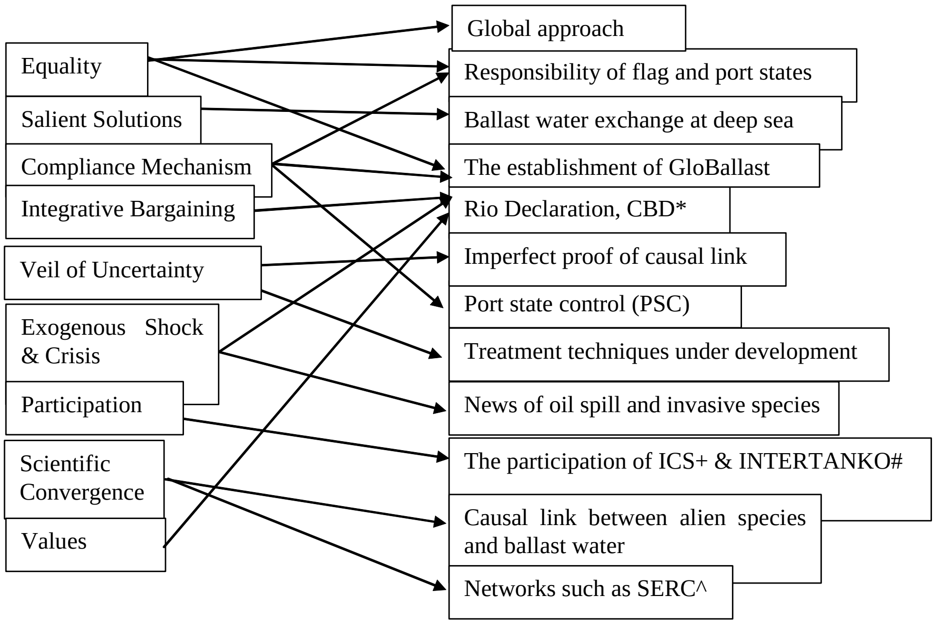
5) Governmental scientific institutes (Smithsonian Environmental Research Centre (SERC), EU Concerted Action, Centre for Research on Introduced Marine Pests (CRIMP), etc.):

The experts have formed an international network among institutes through sharing information or cooperating on research projects, such as SERC with research institutes in Australia, Germany, Israel, Italy, The Netherlands and New Zealand (Gollasch, 1997). The researchers contributed indirectly by giving policy advice or participating as delegates. Ecological experts claimed that only 100% treatment/ sterilisation could be effective to avoid the establishment of unwanted species.

Development of a regime: relation between general factors and specific issues in the ballast water regime

The relationship between general factors in the development of a regime and specific issues in the case of the ballast water regime are mapped out in Fig. 4, based on the regime model of Young and Osherenko (1993).. Such general factors are, for examples, "scientific convergence", "veil of uncertainty", "salient solutions", "exogenous shock and crisis", "participation", "value", "equality", "compliance mechanisms" and "integrative bargaining" and they are shown in relation to the ballast water specific issues (Fig. 4) .

Our analysis shows that the first stage of the development of the ballast water regime strongly experienced "exogenous shock" due to the damage caused by invasive alien species and pathogens. Then, "scientific convergence" emerged, that is, a common belief in a causal relationship between ballast water and the damage. However, the belief still lacked full scientific proof at this stage, so that the "veil of uncertainty" still functioned. This factor motivated various actors to protect their interests and aim for development of a regime incorporating and dealing with uncertainty. The existence of a "salient solution" (such as ballast water exchange in the open sea), based on a common understanding of marine ecosystems, persuaded policy makers to support a pro-active approach.



* CBD - Convention on Biological Diversity
+ ICS – International Chamber of Shipping
# INTERTANKO – International Association of Independent Tanker Owners
^ SERC - The Smithsonian Environmental Research Centre

Figure 4 Development of a regime: relationship between general factors and specific issues in the ballast water regime development (Source: Hijikata and Ikeda 2003).

At the second stage, a change in "values" caused by the adoption of the Rio Declaration and the Convention on Biological Diversity (CBD) in 1992 was a very important factor that enabled integrative bargaining among actors and stakeholders in the next stage. The values held by the actors worked toward the integration of the conservation of biodiversity into considerations at a global level, although it was not fully integrated in considerations at a local level. In addition, "participation" of the ship owners association in the development of the regime was important. The International Chamber of Shipping (ICS) and the International Association of Independent Tanker Owners (INTERTANKO) actively participated in the development of the regime and presented the viewpoint of ship operators.

Finally, in the stage of the regime formation, the "equality" factor contributed, as in response to the increase in different voluntary regulations by individual states and ports, some actors - especially ship owners - began to ask for an international, global, convention. "integrative bargaining" was achieved by the change in values at this stage so that actors benefited from the formation of the convention to conserve biodiversity, which was the common value. "compliancy mechanisms" that could be guaranteed primarily though port state control (PSC) promoted the formation of legally binding international rules in the third stage. Preceding success in PSC (port state control) increased the confidence in effective compliancy.

Our analysis revealed that there are interactions

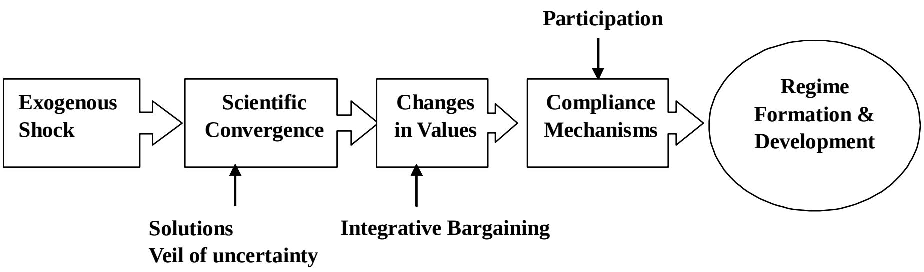


Figure 5 Critical factors and issues in the development of the ballast water regime (Source: Hijikata and Ikeda 2003).

between factors. For example, “changes in values” enable “institutional bargaining” to be an “integrative” one. Also, “scientific convergence” stimulated the emergence of “solutions” and a way to lift the “veil of uncertainty”. “Compliancy mechanisms” functioned, partly because of participation of ICS and INTERTANKO. The most influential factors (keeping in mind interaction) in the development of the ballast water regime, are outlined in the flow diagram in Fig. 5.

CONCLUDING REMARKS: TOWARDS BETTER RISK GOVERNANCE

An interdisciplinary framework for “risk analysis” is presented focusing on the issue of stakeholder participation in decision making when dealing with risks from invasive alien species. Such risks include biodiversity loss and global expansion of new infectious diseases. Since these risks are likely to be latent or inter-generational and likely to be catastrophic, but with low probability, they are often thought of as “virtual-reality” in terms of the difference between the potential and the reality in our generation. To deal with such risks, an interdisciplinary concept has been developed that allows us to include a wide range of socio-cultural aspects of emerging risks from biological hazards.

The expression of inter-disciplinary concept of risk as a “risk triplet: R {< S_i, P_i, D_i>}” (Kaplan and Garrick 1981) asks risk assessors or risk managers to answer the following three basic questions:

1) S_i: risk scenario: What is the nature of the events that can happen?
2) P_i: probability: How likely is it?
3) D_i: damages: What are the consequences?

Among them, the scenario Si shall put questions and conditions on “acceptable level of risks” and “management options for better risk governance” in relation to our anthropocentric activities in the complex and uncertain world. A clear representation of the risk scenarios can work as a catalyst at the interface between risk assessment and risk management under the condition of high scientific uncertainty and complexity of values and issues among stakeholders in our post-industrial society.

The precautionary approach is fundamental in the decision making where scientific uncertainty and complexity, both ecological and socio-cultural, need to be addressed. This was examined in the context of the development of the “Ballast Water Convention”. It seems that this convention has reflected most of core elements of the precautionary approach: first, it takes some “mandatory preventive actions” before scientific proof: second, it attempts to shift the burden of proof ships’ administration: and third, it allows for a “proportional response” to socio-economic issues in terms of cost-effectiveness during the management process. In fact, when the precautionary approach is practiced, decision makers or risk managers have to address the fact that actions to avoid biological invasions, such as the exchange of ballast water in the open sea may produce operational safety risks for crews and ships.

Our analysis showed that there are several critical interactions between the difficult regime factors and the ballast water specific issues that influence the development of the ballast water regime. Those includes such interactions among “changes in values”, “scientific convergence”, “veil of uncertainty, “compliance mechanism” and “integrative bargaining” as illustrated in Fig. 5 that set the stage for the remaining issues to be explored for attaining better risk governance in the context of the emerging risks from invasive alien species. That is to say:

1) The public, including in fishery and marine industries, NGO’s, and researchers were mobilised due to global networking and access to information and data on risks, impacts and concerns. Those networks, and the proliferation of a wide range of stakeholders, will facilitate “changes in values” and “scientific convergence” among stakeholders which will allow better risk governance.

2) The development of education or training programmes such as the Global Ballast Water Management Programme for developing countries initiated in 2000 by IMO (GloBallast 2000) facilitated port regulatory systems. This could also assist “changes in values” as well as increase the reliability of the regime in the legal sense.

3) The revised guidelines 1997 were not legally binding. Different country-based regulations created potential confusion and an incentive to develop further a variety of “compliance mechanisms” that clarified the responsibility of ship-owners and state authorities.

4) New investment in research to decrease the “veil of ignorance” associated with ballast water treatment can lead to effective regulatory measures.

In the East Asian Sea, it seems that we are just at the beginning stage of “scientific convergence” in the case of the ballast water regime, since experts have just started to discuss risk scenarios and issues associated with ballast water transportation into Asian coastal waters. As indicated at the first East Asia regional workshop on ballast water, the issues are not yet studied well (GloBallast 2003). In order to reach the final stage of regime formation, there must be a "change in values" and enough capability for "integrative bargaining" to end up with better governance. This could be done by promoting the sharing of information on scientific research on harmful algae blooms and on other marine impacts through a variety of networks from the grass roots up to government agencies. Risk analysis in the context of an interdisciplinary framework is likely to contribute to lifting "the veil of ignorance" by making risk scenarios more explicit through identification of sources and pathways of biohazards, and characterisation of the risks to biodiversity and ecosystems. It could also serve as an indispensable tool for regulatory decisions and for collaboration among stakeholders, at bilateral as well as multilateral levels.

ACKNOWLEDGEMENTS

This work was funded in part by both Grand-in-aid for scientific research from Ministry of Education, Culture, Sports and Technology, Japan, and Research Institute of Science and Technology for Society, Japan Science and Technology Agency. The author thanks Dr. Toshiaki Mizuno, WWF Japan and Naomi Hijikata, University of Tsukuba for their valuable assistance in preparing this manuscript.

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Legal strategies for combating invasive alien species

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Abstract With only a few exceptions, nations lack adequate legal mechanisms for addressing the significant environmental and economic impacts created by invasive alien species. Statutes are too limited in scope and lack flexibility to deal with changing information. Common-law tort schemes provide the flexibility that statutes lack, but have proof of harm requirements that can be insurmountable in the context of invasive species. Creating a tort cause of action through statutes would strengthen the common-law scheme without losing any of the flexibility. When combined with permitting requirements that impose technological controls and "dirty lists" (and proven "clean list" exceptions), as well as citizen enforcement, such a statutory scheme would allow governments and individuals to effectively prevent or control the introduction of invasive alien species. Because invasive alien species do not recognise international borders, the same legal concepts should be reflected to the extent possible in international treaties addressing this significant and growing ecological and economic issue.

Keywords: legal mechanisms to address invasive alien species; invasive species (USA); Boundary Waters Treaty; NAWS

INTRODUCTION

Over the last two centuries, advances in human technology have eroded the natural geological barriers that separated distinct ecosystems. As a result, wildlife that historically evolved discretely has begun to mingle. The effects of the invasion of alien (= non-native) species have been devastating. Invasive alien species may compete with native species for natural resources.[1] Predation on native species can also be a problem.[2] Furthermore, because alien species may lack natural predators, other limiting factors, in their new ecosystem, population growth and/or the spread of alien species can go unfettered, and they can become invasive.[3] The resultant impacts on the environment can trigger staggering economic consequences. For instance, zebra mussels, invaders in the Great Lakes, clog water intake pipes.[4] Indeed, reports estimate that in the United States alone, over $137 billion is lost annually in crop, rangeland and waterways damage from invasive alien species, and the associated control costs,[5] resources damage and lost economic production, and approximately $97 billion of these losses are associated with only 97 aquatic nuisance species.[6]

Unfortunately, with only a few exceptions, nations lack adequate legal mechanisms for addressing this significant environmental and economic issue. In order to prevent the introduction and spread of invasive alien species, an effective legal remedy must meet several criteria. First and foremost, an ideal legal response should be forward-looking in nature, while also providing sufficient deterrence against the introduction of invasive alien species. Because "it may not be possible to contain or remove"[7] an invader once it has been introduced into the wild, an effective solution should emphasise preventing the introduction of invasive alien species.[8] Additionally, the legal remedy must be broad enough to encompass not only species that are known to be invasive, but also those that have the potential to wreak environmental damage.[9] At the same time, the legal framework must accommodate introductions of alien species that have been determined to be "safe." Finally, because biological invasions respect neither state nor national sovereignty, an ideal legal remedy will be able to prevent or address threats without regard to political boundaries.

In addition to these substantive requirements, any legal remedy must be procedurally reasonable and feasible to implement and enforce. It also must make any remedies readily available so that invasions - or potential invasions - can be dealt with quickly, before major environmental or financial damage has occurred.

In the United States, an effective legal framework for addressing invasive alien species issues has yet to be developed. Statutes adopted to address the problem afford little, if any, prospective relief.[10] The relief that is available too often uses "dirty lists," which prohibit importation of species already known to be pests, instead of instituting broader guidelines to curb potentially damaging introductions.[11] State law varies considerably,[12] and international law that could

Pages 27-34. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks.* Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

otherwise be applied within the United States essentially does not exist.[13]

This paper discusses potential legal approaches to combating invasive alien species. Part I sets the stage by briefly describing some examples of potential and actual alien species invasions that illustrate the significant complexity surrounding this problem. Part II describes the adequacy of current statutory responses to invasive alien species and difficulties with drafting statutes to deal with the issue. Part III analyses possible common-law[14] tort[15] responses to the problem and explains why they are inadequate. Finally, Part IV argues that a legal framework that combines statutory, international, and common-law tools would be most effective against an invasive alien species threat.

EXAMPLES OF THE CHALLENGE

The Potential Introduction of Alien Species into the Hudson Bay Basin and Ballast Water Regulation

Combating invasive alien species through legal means can occur proactively and reactively. What follows is a description of an example of each.

Several planned projects along the northern border of the United States have the potential to introduce invasive alien species into the Hudson Bay drainage basin.[16] The most significant in this context, the Northwest Area Water Supply Project ("NAWS"), proposes that water be piped forty-five miles from Lake Sakakawea, in the Missouri River basin, to rural communities in North Dakota.[17] The proposal will require water to be piped over the Continental Divide into the Hudson Bay basin.[18] Because the two aquatic ecosystems have been separated for millennia,[19] potential problems with invasive alien species may arise,[20] including the potential introduction of zebra mussels into the Hudson Bay.[21]

NAWS is currently the subject of a lawsuit filed in federal district court by the Government of the Province of Manitoba ("Manitoba").[22] The suit alleges that the Environmental Assessment of NAWS was inadequate,[23] and asks that an Environmental Impact Statement ("EIS") be prepared pursuant to the National Environmental Policy Act.[24] But even if an EIS were prepared, and if it found that there was a risk that invasive alien species would be introduced, NAWS could still go forward.

Manitoba has also challenged NAWS on the basis of the 1909 Boundary Waters Treaty,[25] which states that "waters flowing across the boundary shall not be polluted on either side to the injury of health or property on the other side."[26] This matter is presently before the International Joint Commission ("IJC"), a bilateral organisation formed by the Boundary Waters Treaty that monitors transboundary water quality. Unfortunately, it is unclear whether the treaty authorises IJC to impose a substantive remedy with respect to alien species.[27]

NAWS represents an example of a threat of unintentional introduction of a potential invasive species and proactive legal steps taken to prevent it. While procedural devices may exist to hamper the project, ultimately, there appears to be little in the way of substantive relief available.

A more reactive, but substantive, legal approach is taking place in the context of ballast water discharges and attempts to use the federal Clean Water Act ("CWA"), 33 U.S.C. §§ 1251 *et seq.*, to combat “aquatic nuisance species” introduced into waterways from these discharges. Since 1998, a coalition of public interest groups, Native American tribes, and water utilities has been pressuring the United States Environmental Protection Agency ("EPA") to regulate the discharge of ballast water under the Clean Water Act ("CWA"). These efforts have culminated in the filing of several lawsuits, the most recent of which has resulted in a federal court holding that EPA must apply the CWA permitting program to ballast water discharges due to the “aquatic nuisance species”–pollutants–often contained therein.[28] The court in this case has ordered EPA to create a CWA permitting program to address ballast water discharges. Once the program is created and permits issued, if a discharger violates the terms of its CWA permit, the permittee could face injunctive sanctions and penalties imposed through a CWA citizen suit or government enforcement action.

STATUTES, TREATIES AND REGULATIONS CONCERNING INVASIVE ALIEN SPECIES

To date, codified approaches to combating invasive alien species have for the most part been ineffective.[29] Some statutes may only focus on certain species, like plant pests[30] or aquatic invasive alien species.[31] Statutes also tend to limit their scope to specific species, maintaining "dirty lists" of species that cannot be imported, but ignoring others that have merely not yet caused harm.[32]

International law is generally silent on the issue of invasive alien species.[33] While one international treaty does prohibit the introduction of invasive alien species, courts have not interpreted the provision as creating an enforceable duty.[34] While other treaties, such as the Boundary Waters Treaty, are occasionally invoked to protect against invasive alien species, the lack of specificity hinders their effectiveness.[35]

Currently, member nations of the International Maritime Organisation ("IMO") are debating the endorsement of the International Convention for the Control and Management of Ships' Ballast Water and Sediments.[36] While the treaty represents an unprecedented international attempt to deal with a significant problem, its scope is limited to the treatment and management of ballast water.

National legislation, while slightly more prolific, is not significantly more effective. The vast majority of United States statutes have been aimed at protecting agricultural and aquatic species using "dirty list" approaches, but have not created any more effective legal processes.[37]

Finally, state statutes addressing the invasive alien species problem are inconsistent and for the most part nonexistent.[38] Because species can spread across state boundaries with ease, the inadequacy of one state's statute (or enforcement) can undermine the legislative efforts of its neighbors.[39]

POTENTIAL COMMON-LAW CAUSES OF ACTION

Because of the inadequacy of statutory schemes, commentators have suggested that tort law may be more effective at combating invasive alien species.[40] While there are no purely common-law cases that impose liability[41] for invasive alien species, courts have adjudicated disputes about bugs,[42] weeds,[43] and cattle escaping from property,[44] each of which provide useful analogies for hypothesising how to apply tort law to this issue.

Trespass

A *trespass* occurs if someone *intentionally* "enters land in the possession of the other, or causes a thing or third person to do so." Restatement (Second) of Torts, § 158(a) (1965). *Intentional* as used in this article means that the action was either done on *purpose* or with *knowledge*.

Relief under a trespass theory against a person who introduces an invasive alien species would take the form of money damages[45] or an injunction.[46] Accordingly, a person who introduces an invasive alien species has caused a thing to enter land belonging to another person. This would unambiguously be a trespass, were it not for the fact that trespass is an intentional tort.[47]

In other words, for liability to attach, the wrongdoer must intend a very specific action: that land in the possession of another be entered.[48] And for relief to apply in a practical sense, the invasion must result in some sort of identifiable colonisation or other damage to the property.

Even if the introduction of the species were intended, the subsequent trespass by the invasive species is only intentional if the introducer intended that the species damage (colonize) the land of the person who is complaining. Because the extent to which a species will flourish is often unknown, proving[49] that the introducer intended to cause a species to enter another person's land will often be difficult. Thus, in the context of invasive alien species introductions, the requisite elements of a trespass rarely exist.

Negligence

Negligence is a tort that asserts that a wrongdoer neglected to do something. This neglect must cause an injury. In order to prove that a wrongdoer was negligent, you must establish that the wrongdoer had a duty to do something, that the wrongdoer failed to do it, and that the failure caused a foreseeable injury. For example, in *Merriam v. McConnell,*[50] a court found that there was no duty to prevent box elder bugs from infesting a neighbor's land, even though the bugs originated from the defendant's property.[51]

However, courts have found that there is a duty not to unnaturally burden neighboring land. In *Kukowski v. Simonson Farm, Inc.*,[52] the Simonsons experienced an influx of weeds on their land, particularly kochia and Russian thistle.[53] The Simonsons had no statutory or common-law duty to control these weeds.[54] However, the Simonsons used a combine to remove the weeds.[55] The combine broke the weeds apart, tossing bits of weed (and weed seed) onto the Kukowski's property.[56] Subsequently, the Kukowskis experienced increased weed growth.[57]

The court found that "when the result is traceable to artificial causes, or where the hand of man has, in any essential measure, contributed thereto, the person committing the wrongful act cannot excuse himself from liability upon the ground that natural causes conspired with his will to produce the ill results."[58]

Although the court narrowly held that there was "a duty to use ordinary care when attempting to control or remove weeds," the reasoning can be analogized to the invasive alien species problem.[59] Thus, *Kukowski* could also be read as implying that an original artificial cause—whether it be scattering weed seeds through negligent combining or planting kudzu or releasing gypsy moths—that naturally proliferates may expose the person to potential liability.[60] In other words, the proliferation of a foreign species is undoubtedly a natural cause, which the introducer ceases to prevent with ordinary care.

Unfortunately, some lawmakers, particularly in the U.S., do not recognize the proliferation of and resultant environmental damage from an invasive alien species as immediate or foreseeable (e.g., the accidental introduction of the European gypsy moth into the United States).[61] This narrow viewpoint fails to account for incidents in other countries when the species at issue has proven to be invasive. Such lack of foresight at the policy level should diminish as scientific certainty about the risks posed by certain species develops. Additionally, courts do not require people to make every possible effort to avoid harm.[62] Instead, they require only that people act in a reasonable manner, which is unfortunately difficult to define in the context of invasive alien species.[63] Thus, unless the introducer of a species knew or should have known that the particular species was likely to cause harm, it seems unlikely that courts will impose a duty not to release alien species, even if the species has exhibited invasive characteristics in other settings (unless the introducer should have known this fact).

Strict Liability for Individual Harms

Strict liability, unlike trespass or negligence, does not depend on whether the person intended the action or acted reasonably. If a person is held strictly liable for a certain type of action, they are liable for the harm that may result from that action no matter what the person's intentions or how reasonably they acted. Humans holding wild animals in captivity have been held strictly liable for the actions of those animals.[64] This liability has been expressed as "strict liability for the resulting harm, although it would not have occurred but for the unexpectable . . . operation of a force of nature."[65] The definition of wild animal is sufficiently broad to encompass most animal invasive alien species above the microbial level.[66]

The harm that an invasive alien species causes to the ecosystem is entirely due to the "operation of a force of nature."[67] It is predictable that wildlife would eat and reproduce. However, strict liability has been imposed only in cases where the animal *itself* directly attacks or otherwise injures a human or property. It may not follow that the introducer of an alien species can be found strictly liable for all harms done by a wild animal's subsequent progeny. No common-law cases exist that extend liability to a wild animal's reproduction, let alone to a plant or microbial species. Nevertheless, at least the same rationale for applying strict liability to damage caused by an animal's ferocious act exists for extending liability to an animal's injurious reproductive action.

Private Nuisance

"A private nuisance is a nontrespassory invasion of another's interest in the private use and enjoyment of his land." [68] Unlike negligence, nuisances can be intentional or negligent and do not require a duty owed. Because invasive alien species indirectly damage the land, a person whose land is invaded by such species may claim that a private nuisance has afflicted them.
Although the *Kukowski* court initially imposed a duty of reasonable care in controlling weeds, a later court extended *Kukowski* to apply to a private nuisance as well.[69] The court there claimed that extensive weed growth, which blew weed seeds onto a neighboring farm in large quantities, could constitute a private nuisance if the weed growth resulted from "farming, cultivation, or other activities." [70] Accordingly, some precedent exists for imposing liability for the actual propagation of a species that results from human activities if that propagation ultimately injures another's interest in their land.

Public Nuisance

As solutions to the invasive alien species problem, however, all of the above remedies—trespass, negligence, strict liability, and private nuisance—suffer from similar problems. First, recovery is limited to plaintiffs whose land has been damaged.[71] Furthermore, simply examining damage to one plaintiff's land cannot recompense damage to the larger ecosystem.[72] Finally, because invasive species may harm many plaintiffs, recovery via legal means may be plagued by the difficulties and failures associated with collective action.[73]

Arguably, the most effective common-law means of combating invasive alien species is under the public nuisance doctrine. [74] "A public nuisance is an unreasonable interference with a right common to the general public."[75]

A public nuisance suit can be brought either by an official public representative or by an individual who has suffered a special injury.[76] This flexibility may be important in the invasive alien species context. Because nuisance suits brought by official public representatives employ strict liability, [77] many of the prerequisites associated with trespass and negligence disappear.

Public nuisance suits also provide greater flexibility in terms of remedies. While damages may still be available, public nuisance suits also provide a greater possibility of effective injunctions,[78] which may either prevent the original introduction of invasive species or control the subsequent propagation.

Some problems with imposing liability under this theory may nevertheless exist. The invasive alien species problem is rarely as clear-cut as an infestation of weeds or bugs from a neighbor's property or pollution from a factory. In many cases, it is hard to determine who introduced an invasive alien species—or who should be blamed for its spread. [79] These shortcomings are common to any system—whether statutory or common-law—that attempts to address the problem.

The biggest challenge to proving that a threatened introduction of an alien species would be invasive, i.e. constitute a public nuisance would be establishing that the species would interfere unreasonably with a right common to the general public. In order to enjoin the introduction of a potentially invasive species, a plaintiff must prove that the potentially invasive species will unreasonably invade a public right. Here, the same reasons why the statutory "dirty-list" approach fails will come into play. Proof of environmental damage is generally not available until a species has invaded. At this point, it is often too late to rectify the damage.

A COMBINED APPROACH

One success in combating invasive alien species through the common law involved a regulation that directly declared invasive alien wildlife a public nuisance. Specifically, in a case in Colorado, red deer, Barbary sheep and ibex - all alien species - escaped the physical boundaries of a ranch.[80] The animals were declared a public nuisance *per se* and the owner was subjected to various abatement orders.[81] The public nuisance claim in the case arose from an innovative regulation that states:

> Wildlife which are illegally possessed or have escaped the owner's control and which are determined by the division to be detrimental to native wildlife, habitat or other wildlife resources by threat of predation, the spread of disease, habitat competition, interbreeding with native wildlife, or other significant damage . . . shall [be considered] . . . to be a public nuisance.[82]

This case gives some insight into a regulatory scheme that could drive a common-law challenge to the invasive species problem. While it is unlikely that a court will find that introducing a potentially invasive species would interfere with a right common to the general public on its own, it is well established that lawmakers, through regulations and statutes, can declare certain circumstances to be nuisances *per se*.

Furthermore, by not only designating the introduction and/or propagation of an invasive species as a nuisance *per se*, but also providing a private cause of action through a citizen suit provision, as the CWA does, a legislature could create effective deterrents and enforcement mechanisms.

Unfortunately, only a scattered handful of such regulations and statutory schemes exist. Most of the existing regulations are limited in scope. The Colorado regulation, noted above, applies to alien wildlife. Other state statutes have declared plant pests to be public nuisances.[83] However, a generalised regulatory scheme that both declares that introducing potential invasive alien species of all types into the environment constitutes a public nuisance, and that provides for citizen enforcement, does not exist.

This sort of regulatory scheme would provide the flexibility that an effective challenge to the invasive alien species problem requires. On the one hand, threatened introductions of a known or potentially invasive alien species could be prevented by means of an injunction. Instead of requiring proof of the common-law requirement of an invasion of a right common to the general public, statutes would designate a threatened introduction of a known or potential invasive alien species as a *per se* nuisance. Such a framework would at a minimum provide rights to private property owners and even to states against individuals, and could even provide a cause of action against the federal government if the government waives its immunity to lawsuits and/or acts unreasonably in authorising the introduction of an invasive alien species. On the other hand, if the introduction had already taken place, the regulatory scheme could also allow abatement and control measures to be taken at the expense of the introducer. This threat of financial burden would also encourage those who might otherwise introduce potentially invasive alien species to change their behaviors.

Furthermore, while it might be easier for public officials to bring the public nuisance claim, individuals who suffered particular harm—for instance, local fishermen whose resource would be threatened by the introduction of a species— might also be able to bring suits against potential action by governments which could introduce invasive alien species. Moreover, as has proved true in other environmental statutory schemes that contain citizen suit provisions, the mere threat of citizen enforcement proves to be an important deterrent to both public and private entities. Finally, such a regulatory scheme could prove most effective when combined with a "dirty list" approach (e.g., prohibitions on importing certain terrestrial species) and, when relevant, permitting that imposes technological controls (e.g., CWA permits for ballast water discharges). In fact, flexibility could be afforded through a supplemental "clean list" system that provides exceptions to the law for alien species that science has "proven" do not exhibit invasive characteristics.

CONCLUSION

The current regulatory scheme worldwide is ineffective at combating invasive species. Statutes are too limited in scope and lack flexibility to deal with changing information. Common-law tort schemes provide the flexibility that statutes lack. However, the strength of the tort scheme depends on the proof of harm. Because it may be difficult to prove that a species that has not yet been introduced will cause significant harm, what the tort scheme gains in flexibility it loses in proactive efficacy.

Creating a tort cause of action through statutes would strengthen the common-law scheme without losing any of the flexibility. When combined with permitting requirements that impose technological controls and "dirty lists" (and proven "clean list" exceptions), as well as citizen enforcement, such a statutory scheme would allow governments and individuals to effectively prevent or control the introduction of invasive alien species, and would provide powerful financial incentives for individuals to consider the ecological ramifications of their actions. Finally, because invasive alien species do not recognise international borders, these elements would need to be included in international treaties in order to effectively address the significant risk posed by invasive alien species worldwide.

ACKNOWLEDGEMENT

The author would like to acknowledge Heidi Bond for her significant contributions to this article.

REFERENCES

This article uses the legal citation form common in the United States, "Bluebook: A Uniform System of Citation" (2003 ed.). References are accordingly provided below as endnotes.

[1] *See* Julianne Kurdila, Comment, *The Introduction of Exotic Species into the United States: There Goes the Neighborhood!*, 16 B.C. ENVTL. AFF. L. REV. 95, 100 (1988) (describing competition between native and non-native species for food and habitat).

[2] *Id.* at 101.

[3] *See, e.g.*, U.S. GAO, *Invasive Species: Obstacles Hindering Federal Rapid Response to Growing Threat* (GAO-01-724, July 2001) at 3 ("*GAO Report*"); John A. Ruiter, *Combating the Non-Native Species Invasion of the United States*, 2 DRAKE J. AGRIC. L. 259, 263 (1997) (describing reproductive rate of zebra mussels); *see also* Daniel A. Larsen, *Combatting the Invasive Species Invasion: The Role of Tort Liability*, 5 DUKE ENVTL. L. & POL'Y F. 21, 21 (1995), at 23 (discussing the spread of wild boars).

[4] *Id.; Id* at 263.

[5] *GAO Report* at 3 and 8.

[6] *Reauthorization of the 1990 Nonindigenous Aquatic Nuisance Prevention and Control Act: Hearings on S. 1660 Before the Subcomm. on Drinking Water, Fisheries and Wildlife of the Senate Comm. on Environment*, 104th Cong., 2nd Sess. (1996) (testimony of Rowan W. Gould, Deputy Assistant Director—Fisheries, U.S. Fish and Wildlife Service, Department of Interior).

[7] Lori H. Peoples, Recent Developments, *A Call for Uniform Regulation of International Introductions of Non-Indigenous Species: The Suminoe Oyster*, 81 N.C. L. REV. 2433, 2446-47 (2003).

[8] *Id.* A prospective legal scheme that focuses on preventing projects or behaviors that may result in the introduction of an intruder arguably presents fewer issues than combating purely accidental introduction of invasive species. This is in large part because accidental introductions of invasive species are hard to detect until it is too late, making it difficult to determine who to hold accountable. Other articles have discussed this difficulty; *see also* Eric Biber, Notes, *Exploring Regulatory Options for Controlling the Introduction of Non-Indigenous Species to the United States*, 18 VA. ENVTL. L.J. 375, 396 (1999), note 10, at 453.

[9] *Id.*

[10] Biber, *supra* note 8, at 396-97.

[11] *Id.*

[12] *Id.* at 406 (describing state law addressing the invasive species problem as "patchwork").

[13] *Id.* at 407 (noting that the only international treaty to address invasive species "has been interpreted as hortatory").

[14] *Common-law* refers to law that is based on judicial custom and precedent, and not on codified statutes or regulations.

[15] A *tort* refers to an injury or wrong committed outside of a contractual relationship. It is also used to describe a lawsuit for the injury or wrong suffered.

[16] Clifford Krause, *Water Everywhere, but is it Good for Fish?*, N.Y. TIMES, June 6, 2004, *available at* http://www.nytimes.com/2004/06/06/international/americas/06cana.html.

[17] *Officials Hope to Get Response on Water Project*, ABERDEEN AM. NEWS, Sept. 10, 2001, *available at* 2001 WL 27096034.

[18] Government of the Province of Manitoba, *Manitoba's Concerns: Garrison Projects and Devils Lake Outlet*, *at* http://www.gov.mb.ca/waterstewardship/transboundary/positions/ man-position/backgr.html.

[19] *Id*.

[20] *Id. But see Officials Hope to get Response on Water Project, supra* note 17 (quoting state engineer who says that water treatment eliminates fear of foreign biota being introduced into Canadian waters).

[21] Government of the Province of Manitoba, *supra* note 18.

[22] *U.S., North Dakota Governments Want Manitoba Water Project Lawsuit Dropped*, THE CANADIAN PRESS, Aug. 27, 2003, available at 2003 WL 62167611; *see also* Complaint for Declaratory and Injunctive Relief, The Government of the Province of Manitoba v. Norton, filed Oct. 22, 2002 (No. 02CV02057), *available at* http://www.gov.mb.ca/waterstewardship/transboundary/pdf/naws-complaint.pdf.

[23] Complaint at 38, *The Government of the Province of Manitoba* (No 02CV02057).

[24] *Id.* at 39.

[25] Treaty Between the United States and Great Britain Relating to Boundary Waters Between the United States and Canada, Jan. 11, 1909, U.S.-Can., 36 Stat. 2448 (hereinafter "Boundary Waters Treaty").

[26] *Id.*

[27] Some commentators have noted that invasive species can be thought of as pollutants. *See* Larsen, *supra* note 3, at 21. However, IJC press releases tend to indicate that the IJC believes that invasive species are a problem that must be solved by additional legislation or treaties. *See, e.g.*, Press Release, International Joint Commission, Great Lakes Governors Urged to Share Cost of Electric Fish Barrier, May 24, 2004, *available at* http://www.ijc.org/rel/news/040524_e.htm (explaining that others should take action to prevent transboundary introduction of Asian carp); Press Release, International Joint Commission, IJC Calls on Congress to Protect the Great Lakes: First Action Needed to Prevent Ecosystem from Becoming "Invader Zoo," March 25, 2004, *available at* http://www.ijc.org/rel/news/040325_e.htm (calling on Congress to adopt treaties to protect against invasive species).

[28] *See Northwest Environmental Advocates, et al.. v. EPA*, No. CV 03-05760SI (N.D. Cal., December 2003).

[29] A lengthier list of statutory shortcomings can be found elsewhere. *See, e.g.*, Biber, supra note 8.

[30] *See, e.g.*, Plant Pest Act, 7 U.S.C. §§ 147a, 149, 150aa-150jj; Plant Quarantine Act, 7 U.S.C. §§ 151-167; Federal Noxious Weed Act, 7 U.S.C. §§ 2801-2814; Federal Seed Act, 7 U.S.C. §§ 1551-1610.

[31] Nonindigenous Aquatic Nuisance Prevention and Control Act (National Invasive Species Act), 16 U.S.C. §§ 4701-4751.

[32] *See also* The Lacey Act, 16 U.S.C. §§ 3371-78; 18 U.S.C. § 42 (regulating animal introductions); Biber, *supra* note 8, at 398.

[33] *Biber, supra* note 8, at 407.

[34] *Id*. (referencing the 1982 Law of the Sea Convention).

[35] *Id.* (generally referencing other treaties).

[36] International Maritime Organisation, International Convention for the Control and Management of Ships' Ballast Water and Sediments, *available at* http://www.imo.org/Conventions/mainframe.asp?topic_id=867 (last visited July 28, 2004).

[37] Biber, *supra* note 8, at 399.

[38] *See* Biber, *supra* note 8, at 405 (referring to 1993 report). The authors also conducted a limited survey of state law, which determined that many states still lack laws regarding invasive species. As in 1993, many statutes focus on agricultural pests. However, some states have added laws dealing with aquatic invasive species. The vast majority of these statutes adopt a "dirty" list approach.

[39] *Id.* at 406.

[40] Larsen, *supra* note 3.

[41] A wrongdoer is said to be *liable* to another if the wrongdoer must compensate the person they have wronged for the harm done. It is similar to being guilty in the criminal context; if you are guilty of a criminal offense, you might get a prison sentence. If you are liable of a civil offense, you must make some kind of remedy to the person you have wronged.

[42] *Merriam v. McConnell*, 175 N.E.2d 293 (Ill. App. 1961).

[43] *Kukowski v. Simonson Farm, Inc.*, 507 N.W.2d 68 (N.D. 1993).

[44] *See, e.g., Colorado Division of Wildlife v. Cox*, 843 P.2d 662 (Co. App. 1992).

[45] *Damages* are awarded by courts. Typically, money damages are *compensatory* in nature: the court awards money damages sufficient to rectify the harm done to the person, either by repairing any injury done, or estimating the market value of the injury. Money damages in the invasive alien species context might be an award of money sufficient to control or eradicate the invader. On rare occasions, courts award *punitive* damages, to punish a wrongdoer who has acted particularly maliciously. In the invasive alien species context, this would be very rare.

[46] An *injunction* is a court order that tells a person to take or refrain from taking certain actions.

[47] There also exists an action for negligent trespass (e.g., you did not intend to hit your neighbor's wall with your car, but you did not notice it because you were driving with your eyes shut). As discussed in the next section, courts rarely declare that allowing a species to grow is negligent.

[48] Restatement (Second) of Torts, § 158(a) (1965).

[49] In the United States, liability for torts usually requires that you prove your case by a "preponderance of the evidence," meaning that it is more likely than not that you are correct.

[50] 175 N.E.2d 293, 297 (Ill. App. 1961).

[51] *Id.*

[52] *Kukowski v. Simonson Farm, Inc.*, 507 N.W.2d 68 (N.D. 1993).

[53] *Id.* at 69.

[54] *Id.* at 70.

[55] *Id.* at 69.

[56] *Id.*

[57] *Id.* at 71.

[58] *Id.* at 70 (quoting Merriam, 175 N.E.2d at 296).

[59] *Cf.* 507 N.W.2d at 70.

[60] *Cf. id.*

[61] *See* Ruiter, *supra* note 3, at 259 (noting that although there are 4,500 invasive species, only 675 have adverse economic effects, and only 97 of the most damaging invasive species have caused $97 billion in damages).

[62] Restatement (Second) of Torts, § 283.

[63] *Id.* Technically, the standard is that a person must act "like a reasonable man in like circumstances."

[64] Restatement (Second) of Torts, § 510.

[65] *Id.*

[66] Restatement (Second) of Torts, § 506.

[67] Restatement (Second) of Torts, § 510.

[68] Restatement (Second) of Torts, § 821D. Examples of a private nuisance in other contexts are things such as your neighbor continually playing loud music. Typically, the invasions are indirect or non-physical in nature.

[69] *See Collins v. Barker*, 668 N.W.2d 548, 555 (S.D. 2003).

[70] *Id.*

[71] Most damages are compensatory in nature. Compensatory damages are limited to compensation for the harm suffered by the plaintiff. Restatement (Second) of Torts, § 903.

[72] *Cf.* Biber, *supra* note 8, at 446 (noting that "most individuals only suffer a small marginal harm" for environmental damage).

[73] *Id.*

[74] *See* Larsen, *supra* note 3, at 39-40.

[75] Restatement (Second) of Torts, § 821B. A typical example of a "public nuisance" might be a factory that is releasing unwarranted amounts of toxic pollution into ground water, or the stench from a cattleyard.

[76] *See* Larsen, *supra* note 3, at 41.

[77] *Id.* Strict liability in a nuisance suit means that if the wrongdoer caused (or could cause) a public nuisance, liability will be imposed no matter what the wrongdoer's intentions.

[78] This is not to imply that injunctions cannot be obtained for other actions. However, it is much harder to tell a judge that a person must be restrained from introducing a gypsy moth because the moth might fly onto your land than it is to convince a court that the propagation of moths might pose a danger to the community at large. Injunctions for nuisance suits are relatively common. However, the circumstances under which injunctions are given vary from state to state. For a lengthy discussion of injunctions and damages and the relations between them, *see* Guido Calabresi & A. Douglas Melamed, *Property Rules, Liability Rules and Inalienability: One View of the Cathedral*, 85 Harv. L. Rev. 1089 (1972).

[79] *See* Biber, *supra* note 8, at 453.

[80] *Colorado Division of Wildlife v. Cox*, 843 P.2d 662, 663 (Col. App. 1992).

[81] *Id.*

[82] 2 COLO. CODE REGS. § 1107.b (1992 Cum. Supp.).

[83] "Any premises, plant, appliance, conveyance, or article that is infected or infested with plant pests that may cause significant damage or harm and any premises where any plant pest is found is a public nuisance and must be prosecuted as a public nuisance in all actions and proceedings. All legal remedies for the prevention and abatement of a nuisance apply to a public nuisance under this section. It is unlawful for any person to maintain a public nuisance." Minn. Stat. Ann. § 18G.04 (West 2004).

Establishment and enforcement of the new Invasive Alien Species Act in Japan

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Abstract Hundreds of millions of alien organisms are introduced into Japan every year. Some have become 'invasive,' having adverse effects on ecosystems, human safety, or agriculture, forestry and fisheries. Considering that the Japanese regulatory system has not dealt with the issue comprehensively, taking into account various suggestions from many entities concerned, and aiming to implement the provisions of Article 8 (h) stipulated by the Convention on Biological Diversity, the Japanese Cabinet finalised a draft of the Invasive Alien Species Act and submitted it to the Japanese Diet (= Parliament) in March, 2004.

The framework consists of the following three main points: (1) "Invasive Alien Species" (IAS) shall be designated by the Act as having adverse effects on ecosystems, human safety, or agriculture, forestry and fisheries. Raising, planting, storing, carrying, and importing IAS shall be prohibited with the exception of specified cases such as by obtaining permission from the competent ministers. (2) National/local governments, Non-Profit Organisations, and other entities concerned shall take appropriate measures to mitigate the impact of IAS that already exist in Japan. (3) "Uncategorised Alien Species" (UAS), which are suspected IAS, shall need detailed investigation by the Japanese government for up to six months in order to be allowed to be imported into Japan.

The Diet passed the draft without amendments and the new Invasive Alien Species Act was promulgated as of 2 June, 2004. Following the establishment, the Japanese Cabinet made a basic policy on 15 October, 2004, for effective implementation of the Act. In addition to designation of UAS, 37 different types of alien species were selected as IAS for the first step, based on suggestions by academic experts and the Act was enforced on 1 June, 2005. The second round of designation was completed on 1 February, 2006, with 43 additional IAS. We continue our consideration of additional designation as well as strategies for eradication and control of IAS.

Keywords: alien species; biodiversity; IAS; invasive alien species; importation; LORCA; mitigation, regulation; UAS; uncategorised alien species.

INTRODUCTION

Japan is a country of mass importation of living organisms. In 2003 alone, for example, about 620 million live animals were brought into Japan (Japanese Customs 2003). Although more than 90% of these are classified as worms for fishing bait, a variety of species ranging from vertebrates to insects are brought in every year. Also, as the Japanese economy is strongly sustained by international trade, it is thought that many alien species are unintentionally introduced into Japan via seaports and airports along with imported goods and containers, and by staff involved in international trade.

As a result, some of these alien species have established in Japan. So far, according to preliminary statistics gathered by the Japanese Ministry of the Environment and announced on 27 October 2004, 111 species of vertebrates (17 mammals, 38 birds, 3 amphibians, 11 reptiles, and 42 fish), 584 species of invertebrates (including 433 insects) and 1,556 species of plants (including 1,552 vascular plants) have been recognised as established in Japan or found in the Japanese wild (Japanese Ministry of the Environment, 2004a).

Due to their dissimilar properties from native species, some of the alien species introduced into Japan, have been confirmed to cause the following types of damage:

1) Destruction of ecosystems
   A typical example is the Java mongoose (*Herpestes javanicus*). It was intentionally introduced to the southern islands of Japan in order to eradicate the yellow-spotted lance-head snake (*Trimeresurus flavoviridis*), but the effort resulted in decreased habitats for endangered native animals, such as the Okinawa rail (*Rallus okinawae*) (Ishii 2003, Yamada 2002).
2) Threat to human safety
   Snapping turtles (*Chelydra serpentina*) were originally introduced as pets from the American

Pages 35-44. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

continent but some were, and still are, discarded into the wild when they become difficult to care for. The turtles pose a risk to human safety (biting and injuring humans).

3) Damage to agriculture, forestry and fisheries
The raccoon (*Procyon lotor*) was intentionally introduced as a pet but, like the snapping turtle, some were discarded into the wild due to their fierce temperament. Some agricultural crops are impacted by raccoons in the wild.

The existing legal framework was rather fragmentary with regards to the issue of problematic alien species and it was insufficient for dealing comprehensively with all impacts. For example, the Phytosanitary Act only prevents damage to agricultural plants and the Infectious Diseases Prevention Act only prevents damage to human health caused by infectious diseases. Similarly, the Living Modified Organisms (LMOs) Act controls importation of all living modified organisms, but the Act only targets damage to ecosystems and it does not restrict living 'unmodified' organisms at all.

Additionally, importation control officers have no comprehensive authority or mandate to prevent introduction of those alien species. Importation control is conducted mainly by customs officers, phytosanitary officers and quarantine officers. For instance, in the fiscal year (FY) 2004, 8,427 customs officers were working at headquarters and branch offices. Plant protection stations consist of five main offices with more than 50 branch offices. Quarantine stations for human health have ten main offices and ten branch offices. Additionally, there were 312 animal quarantine officers in FY 2004, working at headquarters and six branch offices.

As already stated, there was no comprehensive regulation to control the importation of alien species and border control officers had insufficient power to prevent the importation of unwanted alien species. This is why a new regulation to control all aspects of impacts by those alien species has been proposed.

INVASIVE ALIEN SPECIES

Article 8 (h) of the Convention on Biological Diversity (CBD) encourages each contracting party to prevent the introduction of alien species, and to control or eradicate those alien species which threaten ecosystems, habitats or species as far as possible and as appropriate.

Decision VI/23, adopted by the Sixth Meeting of the Conference of Parties under the CBD contains detailed guiding principles (CBD, 2002). The primary principle is stated as follows in the annex:

> *Guiding principle 2: Three-stage hierarchical approach*
> *2. Priority should be given to preventing the introduction of invasive alien species, between and within States. If an invasive alien species has been introduced, early detection and rapid action are crucial to prevent its establishment. The preferred response if often to eradicate the organisms as soon as possible… In the event that eradication is not feasible or resources are not available for its eradication, containment…and long-term control measures…should be implemented. Any examination of benefits and costs (environmental, economic and social) should be done on a long-term basis.*

In December 2003, the Central Environment Council, an official advisory group to the Japanese Government, reported that a new framework would be necessary to tackle issues regarding alien species in Japan efficiently. The report also presented key elements which the new system needed to consider, such as regulating importation and possession of undesirable alien species. Under the leadership of the Japanese Ministry of the Environment and the Ministry of Agriculture, Forestry and Fisheries of Japan, the Japanese Cabinet finalised a draft of the new Invasive Alien Species Act on 9 March, 2004 and submitted it to the Japanese Diet.

Establishment of the Invasive Alien Species Act

The Invasive Alien Species Act was approved by the Japanese Diet without amendments at the end of May 2004 and promulgated on 2 June of the same year (Law No. 78). The Act defines those alien species that are recognised or feared to cause the previously mentioned three types of damage as "invasive alien species (IAS)" - the Act only focuses on these three aspects because general economic damage by IAS has not been confirmed as severe enough to be regulated in Japan. The competent ministers (i.e. the Minister of the Environment, but for matters related to the prevention of adverse effects on agriculture, forestry and fisheries, the Minister of Agriculture, Forestry and Fisheries shall be added) have authority and responsibility to decide, based on scientific facts, which alien species should be selected as IAS. It would be against the WTO Treaty to regulate *all* non-native species without sufficient scientific evidence (Article 2, Paragraph 2 of SPS Agreement; World Trade Organisation, 1994).

The purposes of this Act are to control IAS properly and to prevent damage to ecosystems, human safety, and agriculture, forestry and fisheries. The framework consists of three main points: 1) A

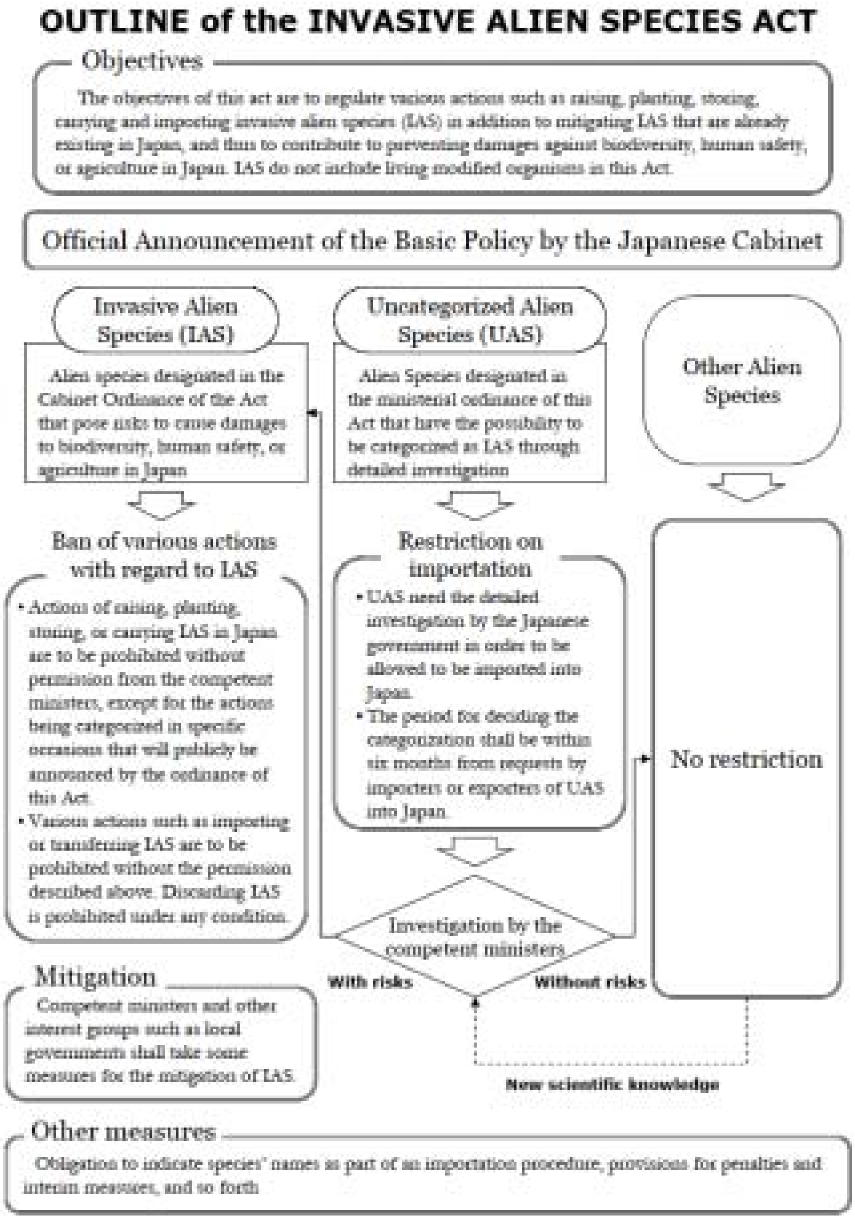

Figure 1 Outline of IAS act

ban on various actions with regard to IAS, 2) Mitigation of IAS in Japan, and 3) Judgment of Uncategorised Alien Species before their importation (see Fig. 1 for the outline).

The first point is to ban various actions regarding IAS. Raising, planting, storing or carrying IAS will be prohibited unless competent ministers give permission for these actions. Permission will be issued only when the applicants can prevent invasion of the IAS into the ecosystems of Japan. Permission is also a prerequisite for importing or transferring IAS. Discarding IAS into the wild in Japan will not be allowed at any time. Additionally, microchips will be required to be implanted in some IAS for identification.

Imposing severe penalties is another characteristic of this first main point. Offenders face imprisonment of up to three years or fines of up to three million yen (equivalent to US$28,000). Corporations that violate the Act will be charged up to 100 million yen (equivalent to US$940,000). These penalties are much more stringent than existing regulations for protecting biodiversity. It seems that the importance for protecting ecosystems is gradually becoming better recognised in Japan.

The second point is to mitigate IAS that already exists in Japanese ecosystems. The competent ministers will announce official national mitigation strategies for respective IAS to conduct efficient mitigation. Various mitigation measures, such as capturing, collecting, or killing IAS will be taken based on those strategies. The mitigation will be performed by the competent ministers with other active organisations such as other national government agencies, local governments and private organisations. When setting up the mitigation strategies, priority-setting is recognised as crucial to effective enforcement of the mitigation due to limited funds and staff members necessary for the implementation.

The third point is to judge Uncategorised Alien Species (UAS) before their importation. UAS are alien species which may be categorised as IAS after detailed investigation. In short, UAS are suspected of being IAS. Basically, UAS are designated in groups of species (see Fig. 1 and Appendix) because the names of some UAS are unknown. According to this Act, importers and exporters of UAS into Japan must request a detailed investigation of UAS from the competent ministers. The request must be delivered with information on ecological properties of the targeted UAS. Then, for up to six months, importation of the UAS will be restricted while the investigation is completed. After the investigation, UAS posing risks of damage will be designated as IAS immediately, while UAS posing no risks will not be regulated by this Act. However, if new evidence shows up that indicates that the alien species pose risks of damage, the species shall be re-designated as IAS.

In addition to the three main points mentioned above, the new Act has two features. One is the establishment of basic policy. The Japanese Cabinet will determine this policy for efficient implementation of this Act. The contents include the basic framework of the regulation, principles concerning the selection of IAS, principles concerning the handling of IAS, principles concerning the mitigation of IAS and so on.

The second feature is the attachment of a certificate for import. IAS, UAS and similar alien species (LORCA, or Living Organisms Required to have a Certificate Attached during their importation) will be required to have a certificate attached verifying their types as part of importation procedures. It is hoped that this certificate will become a powerful tool for customs officers in preventing illegal importation of IAS and UAS.

The full text of the Act and the basic policy is available on the website below:
http://www.env.go.jp/en/nature/as.html

Efficient Enforcement of the Act

The IAS Act was promulgated on 2 June, 2004 and the Cabinet Decision on the basic policy was made in October of the same year. The first selection of IAS, UAS and LORCA was completed in the spring of 2005 and the Act entered into force on 1 June, 2005 (see Fig. 2).

June 2, 2004	Promulgation of the IAS Act
October, 2004	Cabinet Decision on Basic Policy
Fall 2004 – End April 2005	1st Designation of IAS, UAS & LORCA (37 types of IAS)
June 1, 2005	Enforcement of the IAS Act
June 3, 2005	Announcement of mitigation strategies (for 20 IAS)
Around the end of 2005	Expected to complete 2nd Designation

Figure 2 Timetable of IAS act implementation

When the public consultation procedure on the first draft of the basic policy was conducted between July 8 and August 7, we received around 10,000 comments agreeing or disagreeing with the draft. This shows that public awareness on this issue is very strong, as usually only a few comments are made during the public consultation procedures. The Appendix shows key elements from the finalised basic policy (Japanese Ministry of the Environment, 2004b).

Section 2.3 of the basic policy states that alien species that cause damage or pose risks shall be selected as IAS without exception but the timing of designation might be deferred when establishment of a corresponding effective law enforcement system takes time. Due to the fact that some non-native species have been introduced for the purpose of economic and other benefits, adding this provision was inevitable to avoid confusion caused by enforcement of the Act. Although the meaning of "an effective law enforcement system" is rather arbitrary, this should not be used as an excuse to postpone designation unnecessarily.

Based on the said basic policy, the first general expert meeting to discuss what should be designated as IAS, UAS and LORCA was held on 27 October, 2004. The meeting decided to establish six working groups to deal with each group of alien species (i.e. mammals and birds, reptiles and amphibians, fish, insects, invertebrates, and plants). Moreover, two small special working groups have been set up in order to discuss the selection of the largemouth bass (*Micropterus salmoides*) and the large earth bumblebee (*Bombus terrestris*) as IAS. These two species are already introduced into Japan and their designation as IAS caused concerns for the sports fishing industry and the farming industry. Each working group had two to four meetings from November 2004 to the end of January 2005 and made recommendations on the designation to the general expert meeting. Following the final report of the general expert meeting, the Japanese Cabinet set up a Cabinet ordinance to designate IAS at the end of April.

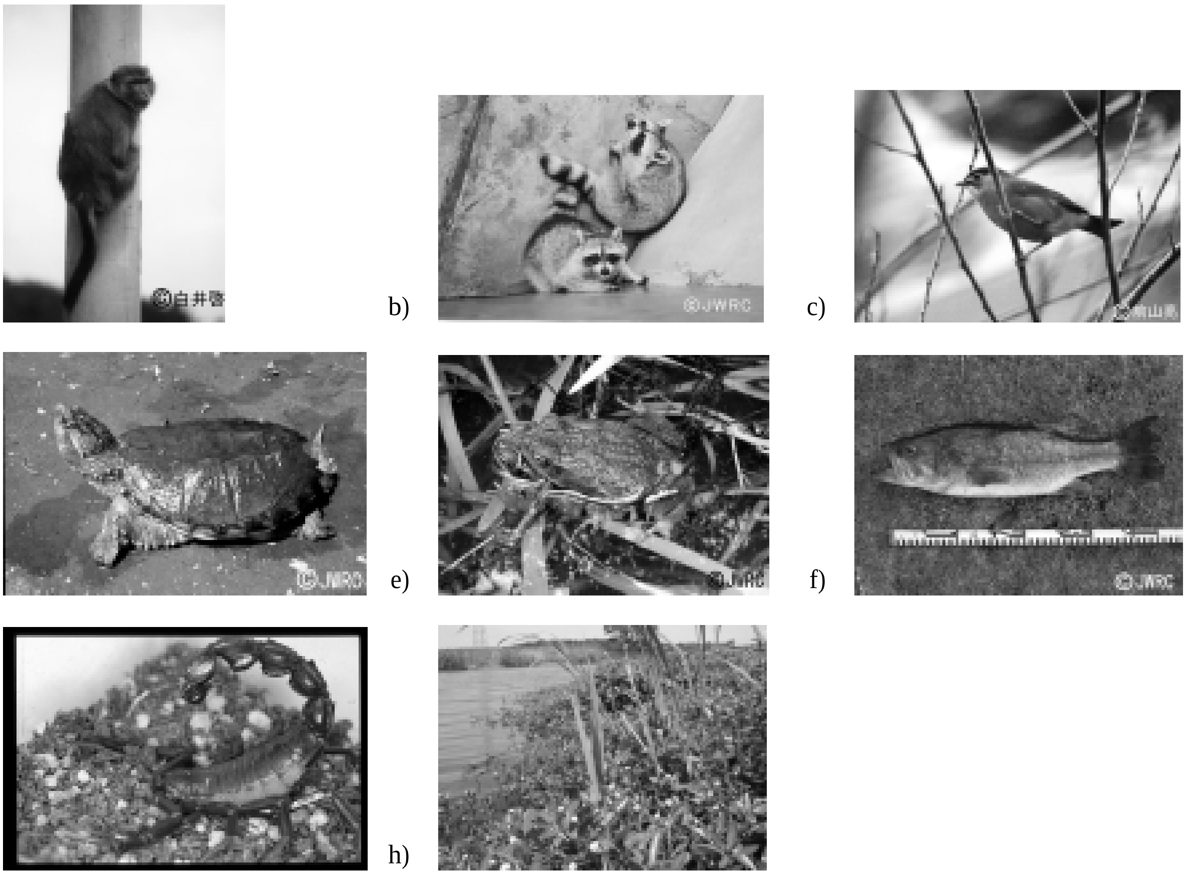

Figure 3 Photos of some IAS (copyright: JWRC); a) Taiwan macaque (*Macaca cyclopis*); b) Raccoon (*Procyon lotor*); c) Red-billed mesia (*Leiothrix lutea*); d) Snapping turtle (*Chelydra serpentina*); e) Cane toad (*Bufo marinus*); f) Largemouth bass (*Micropterus salmoides*); g) Death Stalker Scorpion (*Androctonus bicolor*); h) Alligatorweed (*Alternanthera philoxeroides*)

Designation of UAS and LORCA by ministerial ordinance followed at the end of May, just before the enforcement of the Act. Additionally, public consultation procedures and World Trade Organisation Treaty notification procedures were conducted before the ordinances were established. A record figure of more than 110,000 comments was received from the general public; most of them were from anglers and the sports fishing industry, against designation of largemouth bass as an IAS.

Considering the fact that the first round of selection of IAS had to be completed before enforcement of the Act (1 June, 2005), there was not enough time to establish a comprehensive assessment system for selecting IAS, UAS and LORCA. The first designation was conducted by picking up, with the help of the said academic experts, alien species whose known or likely adverse effects are already been reported by scientific publications from around the world. As a result, 37 different types of animals and plants were selected as IAS (Appendix and Fig. 3). Realising that this selection is not sufficient, consideration of the second round of selection started immediately after the first. Through the working groups and the expert meetings, 43 types of organisms were added to the IAS list on 1 February, 2006 (Appendix). UAS, in the meantime, were selected from alien species that have similar ecological properties to IAS. More specifically, in principle, alien species that belong to the same family or genus of IAS were selected as UAS in the first instance. Also, LORCA were selected, as defined, to cover a wider range of living organisms that physically look like IAS or UAS. The list of all the regulated living organisms is shown in Appendix and available at:
http://www.env.go.jp/nature/intro/siteisyu_list_e.pdf

Additionally, at the second round of designation, we created and publicised a so-called 'alien species alert list', which includes alien species for which there is currently not enough scientific evidence to be regulated by the IAS Act but that should be treated carefully in order not to be dispersed as some scientists believe that they pose risks to Japanese ecosystems.

Meanwhile, the first mitigation strategies were publicly announced on 3 June 2005 for 20 IAS. These 20 IAS are already established in Japan among the 37 IAS designated in 2005 and we need to conduct concrete and feasible eradication projects. The second set of mitigation strategies were announced on 1 February 2006 for the remaining 17 IAS and the 43 added IAS.

In order to realise efficient implementation of this new Act, some challenges must be overcome. For instance, we need to widen the Act's target range as much as possible. Further selections are expected. In the same way, there are several challenges, such as arranging sufficient budget and staff members for enforcement of this Act, preparing effective measures against unintentional introduction of IAS and UAS, setting permission criteria that ensure that IAS are kept out of ecosystems and the establishment of efficient mitigation techniques. For example, the Japanese Ministry of the Environment obtained about 204 million yen, or US$1.8 million, for the FY 2005 in order to implement mitigation projects.

It will be essential to address the above issues, thorough research, database creation, consensus building, fund raising, education, international cooperation, etc. In view of the fact that all IAS come from abroad, international cooperation for exchanging related knowledge, data, techniques, and so forth, are especially critical. The Japanese government hopes to implement this new regulation both nationally and internationally and to obtain international understanding and cooperation.

Additionally, experts point out that some species, native to Japan, introduced to a different habitat or ecosystem in Japan, where they do not occur naturally can also be invasive and cause damage to the ecosystem. As IAS in the Act are defined as species from abroad, the new Invasive Alien Species Act cannot deal with such invasive species of domestic alien origin. Generally speaking, protecting local ecosystems from introduction of invasive species of domestic alien origin is extremely difficult as we cannot assume that the introductions are focused at locations such as seaports and airports. The Japanese Government, however, will revise existing regulations such as the Natural Parks Law to prevent adverse effects and preserve valuable ecosystems as much as possible. This revision will help improve the current alien species policy together with enforcement of the Invasive Alien Species Act. Also, establishment of related bylaws by local government might be helpful to solve this domestic issue.

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APPENDIX

List of Regulated Living Organisms under the IAS Act of Japan. IAS: designated Invasive Alien Species; UAS: Uncategorized Alien Species; LORCA: Living Organisms Required to have a Certificate Attached during their importation in order to verify their types. [1]Designated on June 1, 2005; [2]designated on February 1, 2006; [3]designated on September 1, 2006.

1. Animal Kingdom

Class	Order	Family	Genus	IAS	UAS	LORCA
Mammalia	Marsupialia	Didelphidae	*Didelphis*	None	Any species of the genus *Didelphis*[1]	Any member of the family Didelphidae and Phalangeridae[1]
			All other genera of Didelphidae	None	None	
		Phalangeridae	*Trichosurus*	Brushtail possum (*T. vulpecula*)[1]	Any species of the family Phalangeridae excluding brushtail possum (*T. vulpecula*)[1]	
			All other genera of Phalangeridae	None		
	Insectivora	Erinaceidae	*Erinaceus*	Any species of the genus *Erinaceus*[2]	None	Any species of the genus *Erinaceus, Atelerix, Hemiechinus* and *Mesechinus*[2]
			Atelerix *Hemiechinus* *Mesechinus*	None	Any species of the genus *Atelerix*, *Hemiechinus* and *Mesechinus* excluding *Atelerix albiventris*[2]	
	Primates	Cercopithecidae	*Macaca*	Taiwan macaque (*M. cyclopis*)[1]	Any species of the genus *Macaca* excluding Japanese macaque (*M. fuscata*), Taiwan macaque (*M. cyclopis*), crabeating macaque (*M. fascicularis*), and rhesus macaque (*M. mulatta*)[1]	Any species of the genus *Macaca*[1]
				Crab-eating macaque (*M. fascicularis*)[1]		
				Rhesus macaque (*M. mulatta*)[1]		
	Rodentia	Agoutidae	All genera of Agoutidae	None	None	Any member of the families Agoutidae, Capromyidae, Dinomyidae and Myicastoridae[1]
		Capromyidae	All genera of Capromyidae	None	None	
		Dinomyidae	All genera of Dinomyidae	None	None	
		Myicastoridae	*Myocastor*	Coypu or Nutria (*M. coypus*)[1]	None	
		Sciuridae	*Callosciurus*	Pallas's squirrel or Taiwan squirrel (*C. erythraeus*)[1]	Any species of the genus *Callosciurus* excluding Pallas's squirrel (*C. erythraeus*)[1]	Any member of the family Sciuridae[1]
			Pteromys	Russian (or Siberian) flying squirrel (*P. volans*) excluding Japanese subspecies (*P. volans orii*)[2]	None	
			Sciurus	Gray squirrel (*S. carolinensis*)[1]	Any species of the genus *Sciurus* excluding gray squirrel (*S.carolinensis*), Japanese squirrel (*S. lis*) and red squirrel (*S. vulgaris*)[1]	
				Eurasian red squirrel (*S. vulgaris*) excluding Japanese subspecies (*S. vulgaris orientis*)[2]		
			All other genera of Sciuridae	None	None	
		Muridae	*Ondratra*	Muskrat (*O. zibethicus*)[2]	None	Any species of the genus *Ondatra*[2]
	Carnivora	Procyonidae	*Procyon*	Raccoon (*P. lotor*)[1]	None	Any species of the genus *Procyon*[1]
				Crab-eating raccoon (*P. cancrivorus*)[1]	None	
		Mustelidae	*Mustela*	American mink (*M. vison*)[2]	Any species of genus *Mustela* excluding American mink, Japanese native species (*M. itatsi*, *M. sibilica*, *M. nivalis* and *M. erminea*) and ferret (*M. putoriusfuro*)[2]	Any species of the genus *Mustela*[2]
		Herpestidae	*Herpestes*	Javan mongoose (*H. javanicus*)[1]	Any member of the family Herpestidae excluding meerkat (*Suricata suricatta*) and Javan mongoose (*H. javanicus*)[1]	Any member of the family Herpestidae[1]
			All other genera of Herpestidae	None		
	Artiodactyla	Cervidae	*Axis*	All species of the genus *Axis*[2]	None	Any species of the genera *Axis*, *Cervus* and *Dama* and *E. davidianus*[2]
			Cervus	All species of the genus *Cervus* excluding *C. nippon centralis C. nippon keramae C. nippon mageshimae C. nippon nippon C. nippon pulchellus C. nippon yakushimae C. nippon yesoensis*[2]	None	
			Dama	All species of the genus *Dama*[2]	None	
			Elaphurus	Pere David's deer (*E. davidianus*)[2]	None	
			Muntiacus	Reeves's muntjac (*M. reevesi*)[1]	Any species of the genus *Muntiacus* excluding Reeves's muntjac (*M. reevesi*)[1]	Any species of the genus *Muntiacus*[1]

(APPENDIX continued)

Class	Order	Family	Genus	IAS	UAS	LORCA
Aves	Passeriformes	Timaliidae	*Garrulax*	Laughing thrushes (*G. canorus*)[1]	Any member of the family Timaliidae excluding laughing thrushes (*G. canorus*), whitebrowed laughingthrush (*G. sannio*), masked laughingthrush (*G. perspicillatus*), and redbilled mesia (*L. lutea*)[1]	Any member of the family Timaliidae[1]
				White-browed laughingthrush (*G. sannio*)[1]		
				Masked laughingthrush (*G. perspicillatus*)[1]		
			Leiothrix	Red-billed mesia (*L. lutea*)[1]		
			All other genera of Timaliidae	None		
Reptilia	Testudinata	Chelydridae	*Chelydra*	Snapping turtle (*C. serpentina*)[1]	None	Any member of the family Chelydridae[1]
			All other genera of Chelydridae	None	None	
	Squamata	Iguanidae (Polychrotidae)	*Anolis*	Green anole (*A. carolinensis*)[1]	Any species of the genus *Anolis* (and/or *Norops*) excluding green anole (*A. carolinensis*) and brown anole (*A. sagrei*)[1]	Any species of the genus *Anolis* (and/or *Norops*)[1]
				Brown anole (*A. sagrei*)[1]		
			Norops	None		
		Colubridae	*Bioga*	Brown tree snake (*B. irregularis*)[1]	Any species of the genus *Bioga* excluding brown tree snake (*B. irregularis*)[1]	Any species of the genera *Bioga* and *Psammodynastes*[1]
			Psammodynastes	None	None	
			Elaphe	Taiwan beauty snake (*E. taeniura friesi*)[1]	Any species of *E. taeniura* excluding *E. taeniura schmackeri* and *E. taeniura friesi*[1]	*E. taeniura* and *E. radiata*[1]
		Viperidae	*Protobothrops*	Taiwan pit vipers (*P. mucrosquamatus*)[1]	Any species of the genus *Protobothrops* excluding *P. elegans* and *P. mucrosquamatus*[1]	Any species of the genera *Protobothrops* and *Bothrops*[1]
			Bothrops	None	None	
Amphibia	Anura	Bufonidae	*Bufo*	Cane toad (*B. marinus*)[1]	Any species of the genus *Bufo* excluding *B. marinus*, *B. viridis*, *B. debilis*, *B. terrestris*, *B. valliceps* and *B. paracnemis*[1]	Any species of the genus *Bufo* (Any larvae, or tadpoles of the order Anura)[1]
		Hylidae	*Osteopilus*	Cuban treefrog (*O. septentrionalis*)[2]	Any species of the genus *Osteopilus* excluding Cuban treefrog (*O. septentrionalis*)[2]	Any species of the genus *Osteopilus* (Any larvae, or tadpoles of the order Anura)[2]
		Leptodactylidae	*Eleutherodactylus*	Puerto Rican coqui (*E. coqui*)[2]	Greenhouse Frog (*E. planirostris*)[2]	*E. coqui* and *E. planirostris* (Any larvae, or tadpoles of the order Anura)[2]
		Ranidae	*Rana*	Bullfrog (*R. catesbeiana*)[2]	Green frog(*R. clamitans*), Pig frog (*R. grylio*), River frog (*R. heckscheri*), Florida bog frog (*R. okaloosae*), Mink frog (*R. septentrionalis*), and Carpenter frog[2]	*R. catesbeiana*, *R. clamitans*, *R. grylio*, *R. heckscheri*, *R. okaloosae*, *R. septentrionalis*, and *R. virgitipes* (Any larvae, or tadpoles of the order Anura)[2]
		Rhacorhoridae	*Polypedates*	Asian tree frog (*P. leucomystax*)[2]	Any species of the genus *Polypedates* excluding *P. leucomystax*[2]	Any species of the genus *Polypedates* (Any larvae, or tadpoles of the order Anura)[2]
Osteichthyes	Siluriformes	Ictaluridae	*Ictalurus*	Channel catfish (*I. punctatus*)[1]	Any species of the genus *Ictalurus*[1]	Any species of the genera *Ictalurus* and *Ameiurus*[1]
			Ameiurus	None	Any species of the genus *Ameiurus*[1]	
	Esociformes	Esocidae	*Esox*	Northern pike (*E. lucius*)[2]	Any species of the genus *Esox* excluding northern pike (*E. lucius*) and muskellunge (*E. masquinongy*)[2]	Any sprcies of the genus *Esox*[2]
				Muskellunge (*E. masquinongy*)[2]		
	Cyprinodontiformes	Poeciliidae	*Gambusia*	Western mosquito fish (*G. affinis*)[2]	*G. holbrooki*[2]	*G. affinis* and *G. holbrooki*[2]
	Perciformes (Percoidei)	Centrarchidae	*Lepomis*	Bluegill (*L. macrochirus*)[1]	Any member of the family Centrarchidae excluding largemouth bass (*M. salmoides*),smallmouth bass (*M. dolomieu*), and bluegill (*L. macrochirus*)[1]	Any member of the families of Centrarchidae, Centropomidae, and Nandidae[1]
			Micropterus	Smallmouth bass (*M. dolomieu*)[1]		
				Largemouth bass (*M. salmoides*)[1]		
			All genera of Centrarchidae	None		
		Centropomidae	All genera of Centropomidae	None	None	
		Nandidae	All genera of Nandidae	None	None	
		Moronidae	*Morone*	Striped bass (*M. saxatilis*)[2]	Any member of the family Moronidae excluding *M. saxatilis* and *M. chrysops*[1]	Any member of the family Moronidae[1]
				White bass (*M. chrysops*)[2]		
			All genera of Moronidae	None		
		Percichthyidae	*Gadopsis*	None	Any species of the genera *Gadopsis*, *Maccullochella*, *Macquaria* and *Percichthys* excluding Murray cod (*Maccullochella peelii*) and Golden perch (*Macquaria ambigua*)[1]	Any species of the genera *Gadopsis*, *Maccullochella*, *Macquaria*, and *Percichthys*[1]
			Maccullochella	None		
			Macquaria	None		
			Percichthys	None		

(APPENDIX continued)

Class	Order	Family	Genus	IAS	UAS	LORCA
Osteichthyes	Perciformes (Percoidei)	Percidae	*Gymnocephalus*	None	Any species of the genera *Gymnocephalus*, *Perca*, *Sander* (*Stizostedion*), and *Zingel* excluding pikeperch (*S. licioperca*) and Eurasian perch (*P. fluviatilis*)[1]	Any species of the genera *Gymnocephalus*, *Perca*, *Sander* (*Stizostedion*), and *Zingel*[1]
			Perca	Eurasian perch (*P. fluviatilis*)[2]		
			Sander (*Stizostedion*)	Pikeperch (*S. lucioperca*)[2]		
			Zingel	None		
		Sinipercidae	*Siniperca*	Mandarin fish (*S. chuatsi*)[2]	Any species of the genus *Siniperca* excluding *S. chuatsi* and *S. sherzeri*[1]	Any species of the genus *Siniperca*[1]
				Mandarin fish (*S. scherzeri*)[2]		
Arachnid	Scorpiones	Buthidae	All genera of Buthidae	Any species of the family Buthidae[1]	None	Any member of the Class Scorpiones[1]
	Araneae	Hexathelidae	*Atrax*	Sydney funnelweb spider (*A. robustus*) and all other species of the genus *Atrax*[1]	None	Any species of the genera *Atrax* and *Hadronyche*[1]
			Hadronyche	Any species of the genus *Hadronyche*[1]	None	
		Loxoscelidae	*Loxosceles*	*L. reclusa*[1]	None	Any species of the genus *Loxosceles*[1]
				L. laeta[1]	None	
				L. gaucho[1]	None	
		Theridiidae	*Latrodectus*	Brown widow spider (*L. geometricus*)[1]	Any species of the genus *Latrodectus* excluding *L. indicus*, *L. hasseltii*, *L. geometricus*, *L. tredecimguttatus*, and *L. mactans*[1]	Any species of the genus *Latrodectus*[1]
				Red back spider (*L. hasseltii*)[1]		
				Black widow spider (*L. mactans*)[1]		
				Mediterranean black widow spider (*L. tredecimguttatus*)[1]		
Crustacea	Decapoda	Astacidae	*Astacus*	Any species of the genus *Astacus*[2]	Any species of the family Astacidae excluding the genus *Astacus* and signal crayfish (*P. leniusculus*)[2]	Any species of the families Astacidae, Cambaridae, and Parastacidae[2]
			Atlantoastacus *Austropotamobius* *Caspiastacus* *Pacifastacus*	Signal crayfish (*P. leniusculus*)[2]		
		Cambaridae	*Orconectes*	Rusty crayfish (*O. rusticus*)[2]	Any species of the familyCambaridae excluding *O. rusticus*, zarigani (*Cambaroides japonicus*) and red swamp crayfish (*Procambarus clarkii*)[2]	
			All genera of Cambaridae	None		
		Parastacidae	*Cherax*	Any species of the genus *Cherax*[2]	Any species of the family Parastacidae excluding the genus *Cherax*[2]	
			All genera of Parastacidae	None		
		Varunidae	*Eriocheir*	Any species of the genus *Eriocheir* excluding Japanese mitten crab (*E. japonica*)[2]	None	Any species of the genus *Eriocheir*[2]
Insecta	Coleoptera	Scarabaeidae	*Cheirotonus*	Any species of the genus *Cheirotonus* excluding Yanbaru Long-armed scarab (*C. jambar*)[2]	None	Any species of the families Bolboceratidae, Ceratocanthidae, Diphyllostomatidae, Geotrupidae, Glaphyridae, Glaresidae, Hybosoridae, Lucanidae, Ochodaeidae, Passalidae, Plecomidae, Scarabaeidae and Trogidae[2]
			Euchirus	Any species of the genus *Euchirus*[3]	None	
			Propomacrus	Any species of the genus *Propomacrus*[3]	None	
	Hymenoptera	Apidae	*Bombus*	Large earth bumblebee (*B. terrestris*)[3]	Any species of the genus *Bombus* excluding *Bombus ardens ardens*, *Bombus ardens sakagamii*, *Bombus ardens tsushimanus*, *Bombus beaticola beaticola*, *Bombus beaticola moshkarareppus*, *Bombus beaticola shikotanensis*, *Bombus consobrinus wittenburgi*, *Bombus deuteronymus deuteronymus*, *Bombus deuteronymus maruhanabachi*, *Bombus diversus diversus*, *Bombus diversus tersatus*, *Bombus florilegus*, *Bombus honshuensis honshuensis*, *Bombus honshuensis tkalcui*, *Bombus hypnorum koropokkrus*, *Bombus hypocrita hupocrita*, *Bombus hypocrita sapporensis*, *Bombus ignitus Bombus oceanicus Bombus pseudobaicalensis Bombus schrencki albidopleuralis Bombus schrencki konakovi Bombus schrencki kuwayamai Bombus terrestris*, *Bombus ussurensis*, *Bombus yezoensis*[3]	Any species of the genus *Bombus*[3]

(APPENDIX continued)

Class	Order	Family	Genus	IAS	UAS	LORCA
Insecta	Hymenoptera	Formicidae	*Solenopsis*	Red imported fire ant (*S. invicta*) [1]	None	*S. invicta*[1]
				Fire ant (*S. geminata*) [1]	None	*S. geminata*[1]
			Linepithema	Argentine ant or Tropical fire ant (*L. humile*) [1]	None	*L. humile*[1]
			Wasmannia	Little fire ant (*W. auropunctata*) [2]	None	*W. auropunctata*[2]
Mollusca	Mytiloida	Mytilidae	*Limnoperna*	Any species of the genus *Limnoperna*[2]	None	Any species of the genus *Limnoperna*[2]
	Veneroida	Dreissenidae	*Dreissena*	Quagga mussel (*D. bugensis*) [2]	None	*D. bugensis*[2]
				Zebra mussel (*D. polymorpha*) [2]	None	*D. polymorpha*[2]
	Stylommatophora	Haplotrematidae	*Ancotrema* *Haplotrema*	None	Any species of the family Haplotrematidae[2]	Any species of the families Spiraxidae, Haplotrematidae, Oleacinidae, Rhytididae, Streptaxidae, and Subulinidae[2]
		Oleacinidae	All genera of Oleacinidae	None	Any species of the family Oleacinidae[2]	
		Rhytididae	All genera of Rhytididae	None	Any species of the family Rhytididae[2]	
		Spiraxidae	*Euglandina*	Cannibal snail (*E. rosea*) [2]	Any species of the family Spiraxidae excluding *E. rosea*[2]	
			All genera of Spiraxidae	None		
		Streptaxidae	All genera of Streptaxidae	None	Any member of the family Streptaxidae excluding the genus *Sinoennea* and *Indoennea bicolor*[2]	
		Subulinidae	All genera of Subulinidae	None	Any member of the family Subulinidae excluding *Allopeas brevispirum*, *A. clavulinum kyotoense*, *A. gracilis*, *A. heudei*, *A. javanicum*, *A. mauritianum obesispira*, *A. pyrgula*, *A. satsumense*, *Rumina decollata*, *Subulina octona*[2]	
Platyhelminthes	Tricladida	Rhynchodemidae	*Platydemus*	Predatory flatworm (*P. manokwari*) [2]	None	*P. manokwari*[2]

2. Plant Kingdom

Class	Order	Family	Genus	IAS	UAS	LORCA
Tracheophyte	Sympetalae	Compositae	*Coreopsis*	Lanceleaf tickseed (excluding cut flowers) (*C. lanceolata*) [2]	None	Any species of the genus *Coreopsis*[2]
			Gymnocoronis	Senegal tea plant (*G. spilanthoides*) [1]	None	Any species of the genus *Gymnocoronis*[1]
			Rudbeckia	Cutleaf coneflower (excluding cut flowers) (*R. laciniata*) [2]	None	Any species of the genus *Rudbeckia*[2]
			Senecio	Madagascar ragwort (*S. madagascariensis*) [2]	None	Any species of the genus *Senecio*[2]
		Scrophulariaceae	*Veronica*	Water speedwell (excluding cut flowers) (*V. anagallis-aquatica*) [2]	None	Any species of the genus *Veronica*[2]
	Caryophyllales	Amaranthaceae	*Alternanthera*	Alligatorweed (*A. philoxeroides*) [1]	None	Any species of the genus *Alternanthera*[1]
		Apiaceae	*Hydrocotyle*	Floating marshpennywort or Pennywort (*H. ranunculoides*) [1]	*H. bonariensis*, *H. umbellata*	Any species of the genus *Hydrocotyle*[1]
		Cucurbitaceae	*Sicyos*	Bur cucumber (*S. angulatus*) [2]	None	Any species of the genus *Sicyos*[2]
		Haloragaceae	*Myriophyllum*	Parrotfeather (*M. aquaticum*) [2]	None	Any species of the genus *Myriophyllum*[2]
	Liliopsida	Poaceae	*Spartina*	Common cord grass (*S. anglica*) [2]	None	Any species of the genus *Spartina*[2]
		Araceae	*Pistia*	Water lettuce (*P. stratiotes*) [2]	None	*P. stratiotes*[2]
	Pteridophyta	Azollaceae	*Azolla*	Water fern (*A. cristata*) [2]	None	Any species of the genus *Azolla*[2]

A comparison of legal policy against alien species in New Zealand, the United States and Japan - can a better regulatory system be developed?

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Abstract Legal policy to regulate alien species is discussed. Legal approaches of New Zealand, the United States and Japan are reviewed and compared and legal and structural strengths and weaknesses of the current systems are identified. Based on this, a comprehensive regulatory system is proposed. Regulations on GMO and LMOs are included insofar as they are relevant to this discussion.

In the United States economic damage caused by invasive alien species, especially insect pests, aquatic organism and weeds, has lead to some strengthening of legal measures. For example, the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990, and the National Invasive Species Act of 1996 were enacted to regulate and prevent the introduction and spread of aquatic nuisance species, such as the zebra mussel, through ballast water exchange in the Great Lakes but have not been applied nationally. Japan promulgated the Invasive Alien Species Act in 2004 as a comprehensive measure against alien species. However, these statutes are insufficient because of the "dirty list" approach and bureaucratic sectionalism.

New Zealand tries to solve invasive alien species problems by implementing a "Biosecurity Strategy". The strategy includes comprehensive legislation such as the Biosecurity Act and Hazardous Substances and New Organisms (HSNO) Act and is implemented with tight intra-agency coordination. Especially, New Zealand's HSNO Act calls for the application of the "precautionary approach," and implements a "clean list" approach. However, as shown by the recent debate on genetic modification (GM) in New Zealand, making public decisions embodying the precautionary approach is a complex issue.

In conclusion, the author recommends precautionary risk management through a "Clean list" approach, comprehensive regulatory regimes and public input, as critical components in the development of legal structures to address alien species.

Keywords: Biosecurity; Alien species regulation; Biosecurity Act; HSNO Act; ERMA; Invasive Species Act.

INTRODUCTION

Of the many causes of biodiversity loss, invasive alien species are amongst the most difficult for the general public to fully comprehend even though they are amongst the most destructive. Due to the complexity and often incremental nature of such biodiversity loss, scientists have played a leading role in promoting public awareness and proposing measures to address biological invaders. Although the problem has become more apparent to policy makers within the last decade or so, many jurisdictions are still searching for effective ways to prevent and address invasive alien species.

"Biological invasions by invasive alien species[1] are one of the major reasons for the loss of biodiversity, globally. [2] Huge numbers of organisms have been moved intentionally and unintentionally. This trend has been greatly accelerated by globalisation of trade and travel. Impacts include a progressive biological homogenisation or 'biosimilarity' which could be described as "evolution in reverse". Impacts are particularly devastating on oceanic islands such as Hawaii[3], and in island states like Japan (which is not an oceanic island in strict bio-geographical terms). The effects are also evident in the "New World", such as the Americas. One of the most challenging regions, however, is Oceania, particularly New Zealand and Australia, which have been largely isolated with rich unique biodiversity and where human presence is relatively new.

The biological disturbance of ecosystems has progressed now to the point where it is causing serious economic damage to both developed and developing nations. In the United States, scientists at Cornell University have estimated the total annual losses and

Pages 45-55. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks.* Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

control costs of such problems to be approximately 137 billion dollars a year.[4]

The emergence of bio-technology and the use of genetically modified organism (GMO) and living modified organism (LMO) technology has led to a new set of concerns. There are many divergent opinions toward GMOs, including in the scientific community, and policy makers are struggling to find a resolution.

In recent years New Zealand has introduced a number of comprehensive measures and innovations, and is recognised as something of a world-leader in the field[5]. In this paper I will compare the legal and decision-making systems of New Zealand, the United States and Japan, and discuss options for a better regulatory system to prevent and address the risks of alien species.

HISTORY AND CURRENT STATUS OF LEGAL MEASURES

Although they are now better understood, the problems of invasive alien species are not new. They were first recognised in the agricultural and economic context. As a consequence, early legal measures focused on quarantine and import restrictions to protect primary production from pests, weeds, and diseases. Plant and animal quarantine has in fact been in existence since the late 19th century.

As globalisation has expanded, these conventional legal measures, focusing on the primary sector or on human health interests, have been criticised for their inability to address the comprehensive risks and damages caused to biodiversity by invasive alien species. GMOs have also become an issue of discussion in relation to biodiversity conservation.

United States of America

In the United States, the Plant Quarantine Act,[6] which is the predecessor of the Plant Protection Act of 2000,[7] was enacted by Congress in 1912. Concerning wildlife and natural resources, many states (which traditionally have primary, but not exclusive, jurisdiction over wildlife) have been regulating the deliberate release and importation of certain animals within their jurisdiction. At the federal level, the Lacey Act,[8] which was passed in 1900 as the first federal law on wildlife protection, regulates interstate and international trade in a limited number of wildlife species by prohibiting the importation or transportation of about 95 listed “injurious” species and genera.[9]

As already mentioned, management of wildlife in the USA is fundamentally the role of the local government (states) while the national (federal) government’s role is somewhat limited, especially once the invasive alien species are introduced. Once an alien species establishes itself and becomes invasive, its management is often dealt with separately by local authorities. The role of the federal government is still crucial, however, in strengthening border control and to act as coordinator between numerous local and national agencies.

In recent decades, economic damage caused by invasive alien species, especially insect pests, aquatic organism and weeds, has lead to further federal and state statutes.[10] For example, in reaction to the exploding propagation of zebra mussels in the Great Lakes, the Nonindeginous Aquatic Nuisance Prevention and Control Act of 1990,[11] and National Invasive Species Act of 1996[12] were enacted to regulate and prevent the introduction and spread of aquatic nuisance species in the Great Lakes through ballast water exchange.[13] However, the coverage of these is very limited, such that a major legislative effort to pass more comprehensive proactive aquatic invasive legislation has been underway but without success so far.

In February 1999, President Clinton issued Executive Order 13112.[14] It requires Federal agencies to refrain from actions likely to increase invasive alien species, creates an interagency Invasive Species Council and calls for a National Invasive Species Management Plan to be completed by 2000.[15] The National Invasive Species Management Plan, *Meeting the Invasive Species Challenge*,[16] was released in January 2001. Nevertheless, actual implementation of that plan has been weak and has missed many deadlines.

Even though there are similarities in the potential threats to biodiversity, GMOs are treated differently from invasive alien species. Regulation of GMOs in the United States is fragmented and complex.[17] Regulatory activities are divided between at least three central agencies. For LMOs the Department of Agriculture (USDA) and the Environmental Protection Agency (EPA) are involved. Under the authority of the Plant Protection Act of 2000, the USDA issues permits through the Animal Plant Health Inspection Service (APHIS) for the “import, interstate movement, and field testing of genetically altered plants, microorganisms, and invertebrates”[18]. The EPA is involved in regulating bio-engineered pesticides pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act 1947 (FIFRA)[19] and other provisions. The United States has consistently maintained a pro-

GMO position, but some suggest that the fragmented regulatory system "has failed to adequately determine the effects of GMOs on humans and the environment".[20]

Japan

In Japan, measures towards alien species started with a quarantine system which was introduced shortly after the modernisation of the late 19th century. Before its modernisation, Japan maintained a national isolation policy, which may have helped to protect its biodiversity. Following several discrete measures to prevent animal epidemics and pests, the animal quarantine system was established by 1896, and the plant quarantine was started in 1914. Currently, the quarantine system is operated and administered under the Domestic Animal Infectious Diseases Control Law, Plant Quarantine Law, and related statutes. These statutes are administered and enforced in accordance with international treaties and standards. As Japan is a major importer of agricultural products whilst sustaining its economy through export of industrial materials, it constantly faces demands of deregulation from the agro-industry and exporting countries including the USA. Apart from pest control and quarantine issues, wildlife and conservation laws basically do not address the introduction and/or release of invasive alien species at all.

In recent years, scientists and environmentalists have started to express concern about invasive alien species such as raccoon (*Procyon lotor*) and black bass (*Micropterus*). Ratification of the Convention on Biological Diversity (CBD) led the Japanese government to address the invasive alien species issue. In 2004, the Invasive Alien Species Act was promulgated as a comprehensive measure against alien species in protecting biodiversity.[21] It bans importation, transfer, raising, planting, and carrying of invasive alien species which are specified as invasive by Cabinet Order. At the beginning of September 2006, about 90 species and genera has been designated as invasive by Cabinet Order and the Ministry of Environment is working on reviewing other potentially harmful species, but it still has a long list of species yet to be reviewed.

LMOs also have become a subject for regulation after the ratification of the CBD. In 2003, the Japanese government enacted the Law Concerning the Conservation and Sustainable Use of Biological Diversity through Regulations on the Use of Living Modified Organisms (commonly called the "Cartagena Law").[22] As its common name tells, it was promulgated to fulfil the obligation under the Cartagena Protocol on Biosafety to the CBD.

New Zealand

Consistent with the popular image of a "clean and green" country, New Zealand has been less affected by industrial pollution and over-population, thus enabling the country to learn from the experiences from other countries to prevent future environmental problems.[23] Despite relatively low involvement in some areas, such as pollution control, it has been noted that "New Zealand has introduced one of the most integrated approaches to the management of natural resources in the World".[24] Some of these innovations include: the Resource Management Act of 1991,[25] the establishment of the Environment Court, the establishment of an Environmental Ombudsman, and other bold policies.

Owing to its unique natural environment and its dependence on primary industry, New Zealand has traditionally taken a very strict approach to quarantine and border control issues.[26] The Biosecurity Council of the New Zealand government notes that "in many cases, our competitive edge stems from our relatively natural production systems and relative freedom from pests and diseases."[27] New Zealand has defined its policy against biological pollution as "biosecurity". Within this policy it is aiming to implement an integrated comprehensive strategy based on sound scientific knowledge and research and on applying precautionary measures.[28]

Biosecurity Act

The Biosecurity Act 1993,[29] along with the Hazardous Substances and New Organism Act 1996 (HSNO Act),[30] defines New Zealand's current biosecurity framework. The Biosecurity Act is the primary source of biosecurity, dealing with unintentional introductions, and also covering regulatory programs. These include quarantine of flora and fauna, quarantine of humans where necessary, and pest management of species that have already been introduced and/or became established in New Zealand. In most other countries, the legal framework is often fragmented, e.g. between plant health and animal health or between the sector interests, such as agriculture, conservation and health. In New Zealand however, the Biosecurity Act is comprehensive and integrated, and as such it is a model statute for dealing with biological pollution.

The major objective of the Biosecurity Act is the management of “unwanted organisms” through prevention, quarantine and pest management strategies . “Unwanted organisms” are determined by the Chief Technical Officer of the related Ministries having responsibilities under the Biosecurity Act in their respective areas of responsibility.[31] Once an animal, plant, or any other organism is determined as an unwanted organism, it may not be released in the field, and any cargo containing such material is defined as “risk goods” and may not be imported except when in compliance with a specific import health standard.[32] For unwanted organisms that have already come into the country, pest control plans are promulgated and implemented according to their urgency and necessity by responsible Ministries or Regional Councils. Information on unwanted organisms is widely provided to the public through MAF’s Biosecurity’s internet Web site (*http://www.biosecurity.govt.nz/*).

The HSNO Act

The Hazardous Substances and New Organisms (HSNO) Act created a comprehensive regulatory regime for “new organisms” under a single agency, the Environmental Risk Management Authority (ERMA). The HSNO Act bans any “new organism” not approved by ERMA, from being imported, field tested, or released in to the field.[33] The HSNO Act defines new organism as, inter alia, a species of any organism not present in New Zealand on the date of commencement of the HSNO Act (July 1998); or an organism which is in containment; an organism that has been conditionally released under the Act; a genetically modified organism which is not previously been approved for importation or release; or a an organism that belongs to a species, subspecies, or variety that has been eradicated from New Zealand.[34] Any new organism may only be imported or released in the field after approval.[35]As such, this act applies to proposed intentional introductions of alien species (but not to unintentional introductions that are covered under the biosecurity act). The approval of a new organism is issued by ERMA.

Before making its decision on importation or release, ERMA must carry out a risk assessment at the applicant’s expense.[36] The HSNO Act requires most applications to be notified to the public.[37] “Any person” may submit a written submission to the application.[38] Also, ERMA may hold a public hearing when it considers necessary.[39] ERMA’s decision is discretionary. There is no general right of appeal against ERMA’s decision on an application, except on questions of law which may be appealed to the High Court (not to the District Court or the Environment Court).[40]

ERMA

Although ERMA does not have any operational responsibilities in the Biosecurity Act, it plays an important role in New Zealand’s biosecurity by assessing all alien species and GMOs, that are proposed for intentional introduction and then deciding to ban or approve their introduction. ERMA also administers approvals of hazardous substances as provided by the HSNO Act.

ERMA is a quasi-independent multimember agency comprising of Authority Members appointed by the Minister for the Environment,[41] assisted by staff headed by a Chief Executive and senior executives. Also, to provide a forum for Maori input, an advisory committee (Nga Kaihatu Tikanga Taiao) is established in the ERMA by section 9 of the HSNO Act.

The backgrounds of the Authority Members are diverse an represent the various interests involved. Out of eight Members currently serving (Sept. 2006) four are reported to have a scientific background, others include legal, political, Maori philosophy, and hazardous substances expertise. The science backgrounds of the four Members included bio-chemistry and entomology, biological control, and resource management.[42] Many of the staff are highly qualified scientists, ensuring ERMA’s scientific and technical capabilities to meet its statutory responsibilities.

Government agencies in charge of biosecurity

Under the Biosecurity Act, one of the Ministers of the Crown in Cabinet is appointed as the Minister for Biosecurity. The Minister for Biosecurity takes responsibility for providing for the coordinated implementation of the Act; recording and coordinating reports of suspected new organisms; and managing appropriate responses to such reports.[43] However, the Minister of Biosecurity does not maintain an independent Ministry to carry out his or her duties, but other central government agencies and regional councils hold statutory responsibilities for operations.

The Ministry of Agriculture and Forestry holds overall responsibility for biosecurity as the “lead agency”,[44] but it is just one part of New Zealand’s biosecurity system.[45] The central government is responsible for border management, national-scale events, agency co-ordination, and the legislative framework. The four main biosecurity agencies are: the Ministry of Agriculture and Forestry (MAF), the Department of Conservation (DOC) , the Ministry of Fisheries (MFish) and the Ministry of Health (MOH). These agencies have a memorandum of understanding (MOU) that sets out how they work together on biosecurity matters. [46] Biosecurity New Zealand, established in 2004, is the division of the Ministry of Agriculture and Forestry (MAF) that now has the lead role in preventing unwanted pests and diseases being imported, and for controlling, managing or eradicating them should they arrive. It is charged with biosecurity protection, not only of economic interests, but also health, the natural environment that is unique and special to New Zealand, native flora and fauna, biodiversity, marine areas and a range of resources uniquely important to Maori. [47] Another agency with biosecurity interests is the Environmental Risk Management Authority (ERMA), which makes decisions on applications to introduce hazardous substances or new organisms, including genetically modified organisms (GMOs).

The central agencies and regional councils that have statutory obligations under the Biosecurity Act are responsible for developing national pest management strategies, [48] proposing regional pest management strategies,[49] and taking action in relation to biosecurity emergencies.[50]

A Biosecurity strategy was developed for New Zealand, to promote a comprehensive approach. The Minister of Biosecurity established the Biosecurity Council, [51] as an inter-agency forum, a non-statutory advisory group with an independent chair and comprising the chief executives of the four central agencies and the Ministry for the Environment, the Ministry of Research Science and Technology, ERMA and representative of the various regional councils. Its main task has been the designing the Biosecurity Strategy, which was published in August 2003, [52] and has been fully endorsed by the Government. The Biosecurity New Zealand website (*http://www.biosecurity.govt.nz/bio-strategy/*) describes the strategy as "propos[ing] a new direction for New Zealand's biosecurity, to deal with the mounting pressures on the biosecurity system."[53]

This framework itself is like a general mobilisation order with many ongoing governmental campaigns. However, the resources and business of biosecurity is overwhelmingly concentrated in the hands of MAF – and while there are many advantages to having a "lead agency" there are also potential drawbacks, for instance due to either real, or perceived organisational bias.

For instance, before the restructuring, in 2000 the relative proportions of the government funding administered by the four Ministries were: MAF 93%, DOC 3%, MFish 2.4%, and MOH 1.7%. [54] The concentration of resources and funding with MAF could be explained by the importance of pre-border measures which MAF administers. Within the funding for MAF, 48% was allocated to border inspection[55] and there has been criticism that this concentration reflected the political priorities of government, and MAF's institutional bias toward agricultural risks.[56]

The Biosecurity Council's recommendation was that MAF should be "clearly accountable for the overall management of the whole biosecurity system" [57] and in spite of the biosecurity strategy's incorporation of this concept, and the clear statement that Biosecurity New Zealand's role includes the protection of environmental and human health aspects as well as agricultural protection, not everyone agrees that MAF has been able to significantly change its former agricultural focus yet. As an example, the Royal Forest and Bird Protection Society states that "[d]espite the fact that the scope of biosecurity has been broadened to include environment, marine and health impacts, MAF is the lead agency and a strong bias towards agriculture, horticulture and forestry remains". [58]

PROBLEMS WITH CONVENTIONAL LEGAL MEASURES

Measures to address invasive alien species must be scientifically sound to be efficient and effective. However, most of the legal instruments arose from old structures and often they are overridden by existing vested interests or funding shortages.

Insufficient coverage of unwanted organisms

In the United States, the Lacey Act has been addressing "injurious species" since 1900 and quarantine laws have been around from the early 20th century. However, many categories of organism were not dealt with at all until quite recently and, on the whole, it is a "dirty" list approach. Many nations still lack a regulatory regime of any sort for the management and control of alien species. For example, Japan is one of the largest importers of agricultural products in the World. Yet Japan does not regulate seeds, weeds, or trees in their plant quarantine laws. Japanese plant quarantine deals only with pests such as insects, fungi, and parasites. Owing to the low interest in nature conservation and outdoor recreation generally in Japan, wildlife laws and park management regulations lack sufficient provisions against the release of animals in the wild, and the importation of wild animals.[59] Nevertheless, after growing awareness on the concept of biodiversity in recent years, Japan has enacted the Invasive Alien Species Act in 2004, to protect biodiversity. Many nations are now starting to take legal measures against invasive alien species, and the threats are also being addressed by an increasing number of international conventions, such as the Convention for Biological Diversity. However, those legislations are often a patchwork of existing regulations and thus difficult to comprehend, moreover, often there will be gaps and/or loopholes. The most comprehensive, yet easily understood, statutes are found in New Zealand.

"Dirty List" approach

Existing legislation has often been reactive in approach, addressing only those species known to be dangerous ("dirty list"), thus making the regulatory range very narrow.[60] The importance of a "precautionary approach" cannot be overstated in combating invasive alien species. It is a truism that alien species that are inconspicuous in their native habitat often cause major disruptions to the balance of local ecosystems when introduced to a habitat where they are alien. It is very difficult to predict the effects of such introductions. Therefore, a "clean list" approach is suggested, imposing a general ban on introductions of alien species except those that have been assessed and approved.[61] Prior to any decision on such introductions, an environment risk analysis should be executed, and approval should only be granted to species that pose a low risk compared to the benefit of their introduction. Regrettably, however, the precautionary approach has not been widely implemented. Japan's Invasive Species Act regulates only about 90 species so far, which have been gazetted as invasive. In the United States, the list of regulated exotic species is longer, but the basic system of requiring a listing before regulation is the same and it can take many years to reach a decision. On the other hand, New Zealand has the "clean list" approach, at least, in theory. In reality, New Zealand's "clean list" standard is not always so rigorous, but the different paradigm is nevertheless crucial.

Coordination between regulatory agencies

Statutes and measures for biosecurity have, in the past, been ad hoc and uncoordinated between independent agencies and stakeholders.[62] Ecologically, issues associated with alien species are often interrelated within the ecosystem. However, the current legal and administrative systems, inter-jurisdictional and intra-jurisdictionally, are not designed to address ecological issues comprehensively or in an integrated fashion.[63] Integration between regulatory agencies at national level and local government is also indispensable.[64]

New Zealand has a Minister of Biosecurity responsible for providing coordination in the implementation of the Biosecurity Act, and the Ministry of Agriculture and Forestry as the lead agency. However, the USA and Japan do not have a single responsible agency for within- government coordination and as a result they often fall into bureaucratic sectionalism.

Internationally, efforts are being made to construct clearing house mechanisms for the sharing of information and to improve coordination.[65]

Difficulties in applying the precautionary approach

Under the New Zealand HSNO Act, the regulatory agency (ERMA) may, at its discretion, decline an application to introduce a new organism "if insufficient information is available to enable the Authority to assess the adverse effects of the organism".[66] This HSNO Act has the first incorporation of the term "precautionary approach" in New Zealand domestic legislation.[67] It states:

> "Precautionary approach – All persons exercising functions, powers, and duties under this Act . . . shall take into account the need for caution in managing adverse affects where there is scientific and technical uncertainty about those effects".[68]

Even in New Zealand, however, the application of the precautionary principle is not yet well established, as the following examples show. The Planning Tribunal (which is the predecessor of the Environment Court) in an early Resource Management Act (RMA) case held that the lack of knowledge does not automatically leads to a phase-out: "It is not to reject the precautionary approach, but there needs to be some plausible basis, not mere suspicion or innuendo for adopting that approach".[69] In another RMA case, concerning alleged harmful health effects from a mobile phone transmitter, the Planning Tribunal in *McIntyre v Christchurch City Council* struck down the case after studying the U. S. Supreme Court's decision in *Daubert v Merrill Dow Pharmaceuticals, Inc.,* [70] and other cases from England, Canada and New South Wales, largely on the question of the reliability of the contested scientific evidence.[71] Christensen & Williams suggest that the Environment Court (the Planning Tribunal's successor) should follow *McIntyre* in this type of situation and be satisfied on the "balance of probabilities" that "reliable evidence" in support of the allegation is presented, "[t]he onus will then be on the other party to prove to the balance of probabilities that the alleged effect would not occur". [72]

Biosecurity is essentially aimed at applying the precautionary approach towards biohazards. However, the concept of a "precautionary approach" is still new and not well established. The tools designated to carry out the precautionary approach are numerous, and this application can vary widely, depending upon the context. In some circumstances the precautionary approach could be "weak" and undistinguishable from conventional approaches, and in other circumstances it could be an absolute ban or phase-out.

Difficulties in decision making based on both technical and non-technical issues

Accounting for both technical and non-technical issues simultaneously is difficult even in New Zealand. The MAF Biosecurity Authority Chief Director and executives are scientists (including veterinarians). ERMA's Members and executive staff, including the Chief Executive, are also scientists. Under the Biosecurity Act, Chief Technical Officers of the responsible agencies, (who are also often scientists), are given the power to determine "unwanted organisms". These decisions are framed to be technical, but involve value judgments and can lead to "unresolved disagreement between agencies."[73]

Often scientists are obliged to make difficult decisions. Decisions on Import Health Standards, which are required by the Biosecurity Act for importation of risk goods, are delegated by statute to technical officers. Even though the original aim was to avoid political interference in the decision-making process, the *Biosecurity Draft 2002* apprehended their decisions as narrow and technical, recognising that where "societal, environmental or other (non-technical) concerns are under consideration . . . technical officers [of specific agency] . . . may not be the appropriate final decision maker". [74] Although the *Biosecurity Strategy* issued in 2003 did not make the point clear, the *Biosecurity Draft 2002* stated that the decision should be escalated in the following cases: where conflicting interests exist; where there is uncertainty in risk; or where society's desired level of protection is unclear.[75] The words of the *Biosecurity Draft 2002* may fit well with the idea of separating technical (scientific) findings and policy decisions.[76]

Separating the functions of science on one hand and policy decisions on the other is legitimate in a general context, especially when socio-economic issues are in stake, but there is a risk that legislative intent is eviscerated in the implementation of specific cases. Ultimately, New Zealand intends to base decisions on cost (risk)/ benefit analysis,[77] and, combined with its international obligations, it is not clear on how much the legislator intended to defer to scientific decisions. It is a reality that regulatory decisions for environmental protection may not be consistent with scientific priorities. Often, the legislators' decisions are not based on scientific priorities or reasons, but rather on political interests and expediency. As a result, ecologically crucial cases can be left with little or no resources to address them.

Neutrality and challengeability of decisions by specialists

In New Zealand, decisions made by agencies responsible for biosecurity such as ERMA are generally made by in-house scientists, and are not subject to external peer review. As already mentioned, ERMA's scientific decisions are judicially unchallengeable. According to the fields of expertise and disciplines of staff scientists in the agencies, such as biotechnology in ERMA and agro science in MAF, there is the opportunity for unintentional bias in their scientific values and decisions. In an interview that the author had with Dr. Walker, the then Chief Executive of ERMA and his senior scientist, they expressed confidence in the biotechnology science, and expressed a sense of mission in promoting a responsible and sustainable bio industry in New Zealand.[78] Their views are legitimate and there was no intention of bias in them. Often, however, scientists have a zealous attachment to their practice and work and sometimes promote it, at the expense of other values.

In New Zealand, the relatively small size of the scientific community and the high turnover in the career market may act to minimise biased scientific decisions. Since 1984, when the Fourth Labour Government came into power on a platform of New Right economic principles, New Zealand has been experiencing drastic institutional reform and privatisation, even in environmental policies.[79] This reform has stimulated career mobility and many private consultants or former NGO staff are hired by the government to deal with environmental or biosecurity affairs. This, however, does not vindicate the concerns that government technocrats may monopolise scientific decisions. It is the role of the judiciary to check on the administrative branch of the government, but, even assuming that a specialised environmental court exists, such as in New Zealand,[80] it is unlikely that a judicial branch would be able to review technical decisions on biosecurity.

Public Input

Although decisions in the biosecurity context tend to be technical, policy decisions should be made with public input. The HSNO Act in New Zealand provides wide opportunity for public participation as the applications are generally required to be notified to the public.[81] Any person may make a submission in writing to a notified application.[82] When the submitter requests a hearing, ERMA has to conduct the hearing at the expense of the applicant seeking approval.

The drafting of the Biosecurity Strategy involved the participation of conservation groups such as the Royal Forest and Bird Protection Society of New Zealand and also received many submissions from the public. Opinions relating to invasive alien species are not all that divergent, as the public is keen to preserve New Zealand's unique biodiversity. However, there are very divergent views and heated debate on the issue of genetic engineering (commonly called "GE" in New Zealand) and genetically modified organisms.

New Zealand's leading newspaper, The New Zealand Herald, has been carrying pro-GE campaign articles and has condemned ERMA procedures as being "tipped heavily against the scientific community". It stated "[s]o much is the process weighted towards public participation that it can easily be hijacked by anti-GM groups".[83] This, the editorial alleges, creates a risk that New Zealand may lose competitive power in the bio industry and it may also result in the loss of scientists who will leave the country for better positions overseas.[84] There is no doubt that the approval process of ERMA has constantly been longer than set out by statutory deadlines. In the year ending June 30, 2001, before the aftermentioned moratorium, the average time period for processing was 203 working days for GMO application and 135 working days for conventional new organisms.[85] Although records show that the statutory timeline was exceeded, this was not unreasonable. Also, there were only three hearings in the business year 2001, of which one was a GMO application with a three-day hearing, and another was a one-day hearing for a non-GMO application.[86] Thus, the image of activists clogging the agency with hearings and cross-examinations is not accurate.

Despite assurances of participation in the procedures, public anxiety toward genetic engineering and concern about the effects upon biodiversity was overwhelming in New Zealand, leading to a big political debate. In October of 2001, after a year of protracted discussions, the Labour government announced its decision to put in place a moratorium of two years – which expired on October 2003 - and to prohibit the release of genetically modified organisms in the environment, while allowing time for further research.[87] To attempt to resolve this controversial issue, the government formed the Royal Commission on Genetic Modification as a forum for debate and investigation into genetic engineering and genetically modified organisms. The Commission received over 10,000 written submissions and conducted 13 weeks of hearings.

Maori groups, along with environmentalists including NGO's and the Green Party, provided very strong opposition against releasing GMO's in the environment, whilst the mainstream of the Labour Party (currently the leading party of the coalition government), were not in favour of the moratorium. The political and legal rights of Maori under the Treaty of Waitangi gives them a position as important stakeholders in issues relating to natural resources and heritage, which the government cannot ignore.[88] This unique position of Maori is often described as a "guardianship" role for the environment, while it is seen by detractors as a roadblock to the biotech industry.[89] The NZ Herald editorial criticised Ngai Tahu, one of the major Maori iwi (tribe) and the most active one on GM issues, to be "largely close minded" and suggested that ERMA is giving too much weight to spiritual values of Maori rather than to science.[90] However, Ngai Tahu member Whiti Reia has been reported as saying, "consultation with Maori was required because 'the public does not trust scientists' [as well as by] treaty obligations".[91] One of the reasons for the conspicuous role of Maori may be the barrenness of scientific debate over GE. The environmentalists, whose views are generally regarded as being represented by the Green Party, are alleging that the risk of GE comes from scientific uncertainty. Winning the risk debate on scientific grounds is risky for the environmentalists, because the argument fundamentally arises from a difference of values. Therefore, the frontal attack by Maori based on cultural values, rather than science, made a significant difference especially when the biosecurity policy's aim involves defending native biodiversity.

Cost of science and public input

Science is expensive. Financial demands for better science are potentially infinite. The Biosecurity Strategy notes "[a]vailable sources are not well marshalled, most New Zealand sources of data are hard to access and many are poorly maintained".[92] To pay the cost, the New Zealand government, has abided by the New Right's "user pays" principle.

In ERMA's approval procedure, the cost is borne by the applicants, including the public notification and hearings. The imposition of costs is disputable when considering the uncertainty of the potential advantages which the applicant may gain from the approval of the new organism, given that such approval is generally nonexclusive.

Although acknowledging these financial realities, Dr. Walker, the then Chief Executive of ERMA, has noted the difficulty in coordinating with applicants to fund in depth investigation and supplemental procedures that the agency deems necessary.[93] In Dr. Walker's opinion governmental funding is reasonable when pursuing public benefits. On the other hand, internalisation of costs is expected to have the effect of discouraging the introduction of dangerous alien species, as well as other irresponsible projects.

THE IDEAL REGULATORY SYSTEM

New Zealand's biosecurity system can be acknowledged as one of the boldest experiments to bring science and the precautionary approach into policy, regulation and operational decision-making at the domestic level. It has a comprehensive structure and the responsible agencies are relatively well staffed by scientists and professionals. The decisions are made after risk assessments and public input. The legislature has intended to give appropriate weight to scientific and technical data and to analysis to arrive at decisions that are politically neutral. New Zealand's comprehensive structure and the internalisation of the precautionary approach, among other things, provide a model that other countries could learn from.

Although New Zealand's Biosecurity Act and HSNO Act do not necessarily provide the ultimate solution, and although the precautionary approach is not always the panacea it is often made out to be, the attitude of boldly internalising science in legislation should be used as a model in the post-modern state.

All things considered, the future ideal policy in dealing invasive alien species is complex, but the basics should include:

1. Precautionary risk management through a "Clean list" approach:
   Alien species, unknown and/or new organism should be suspected as "injurious". Introduction and usage should be approved only after the risks are assessed. The approval should be granted only after the expected benefits, including social, exceed the risks, and the applicant (user) should carry the burden of proof.

2. A comprehensive regulatory regime:
   In designing the regulatory scheme and roles of agencies, it is easy to end up with a haphazard patchwork by using existing statutes and agencies. Of course it is better to use every tool one can use, but it is the best to design a regulatory regime with a single lead agency under a comprehensive statute. The regulatory agency does not have to be literally a single agency as this may not be realistic, but one agency should act as a lead agency to coordinate multiple others. This agency needs to have capacity to take a science based approach to its decisions.

3. Public input:
   Even with science playing a key role in decision-making, public input is a key requirement in the decision making on the introduction of alien species. The New Zealand example shows that an evaluation of scientific argument is not always easy for the general public. Furthermore, the public does not make up its mind solely based on science, but includes other values, including spiritual and cultural ones. The New Zealand government engaged in a nationwide debate on GE only after the HSNO Act and ERMA were put in place. A lesson for Japan is that input on values and public opinion should be sought at an early stage. In addition, government policy should be determined on the basis of such consultation, before implementation takes place in the form of legislation and delegation of decision-making to the scientific community or others. It would be very unwise for technical agencies and scientists to make decisions before the value argument is settled. Although addressing alien species must rely on science, in a democracy, the preferences and values of the general public must be

incorporated in the system in a balanced and representative way.

ACKNOWLEDGEMENTS

The author would like to thank the Japan Fund for Global Environment for funding the research trip to New Zealand, and to David Grinlinton (University of Auckland) and Aaron Courtney for their invaluable comments as well as Peter Jenkins from the International Center for Technology Assessment, USA and Maj De Poorter from the Invasive Species Specialist Group. The responsibility for any errors resides with the author.

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[12] 16 U.S.C. §§4701-51 (amending the Nonindeginous Aquatic Nuisance Prevention and Control Act of 1990).

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[30] Hazardous Substances and New Organism Act 1996 (Public Law 1993 No30).

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[34] *Id.* §2 (1).

[35] *Id.* §34.

[36] *Id.* §21.

[37] *Id.* §53.

[38] *Id.* §54.

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[40] *See* Justine Sefton, *The Hazardous Substances and New Organism Act 1993 – Legislation and Treaty Notes,* N. Z. Envtl. L. J 2:263, 265 (1998).

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[43] Biosecurity Act §8.

[44] *The Biosecurity Strategy*, at 17.

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[49] *Id.*§13, 71 & 73.

[50] *Id.*§11.

[51] *See* MAF, Biosecurity Forums, at *http://www.maf.govt.nz/ biosecurity-strategic-unit/biosecurity-forums/ index.htm* (last visited Oct. 12, 2005).

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The role of the International Plant Protection Convention in the prevention and management of invasive alien species

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Abstract The International Plant Protection Convention (IPPC) is an international treaty relating to plant health. The purpose of the IPPC is "to secure common and effective action to prevent the spread and introduction of pests of plants and plant products, and to promote appropriate measures for their control" (Article I of the IPPC).

One of the IPPC's objectives is to protect cultivated/unmanaged plants, wild flora, habitats and ecosystems with respect to invasive alien species (IAS) that are plant pests. Several national obligations under the IPPC that are relevant to IAS are:

- establishment of an official national plant protection organization (NPPO);
- conduct of pest risk analysis;
- surveillance of plants and plant products with the object of reporting;
- eradication or control of pests.

An IAS which has been identified, through pest risk analysis, to have potential economic importance (including environmental considerations) can be nationally listed as a regulated pest and the NPPO can put measures in place to prevent its introduction.

The Commission on Phytosanitary Measures (CPM) is the governing body for the IPPC and provides a forum for discussions on international plant protection issues. The IPPC framework facilitates relationships between countries, encourages them to cooperate at a regional level and gives them guidance in developing their own national plant protection systems.

In addition, the IPPC is developing closer cooperation with other international organisations that are also concerned with the management of IAS such as the Convention on Biological Diversity (CBD). In this case, the Secretariats of the IPPC and CBD have achieved closer cooperation with the signing of a Memorandum of Understanding and the development of a joint work programme in an effort to promote synergy and avoid unnecessary duplication.

The prevention and management of IAS relevant to plants requires close cooperation at the national, regional and international level.

This paper clarifies how the purpose of the IPPC applies to IAS that are pest of plants, outlines the key national obligations under the IPPC, and describes regional and international cooperation in regard to IAS. All of these elements help contribute to the prevention and management of IAS.

Key words: Commission on Phytosanitary Measures (CPM), International Plant Protection Convention (IPPC), International Standards for Phytosanitary Measure (ISPM), invasive alien species (IAS), national plant protection organization (NPPO), pest (pest of plants and plant products), pest risk analysis (PRA), regional plant protection organization (RPPO)

INTRODUCTION TO THE INTERNATIONAL PLANT PROTECTION CONVENTION (IPPC)

The International Plant Protection Convention (IPPC) is an international treaty "with the purpose of securing common and effective action to prevent the spread and introduction of pests of plants and plant products, and to promote appropriate measures for their control" (Article I of the IPPC). The IPPC has been deposited with the Director-General of the Food and Agriculture Organization of the United Nations (FAO) and is administered through the IPPC Secretariat, located in FAO's Plant Protection Service.

The IPPC was adopted by the Sixth Session of the FAO Conference in 1951 and came into force in 1952. Subsequently, it was revised in 1979, with this revision coming into force in 1991. The IPPC was further amended in 1997 and on 2 October 2005, these amendments came into force with respect to all

Pages 56-62. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

contracting parties to the IPPC. The 1997 amendments update the Convention and reflects the role of the IPPC in relation to the *Agreement on the Application of Sanitary and Phytosanitary Measures* of the World Trade Organization (the WTO-SPS Agreement). As of December 2006, 158 countries are contracting parties to the IPPC.

The IPPC lays out obligations for countries which are contracting parties. Contracting parties, have the sovereign right to use phytosanitary measures to regulate plants and plant products, and other articles capable of harbouring plant pests. In addition, the Commission on Phytosanitary Measures (CPM), which is the governing body for the IPPC, provides a forum for international cooperation, harmonisation of phytosanitary measures, exchange of information and provision of technical assistance in collaboration with national and regional plant protection organizations.

THE SCOPE OF THE IPCC INCLUDES INVASIVE ALIEN SPECIES (IAS)

The IPPC not only applies to the protection of cultivated plants and plant products, but also extends to the protection of uncultivated/unmanaged plants, wild flora, habitats and ecosystems. Its scope covers organisms that can cause indirect damage to plants, as well as invasive alien species such as weeds. The provisions of the IPPC also cover conveyances, containers, storage places, soil and any other objects or material capable of harbouring plant pests.

The scope of the IPPC thus extends to organisms which are pests because they:

- directly affect uncultivated/unmanaged plants
- indirectly affect plants (e.g. weeds or invasive plants cause damage through competition for light, water or other sources)
- indirectly affect plants through effects on other organisms (for example, parasites of beneficial organisms such as biological control agents or pollinators could eliminate or reduce the benefits of these organisms by reducing predation or causing ineffective pollination).

KEY ACTIVITIES AND ELEMENTS OF THE IPPC FRAMEWORK

The IPPC lays out several obligations for countries in order to prevent the introduction and/or spread of pests of plants and plant products. Many of these obligations, and International Standards for Phytosanitary Measures (ISPMs) developed under the IPPC framework, overlap with the scope and intention of the *Guiding principles for the prevention, introduction and mitigation of impacts of alien species that threaten ecosystems, habitats or species* (Decision VI/23) of the Convention on Biological Diversity (CBD/COP, 2002).

Provisions of the IPPC and ISPMs that are relevant to the Guiding Principles require countries to:

- adopt phytosanitary legislation, regulation, or official procedures;
- assess and manage potential plant pest risks;
- protect areas that may be threatened by plant pests;
- build capacity and technical assistance for developing countries;
- apply measures to prevent unintentional introduction of plant pests;
- certify that exports have met importing countries requirements;
- assess and manage the intentional introduction of organisms that may be pests of plants, including claimed beneficial and biological control organisms;
- exchange scientific and regulatory information relevant to plant pest;
- cooperate between countries to minimise the impact of plant pests;
- detect, control and eradicate pests in areas under cultivation and in wild flora.

For plant protection, there are three main stages for the prevention and management of pests, which are prevention of the introduction, early detection and thirdly the mitigation of impacts. These three stages are covered and implemented by a range of national obligations that countries have as contracting parties to the IPPC. ISPMs provide guidance for countries to meet these obligations. In meeting their obligations under the IPPC, most countries have established regulatory organisations which assess and manage the risk(s) of pests that threaten plant health.

International Standard Setting

Under the IPPC (i.e. Article X), countries agree to cooperate in the development of ISPMs. These ISPMs, which are adopted by the governing body for the IPPC, provide guidance to countries to meet their IPPC obligations (as of April 2006 27 ISPMs are adopted). ISPMs are intended to harmonise phytosanitary measures applied in international trade. In addition, according to the WTO-SPS Agreement, international standards, guidelines and recommendations developed under the auspices of the Secretariat of the IPPC (in cooperation with regional plant protection organizations) are recognised by members of the WTO, and member countries are required to base their phytosanitary measures on them (WTO, 1994). This recognition by the WTO-SPS Agreement gives strength to international standards, guidelines and recommendations developed under the IPPC framework.

The following ISPMs which are most relevant to the prevention and management of IAS are:

- *Guidelines for pest risk analysis* (ISPMs No.2, No.11 and No.21);
- *Guidelines for the export, shipment, import and release of biological control agents and other beneficial organisms* (ISPM No.3);
- *Requirement of the establishment of pest free areas* (ISPM No.4);
- *Glossary of phytosanitary terms* (ISPM No.5);
- *Guidelines for surveillance* (ISPM No.6);
- *Export certification system* (ISPM No.7);
- *Determination of pest status in an area* (ISPM No.8);
- *Guidelines for pest eradication programmes* (ISPM No.9);
- *Requirements for the establishment of pest free places of production and pest free production sites* (ISPM No.10);
- *Guidelines for phytosanitary certificates* (ISPM No.12);
- *Guidelines for the notification of non-compliance and emergency action* (ISPM No. 13);
- *The use of integrated measures in a systems approach for pest risk management* (ISPM No.14);
- *Guidelines for regulating wood packaging material in international trade* (ISPM No.15);
- *Pest reporting* (ISPM No.17);
- *Guidelines on lists of regulated pests* (ISPM No.19);
- *Guidelines for a phytosanitary import regulatory system* (ISPM No.20);
- *Requirements for the establishment of areas of low pest prevalence* (ISPM No.22);
- *Guidelines for inspection* (ISPM No.23);
- *Diagnostic protocols for regulated pests* (ISPM No.27).

(ISPMs are available from *http://www.ippc.int*)

Technical Assistance

Technical assistance is important in aiding developing countries with the implementation of the IPPC (i.e. Article XX). The IPPC Secretariat has a programme to facilitate technical assistance. It devotes considerable resources in providing technical assistance to countries in order to build phytosanitary capacity. Some of these activities include sending officers to work with countries in developing phytosanitary legislation, organising regional workshops (e.g. reviewing draft ISPMs, evaluating phytosanitary capacity), and providing travel assistance to enable participants from developing countries to attend relevant workshops and meetings.

Technical assistance and capacity-building relevant to IAS and analysis of environmental risks are included in appropriate IPPC activities. In 2003, the IPPC Secretariat organised a workshop on IAS in cooperation with Germany to seek to explain the role of the IPPC and how its framework can contribute to the management and mitigation of risks posed by IAS relevant for plants. 110 participants from both phytosanitary services and environmental protection agencies attended, with over half of the participants coming from developing countries.

(The information of the workshop is available from *https://www.ippc.int/servlet/CDSServlet?status=ND0yNjkwMSY2PWVuJjMzPSomMzc9a29z*) A handbook on the management of IAS through the use of the IPPC framework, based on discussions and presentations from the workshop, was published in 2005 (FAO, 2005).

(This handbook is available from *http://www.fao.org/docrep/008/y5968e/y5968e00.htm*)

Information Exchange

Under the IPPC (i.e. Article VIII), each country is obliged to designate a contact point for the exchange of information connected with the implementation of the IPPC. In particular, countries are required to cooperate in the exchange of information on plant pests, such as the reporting of the occurrence, outbreak and spread of a pest that may be of immediate or potential danger. Pest reports and other relevant information should be submitted directly to these contact points, as these are the countries representatives established to communicate such information.

The IPPC Secretariat has set up the International Phytosanitary Portal (IPP) - *http://www.ippc.int* which provides a mechanism for countries to exchange relevant phytosanitary information. The IPP includes contact details for national and regional plant protection organizations, and the IPPC Secretariat. It also provides documents developed under the auspices of the IPPC Secretariat such as the IPPC text, ISPMs and documents and meeting reports relevant to the meetings organised by the IPPC Secretariat. Developments for electronic information exchange among countries are under way, including an official pest reporting system.

National Plant Protection Organizations (NPPOs)

Under the IPPC (i.e. Article IV), each country is required to establish a national plant protection organization (NPPO).

The key responsibilities of a NPPO include:

- surveillance of growing plants (cultivated and non-cultivated), and plants and plant products in storage/transportation, with the object of

reporting the occurrence, outbreak and spread of pests, and of controlling those pests;
- conduct of pest risk analysis;
- protection of endangered areas;
- disinfestations/disinfection of consignments (of plants and plant products) moving in internationally, to meet phytosanitary measures;
- issuance of certificates relating to the phytosanitary regulations of the importing country;
- inspection of consignments.

Pest Risk Analysis (PRA)

Pest risk analysis (PRA) is "the process of evaluating biological or other scientific and economic evidence to determine whether a pest should be regulated and the strength of any phytosanitary measures to be taken against it" (IPPC definition). The objectives of PRA are, for a specified area (which could be a country, part of a country, or parts of several countries), to identify pests and/or pathways of phytosanitary concern and evaluate their risk, to identify endangered areas, and, if appropriate, to identify risk management options. According to the IPPC (i.e. Article IV), NPPOs have the responsibility for the conduct of PRA.

Phytosanitary measures that countries institute should be technically justified, through the application of PRA. PRA will determine if a plant, plant product or article should be regulated and the strength of phytosanitary measures to be taken to mitigate the risk. Most NPPOs have set up a comprehensive screening system based on PRA to evaluate proposed introductions.

The decision to allow the intentional introduction of alien species into a county, or into new ecological areas within the country, should only take place after the alien species have been evaluated through PRA. This evaluation includes the assessment of organisms such as plants for planting, beneficial organisms and living modified organisms (living modified organism is defined as "Any living organism that possesses a novel combination of genetic material obtained through the use of modern biotechnology": Cartagena Protocol on Biosafety to the CBD, 2000), to determine whether they are potentially injurious to plants in the PRA area. For an intentionally introduced organism, establishment within defined areas is normally what is intended but the risk of unintentional spread beyond these should be considered. Unintentional introduction of organisms associated with a particular pathway may also be evaluated through PRA.

The scope of the IPPC covers environmental risks including impacts of IAS that affect plants. Until recently, there was no specific guidance on the analysis of risks of plant pests to the environment and biological diversity nor on evaluating potential phytosanitary risks to plants by living modified organisms. In addition, there was no clear understanding of potential economic importance with reference to environmental considerations. Two supplements to ISPM No.11 were developed to add criteria to the PRA process (i.e. ISPM No.11 "*Pest risk analysis for quarantine pests, including analysis of environmental risks and living modified organisms*") and a supplement to ISPM No 5 "*Glossary of phytosanitary terms*" clarifies the relationship between environmental considerations and potential economic importance as used in PRA.

When evaluating environmental risks, including IAS threats, the sources of information available to the NPPO will generally be wider than those traditionally used. On the other hand, little information may exist for IAS that do not affect the agriculture/horticulture sectors and so the impacts of these IAS will be largely unknown. Partnerships between NPPOs and national environmental protection authorities and researchers (ecologists) may support the NPPOs in conducting PRAs by providing additional information gathered by researchers (including the identification of 'triggers' for invasion). This type of information will be extremely useful when conducting PRAs for new import requests of alien species where limited traditional information is available.

KEY OBLIGATIONS UNDER THE IPPC, RELEVANT TO IAS

For plant protection, there are three main stages that help to prevent the introduction and spread of pests including IAS. These stages are prevention of the introduction, early detection and mitigation of impacts of pests. This section describes key national obligations under the IPPC and ISPMs that are relevant to these three stages respectively.

Prevention of Introduction of IAS

Import Regulatory System that covers several IPPC obligations

Under the IPPC (i.e. Article VII), with the aim of preventing the introduction and spread of pests into their territories, countries have the sovereign authority to regulate imports (the entry of plants, plant products and other regulated articles) in order to achieve their appropriate level of protection.

ISPM No.20 "*Guidelines for a phytosanitary import regulatory system*" describes the structure and operation of a phytosanitary import regulatory system. The objective of a phytosanitary import regulatory system is to prevent the introduction of pests with imported plants, plant products and other regulated articles.

An import regulatory system should consist of two main components:

- a regulatory framework of phytosanitary legislation, regulations and procedures;
- an NPPO that is responsible for operation or oversight (organisation and management) of the system.

The issuing of regulations is a country responsibility (i.e. Article IV). The phytosanitary legal framework should include:

- legal authority to enable NPPO to carry out its responsibilities and functions with respect to the import regulatory system;
- authority and procedures, such as through PRA, to determine phytosanitary measures;
- phytosanitary measures that apply to imported plants, plant products and other regulated articles;
- prohibitions that apply to the import of plants, plant products and other regulated articles;
- legal authority for action with respect to non-compliance and for emergency action.

Countries may make special provisions for the import of biological control agents for research and field release (i.e. Article VII and ISPM No.3 "*Guidelines for the export, shipment, import and release of biological control agents and other beneficial organisms*"). Such imports may be authorised subject to the provision of adequate safeguards.

In operating an import regulatory system, the NPPO has a number of responsibilities. These include:

- carrying out surveillance to maintain information on pest status (presence or absence) for technical justification of phytosanitary measures;
- conducting pest risk analysis for technical justification of phytosanitary measures;
- listing of pests;
- auditing procedures in the exporting country and checking compliance at import.

The IPPC requires importing countries to establish and update lists of regulated pests (i.e. Article VII). These lists should be made available, on request, and can help exporting countries ensure that regulated IAS are not exported.

Import regulations often include specific requirements that should be applied in the exporting country, such as production procedures (e.g. inspection of plants while growing) or specialised treatment procedures (e.g. laboratory testing, cold treatments or fumigation).

In addition, certification systems are useful to ensure compliance with relevant phytosanitary measures of the importing country for imported plants and plant products. The NPPO is responsible for inspection, laboratory testing, verifying documents and other related activities leading to the issuance of the certification.

Preventing the Introduction of IAS through Wood Packaging Material

ISPM No.15 "*Guidelines for regulating wood packaging material in international trade*" sets out technical and labelling measures to reduce the risk of introduction and spread of pests associated with wood packaging material in international trade. This standard helps to prevent unintentional introductions of pests that could have a negative impact on forests.

Early Detection of IAS

Under the IPPC (i.e. Article IV and VII), countries are obliged to undertake "surveillance of growing plants, including both areas under cultivation (*inter alia* fields, plantations, nurseries, gardens, greenhouses and laboratories) and wild flora, and of plants and plant products in storage or in transportation, particularly with the object of reporting the occurrence, outbreak and spread of pests, and of controlling those pests". Countries are also required to conduct surveillance for pests and to develop and maintain adequate information on pest status in order to support categorisation of pests, and for the development of appropriate phytosanitary measures.

According to ISPM No. 6 "*Guidelines for surveillance*", there are two major types of methods to gather information. The first method is through general surveillance, which is a process whereby information on pests is gathered from many sources, and the second method is through specific surveys where NPPOs obtain information on pests of concern, at specific sites and in a specific area over a defined period of time.

Information gathered through such systems will most often be used: to aid early detection of new pests; to support NPPO declarations of pest freedom in an area (pest free area); in the compilation of pest lists; and to support the development of appropriate phytosanitary measures.

Mitigation of Impacts of IAS

The ability to eradicate pests from an area, and the ability to control pests and restrict their spread are essential components of a phytosanitary system. Several ISPMs provide partial guidelines on limiting the spread of pests (i.e. ISPM No.4 "*Requirements for the establishment of pest free areas*", ISPM No.9 "*Guidelines for pest eradication programmes*", ISPM No.10 "*Requirements for the establishment of pest free places of* *production and pest free production sites*", ISPM No.14 "*The use of integrated measures in a systems approach for pest risk management*", and ISPM No.22 "*Requirements for the establishment of areas of low pest prevalence*").

Eradication

ISPM No.9 "*Guidelines for pest eradication programmes*" provides provisions concerning the establishment of pest eradication programmes, which specifies administrative and technical components of such programmes.

Examples of eradication programmes in Japan are:

(1) The oriental fruit fly:

The oriental fruit fly, which had inhabited the South western Islands (Okinawa and Amami) and Ogasawara Islands, was eradicated in 1986 with the total expenditure of 5 billion yen over 18 years.

(2) The melon fly:

The melon fly in the South western Islands was exterminated in 1993 at the total cost of 20.4 billion yen over 22 years. Farmers in these regions now have a wider choice of tropical and subtropical crops to grow for the mainland market.

(3) At present, the eradication of the sweet potato weevil is underway in the South western Islands.

Emergency action

The IPPC (i.e. Article VII) allows for countries to take appropriate emergency action upon the detection of a pest posing a potential threat to its territory or the report of such detection. Table 1 shows examples of emergency control in Japan. (The information of the Japan's example is available from http://www. pps.go.jp /english/jobs/index.htmel).

Table 1 Examples of emergency controls in Japan.

Year	Emergency control
1954-65	Potato tuber moth
1965-69	Sweet potato weevil (Eradicated in Kagoshima)
1967-69	Citrus burrowing nematode (Eradicated in Hachijo Island, Tokyo)
1991-98	Sweet potato weevil (Eradicated in Nishino-omote city (Tanegashima Island), Kagoshima)
1995	Sweet potato weevil (Eradicated in Satsuma Peninsula, Kagoshima)
1995-99	Bacterial shoot blight (Eradicated in a part of Hokkaido)
1996-98	Sweet potato weevil (Eradicated in Muroto City, Kochi)
1998-2000	Sweet potato weevil (Eradicated in Yakushima, Kagoshima)
1998-2004	West Indian sweet potato weevil (Eradicated in Yakushima, Kagoshima)

COOPERATION WITH RELEVANT ORGANISATIONS ON IAS

Regional Plant Protection Organizations (RPPOs)

Regional plant protection organizations (RPPOs) may be drawn into the international framework dealing with IAS that are plant pests. Under the IPPC (i.e. Article IX), RPPOs function as regional coordinating bodies and participate in various activities to achieve the objectives of the IPPC. There are nine RPPOs, including the Asia and Pacific Plant Protection Commission.

Some RPPOs are active in relation to IAS, such as the European and Mediterranean Plant Protection Organization (EPPO) which has set up a working group on invasive species, and is compiling a common EPPO list of invasive alien plants.

The North American Plant Protection Organization (NAPPO) is conducting a pathway analysis for weed seeds in grain and identifying risk management options to prevent the spread of weed seeds. This work was previously limited to the national level and has now been expanded to cover North America. NAPPO has also developed a bilingual phytosanitary alert system on its home page in order to provide pest alerts and news of emerging plant pests of significance. This system facilitates awareness, and aids countries in the early detection, prevention and management of plant pests, which include IAS. (The information of NAPPO is available from *http://www.pestalert.org/index.cfm?NAPPO LanguagePref='English'; http://www.nappo.org/currentact.htm*)

IAS often threaten other species, habitats and ecosystems at a regional rather than national level. In particular, where developing countries are concerned, there will be many potential benefits to regional approaches, because, at a regional level, there is tremendous potential for the sharing of common resources, information, and approaches for the prevention and management of IAS.

Cooperation with the Convention on Biological Diversity (CBD)

In 2001, the governing body for the IPPC clarified the role of the IPPC regarding its liaison to the CBD, including the relationship between IAS and plant pests and the scope of the IPPC regarding IAS. It was noted that the IPPC creates obligations for countries, and has established standards and procedures that are designed to prevent the introduction of plant pests, which include IAS. It was done in an effort to clarify that a framework already exists under the IPPC which can be used by countries helps to combat IAS that affect plants.

In 2003, the two Secretariats of the IPPC and CBD signed a Memorandum of Cooperation and in 2004 a joint work programme was developed which is updated regularly. This work programme may include the possible establishment of a liaison group on IAS, input into the development of relevant standards, organisation of a PRA workshop and participation in each other's meetings (e.g. Glossary of terms, gaps and inconsistencies in the international framework).

CONCLUSIONS

To summarise, the IPPC provides countries with rights and obligations that could contribute to the prevention and management of IAS. Although the implementation of the IPPC has historically focused on the protection of agriculture, the scope of the IPPC is not limited to this and efforts have been made to clarify that the IPPC's objectives include the protection of uncultivated/unmanaged plants, wild flora, habitats and ecosystems with respect to IAS that are plant pest.

The IPPC has been in place for more than 50 years. Currently infrastructures have been set up by NPPOs, such as import regulatory systems, pest risk analysis, surveillance, eradication programmes and diagnostic laboratories. These all help to prevent the introduction and spread of plant pests. As resources are limited, every effort should be made to increase the capacity of existing infrastructures rather than creating duplicate systems.

In order to prevent and manage IAS more effectively, organisations that have information on IAS, such as scientific organisations (e.g. universities) and governmental organisations (e.g. environmental and plant protection agencies), need to work together at a national level. Additionally, neighbouring countries need to cooperate at the regional and international level and, finally, international organisations such as the IPPC and CBD should continue to develop joint work programmes to promote synergy and avoid unnecessary overlaps.

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Part 2

Risk Assessment

Effectiveness of the weed risk assessment system for the Bonin Islands

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Abstract Ecosystems of the Bonin (Ogasawara) Islands have been greatly affected by invasive alien species. Moreover, it is likely that additional pest plants will be introduced and become established. We verified the performance of the Hawaiian weed risk assessment (WRA) system, which is a modification of the Australian and New Zealand system, for predicting potential pest plants in the Bonin Islands. We applied the WRA to about 130 introduced plant species in the Bonin Islands. In order to validate the WRA system, the outcomes of the WRA were compared with the opinions of three expert botanists for each of the species. The system successfully identified most pest plants and many non-pest plants. It is, therefore, likely to be useful for assessing the invasiveness of each plant species prior to introduction to the Islands, and thereby provide a rationale to impose a regional quarantine. This system also has a significant potential use for prioritising control and/or eradication programmes of invasive plants that are already present in the Islands.

Keywords: Bonin; Ogasawara Islands; invasive plants; weed risk assessment; plant introduction

INTRODUCTION

Oceanic islands have been recognised as being highly vulnerable to invasions by alien animal and plant species (Vitousek 1988). Introduced plants and animals are more likely to establish and spread in insular ecosystems than in most continental ones due to vacant niches and the absence of defensive traits in the island biota (Cronk and Fuller 2001). In many oceanic islands that have been damaged by invasive alien species, such as the Hawaiian Islands, Galapagos Islands and Juan Fernández Islands, active human management has been required (e.g. Smith 1985, Mauchamp 1997, Greimler *et al.* 2002), and on some islands the problem of alien species has been tackled through practical strategies, including prevention, control and eradication (e.g. Thomas 2001, Tye *et al.* 2002, Soria *et al.* 2002).

The Bonin (Ogasawara) Islands are located in the western Pacific Ocean about 1000km south of the Japanese mainland (Fig. 1). They consist of about 30 small islands, mostly scattered within a 500km range from north to south. The climate of the islands is subtropical with an annual mean temperature of 23.0°C and annual precipitation of nearly 1300 mm. Many of the islands emerged at least one million years ago, and have never been connected to any continent or archipelago (Kaizuka 1977).

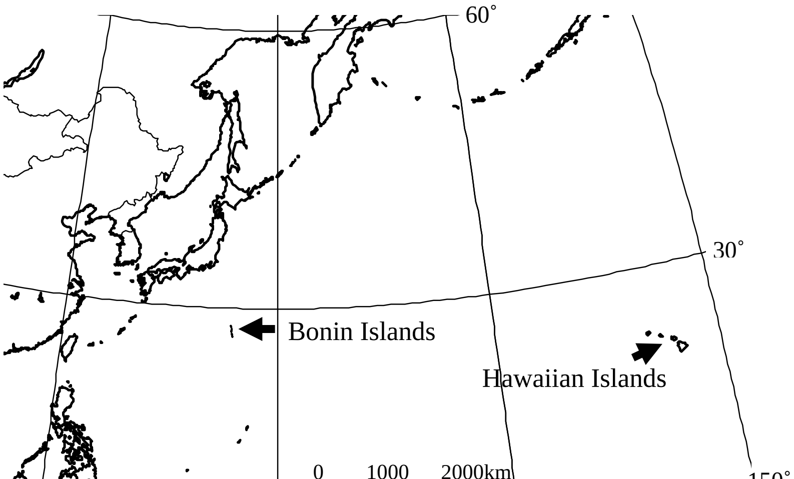

Figure 1 Map showing the location of the Bonin Islands and Hawaiian Islands.

Pages 65-72. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

Consequently, the flora is very disharmonic and a high proportion of endemism has been recognised. The flora of the Bonin Islands comprises approximately 420 native vascular plant taxa, of which about 150 are endemic (Kobayashi and Ono 1987). This includes some species of uncertain origin, such as pantropical weeds, which may have been introduced by early human visitors to the islands.

The islands were first inhabited by Western and Polynesian people in 1830. After the islands became Japanese territory in 1876, many Japanese immigrants settled on the islands, generating impacts associated with human settlement of previously uninhabited islands: introduction of alien species, deforestation and destruction of native biota. The islands came under U.S. governmental administration at the end of World War II and were returned to Japan in 1968. During this period, the islanders were forced to relocate to the mainland of Japan, and as a result, the islands almost returned to an uninhabited condition (Shimizu 2003). Thereafter, many introduced weeds spread in the disturbed areas and invaded the natural vegetation. They included: *Pinus luchuensis*, *Casuarina equisetifolia*, *Leucaena leucocephara*, *Acacia confusa*, *Derris elliptica*, *Bischofia javanica*, *Psidium cattleianum*, *Lantana camara*, *Stachytarpheta jamaicensis* and *Bidens pilosa* (Ono 1998, Shimizu 2003).

Ecosystems of the Bonin Islands have been greatly damaged by invasive alien plants. Furthermore, additional species have been introduced and some of them are likely to become serious pests in the future, since there is no official quarantine system to limit the introduction of non-native species to the islands. Most invasive alien plant species that pose a threat to native ecosystems have been deliberately introduced as useful plants, for their ornamental, timber, agricultural, windbreaker and repellent value (Toyoda 1981). Therefore, future ecological harm could be minimised if the likely invasiveness of plant species could be predicted prior to introduction.

Numerous studies have been published in the past decade on the characteristics of invading alien species, and these have been useful for building rigorous quantitative methods for predicting the invasiveness of alien species (Kolar and Lodge 2001). The weed risk assessment (WRA) system is an empirical tool for identifying likely pest plants (Groves *et al.* 2001). The WRA system was adapted and developed for Australia and New Zealand (Pheloung *et al.* 1999). This system uses 49 questions covering factors affecting the likelihood that the proposed introduced species will become a pest. It generates a score that quantitatively classifies each species assessed as ‘pest’, ‘non-pest’ or ‘evaluate’ (requiring further study). Daehler and Carino (2000) found that a modified version (H-WRA) of the Australian and New Zealand WRA system showed excellent potential as a tool for identifying pest plants in Hawaii and other Pacific Islands. Furthermore, Daehler *et al.* (2004) developed an additional second screening system for use with species with ambiguous outcomes from the H-WRA system. They tested the accuracy of the screening systems, and showed that the H-WRA system successfully identified most pest plants and that the second screening reduced the rate of indecision (Daehler *et al.* 2004). Since these systems were developed for use in tropical and subtropical oceanic islands, it is likely that they are applicable to the Bonin Islands.

In order to evaluate the performance of the Hawaiian WRA system for its application to the Bonin Islands, we applied the WRA to plant species that had been intentionally introduced to the Bonin Islands. To assure the validity of this system, we compared its decisions to the opinions of experts with substantial field experience in the Bonin Islands following Daehler *et al.* (2004).

METHODS

The WRA screening system and information sources

The 49 questions of the original WRA system (Pheloung *et al.* 1999) cover a range of species attributes: biogeography (e.g. distribution, climate preferences, history of cultivation, weediness elsewhere), undesirable characteristics (e.g. toxic, growth habit) and ecological traits (e.g. reproductive and dispersal mode, life history). The H-WRA system (Appendix 1) was developed by modifying 4 of the 49 questions from the Australian and New Zealand system to better suit the environmental factors of Hawaii and other Pacific Islands (Daehler *et al.* 2004); for example, “Australian climates” was replaced with “tropical or subtropical climates”. It is not necessary to answer all questions. Based answers to the questions, the WRA system assigns one of three possible outcomes based on the total score for each taxon: a low score (<1) identifies species that are not likely to become pests (recommendation to “accept”), a high score (>6) identifies species as pests (recommendation to “reject”), and intermediate scores (1-6) identify a requirement for further study (Pheloung *et al.* 1999, Daehler and Carino 2000, Daehler *et al.* 2004).

To reduce the number of ambiguous outcomes, a second screening was performed for the species with intermediate scores (1-6) from the H-WRA screening. This second screening consisted of two independent decision trees developed using a subset of questions from the H-WRA (Appendix 1): one for herbs or small shrubs and another for trees or tree-like shrubs (Daehler *et al.* 2004). Climbers (vines and lianas) were assessed using both decision trees, and a final decision was made by choosing the more undesirable of the two results.

We applied the H-WRA system and the second screening system without any modification following Daehler *et al.* (2004). Prior to the start of the screening processes, we computerised the screening system using FileMaker Pro (FileMaker, Inc., USA).

The information necessary to answer each question was obtained through various means such as the primary literature, books and online resources. For information on many species, we consulted the following online resources: online literature abstracting system AGRICOLA database (*http://agricola.nal.usda.gov/*), a plant database on CD-ROM Horticopia (Horticopia, Inc., USA), a weed database on the Internet, such as the Hawaiian Ecosystems at Risk Project (HEAR) (*http://www.hear.org/index.html*), Florida Exotic Pest Council (FLEPPC) (*http://www.fleppc.org/*), the Invasive Species Specialist Group's (ISSG) Global Invasive Species Database (*http://www.issg.org/database*) and other Internet-based resources accessible through Google (*www.google.com*). All sources of information were documented as completely as possible and recorded in the database.

Screened Species and evaluation of WRA decisions

We chose 160 plant species, which had been intentionally introduced to the Bonin Islands, including various plant families and plant forms such as trees, vines and herbs. We screened these species using the H-WRA system, based on information from the literature, the Internet, and personal communications. For species with intermediate scores (1-6), second screenings were carried out. Information concerning plant behaviour on the Bonin Islands was not used for answering the questions, to avoid tautology in evaluating the WRA system for the Bonin Islands. To confirm the answers for each question, the screenings were carried out by at least two people.

Following Daehler *et al.* (2004), independent rankings were made by several experts who have made considerable field observations and have accumulated knowledge of alien plant problems in the Bonin Islands. They rated each species as a "major pest", "minor pest", or "non-pest" based on its impact to the native ecosystem of the Bonin Islands, applying only on their personal field experience. The criteria used for rating impact are as follows: "major pest": species has substantial effect on native ecosystems, or the plant dominates in the wild flora; "minor pest": species may have some minor effects on native ecosystems, or the plant naturalises in the wild flora but does not dominate; and "non-pest": species does not appear to have negative impacts on native ecosystems, or the plant does not naturalise in the wild. Experts were not asked to rate species for which they had no personal experience. In compiling the results, we classified the ranking according to the majority of the expert decisions. We excluded any species that had not been rated by at least two experts. Daehler *et al.* (2004) did not use a simple majority vote because there was no uniformity of opinions among experts who had experience on the different islands and in the different ecosystems. In our study, however, all the experts are inhabitants of Chichijima Island in the Bonin Islands, so uniformity of opinion could be expected.

RESULTS

We were able to acquire survey results from three experts. Of 160 species, we excluded 30 for which there was one or no response from the experts. The results we obtained for the remaining 130 species for the H-WRA, the second screening, and the expert surveys are shown in Appendix 2. The species information, including detailed WRA results, will be released at *http://wwwmakino.shizen.metro-u.ac.jp/bonin.htm* in the future.

Among the 130 remaining species in the analyses, H-WRA without second screening classified 22 species (17%) as "accept", 34 species (26%) as "evaluate further" and 74 species (57%) as "reject", whereas H-WRA with second screening resorted 34

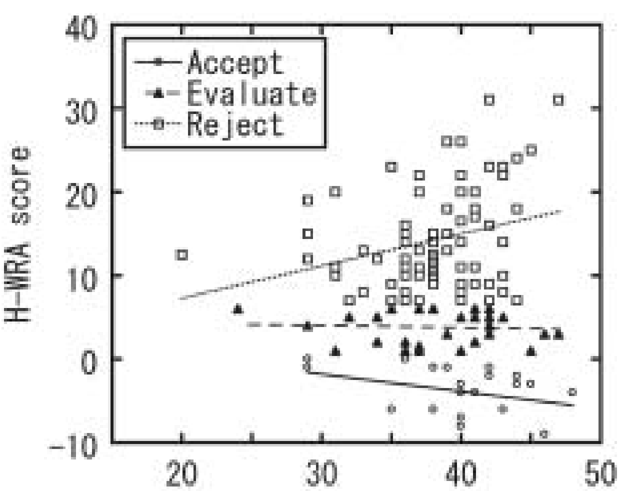

Figure 2 Relationship between the number of questions answered in the weed risk assessment (WRA) and the H-WRA score of each species identified "reject" (open square), "evaluate further" (solid triangle) and "accept" (open circle).

Table 1 Effectiveness of two screening methods for identifying pests of native ecosystems: weed risk assessment (H-WRA) system without the second screening and with second screening.

Screening method	outcome	Expert survey results		
		Major pest	Minor pest	Non-pest
H-WRA	Accept	0 (0%)	3 (7%)	19 (43%)
	Evaluate	5 (12%)	11 (25%)	18 (41%)
	Reject	36 (88%)	30 (68%)	8 (18%)
H-WRA with 2nd screening	Accept	1 (2%)	5 (11%)	28 (64%)
	Evaluate	2 (5%)	4 (9%)	7 (16%)
	Reject	38 (93%)	35 (80%)	10 (23%)

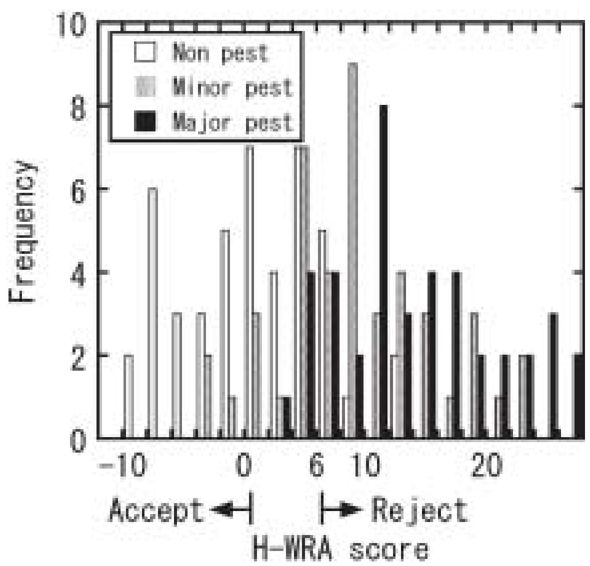

Figure 3 Frequency distributions of the H-WRA scores for major, minor and non-pest plants in the Bonin Islands, as rated by three experts.

species (26%) as "accept", 13 species (10%) as "evaluate further" and 83 (64%) as "reject". There was no statistical significant correlation (r=0.069, p=0.415) as a whole between the number of questions answered and the WRA score (Fig. 2). However, we found a weak positive correlation between the number of questions answered and the score for the species classified as "reject" only (r=0.251, p=0.02) by H-WRA, whereas no correlation was found between the number of questions answered and the scores of the species clarified as "accept" (r=-0.236, p=0.26) or "evaluate further" (r=0.069, p=0.68) by H-WRA (Fig. 2).

There was a high degree of correspondence between the H-WRA results and the expert survey results (Tab. 1, Fig. 3). Of the "major pest" species identified by the experts, the H-WRA system identified 36 species (88%) as "reject", five (12%) as "evaluate further" and none as "accept". The second screening correctly assigned two of the five "major pest" species in the "evaluate further" category to "reject", but one "major pest" species was incorrectly designated "accept". The majority of "minor pest" species, as rated by experts, were also assigned to "reject" by the screening methods: 30 "minor pest" species (68%) according H-WRA alone and 35 (80%) by H-WRA and the second screening. However, three "minor pest" species (7%) were assigned to "accept" by the H-WRA and five by H-WRA and the second screening were assigned to "accept". The H-WRA without second screening identified 19 species (43%) of the "non-pest" species as "accept", and the number increased to 28 (64%) after the second screening, but "non-pest" species were incorrectly assigned to "reject" in 8 cases (18%) without the second screening and in 10 cases (23%) with the second screening.

DISCUSSION

The H-WRA system successfully identified most pest plants in the Bonin Islands, and a second screening using a decision tree reduced the number of ambiguous decisions. Although 34 species (26%) of all species had H-WRA scores between 1 and 6 ("evaluate further"), a second screening succeeded in producing a decision, leaving only 13 species (10%) in the "evaluate further" category. These results strongly support the findings of Daehler *et al.* (2004). Use of both the H-WRA and the second screening system to assess proposed plant introductions to the Bonin Islands would reduce future pest plant problems while allowing the introduction of "non-pest" plant species.

We compared WRA outcomes with expert survey results in order to evaluate the effectiveness of the system, though it is difficult to objectively distinguish between major, minor or non-pest species, because these categories are continuous. Moreover, the actual performance of each introduced plant on the Bonin Islands may not exhibit its intrinsic invasiveness; for example, recently introduced species may not have become invasive yet. It is also possible that through effective management, plant species will not become naturalised. A portion of the errors associated with this subjective evaluation method, such as a "non-pest" species being assigned to "reject", may be due to such factors. However, it is better to make the mistake of denying introduction to a "non-pest" than to allow introduction of a pest, because a decision to allow species introduction is usually irreversible (Daehler *et al.* 2004).

The processes of searching for relevant information necessary for the assessment might often be time-consuming, but the greater availability of literature is likely to improve efficiency. In our study, it took more than five hours to complete the questions for each species. An extensive search for information is necessary to reduce incorrect answers which would lead to inaccurate decisions. Although not all of the 49 questions need to be answered, Pheloung *et al.* (1999) and Daehler and Carino (2000) suggest that answering a greater number of questions reduces the number of ambiguous decisions. Online resources for finding information are highly efficient and provide up-to-date and useful information. The information required for the assessment of invasive plants was easier to obtain because of online resources devoted to these species. It is likely that this contributed to the positive correlation between the number of questions answered and the score for only the species classified as "reject" (Fig. 2). Therefore, sharing of WRA results and sharing information sources through the Internet, such as through Daehler's website (*http://www.botany.hawaii.edu/faculty/daehler/WRA/)* could prove very practical for future assessments.

The economic impact of pest plants is a major concern for agricultural areas, roadsides and other amenity spaces in the Bonin Islands. The WRA system is also applicable for such managed ecosystems (Pheloung *et al.* 1999, Daehler *et al.* 2004), though we focused only on natural ecosystems in our study. In order to re-evaluate this system for managed ecosystems, comprehensive surveys carried out by agriculturalist and horticulturists would be necessary.

Although we used H-WRA and a decision tree developed by Daehler *et al.* (2004) without any modification, some of the questions or criteria for answering could be modified to be more suitable for the Bonin Islands. For example, one major pest species, *Alpinia zerumbet* (Zingiberaceae), was assigned to "evaluate further" by the H-WRA, but was incorrectly designated "accept" by the second screening system, because it was not reported as a weed of cultivated lands. Furthermore, in the case of question 4.10 "tolerates a wide range of soil conditions", water availability should be included in the soil conditions because conspicuous drought conditions are likely to influence the survival of plants in the Bonin Islands (Shimizu, 2003). In order to improve the rates of correct classification for pests or non-pests in the islands besides modification of the questions or criteria, the relative weighting of factors contributing to the total WRA assessment score should be re-examined, both statistically and empirically.

Aside from risk assessment of species that have not yet been introduced in an area, this system has significant potential for prioritising efforts to manage invasive alien species that are already present in an area (Thomas 2001). The accumulated information of assessed species, especially biological characteristics, could be useful for adequate management of introduced plant species. Risk assessment techniques for species that are already present should be modified, taking into consideration the cost effectiveness and/or feasibility of either control or eradication (Thomas 2001).

Preventing the introduction of new invasive alien species is the first step in addressing the problem; it is much easier and cost effective to achieve adequate prevention than to control and/or eradicate species that are already present. Isolated islands are at an advantage for prevention due to their geographic isolation and their small areas. Limited transportation between the Bonin Islands and other areas could also be an advantage given that ships are the only links to the Islands, aside from aerial transport by the Japanese Self-Defence Forces. Nevertheless, it is not easy to administer a quarantine system for restricting the introduction of alien species into the Islands due to economic, political or public issues. For this to be successful, education and/or persuasion of the islanders, administrators and politicians will be required.

ACKNOWLEDGEMENTS

We thank Takaya Yasui, Fuyuo Nobushima, Miho Wada and Katsunori Wada for volunteering for our survey. Tomoko Nishida, Shunji Kurokawa and Paul C. Pheloung provided helpful comments. The present study was partly supported by Grants-in-Aid from the Japanese Ministry of Education, Science and Culture (No. 14340269 to H. Kato).

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Appendix 1-1 H-WRA system following Pheloung *et al.* (1999) with minor modifications for use in Pacific Islands (Daehler and Carino 2000, Daehler *et al.* 2004).

A. Biogeography/historical			Score
1 Domestication/cultivation	1.01	Is the species highly domesticated? If answer is 'no' go to Question 2.01	y=-3, n=0
	1.02	Has the species become naturalised where grown?	y=1, n=-1
	1.03	Does the species have weedy races?	y=1, n=-1
2 Climate and Distribution	2.01	Species suited to tropical or subtropical climate(s) - If island is primarily wet habitat, then substitute "wet tropical" for "tropical or subtropical"	low=0; intermediate=1; high=2
	2.02	Quality of climate match data	low=0; intermediate=1 high=2
	2.03	Broad climate suitability (environmental versatility)	y=1, n=0
	2.04	Native or naturalised in regions with tropical or subtropical climates	y=1, n=0
	2.05	Does the species have a history of repeated introductions outside its natural range?	y=-2, ?=-1, n=0
3 Weed Elsewhere (depends on 2.01and 2.02)	3.01	Naturalised beyond native range	y=1*multiplier (see App. 1-2), n=question 2.05
	3.02	Garden/amenity/disturbance weed	y=1*multiplier (see App. 1-2), n=0
	3.03	Agricultural/forestry/horticultural weed	y=2*multiplier (see App. 1-2), n=0
	3.04	Environmental weed	y = 2*multiplier (see App. 1-2), n=0
	3.05	Congeneric weed	y = 1*multiplier (see App.1-2), n=0
B Biology/Ecology			
4 Undesirable traits	4.01	Produces spines, thorns or burrs	y=1, n=0
	4.02	Allelopathic	y=1, n=0
	4.03	Parasitic	y=1, n=0
	4.04	Unpalatable to grazing animals	y=1, n=-1
	4.05	Toxic to animals	y=1, n=0
	4.06	Host for recognised pests and pathogens	y=1, n=0
	4.07	Causes allergies or is otherwise toxic to humans	y=1, n=0
	4.08	Creates a fire hazard in natural ecosystems	y=1, n=0
	4.09	Is a shade tolerant plant at some stage of its life cycle	y=1, n=0
	4.10	Tolerates a wide range of soil conditions (or limestone conditions if not a volcanic island)	y=1, n=0
	4.11	Climbing or smothering growth habit	y=1, n=0
	4.12	Forms dense thickets	y=1, n=0

(Appendix 1-1 continued)

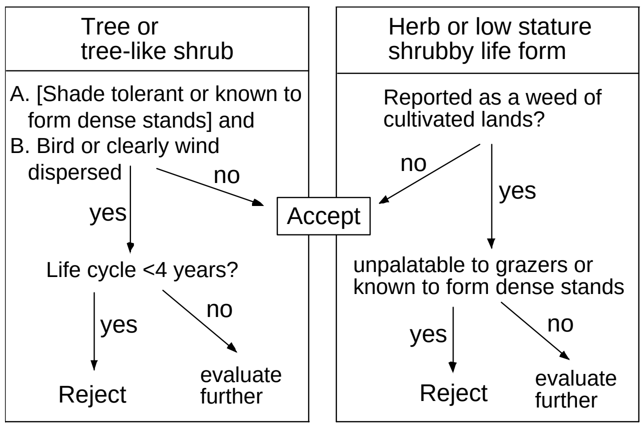
5 Plant type	5.01	Aquatic	y=5, n=0
	5.02	Grass	y=1, n=0
	5.03	Nitrogen fixing woody plant	y=1, n=0
	5.04	Geophyte (herbaceous with underground storage organs -- bulbs, corms, or tubers)	y=1, n=0
6 Reproduction	6.01	Evidence of substantial reproductive failure in native habitat	y=1, n=0
	6.02	Produces viable seed.	y=1, n=-1
	6.03	Hybridises naturally	y=1, n=-1
	6.04	Self-compatible or apomictic	y=1, n=-1
	6.05	Requires specialist pollinators	y=-1, n=0
	6.06	Reproduction by vegetative fragmentation	y=1, n=-1
	6.07	Minimum generative time (years)	1 year = 1, 2 or 3 years = 0, 4+ years = -1
7 Dispersal mechanisms	7.01	Propagules likely to be dispersed unintentionally (plants growing in heavily trafficked areas)	y=1, n=-1
	7.02	Propagules dispersed intentionally by people	y=1, n=-1
	7.03	Propagules likely to disperse as a produce contaminant	y=1, n=-1
	7.04	Propagules adapted to wind dispersal	y=1, n=-1
	7.05	Propagules water dispersed	y=1, n=-1
	7.06	Propagules bird dispersed	y=1, n=-1
	7.07	Propagules dispersed by other animals (externally)	y=1, n=-1
	7.08	Propagules survive passage through the gut	y=1, n=-1
8 Persistence attributes	8.01	Prolific seed production (>1000/m^2)	y=1, n=-1
	8.02	Evidence that a persistent propagule bank is formed (>1 yr)	y=1, n=-1
	8.03	Well controlled by herbicides	y=-1, n=1
	8.04	Tolerates, or benefits from, mutilation, cultivation, or fire	y=1, n=-1
	8.05	Effective natural enemies present locally (e.g. introduced biocontrol agents)	y=-1, n=1

Total score: =sum
Outcome:

Appendix 1-2 Answers to questions 2.01, 2.02 and 2.05 used to determine appropriate scores for questions 3.01-3.05 by looking up a "multiplier" (ranging from 0.5 to 2) in Appendix 1-1 (Daehler *et al.* 2004; *http://www.botany.hawaii.edu/faculty/daehler/WRA/description.htm*)

		question 2.01		
		0	1	2
question 2.02	0	2	2	2
	1	1	1.5	2
	2	0.5	1	2

Appendix 1-3 Decision tree for second screening of pest species that had intermediate scores (between 1 and 6) (Daehler *et al.* 2004).



Appendix 2 Results of weed risk assessment and expert survey for 130 non-native plant species in the Bonin Islands.

Scientific name	Family	H-WRA result score	H-WRA result outcome	2nd screening outcome	Expert survey result
Cycas revolute	CYCADACEAE	5	Evaluate	Evaluate	minor pest
Pinus luchuensis	PINACEAE	9	Reject		major pest
Cunninghamia lanceolata	TAXODIACEAE	1	Evaluate	Evaluate	non-pest
Podocarpus macrophyllus	PODOCARPACEAE	-3	Accept		minor pest
Araucaria cunninghamii	ARAUCARIACEAE	-4	Accept		non-pest
Casuarina equisetifolia	CASUARINACEAE	23	Reject		major pest
Ficus benghalensis	MORACEAE	6	Evaluate	Evaluate	non-pest
Ficus caulocarpa	MORACEAE	6	Evaluate	Evaluate	non-pest
Ficus elastica	MORACEAE	9	Reject		minor pest
Ficus microcarpa	MORACEAE	13	Reject		major pest
Ficus superba var. *japonica*	MORACEAE	7	Reject		non-pest
Morus australis	MORACEAE	20	Reject		major pest
Antigonon leptopus	POLYGONACEAE	11	Reject		minor pest
Rivina humilis	PHYTOLACCACEAE	11	Reject		major pest
Bougainvillea spectabilis	NYCTAGINACEAE	2	Evaluate	Evaluate	non-pest
Mirabilis jalapa	NYCTAGINACEAE	14	Reject		minor pest
Cinnamomum camphora	LAURACEAE	6	Evaluate	Evaluate	minor pest

(Appendix 2 continued)

Scientific name	Family	H-WRA result		2nd Screening	Expert survey
		score	outcome		
Garcinia mangostana	GUTTIFERAE	3	Evaluate	Evaluate	major pest
Kalanchoe pinnata	CRASSULACEAE	18	Reject		major pest
Hydrangea macrophylla f. *normalis*	SAXIFRAGACEAE	5	Evaluate	Reject	major pest
Acacia confusa	FABACEAE	14	Reject		major pest
Adenanthera pavonina	LEGUMINOSAE	13	Reject		minor pest
Amorpha fruticosa	LEGUMINOSAE	20	Reject		minor pest
Derris elliptica	LEGUMINOSAE	15	Reject		major pest
Erythrina crista-galli	LEGUMINOSAE	1	Evaluate	Accept	non-pest
Erythrina variegata	FABACEAE	7	Reject		minor pest
Erythrina x bidwillii	LEGUMINOSAE	-6	Accept		non-pest
Leucaena leucocephala	LEGUMINOSAE	26	Reject		major pest
Medicago polymorpha	LEGUMINOSAE	16.5	Reject		minor pest
Mimosa pudica	LEGUMINOSAE	22	Reject		major pest
Pongamia pinnata	LEGUMINOSAE	9	Reject		minor pest
Senna siamea	LEGUMINOSAE	7	Reject		non-pest
Acalypha godseffiana	EUPHORBIACEAE	5	Evaluate	Reject	minor pest
Bischofia javanica	EUPHORBIACEAE	17	Reject		major pest
Breynia nivosa f.*roseopicta*	EUPHORBIACEAE	-3	Accept		minor pest
Ricinus communis	EUPHORBIACEAE	9	Reject		minor pest
Citrus natsudaidai	RUTACEAE	-7	Accept		non-pest
Citrus sinensis	RUTACEAE	-2	Accept		non-pest
Murraya paniculata	RUTACEAE	5	Evaluate	Accept	non-pest
Mangifera indica	ANACARDIACEAE	2	Evaluate	Accept	minor pest
Rhus succedanea	ANACARDIACEAE	10.5	Reject		major pest
Cardiospermum halicacabum	SAPINDACEAE	16	Reject		minor pest
Koelreuteria henryi	SAPINDACEAE	12	Reject		major pest
Buxus liukiuensis	BUXACEAE	2	Evaluate	Evaluate	minor pest
Hibiscus rosa-sinensis	MALVACEAE	-4	Accept		non-pest
Hibiscus schizopetalus	MALVACEAE	-8	Accept		non-pest
Carica papaya	CARICACEAE	3	Evaluate	Reject	minor pest
Lagerstroemia indica	LYTHRACEAE	8	Reject		non-pest
Lagerstroemia subcostata	LYTHRACEAE	7	Reject		minor pest
Psidium guajava	MYRTACEAE	24	Reject		minor pest
Psidium cattleianum	MYRTACEAE	25	Reject		major pest
Syzygium jambos	MYRTACEAE	12	Reject		major pest
Punica granatum	PUNICACEAE	3	Evaluate	Accept	non-pest
Barringtonia asiatica	LECYTHIDACEAE	-7	Accept		non-pest
Polyscias fruticosa	ARALIACEAE	-1	Accept		non-pest
Diospyros ferrea	EBENACEAE	6	Evaluate	Evaluate	major pest
Diospyros kaki	EBENACEAE	1	Evaluate	Reject	non-pest
Jasminum hemsleyi	OLEACEAE	6	Evaluate	Accept	minor pest
Allamanda cathartica	APOCYNACEAE	9	Reject		non-pest
Catharanthus roseus	APOCYNACEAE	10	Reject		minor pest
Nerium indicum	APOCYNACEAE	4	Evaluate	Accept	non-pest
Coffea arabica	RUBIACEAE	1	Evaluate	Reject	non-pest
Ixora chinensis	RUBIACEAE	1	Evaluate	Evaluate	non-pest
Ipomoea alba	CONVOLVULACEAE	10	Reject		minor pest
Clerodendrum wallichii	VERBENACEAE	5	Evaluate	Reject	minor pest
Clerodendrum japonicum	VERBENACEAE	10	Reject		minor pest
Lantana camara var. *aculeata*	VERBENACEAE	31	Reject		major pest
Stachytarpheta jamaicensis	VERBENACEAE	20	Reject		minor pest
Stachytarpheta urticifolia	VERBENACEAE	22	Reject		major pest
Capsicum frutescens	SOLANACEAE	5	Evaluate	Accept	non-pest
Cestrum nocturnum	SOLANACEAE	14	Reject		non-pest
Nicotiana tabacum	SOLANACEAE	8	Reject		minor pest
Russelia equisetiformis	SCROPHULARIACEAE	1	Evaluate	Accept	non-pest

Scientific name	Family	H-WRA result		2nd Screening	Expert survey
		score	outcome		
Thunbergia alata	ACANTHACEAE	16	Reject		major pest
Thunbergia laurifolia	ACANTHACEAE	22	Reject		minor pest
Wedelia biflora	COMPOSITAE	11	Reject		minor pest
Wedelia trilobata	COMPOSITAE	15	Reject		major pest
Lilium longiflorum	LILIACEAE	10	Reject		minor pest
Agave americana	AGAVACEAE	11	Reject		major pest
Agave sisalana	AGAVACEAE	6	Evaluate	Reject	major pest
Dracaena fragrans	AGAVACEAE	-7	Accept		non-pest
Sansevieria nilotica	AGAVACEAE	15	Reject		major pest
Sansevieria stuckyi	AGAVACEAE	4	Evaluate	Accept	non-pest
Yucca smalliana	AGAVACEAE	7	Reject		non-pest
Hippeastrum x hybridum	AMARYLLIDACEAE	-1	Accept		minor pest
Hymenocallis littoralis	AMARYLLIDACEAE	8	Reject		major pest
Dioscorea bulbifera	DIOSCOREACEAE	11	Reject		major pest
Dioscorea japonica	DIOSCOREACEAE	1.5	Evaluate	Reject	minor pest
Rhoeo spathacea	COMMELINACEAE	15	Reject		minor pest
Arundo donax	GRAMINEAE	19	Reject		minor pest
Bambusa multiplex	GRAMINEAE	9	Reject		minor pest
Bambusa vulgaris	GRAMINEAE	6	Evaluate	Evaluate	minor pest
Chloris gayana	GRAMINEAE	18	Reject		major pest
Cymbopogon citratus	GRAMINEAE	0	Accept		non-pest
Dendrocalamus latiflorus	GRAMINEAE	5	Evaluate	Evaluate	non-pest
Eragrostis curvula	GRAMINEAE	23	Reject		minor pest
Paspalum dilatatum	GRAMINEAE	31	Reject		major pest
Paspalum notatum	GRAMINEAE	20	Reject		major pest
Pennisetum purpureum	GRAMINEAE	18	Reject		major pest
Phyllostachys aurea	GRAMINEAE	12	Reject		major pest
Pleioblastus simonii	GRAMINEAE	7	Reject		major pest
Pseudosasa japonica	GRAMINEAE	12.5	Reject		major pest
Saccharum officinarum	GRAMINEAE	5	Evaluate	Reject	minor pest
Vetiveria zizanioides	GRAMINEAE	-6	Accept		non-pest
Zoysia japonica	GRAMINEAE	8	Reject		non-pest
Archontophoenix alexandrae	PALMAE	5	Evaluate	Evaluate	non-pest
Areca catechu	PALMAE	-2	Accept		non-pest
Arenga engleri	PALMAE	8	Reject		major pest
Caryota urens	PALMAE	7	Reject		major pest
Chrysalidocarpus lutescens	ARECACEAE	7	Reject		minor pest
Cocos nucifera	ARECACEAE	-6	Accept		non-pest
Howea belmoreana	PALMAE	0	Accept		non-pest
Mascarena verschaffeltii	PALMAE	-1	Accept		non-pest
Phoenix roebelenii	ARECACEAE	3	Evaluate	Accept	non-pest
Rhapis excelsa	PALMAE	-3	Accept		non-pest
Veitchia merrillii	PALMAE	-1	Accept		non-pest
Alocasia cucullata	ARACEAE	12	Reject		major pest
Colocasia esculenta var. *aquatilis*	ARACEAE	16	Reject		minor pest
Epipremnum aureum	ARACEAE	11	Reject		minor pest
Monstera deliciosa	ARACEAE	10	Reject		minor pest
Cyperus alternifolius subsp. *flabelliformis*	CYPERACEAE	23	Reject		major pest
Cyperus malaccensis subsp. *monophyllus*	CYPERACEAE	26	Reject		major pest
Schoenoplectus grossus	CYPERACEAE	14	Reject		minor pest
Schoenus brevifolius	CYPERACEAE	12.5	Reject		minor pest
Musa x sapientum	MUSACEAE	-9	Accept		non-pest
Ravenala madagascariensis	MUSACEAE	5	Evaluate	Accept	non-pest
Strelitzia reginae	MUSACEAE	-4	Accept		non-pest
Alpinia zerumbet	ZINGIBERACEAE	5	Evaluate	Accept	major pest
Hedychium coronarium	ZINGIBERACEAE	10	Reject		major pest
Canna indica	CANNACEAE	13	Reject		non-pest

Evaluation of species properties used in weed risk assessment and improvement of systems for invasion risk assessment

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Abstract The weed risk assessment (WRA) system is used in Australia for plant species. Results of applying such weed risk assessment in Bonin (Ogasawara) Islands, Japan were analysed to evaluate the effectiveness of the species properties considered. In addition, a comparison was carried out of various risk assessment systems, such as WRA, weight estimation by linear regression, logistic regression and decision tree analysis, with the aim of improving the systems used for risk assessment.

Among species properties considered in WRA, history of naturalisation (establishing self-perpetuating population in regional vegetation) was the most important property, whereas many biological traits did not contribute effectively. A better result was obtained with a risk assessment system based on four groups of variables: 1) naturalisation elsewhere, 2) special biological traits, such as natural enemies, 3) weediness, 4) habitat specific ecological properties (for forests and for disturbed land).

In the short term, the "biogeography and history" components of traditional WRA may provide the best system, although the threshold value for acceptance and rejection needs to be determined. In the slightly longer term, the regression model with grouped variables proposed in this paper is another candidate, although this should be tested with additional data.

Keywords: invasion risk assessment; Bonin (Ogasawara) Islands; Japan; oceanic island; weed risk analysis; ecological trait

INTRODUCTION

Weed risk assessment (WRA) is an invasion risk assessment system for plant species developed in Australia (Pheloung *et al.* 1999), to evaluate the risk that a species proposed for introduction will become a noxious weed. It has been used in Australia to evaluate invasion risks before introduction and in New Zealand to determine the management of alien plant species already introduced. In WRA, species properties relating to climate, domestication history, naturalisation elsewhere, as well as other ecological traits, are considered. The sum of the values assigned to each property gives a total score which is used to evaluate the likely invasion ability and impacts of the focal species. WRA has been adapted to other regions such as Hawaii (Daehler *et al.* 2004). In the Hawaiian WRA, small decision trees have been added as an additional step to decide on species with an ambivalent outcome; nevertheless the properties considered and their weightings were basically those of the original Australian WRA.

The objectives of this research are to evaluate the effectiveness of the species properties considered in WRA, with the aim of improving the invasion risk assessment system. The WRA results for Bonin (Ogasawara) Islands (Kato *et al.* 2006), Japan were used for analysis. The WRA for the Bonin Islands was successful, but providing the information asked for in the assessment for all species properties is labour intensive and there may be a room to improve weighting for each property.

The Bonin Islands are subtropical oceanic islands with many endemic species. Land use in these islands is different from Australia and grazing in pastures is not common. The shade tolerant tall alien tree *Bischofia javanica* is now the most dominant species in the climax forests, and environmental weeds in natural habitats are a more important issue than weeds in arable lands and pastures. As a result, some properties considered in WRA elsewhere may not be important in the islands. Proper weighting of properties and improvement of formulation will improve evaluation of invasion risk.

Pages 73-83. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

METHODS: DATASET AND ANALYSYS OF SPECIES PROPERTIES

Data set

The WRA results in the Bonin Islands (Kato *et al.* 2006), based on the Hawaiian WRA (Daehler *et al.* 2004), were used for analysis. Seed plant species that have a history of introduction to Bonin Islands were listed for analysis. The data set was a table of species (sample) x properties (variable).

Invasiveness value, representing the impact of alien species, was used as a dependent variable to evaluate the risk assessment models. Since a suitable standardised method to classify the impact of a given alien species on the native ecosystem was not yet established, expert opinion was used. Several plant experts living in the Bonin Islands evaluated the impact of the listed alien species, based on their experience, and classified each species as either a "major pest" (established widely in the wild or having substantial impact on ecosystems), "minor pest" (naturalised but no significant impact), and "non-pest" (not naturalised). The majority vote of these experts was used to assign an invasiveness value to a species. Values assigned in quantitative analysis (correlation coefficient and linear regression) were 0 for non-pest, 1 for minor pest, 2 for major pest. Values assigned in qualitative analyses (logistic regression, decision tree analysis) were 0 for non-pest and 1 for major pest. Minor pest species were not used in qualitative analyses.

Independent variables used to predict invasion risk were bio-geographical, historical and biological properties of species with the list of properties used the same as in the WRA. The value for each property was agreed by botanists based on information available in literature and on the Internet (Kato *et al.* 2006). For many properties, the value was one of three states: "yes", "no" or "unknown". If the shade tolerance is given quantitatively in the information source, the researcher needed to use judgment in giving values as "tolerant" or "intolerant". These processes are like a questionnaire based on personal experience. Results obtained may not be very precise, but it is much quicker than requiring quantitative measurements. To calculate the WRA score, numerical values were assigned for "yes", "no" and "unknown" according to the WRA method (see Appendix 1 in Kato *et al.* 2006, for the list of properties and their weightings).

Species properties were categorised for the two fields of "biogeography and history" and "biology and ecology" (Pheloung *et al.* 1999). "Biogeography and history" was subdivided into "domestication and cultivation", "climate similarity" and "naturalisation elsewhere". "Biology and ecology" was subdivided into "undesirable traits" (such as spines, allelopathy, toxic, etc.), "plant type" (such as aquatic, nitrogen fixer, etc.), "reproduction", "dispersal", and "persistence attributes" (such as huge seed production, propagule bank, herbicide tolerance, etc.). Details for the WRA in the Bonin Islands have been described by Kato *et al.* (2006) in this volume.

Grouping of species properties and correlation to invasiveness

WRA considers many closely related properties. Properties were classified by dichotomous cluster analysis according to responses for species using Minna de GIS software (Koike 2004). In the cluster analysis, properties were subjected to principal component analysis, and then properties were divided into two groups based on the first principal component axis. The divided groups were subdivided iteratively. When the contribution of principal component analysis was less than 10% of total variance, division was stopped.

To evaluate the effectiveness of the species properties considered in the WRA, a simple correlation coefficient with "invasiveness" (expert judge score) was calculated for individual properties, and for the sum of values of each property type (with property types being biogeography, biological traits, etc.).

RISK ASSESSMENT MODELS

Various regression models were evaluated to obtain a model predicting species invasiveness from properties. The expert judge score was used as the dependent variable, and various properties were considered as independent variables. SPSS 13.0J was used for analysis.

Linear regression

All properties considered in WRA were used as independent variables. A stepwise variable selection procedure was applied. In this procedure, the most significant variable (property) was taken as the first step. The best two variables model (including the previously selected, most significant variable) was determined in the next step. Steps were iterated, and the number of variables in the model was increased as long as the addition of the new variable was statistically significant. If there were two similar variables with high mutual correlation, only one of

y	rand1	rand2	rand3	rand4	…	rand200
1	0.921938	0.156568	0.869971	0.192272	…	0.005387
2	0.495474	0.710752	0.508757	0.711095	…	0.366705
3	0.260066	0.036559	0.25835	0.468038	…	0.501751
4	0.323475	0.357433	0.558227	0.560274	…	0.232222
5	0.063304	0.591537	0.694699	0.853888	…	0.198453
6	0.338076	0.452384	0.07734	0.765761	…	0.079108
7	0.259525	0.005843	0.595242	0.080746	…	0.643076
8	0.667955	0.192798	0.680988	0.19825	…	0.765713
9	0.890244	0.295702	0.299972	0.237536	…	0.784451
10	0.860162	0.862589	0.699743	0.385391	…	0.908238

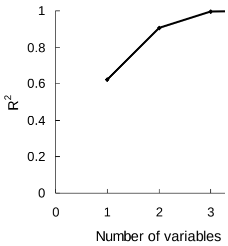

Figure 1 Artificial data of dependent variable *y* (sequential value from 1 to 10), and random number (0.0 to 1.0) for 200 independent variables. The graph represents the results of stepwise regression using a fixed threshold value of significance level at P=0.05. A significant linear regression was obtained. Thus controlling the significance level is essential in the stepwise variable selection procedure.

them was taken.

Although the significance level of P=0.05 is often used as a threshold, if we considered many independent variables simultaneously, even a variable with random values becomes significant. In our example (Fig. 1), we generated a test data set with 10 samples and 200 variables; we gave a sequential value from 1 to 10 to the dependent variable (y), and a random value between 0 to 1 for all independent variables (*rand*1 to *rand*200). Even though no variable should correlate to y, a significant regression model was obtained when using the stepwise linear regression (Fig. 1), if P was assumed to be 0.05. This was because the number of variables considered was too large, and hence at least some exceeded the significance level of $P<0.05$. To avoid this phenomenon, we used a threshold significance level of

$$P = 0.05 \,/\, (n - i + 1) \qquad (1)$$

for the i-th step of the variable selection procedure, where n was the number of whole independent variables considered (Rice 1989). The term, $n - i + 1$, gives the number of variables examined in the i-th step.

Linear regression based on the properties in the category "biogeography and history"

A linear regression based only on the species properties in the category of "biogeography and history" was conducted to evaluate their effectiveness. A step wise variable selection procedure was applied.

Linear regression based on the properties in the category "biology and ecology"

A linear regression based only on species properties in the category "biology and ecology" was conducted to evaluate their effectiveness. A stepwise variable selection procedure was applied.

Logistic regression

A logistic regression with stepwise variable selection was also evaluated. This model gives the probability that the species will become invasive. Two states of "major pest" and "non-pest" were used as dependent variables, and "minor pest" species were not included in this analysis.

Decision tree analysis

Species were divided into two groups depending on a threshold value for a property, and the best combination of property and its threshold value was determined to divide 'major-pest" and "non-pest" species. This process was iterated and a tree-like division of species was obtained. There will be a branch of mainly "major pest" species and one of mainly "non-pest" species. ANSWER TREE 3.0J was used for the analyses.

Linear regression with grouped variables

Data used in WRA is provided using a questionnaire and personal experience will affect this to some extent. When we deal with data containing much noise (error of data), a slight fluctuation of a given variable will cause a large difference in the prediction, if only a few variables were considered. This may be the reason why the WRA considers several properties with much redundancy (Pheloung *et al.* 1999). For example, arable weed species usually have a set of traits, such as a long flowering time, short life cycle, and non-synchronous germination. If we use stepwise variable selection, only one, most significant, trait will be used from many traits with high mutual correlation. However, the selected trait will have error. Adding many mutually correlated variables can reduce random noise as

$$S = \frac{V}{N} \qquad (2)$$

where S is the variance of averaged value of considered variables, V is the variance of a individual variable, N is the number of variables averaged. It is possible to improve the reliability of the prediction model by adding mutually correlated variables.

In this research several groups of properties were selected and the sum of property values for each group was calculated. A linear regression with these summed values was conducted, assuming the expert judge score as a dependent variable.

To define property groups, properties were divided by cluster analysis as mentioned above. The important property in each group was selected, based on the correlation coefficients with the expert judge score. Several variables were added to each group to increase redundancy. These grouped variables were used in linear regression with stepwise variable selection.

METHOD FOR EVALUATION OF ASSESSMENT SYSTEMS

The following regression models, described above, were compared: "traditional" WRA, WRA using only the properties of "biogeography and history", and WRA using the properties of "biology and ecology". The decision tree used for the second screening in the Hawaiian WRA (Daehler *et al.* 2004, see Appendix 1-3 in Kato *et al.* 2006) was not used in this analysis.

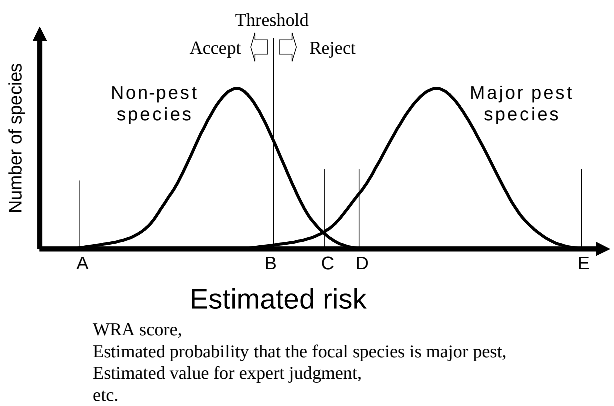

Figure 2 Hypothetical diagram to determine the threshold value that will be used to decide on the acceptance or rejection of a proposed introduction. Many types of scores can be used for the horizontal axis. They are: WRA score in weed risk assessment, estimated probability that the focal species is a major pest in logistic regression, decision tree analysis, and score of expert judge in various linear regression models. **A**: all species will be rejected, **B**: almost all major pest species will be rejected and many non-pest will be accepted (Scenario 2), **C**: maximise the probability that both major pest and non-pest species were judged correctly (Scenario 1), **D**: almost all non-pest species will be accepted and some major pest will also be accepted, and **E**: all species will be accepted.

A score of 6 is usually used as a threshold value in WRA, meaning that a species with a value higher than 6 is not allowed to be introduced (Pheloung *et al.* 1999).

We need to determine such a threshold for various risk assessment systems (Fig. 2). Several non-pest species may score above the threshold and be rejected. Some major-pest species might score below the threshold and be accepted. The probabilities that a major pest species is judged as a pest and that a non-pest species is judged as a non-pest, will vary with the threshold value. The threshold should be set to maximise these probabilities simultaneously. We compared a plot of these probabilities for each risk assessment method (Fig. 5). Minor pest species were not considered in this evaluation.

Two scenarios were considered to set a threshold score. One scenario was to maximise the probability that both major pest and non-pest species were judged correctly (Scenario 1, threshold **C** in Fig. 2). This probability is the product of (correctly judged probability for major pest) × (correctly judged probability for non-pest). In this scenario a model with a high value for this probability is a good risk assessment model. In another scenario, it was assumed that major pests cause unacceptable hazards, thus the probability that the major pest is judged correctly should be as high as 95% (Scenario 2, threshold **B** in Fig. 2). However, this will cause rejection of many non-pest species. A model with a large probability that a non pest is judged correctly is a good risk assessment model under this second scenario.

RESULTS

Structure of species properties

Species properties were classified in six groups (Tab. 1). Group A contains species properties of agricultural and garden weeds. Variables in this group include propagule properties such as likely unintentional dispersal, dispersal as a contaminant, wind dispersal, and short generation time. Group B contains species properties relating to the degree of environmental tolerance. Group C contains species properties of weeds in forests and shrub lands. Group D included records of naturalisation in similar climate. Group E contained various properties. Although this latter group was very divers, contribution of the first principal component was low, and the group was not subdivided. Group F grouped properties of endozoochoric propagule dispersal.

Table 1 Classification of species properties based on the results of weed risk assessment in Bonin (Ogasawara) Islands, Japan (Kato *et al.* 2006). Dichotomous cluster analysis was carried out. Properties were divided into two groups using Division 1 (0 and 1 for Division 1), and each group was subdivided using Division 2 (0 and 1 for Division 2). The second level groups were subdivided using Division 3.

Group	No	Query	Division 1	Division 2	Division 3
A	Q302	Garden/amenity/disturbance weed	0	0	0
	Q303	Agricultural/forestry/horticultural weed	0	0	0
	Q402	Allelopathic	0	0	0
	Q501	Aquatic	0	0	0
	Q502	Grass	0	0	0
	Q504	Geophyte (herbaceous with underground storage organs)	0	0	0
	Q607	Minimum generative time (years)	0	0	0
	Q701	Propagules likely to be dispersed unintentionally	0	0	0
	Q703	Propagules likely to disperse as a produce contaminant	0	0	0
	Q704	Propagules adapted to wind dispersal	0	0	0
	Q707	Propagules dispersed by other animals (externally)	0	0	0
B	Q203	Broad climate suitability (environmental versatility)	0	0	1
	Q410	Tolerates a wide range of soil conditions	0	0	1
	Q606	Reproduction by vegetative fragmentation	0	0	1
C	Q304	Environmental weed	0	1	0
	Q412	Forms dense thickets	0	1	0
	Q801	Prolific seed production (>1000/m2)	0	1	0
	Q804	Tolerates, or benefits from, mutilation, cultivation, or fire	0	1	0
D	Q301	Naturalised beyond native range	0	1	1
	Q305	Congeneric weed	0	1	1
E	Q102	Has the species become naturalised where grown?	1	0	0
	Q205	History of repeated introductions outside its natural range	1	0	0
	Q401	Produces spines, thorns or burrs	1	0	0
	Q406	Host for recognised pests and pathogens	1	0	0
	Q407	Causes allergies or is otherwise toxic to humans	1	0	0
	Q408	Creates a fire hazard in natural ecosystems	1	0	0
	Q411	Climbing or smothering growth habit	1	0	0
	Q503	Nitrogen fixing woody plant	1	0	0
	Q601	Evidence of substantial reproductive failure in native habitat	1	0	0
	Q603	Hybridises naturally	1	0	0
	Q604	Self-compatible or apomictic	1	0	0
	Q101	Is the species highly domesticated?	1	0	0
	Q103	Does the species have weedy races?	1	0	0
	Q201	Species suited to tropical or subtropical climate(s)	1	0	0
	Q202	Quality of climate match data	1	0	0
	Q204	Native or naturalised in regions with tropical or subtropical climates	1	0	0
	Q404	Unpalatable to grazing animals	1	0	0
	Q405	Toxic to animals	1	0	0
	Q409	Is a shade tolerant plant at some stage of its life cycle	1	0	0
	Q602	Produces viable seed.	1	0	0
	Q605	Requires specialist pollinators	1	0	0
	Q702	Propagules dispersed intentionally by people	1	0	0
	Q705	Propagules water dispersed	1	0	0
	Q802	Evidence that a persistent propagule bank is formed (>1 yr)	1	0	0
	Q803	Well controlled by herbicides	1	0	0
	Q805	Effective natural enemies present locally	1	0	0
F	Q706	Propagules bird dispersed	1	1	0
	Q708	Propagules survive passage through the gut	1	1	0

Importance of properties

High correlation to the expert judge score suggests importance of species properties. In our study, naturalisation beyond the native range (Q301) was the most important property in the WRA (Tab. 2). Domestication (Q101), environmental weediness (Q304) and existence of natural enemies (Q805) were the next most important.. High correlation to the WRA score suggests that there were many similar species properties used in the WRA or that the WRA gives high weighting to the property.

The WRA score showed high correlation to the expert judge score (v1, Table 3). However, correlation between the expert judge score and the sum of properties of "biogeography and history" (v3, including naturalisation and domestication) was almost the same as the correlation between the expert judge score and the WRA score. The sum of properties of "biology and ecology" (v4) showed lower correlation than "biogeography and history" (v3). The properties of "biogeography and history" determine a significant part of the WRA score.

In linear regression and logistic regression with stepwise variable selection, naturalisation records (Q301) was the most important variable (Table 4). Existence of natural enemies (Q805) and domestication (Q101) followed it. In the decision tree

Table 2 Correlation coefficient of score by expert judge and WRA score. Variables were sorted according to the correlation with the expert judge score. Correlation for Q403 was not calculated because there was no parasitic plant on the list. High correlation with expert judge score represents importance of the property for WRA, and high correlation with WRA score suggests that the focal property correlated with many other properties, or that the weight of the property in calculating the WRA score was large.

No.	Species property	Expert judge score	WRA score	Group
Q301	Naturalised beyond native range	0.49**	0.68**	D
Q304	Environmental weed	0.38**	0.52**	C
Q101	Is the species highly domesticated?	0.38**	0.43**	E
Q805	Effective natural enemies present locally	0.34**	0.1	E
Q303	Agricultural/forestry/horticultural weed	0.33**	0.60**	A
Q704	Propagules adapted to wind dispersal	0.33**	0.37**	A
Q103	Does the species have weedy races?	0.32**	0.39**	E
Q302	Garden/amenity/disturbance weed	0.29**	0.56**	A
Q804	Tolerates, or benefits from, mutilation, cultivation, or fire	0.29**	0.45**	C
Q607	Minimum generative time (years)	0.27**	0.45**	A
Q605	Requires specialist pollinators	0.27**	0.18*	E
Q703	Propagules likely to disperse as a produce contaminant	0.25**	0.42**	A
Q305	Congeneric weed	0.25**	0.35**	D
Q412	Forms dense thickets	0.24**	0.45**	C
Q801	Prolific seed production (>1000/m2)	0.23**	0.45**	C
Q402	Allelopathic	0.22*	0.49**	A
Q701	Propagules likely to be dispersed unintentionally	0.19*	0.45**	A
Q606	Reproduction by vegetative fragmentation	0.19*	0.28**	B
Q802	Evidence that a persistent propagule bank is formed (>1 yr)	0.18*	0.24**	E
Q602	Produces viable seed.	0.17*	0.22*	E
Q504	Geophyte (herbaceous with underground storage organs)	0.16	0.02	A
Q604	Self-compatible or apomictic	0.15	0.39**	E
Q407	Causes allergies or is otherwise toxic to humans	0.15	0.07	E
Q707	Propagules dispersed by other animals (externally)	0.13	0.34**	A
Q405	Toxic to animals	0.13	0.19*	E
Q204	Native or naturalised in regions with tropical or subtropical climates	0.13	0.17*	E
Q803	Well controlled by herbicides	0.12	0.38**	E
Q408	Creates a fire hazard in natural ecosystems	0.11	0.26**	E
Q203	Broad climate suitability (environmental versatility)	0.11	0.25**	B
Q503	Nitrogen fixing woody plant	0.11	0.18*	E
Q705	Propagules water dispersed	0.1	0.25**	E
Q501	Aquatic	0.1	0.22*	A
Q502	Grass	0.1	0.17*	A
Q201	Species suited to tropical or subtropical climate(s)	0.08	0.13	E
Q601	Evidence of substantial reproductive failure in native habitat	0.08	0.06	E
Q205	History of repeated introductions outside its natural range	0.08	-0.05	E
Q409	Is a shade tolerant plant at some stage of its life cycle	0.07	-0.18*	E
Q410	Tolerates a wide range of soil conditions	0.07	0.1	B
Q404	Unpalatable to grazing animals	0.06	-0.04	E
Q411	Climbing or smothering growth habit	0.04	0.1	E
Q406	Host for recognised pests and pathogens	0.02	0.15	E
Q102	Has the species become naturalised where grown?	0.02	0.13	E
Q202	Quality of climate match data	0.01	0.14	E
Q702	Propagules dispersed intentionally by people	-0.01	-0.06	E
Q603	Hybridises naturally	-0.04	0.11	E
Q708	Propagules survive passage through the gut	-0.08	-0.01	F
Q401	Produces spines, thorns or burrs	-0.08	-0.06	E
Q706	Propagules bird dispersed	-0.09	-0.04	F
Q403	Parasitic			

* P<0.05; **P<0.01

analysis, large values for naturalisation (Q301) and existence of natural enemies (Q805) were selected (Fig. 3).

Regression models

In stepwise linear regression based on all variables, naturalisation beyond native range (Q301) was the most effective variable, and existence of natural enemies (Q805) and domestication level (Q101) followed it (Table 4). The equation was

$$\textit{Estimates of expert judge} = 0.222\ Q301 + 0.463\ Q805 + 0.175\ Q101 + 0.510 \qquad (3)$$

If only the variables of "biogeography and history" were considered, Q301 and Q101 were important variables. Q805 was the most important trait in the analysis of "biology and ecology" type variables. Variables selected in logistic regression were similar to those in linear regression. Naturalisation beyond native range (Q301) was also the most important variable in the decision tree analysis (Fig. 3).

Table 3 Correlation coefficients of summed variables. Sum of individual properties gives the value for higher level category (v5 to v12). Biogeographical score was the sum of categories belonging to biogeography and history (v3 = v5 +v6 +v7). Biological trait (v4) was the sum of categories belonging to it (v4 = v8 +v9 +v10 +v11 +v12). WRA score was the sum of v3 and v4 (v2 = v3 +v4).

Variables					Biogeography and history			Biological trait				
	v1	v2	v3	v4	v5	v6	v7	v8	v9	v10	v11	v12
v1 Expert judge score	1.00											
v2 WRA score	0.61**	1.00										
v3 Biogeography	0.60**	0.88**	1.00									
v4 Biological trait	0.52**	0.82**	0.49**	1.00								
v5 Domestication	0.39**	0.48**	0.62**	0.21*	1.00							
v6 Climate	0.18*	0.29**	0.38**	0.09	0.14	1.00						
v7 Naturalisation	0.57**	0.87**	0.95**	0.51**	0.39**	0.24**	1.00					
v8 Undesirable trait	0.29**	0.37**	0.30**	0.35**	0.05	0.14	0.33**	1.00				
v9 Plant type	0.19*	0.29**	0.14	0.43**	0.06	-0.02	0.16	-0.09	1.00			
v10 Reproduction	0.39**	0.65**	0.38**	0.76**	0.04	0.16	0.44**	0.11	0.25**	1.00		
v11 Dispersal	0.26**	0.56**	0.26**	0.72**	0.22*	-0.04	0.25**	-0.01	0.10	0.47**	1.00	
v12 Persistence	0.43**	0.49**	0.35**	0.62**	0.22*	0.05	0.35**	0.12	0.24**	0.30**	0.25**	1.00

Table 4 Selected variables using a stepwise variable selection procedure. Significance level was determined by equation 1. Species properties are in Tab. 1.

Number of variables	Linear regression	Linear regression with bio-geographical variables	Linear regression with biological traits	Logistic regression	Linear regression with grouped variables
1	Q301	Q301	Q805	Q301	*Naturalisation*
2	Q301, Q805	Q301, Q101	Q805, Q804	Q301, Q805	*Naturalisation, SpecialTrait*
3	Q301, Q805, Q101	ns	Q805, Q804, Q605	ns	*Naturalisation, SpecialTrait, Max Vegetation*
4	ns	ns	ns	ns	*Naturalisation, SpecialTrait, Max Vegetation, Weediness*

Max Vegetation = Max(*Forest*, *DisturbedHabitat*); *Naturalisation* = Z(Q101 + Q301 + Q305); *Weediness* = Z(Q302 + Q303/2 + Q304/2); *SpecialTrait* = Z(Q503 + Q605 + Q805); *Forest* = Z(Q409 + Q602); *DisturbedHabitat* = Z(Q607 + Q701 + Q703 + Q704)
$Z(x)$ represents standardised value of the variable x, using mean (μ_x) and standard deviation (σ_x) of x as $Z(x) = (x-\mu_x) / \sigma_x$.
$\mathrm{Max}(x_1, x_2, x_3, ... x_n)$ gives the maximum value among values of $x_1, x_2, x_3, ... x_n$.

Construction of linear regression model with grouped variables

Variable groups were determined based on the results of the cluster analysis of properties (Tab. 1), regression and correlation to expert judge scores, and empirical experiences. At first, the most closely correlated variable to the expert judge score (Tab. 2) was listed for the groups of A to F. These were Q303, Q304, Q301 and Q101 for Groups of A, C, D and E respectively. Group B and F were not considered because of low correlation. Ecologically similar variables were combined into one group. They were Q301 and Q101 of the variable *Naturalisation*, and Q303 and Q304 of the variable *Weediness*. The variable chosen in the stepwise linear regression was also added (Q805, Table 4) as the variable *SpecialTrait*. Ecologically similar variables were added to each group, and the number of variables in each group was adjusted to about three, to increase redundancy and to reduce stochastic noise (equation 2).

The variable *Naturalisation* represented the naturalisation of the species elsewhere in similar climate, and domestication history. The variable *Weediness* represent records of weediness elsewhere in a similar climate and having an undesirable impact. *SpecialTrait* was collected from Group E. Existence of natural enemies (Q805) was significant in the regression model, and other biological traits such as requirement for specialist pollinator (Q605), and nitrogen fixing woody plant (Q503) were added to it.

Almost all of the above selected variables were properties of the "biogeography and history" type, except for *SpecialTrait*. However, risk assessment based on biological traits is necessary for species not known as invasive yet. Key traits to assess invasion risk into climax forest communities were shade tolerance and maximum tree height (Koike 2001);

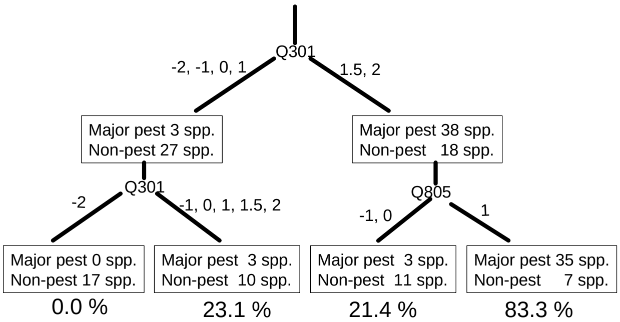

Figure 3 Result of decision tree analysis (C&RT, ANSWER TREE 3.0J). Three variables significant in the linear regression model were used (Q101, Q301, and Q805). The values shown for each branch (-2, -1, 0, 1, 1.5, 2) are the value of the variable assigned by WRA (see Appendix 1-1 in Kato *et al.* 2006). Large value in Q301 suggests that the species have the record of naturalisation, and climate in Bonin Island is suitable for it. Large value of Q805 represents absence of natural enemies. Percentages under the boxes represent the probability that a species in the box is a major pest. If we set the threshold value for this probability, we can make a judgement to accept or reject new species. For example, only the right side box will be rejected if we use the threshold probability of 50%, but three boxes of species, except the left one, will be rejected if we use 5% as threshold value. Correctly judged probability for major pest is 95% in the latter case (Scenario 2).

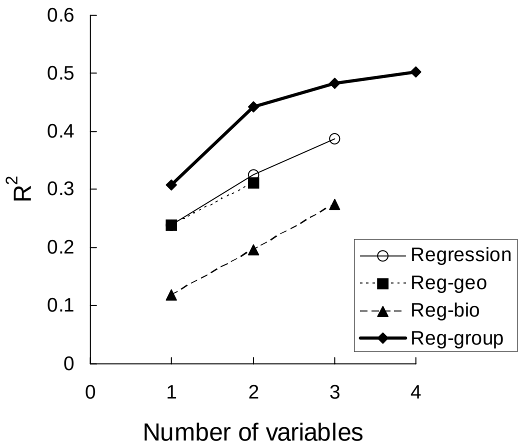

Figure 4 R^2 values for regression models. Abbreviations are as follows; Regression: stepwise linear regression with all variables, Reg-geo: stepwise linear regression with biogeographical variables only, Reg-bio: stepwise linear regression with biological properties only, and Grouped: stepwise linear regression with grouped variables.

however, these traits will prevent invasion into arable weed communities. Shade tolerance will work positively for forest habitats, but negatively for disturbed habitats. A simple sum including such variables may cause an underestimation of the invasiveness risk for species originating from disturbed habitat, when using WRA. Because of

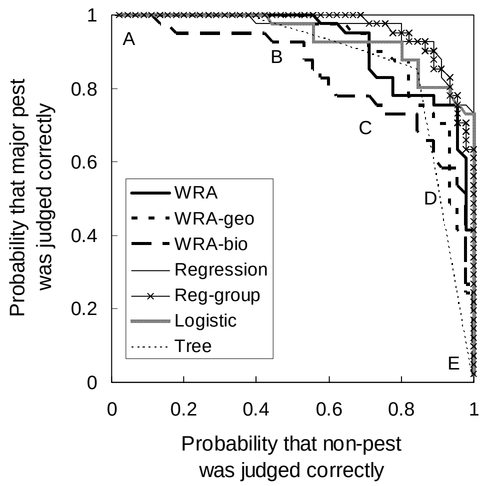

Figure 5 The diagram of two probabilities that a "major pest" is judged correctly, and "non-pest" is judged correctly. Five positions, **A**-**E**, correspond to the approximate position of threshold value shown in Fig. 2. The model closest to the upper and right point (1, 1) is the best one in Scenario 1 (**C**), and **B** corresponds to Scenario 2. Abbreviations are as follows; WRA: WRA score, WRA-geo: sum of WRA scores only for "biogeographical and historical" species properties, WRA-bio: sum of WRA scores only for "biology and ecology" properties, Regression: stepwise linear regression with all variables, Reg-group: stepwise linear regression with grouped variables, Logistic: stepwise logistic regression with all variables, and Tree: decision tree analysis shown in Fig. 3.

this we evaluated invasiveness into forests (*Forest*) and disturbed land (*DisturbedHabitat*) independently, and the maximum value for invasiveness of the different habitats (*Max Vegetation*) was used. By taking the maximum value, the invasiveness to forest was used for forest species, and invasiveness to disturbed land was used for arable weeds. Shade tolerant (Q409) and viable seed production (Q602) were considered as properties of invasive species to forest (*Forest*). Short lived (Q607) species with specific properties of dispersal (Q701, Q703 and Q704) were considered as invasive species to disturbed land (*DisturbedHabitat*).

The obtained equation was

$$\begin{aligned} \textit{Estimates of expert judge} &= 0.310\ \textit{Naturalisation} \\ &+ 0.260\ \textit{SpecialTrait} \\ &+ 0.191\ \textit{Max Vegetation} + 0.131\ \textit{Weediness} \\ &+ 0.857 \end{aligned} \tag{4}$$

where

$$\textit{Max Vegetation} = \mathrm{Max}(\textit{Forest}, \textit{DisturbedHabitat})$$
$$\textit{Naturalisation} = \mathrm{Z}(Q101 + Q301 + Q305)$$
$$\textit{Weediness} = \mathrm{Z}(Q302 + Q303/2 + Q304/2)$$

SpecialTrait = Z(Q503 + Q605 + Q805)
Forest = Z(Q409 + Q602)
DisturbedHabitat = Z(Q607 + Q701 + Q703 + Q704)

Z(x) represents standardised value of the variable x, using mean (μ_x) and standard deviation (σ_x) of x as $Z(x) = (x-\mu_x) / \sigma_x$. Max($x_1$, x_2, x_3, ... x_n) gives the maximum value among values of x_1, x_2, x_3, ... x_n. Weightings for Q303 and Q304 were reduced because weightings for these variables were heavier than Q302 in the traditional WRA. All four grouped variables (*Naturalisation*, *SpecialTrait*, *Weediness* and *Max Vegetation*) were significant in stepwise variable selection at the significance level of equation 1.

Evaluation of risk assessment systems

In various linear regressions, the regression with grouped variable had highest R^2 value (Fig. 4). Stepwise linear regression with all variables followed it. Linear regression with biological traits had the lowest predictability.

The plot of two probabilities: "major pest was judged correctly", and "non-pest was judged correctly", was used to evaluate various risk assessment models (Fig. 5). Each model gives a single line on this graph. A point on the line in Fig. 5 corresponds to the threshold value used (for example, threshold in WRA scores, threshold for the predicted value of expert judge in linear regressions, etc).

In Scenario 1, the position closest to the coordinate (1, 1) gives the best threshold value, and models closest to the (1, 1) position are the best (threshold value **C** in Fig. 2 and 5). In Scenario 2, the point where the line crosses the horizontal of [(probability that major pest was judged correctly) = 0.95] gives the best threshold value (**B**). The more to the right this point is, the better the model.

In Scenario 1, the linear regression model and the model with grouped variables were similarly best risk assessment models (Table 5, Fig. 5), although the grouped variables model was the best in the measure

Table 5 Comparisons of risk assessment models based on WRA in Bonin (Ogasawara) Islands (Kato *et al.* 2006). Two scenarios that both major pest and non pest are treated equally (Scenario 1), and that to reduce the risk by major pest up to 0.05% (Scenario 2). Larger probability suggests better risk assessment system in both scenarios. These variables were determined based on the data shown in Fig. 5.

Model	Scenario 1: threshold value to maximise probability that both major pest and non pest were judged correctly	Scenario 2: threshold value at (correctly judged probability for major pest) = 0.95
Traditional WRA	Threshold value = 8.25[a] Maximum probability = 0.72	Threshold value = 3 [a] Correctly judged probability for non-pest = 0.71
WRA-geo	Threshold value = 6.2[b] Maximum probability = 0.72	Threshold value = 5 [b] Correctly judged probability for non-pest = 0.71
WRA-bio	Threshold value = 1[c] Maximum probability = 0.62	Threshold value = -2 [c] Correctly judged probability for non-pest = 0.42
Linear regression	Threshold value = 0.97[d] Maximum probability = 0.82	Threshold value = 0.95 [d] Correctly judged probability for non-pest = 0.80
Reg-geo	Threshold value = 1.13 [d] Maximum probability = 0.68	Threshold value = 1.05 [d] Correctly judged probability for non-pest = 0.71
Reg-bio	Threshold value = 1.1 [d] Maximum probability = 0.63	Threshold value = 0.41 [d] Correctly judged probability for non-pest = 0.18
Reg-group	Threshold value = 0.89 [d] Maximum probability = 0.80	Threshold value = 0.80 [d] Correctly judged probability for non-pest = 0.82
Logistic regression	Threshold value = 0.81[e] Maximum probability = 0.75	Threshold value = 0.22 [e] Correctly judged probability for non-pest = 0.56
Decision tree	Threshold value = any value between 0.23 to 0.83 [e] Maximum probability = 0.72	Threshold value = any value between 0 to 0.21 [e] Correctly judged probability for non-pest = 0.38

[a] WRA score (Pheloung *et al.* 1999)
[b] Sum of values assigned for species properties in WRA , only biogeographical and historical properties were considered.
[c] Sum of values assigned for species properties in WRA , only biological and ecological properties were considered.
[d] Predicted value of expert judge score; 0: non-pest, 1: minor pest, 2: major pest
[e] Predicted probability that the species is major pest

of R^2 (Fig. 4). The probability that both major pest and non pest were judged correctly was 0.82 at the threshold value of [(estimates of expert judge score) = 0.97] in the linear regression model.

In Scenario 2, the model with grouped variables correctly judged 82% of non-pest species at the threshold value of 0.80. The next best model was linear regression with three individual variables selected by stepwise procedure from all of variables.

DISCUSSION

Weed Risk Assessment (WRA)

Bio-geographical and historical variables, especially for naturalisation beyond native range and intensity of domestication, are key species properties in WRA. Biological traits scarcely contribute to risk assessment. Although our research identified "natural enemies" as an important biological variable, it is difficult to estimate it prior to introduction of an alien species and the reliability of this property is hence, low. The high value for prediction of naturalisation history and the low one of biological properties have been described for various species (Williamson 1996, Goodwin *et al.* 1999, Gollasch 2006). WRA can be considered as a system to evaluate the history of prior invasion. Since so many biological properties were not effective predictors, biological properties could possibly be removed from use in WRA. Low effectiveness of these properties may be one reason why WRA is not sensitive to missing values for them.

When social aspects are considered, WRA is not solely a system to predict invasion risk, but also a system for input by stakeholders. All possible hazards should be considered, even if the weighting is low and the system may not estimate that risk correctly. Use of qualitative values in WRA ("yes", "no", "unknown") enables speedy results and could contribute to avoiding controversy among stakeholders on the weighting of properties.

In conclusion, WRA is a well designed system that is robust and provides for stakeholder consideration. It can be applied quickly in various regions because it only requires qualitative information. However, it can not use biological traits effectively in prediction.

Invasion risk assessment systems available for immediate use

We need to consider time scale in the search for suitable systems of invasion risk assessment. The Japanese Government brought into force the Invasive Alien Species Act in 2005, and needs a suitable risk assessment system to determine what species to act on. For such immediate application, the information should be qualitative, because quantitative information of consistent quality is not available. Properties for which information is difficult to obtain should be excluded (e.g. existence of natural enemies). The "biogeography and history" part of WRA may be the best system, although the threshold value for acceptance (of a proposed introduction of an alien species) needs to be re-determined for Japan's purposes (Fig. 5). WRA can be applied immediately to a wide range of species, including animals.

Australia and New Zealand have different biota from the Northern Hemisphere, and Hawaii and the Bonin Islands are oceanic islands with disharmonic biota. Applying the "biogeography and history" component of WRA for mainland situations in the Northern Hemisphere, while adjusting the threshold value (see Fig. 5) will have some effectiveness. It is also possible to evaluate "naturalisation" only for those mainland areas with similar biota, while excluding information from invasion in oceanic islands and in the Southern Hemisphere.

Invasion risk assessment systems for the near future

In the near future, it will be possible to improve the system for weighting and calculation, but qualitative data will need to be used, like in WRA. The regression model with grouped variables proposed in this paper is one candidate (equation 4), although it should be tested with additional data. The simple regression model (equation 3) gave similar results when minor pests were removed from the evaluation. However, the grouped variable model showed a higher R^2 value than the simple regression model did if minor pest species were included, and such inconsistency may be caused by stochastic noises. In the grouped variable model, stochastic error from individuals evaluating the species will be averaged (equation 2). Since the redundancy of this model is lower than in WRA (even if higher than in simple regression model), missing values should be avoided.

Risk assessment based on biological properties is required in the prediction of invasions by species not known as invasive yet. In order to use biological properties, we may need to assess the risk of invasions independently for various habitats (i.e. risk of invasion for arable land, for forests, etc.), as was done in the model with grouped variables in our research. We applied this to two habitats: forests and disturbed land. However, we need further study to determine the key properties for various other habitats, such as dry rocky cliffs or wetlands.

By evaluating risks of invasion for individual habitats, it might be possible to account for the conservation importance of such habitats. The mere establishment of an alien species can be a significant issue in an area of endangered natural vegetation, whereas a certain level of dominance by an alien species might be acceptable in the case of urban situations.

Invasion risk assessment system for use in the distant future

In the distant future, plant properties should be measured quantitatively. Properties of native species in the local flora should be used in the risk assessment, because the success of invasion depends on local flora (Koike 2001). Impact of invasive alien species (dependent variable in regression) on native ecosystems should be measured quantitatively by standardised methods. It might be possible, in future, to estimate the impact on ecosystems using a community prediction model (Koike 2001).

ACKNOWLEDGEMENTS

The author would like to thank K. Iwasaki and S. Kurokawa for their valuable comments.

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Short paper

Importance of intra-specific variation for risk assessment: a case study on a useful and noxious plant species, velvetleaf (*Abutilon theophrasti* Medic., Malvaceae)

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INTRODUCTION

Invasive alien plants can cause serious damage to agriculture or natural ecosystems (Nishida and Shimizu 1999, Shimizu 1998). To prevent such damage, it is necessary to regulate the introduction of invasive plants. As a tool to assess what species may become invasive, Weed Risk Assessment (WRA) systems have been applied in Australia and New Zealand (Pheloung *et al.* 1999). Such systems should also be useful in Japan for determining which alien species to regulate.

In the practice of WRA, we have usually treated the species as the minimum operational taxonomic unit. However, in the case of species that have both invasive and useful strains, it is too difficult to judge if the species should be rejected by WRA. The combination of weedy strains and cultivars of *Abutilon theophrasti* (Malvaceae) is an example of this problem. *Abutilon theophrasti* was once an important fibre crop in Japan (Kurokawa *et al.* 2003). Recently, however, it has developed into one of the most noxious weed species in Japanese forage crop fields (Nishida and Shimizu 1999). In this paper, we describe the case study of *A. theophrasti* and discuss the potential of using WRA when there is intra-species variation in a target species.

ABUTILON THEOPHRASTI INTRODUCED INTENTIONALLY AND UNINTENTIONALLY INTO JAPAN

In Japan, *Abutilon theophrasti* has been introduced both intentionally and unintentionally. Intentional introductions of the species are considered to have started during or prior to the Heian era (794-1185) for fibre production (Kurokawa *et al.* 2003). In the Meiji era (1868-1911), some cultivars were introduced from China to evaluate their suitability for cultivation (Yoshikawa 1919). After that, the cultivation of *A. theophrasti* declined gradually. At present, 11 cultivars are preserved in the Japan Gene Bank. These cultivars are very important strains for use as fibre crops.

On the other hand, unintentional introductions have been identified as seed contaminants of grains imported into Japan (Shimizu *et al.* 1996). Seeds of *A. theophrasti* have been found in soybean, maize and lupin imported from the United States or Australia (Enomoto 1999). The plants obtained from those seeds exhibited strong weedy growth habits (Kurokawa *et al.* 2003).

Judging from this description of *A. theophrasti,* should the species be rejected for introduction into Japan or not? If we use a WRA system treating the species as the minimum operational unit, the answer would probably be to 'reject.' However, if *A. theophrasti* were still an important fibre crop, the economic loss by rejecting the species would potentially be huge. An alternative would be to only accept crop strains for introduction, but this would require differentiating between crop strains from other, weedy, strains.

THREE GENOTYPES OF *ABUTILON THEOPHRASTI*

Using two genetic markers, that is, capsule colour variation and chloroplast DNA (cpDNA) haplotypes, *Abutilon theophrasti* plants can be classified into three genotypes (i.e., I, II and III) (Kurokawa *et al.* 2004). The genotype I, II and III plants produce an 'ivory' capsule and have the cpDNA haplotype 'A', 'ebony' with 'A', and 'ebony' with 'B', respectively.

To compare the morphological characteristics and the growth habits of those genotypes, we conducted a field examination. Principal components analysis (PCA) using 14 variables, based on morphological characteristics and growth habits, showed distinct characteristics for the three genotypes. The first principal component (PC1) may be a useful index of weedy *versus* crop nature, showing a positive eigenvector in seed dormancy rates. In the PCA plot, genotype I plants showed a crop-like nature, distributed in a more negative value in PC1, genotype III plants showed a weedy nature, distributed in a more positive value, and genotype II plants were distributed in the middle. With respect to plant heights three months after sowing, genotype I plants

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showed significantly higher values than the others (I: 147.0cm; II: 110.2cm; III: 111.1cm). The seed dormancy rates of genotype I plants were significantly lower than those of the others (I: 4.4%; II: 25.2%; III: 22.6%). Other characteristics were also significantly different between genotype I and III. Considering the low seed dormancy rate of genotype I, this genotype is likely to have a less weedy nature. Genotype III is likely to be a much more weedy strain than the others. All of the cultivars preserved in the Japan Gene Bank were classified as genotype I.

Genotyping of *A. theophrasti* could hence be a powerful tool to differentiate the lower-risk strains from the higher-risk strains. In particular, genotype I, that is, a low-risk genotype, can be recognised easily from the others using capsule colours.

WRA CONSIDERING INTRA-SPECIES VARIATION

Developing a WRA system to consider intra-species variation will reduce potential economic loss by allowing the acceptance of useful strains. To put this into practice, genetic markers are useful for distinguishing low-risk genotypes. In the case of *A. theophrasti*, a visible capsule colour variation was the most suitable marker to differentiate the cultivars from the wild strains. If there is no difference in morphology between strains, molecular markers could be used. It is also important to clarify the relationship between the genotyping by molecular markers and invasion risk.

In conclusion: in order to reduce economic loss and to maximise benefits WRA systems considering intra-species variation can be developed, using genetic markers to identify low-risk strains of a particular invasive alien species.

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Short paper

Community assembly rules based on plant ecological traits in a rural landscape

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INTRODUCTION

Biological invasions pose a threat to ecological communities and global biodiversity (Lodge 1993). To prevent invasion of alien species into native communities, it is necessary to predict invasion probability before introducing alien species. The assembly rule of a native plant community based on ecological traits represents what types of plants are likely to invade the community (Koike 2001).

In this study, we determined ecological traits of native and alien plants that statistically explained the local communities in a rural landscape (from forest to herbaceous weed communities). We developed and tested statistical models to predict a community's species composition based on ecological traits of the species pool.

METHODS

This study was conducted in a 1km quadrat in Tokyo Japan (35º38'N, 139º25'E; 52–141m above sea level). Rural landscape elements (e.g. deciduous broadleaved forests, evergreen broadleaved forests, meadows at the forest edges, banks by the wayside, crop fields), residential areas, and a school were part of this area. We conducted vegetation surveys in spring, summer, autumn, winter, and early spring at 40 (41 in autumn) plots. Vegetation data was classified into community types based on species composition using TWINSPAN (Hill 1979).

We examined the flora of the study area, and 481 native and alien species were listed. 139 of these species were chosen using random numbers, because the measurement of traits is laborious. Although the aim was to measure 28 ecological traits for all 139 species, this was only possible for 50 species, due to insufficient individuals of other species. Traits measured included shade tolerance, maximum-height in the growing season, maximum height in the dormant season, proportion of dormant biomass, ratio of leaf height to total height (including reproductive organ), self-supporting stem, extension of shoots or stolons on ground surface, spread of crown, spatiality of genet, stem lignification, seed dispersal mode, how often they reproduced over a lifetime, life span, flowering/fruiting/in-leaf duration through the year and flowering/fruiting/in-leaf duration in spring/ summer/autumn/winter.

Logistic regression and decision-tree analysis (classification and regression tree: CART, Breiman *et al.* 1984) were used as statistical methods. The species was considered as the sample unit of statistical analysis; existence or absence of the species in a given community type was assumed as the dependent variable, and species traits as the independent variables. We calculated the similarity of the species composition between the actual and predicted communities using Jaccard similarity coefficient (Jaccard 1901, Kobayashi 1995, Gotelli and Ellison 2004) based on presence/absence of species determined by threshold abundance level.

RESULTS

Seven community types were obtained by TWINSPAN. These were four forest communities (evergreen broadleaved forest, abandoned coppice forest in the growing season, managed coppice forest in the growing season, and coppice forest in the dormant season), one meadow community, and two weed communities (winter-spring weed community and summer-autumn weed community). Different communities appeared at the same site in different seasons. The winter-spring weed community is often found in winter at the same site of the meadow community, and the winter-spring weed community in crop fields is replaced by the summer-autumn weed community in the next season.

Species composition was predicted through stepwise logistic regression and decision-tree analysis. In both analyses, the mean Jaccard similarity coefficient between predicted and actual communities was around 65% in most forest community types, but less than 50% in the herbaceous communities. Both logistic regression and decision tree analysis showed similar predictability.

In forest communities, strong shade tolerance

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was selected by stepwise logistic regression, as the most important trait. Maximum height was also important in some cases. "Long duration of flowering" and "stems that spread on the ground" were recognised as key traits for all weed communities. In the meadow community, "stems that spread on the ground" was selected by logistic regression and "long duration of flowering" by decision-tree analysis, but predictability was low.

DISCUSSION

Our results showed that it was possible to predict species composition of the plant communities from the ecological traits to some extent. The Jaccard similarity coefficient between predicted and actual forest community was 65% and was higher than for other communities. In herbaceous communities, the predictability was considerably lower than in the forest communities. Herbaceous plants that grow in arable land rarely appear in the dark environment of the forest floor, but seedlings of forest tree species sometimes appear in herbaceous communities. The occurrence of such species in herbaceous communities might be responsible for the lower predictive ability in herbaceous communities than in forests.

Shade tolerance was the key trait for forest communities, both for woody plants and for herbaceous species. Koike (2001) showed that a strong shade tolerance was the most important trait required for woody species to exist in climax forest communities, and a large maximum height to be the next. Results of the present study were similar.

A long flowering duration was the key trait in weed communities. Mabry *et al.* (2000) found that long flowering duration was one of the key traits responsible for resistance to disturbance. Ground surface spread was also selected as a key trait. Ground surface spreading enables the survival of individuals in an environment where they are at risk of being trampled or ploughed, and it is also advantageous in terms of rapid horizontal spread.

Prediction of the meadow community was the most difficult. A typical meadow community has a stable species composition, and meadow communities at geographically distant sites have similar composition (Miyawaki 1986; Sone 1991), thus a set of key traits should exist for the community. The true key traits that predicted this community type were not clear in our study.

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Assessment of the introduction potential of aquatic alien species in new environments

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Abstract Biological invasions of alien aquatic species are increasing on a global scale. The most prominent introduction vectors are shipping (i.e. ballast water, tank sediment and hull fouling of all vessel types, including fishing and recreational craft) and aquaculture activities. Many aspects of biological invasions remain unclear - among them are the most wanted answers: which species will invade, when and where will they arrive, and what kind of impact may be caused by the invaders? Matching salinity and climate conditions in donor and recipient region enable a very first estimation of the species' potential to survive in new habitats. Another important component is the voyage duration of a ship (longer voyage durations, in general, cause a drop in species abundance in ballast tanks). The need for an advanced risk assessment becomes clear as many species have the potential to cause major ecological and economic impacts in the receiving region.

Key words: transport vector; ballast water; hull fouling; environmental match; survival potential

INTRODUCTION

Scientists have tried to mathematically calculate the risk of species invasion for a very long time. Darwin (1900) estimated that 5% and Williamson (1989) that 10% of introduced species may form populations that are self-sustaining in the receiving environment for several generations at least. However, this 10´s rule was predominantly based on introductions to terrestrial habitats (Holdgate 1986; Simberloff 1986, 1989; Williamson & Brown, 1986). Williamson (1996) revised the 10´s rule indicating that "10" actually ranges between 5 and 20. Several methods have been developed in order to identify and/or quantify the risks of biological invasions. As reviewed by Hayes (1997, 1998, 1999), quantitative risk assessment methods target specific, predetermined species, and follow a sequence of steps to evaluate the probability that:

(i) the target species will be entrained ("picked up"/delivered, including unintentionally) by the vector/s of interest,
(ii) the target species will survive the transport phase,
(iii) the target species will survive in its new environment,
(iv) the target species will establish a self-sustaining population, and
(v) the target species will cause harm.

It must be recognised that the end point of Hayes' quantitative risk method is the likelihood of discharge of unwanted target species that can survive in the receiving environment. As Hayes has noted several times, the quantitative method cannot predict the establishment, invasion or impact of such species with any useful certainty, e.g. Hayes (1998, 1999), Hayes & Hewitt (1998), Hewitt & Hayes (2002).

METHODS

The evaluation of the invasion risk as carried out during the German shipping study (undertaken 1992-1996, see below) was based upon matching environmental conditions. However, the environmental matching assessment between recipient and donor ports alone will not provide a complete risk assessment as this method does not take account of:

(1) the survival potential of each species of interest during the uptake, transport and release phase,
(2) the minimum number of individuals required to establish a founder population,
(3) the role of vacant or partly vacant ecological niches and the community structure,
(4) the biological capabilities of the species, i.e. tolerance to salinity and temperature as well as adaptation potential of species to new aquatic circumstances, and
(5) predator/prey relations in the new environment.

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Table 1 Colonisation probability of alien species, according to matching salinity in donor and recipient region, after Carlton (1985).

RECIPIENT region	DONOR region		
	Freshwater	Brackish water	Salt water
Freshwater	high	medium	low
Brackish water	medium	high	high
Salt water	low	high	high

On the other hand, species are more likely to become established in environments that are similar to those of their origin. Therefore, if the port of loading of ballast water and the port of discharge are ecologically similar, the risk of a species introduction is relatively high.

An example of the environmental matching procedure is a shipping study carried out in Germany from 1992 to 1996, being the first European study based on ballast water sampling. In total, 132 ballast water samples, 131 hull fouling samples and 71 samples from ballast tank sediments were taken from 186 vessels, predominantly in ports along the German North Sea coast. In total 257 species were identified of which 146 (57%) were considered to be alien to the North Sea region (Gollasch 1996, 2002a,b). For all alien species sampled from the ballast water, tank sediments and ship hulls, the potential to become established in German waters was assessed. This assessment was carried out in accordance with the scheme developed by Carlton (1985) (Tab. 1), i.e. comparing the salinity tolerance of the species and the salinity conditions of the receiving waters. In addition, a similar scheme was used to take into account the climate in the area of origin (donor area) and the potential recipient area (Tab. 2).

RESULTS

All species collected during the German shipping study which are either native to or occur near cold-temperate climate areas outside Europe were included in the category "establishment highly probable" in German waters (Gollasch 1996, 2002a,b). The number of species and the number of individuals decreased with increasing voyage duration. Species native to cold-temperate areas of the northern hemisphere of the Atlantic Ocean (North American east coast and the upwelling area off western Africa) were included as high risk species due to the relatively short duration of the ships voyage and matching climates. A total 32 alien species (22% of all alien species found) were included in this category (Gollasch 1996, 2002a,b).

An example is the decapod *Hemigrapsus penicillatus*, native range from cold-temperate areas of northern Japan to tropical China (Pillay & Ono 1978, Noël 1997, Türkay pers. com). *H. penicillatus* is a recent macro-invertebrate invader in European waters, first recorded from the Atlantic coast of France in 1994 (Noël 1997). Several specimens of the crab were collected from the hull of a vessel during the German shipping study and it is believed that this vessel introduced the crab to Europe (Gollasch 1999). The establishment of *H. penicillatus* in Europe confirms that the applied model to assess the invasion potential, as carried out during the German shipping study gives a useful first estimate of the probability of establishment and therefore confirms the importance of matching climates in donor and recipient areas. The value of environmental matching for marine bioinvasion assessments has been noted by other workers such as Smith *et al.* (1999) and Clarke *et al.* (2003), who noted that "*if any one factor is to be used alone, environmental matching is probably the best single indicator of risk*".

DISCUSSION

A semi-quantitative model (low - medium - high risk) was developed based on environmental matching of donor and recipient region and applied to all alien species collected during the German shipping study. The approach is considered a simplistic first attempt to assess the invasion potential. Other parameters influencing the invasion risk, not being addressed here, include: vessel ballasting characteristics, ballast water treatment, characteristics of donor and receiving

Table 2 Colonisation probability of alien species, according to matching climate in donor and recipient area, after Gollasch (1996)

RECIPIENT region	DONOR region			
	Arctic & Antarctic	Cold-temperate	Warm-temperate	Tropics
Arctic & Antarctic	high	medium	low	low
Cold-temperate	medium	high	medium	low
Warm-temperate	low	medium	high	medium
Tropics	low	low	medium	high

regions, voyage route and duration, and relevant biological information for the key/target species, i.e. environmental requirements such as temperature, salinity, and light/energy requirements during different stages of the life cycle (including resting stages), habitat requirements, and known biotic interactions.

It has to be taken into account that all general rules or models, such as matching environmental conditions as a risk indicator, have their exceptions and cannot be applied for all habitats and/or species. Assessments based on environmental matching of donor and new habitat alone, do not include the potential of species to tolerate or adapt to environmental conditions uncommon in its native range. Also, locally heated waters, e.g. by power plant effluents, may enable species from warmer climates to survive in colder waters. One example is the tube worm *Ficopomatus enigmaticus*, native to tropical and warm-temperate waters (Walford & Wicklund 1973). It was first recorded in northern Europe along the southern coast of England (Zibrowius & Thorp 1989) and was subsequently found in Germany near a power plant in the Port of Emden (Kühl 1977). As a consequence, when carrying out a risk assessment, locally heated waters should be noted, especially in or near ports or other ballast water release zones. The role of vacant or partly vacant ecological niches and the community structure should be included when assessing the invasions potential (Williamson 1996). However, alien species can also invade in areas where no empty ecological niche is available, for example if they are more resistant to pollution or have a more successful reproduction strategy compared to native species. For instance, the zebra mussel (*Dreissena polymorpha*) invaded the freshwater areas of the Baltic Sea and adjacent waters (Dexbach 1935, Thienemann 1950) as well as the North American Great Lakes (Hebert *et al.* 1989). The ecological niche of this species is characterised as a fresh water filter feeder. Native freshwater filter feeders occur in both systems; however, native mussels were regionally driven extinct by the invader (Minchin pers. com.). The lack of an empty niche cannot, therefore, be considered as an excluding factor in general for future species introductions.

It remains impossible to predict with useful levels of certainty which organisms will survive and establish in new habitats (see e.g. Hayes 1998, 1999, Hewitt & Hayes 2002). This unsatisfactory situation is caused by the enormously high number of factors that need to be taken into account when assessing the invasion risk (e.g. minimum individual numbers of founder population, climate, salinity, habitat structure, food availability and predators) (Hayes 1997). Key limiting factors for successful invaders are flexibility in regard to temperature and salinity tolerance, habitat selection and food. As a result, species from similar latitudes, (i.e. comparable environmental conditions) have a greater chance for survival once introduced. Additionally, the ability to survive the introduction process, especially the transport phase and the season and frequency of transportation, are key limiting factors (e.g. Ruiz 2002, Raaymakers & Hilliard 2002, Clarke *et al.* 2003). To survive in a recipient area is important, but only the first step to a sustainable population. The newly introduced species needs to reproduce effectively enough to form a successful population (Gollasch 1996). Species introduction in ballast water is one of the most prominent vectors of species movements. It should be noted that ship mediated species invasions will continue, even when proper ballast water management options are in place, as ships also carry organisms on the hull, in sea chests and associated with the cargo. On a regional scale, hull fouling as invasion vector may be of greater importance than ballast water.

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Important vectors for marine organisms unintentionally introduced to Japanese waters

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Abstract Ships are recognised as a major vector for the introduction of alien marine organisms, either through hull fouling or via ballast water. It is known that 26 species have been unintentionally introduced into Japanese waters and 42.3% of these are presumed to have been introduced by hull fouling. A notable feature of introductions to Japan is that, hull fouling is considered as the most important vector and there are no species that have been introduced solely by ballast water. This is thought to be due to the fact that ballast water, is usually retained within the ship for long enough to kill the organisms within it. The low importance of ballast water as a vector is also a common feature among importers of natural resources. The most significant source regions for species introduced to Japan are the North East Pacific and the East Asian Sea. Meanwhile, introductions from the North West Pacific, which includes countries close to Japan, are few. Because the risk of introduction from the North West Pacific, where the climate is similar to Japan's, can be assumed to be high, care should be taken with introductions, including secondary ones, from this region. Measures that should be taken to prevent or to reduce future introductions to Japanese waters are discussed, taking into account these factors.

Keywords: ballast water; climate similarity; hull fouling; introduction; Japanese waters

INTRODUCTION

Over many centuries, shipping has inadvertently aided the spread of marine organisms. These species are called introduced species, (or alien species) if they end up in areas where they are not native. Some of these introductions have caused economical or ecological problems around the world. An estimate of the cost to the US economy, including the cost of control measures, of alien marine and freshwater organisms exceeded 2.4 billion US dollars a year (Pimentel *et al.* 2000).

Although similar impacts happened in Japan (Arakawa 1980, Anon 2003), neither the current situation nor the cost of restoration or preventative measures are clear. It is said that world trade has increased 14-fold since 1950 and in this period the number of biological invasions of terrestrial, freshwater, and marine habitats has increased exponentially (Hayes 2003). Since this trend is likely to have occurred also in Japan, it is expected that further impacts will be caused by introduced species, now and in the future. To enable us to take measures to prevent or reduce problems caused by introduced species, it is necessary to clarify the current impacts caused by them and to raise public awareness about them.

As already said, an important way of preventing impacts caused by introduced species is by not introducing them, since this is far less costly than eradication of established alien species (e.g., Carlton 2001a). To do this, it is necessary to elucidate the means of introduction, the life histories, and the habitats of current species introduced to Japan. These results may lead to the development of effective measures and technologies to prevent or reduce introductions.

In this study, the vectors by which introduction occur are reviewed with this in mind and, after consideration of their importance for introduction into Japanese waters, a new measure to prevent or reduce introductions is discussed.

A succession of vectors

Before the 19th century, there were only three vectors for the introduction of marine organisms. They were dry or semi-dry ballast, hull fouling, and intentional movement to provide food (Carlton 1999, 2001a, 2001b). The number of vectors has increased over time, with the diversification of marine transport. During the 19th century, three vectors were added: ballast water, importation for aquaculture and the construction of canals, such as the Suez Canal. In the 20th century, the further diversification of marine use, accompanied by rapid economic development, led to the appearance of many new vectors, so that the total number now exceeds 20 (refer Otani 2004). Based on Carlton (2001a), Williamson *et al.* (2002) assigned the

Pages 92-103. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

known vectors into eight broad categories and showed that commercial shipping and aquaculture are the most important ones. Reviewing many studies, Gollasch (2002) also concluded that these two categories provide the main means of introduction of alien species into aquatic ecosystems. Although aquaculture practices may be considered to be the most important vector for certain regions (see Williamson *et al.* 2002), there are many indications that commercial shipping is the most important vector overall (Eno *et al.* 1997, Gollasch 1999, Steneck and Carlton 2001, Fofonoff *et al.* 2003, Hewitt *et al.* 2004, Otani 2004). Before the development of the use of ballast water in the mid-1800s, the category ‘commercial shipping’ consisted of two vectors, dry or semi-dry ballast and hull fouling. However, the use of dry or semi-dry ballast has gradually reduced with the increased use of ballast water. After the changeover to ballast water in the 1950s (Carlton *et al.* 1995), the two vectors, ballast water and hull fouling, have become important for the transfer of marine organisms by shipping (e.g., Carlton 1985, Williamson *et al.* 2002).

Ballast water

Ballast water is used to add weight and so stabilise the ship at times when the weight of cargo is insufficient to do so or to adjust ship’s trim. This system was developed in the mid-1800s and became extensively used over the next decade or so (Carlton *et al.* 1995). The quantity of ballast water has increased with the steady increase in total seaborne trade (Carlton 1985). At present, “it can be concluded that the average annual ballast water discharge worldwide is nearing 3 billion tonnes, whilst the annual ballast water discharge worldwide has changed by small increments since 1996” (Karaminas 2002). The variation in the quantity of ballast water discharged worldwide is expected to be small because shipping capacity is almost constant (Karaminas 2002). Ballast water has received much attention as a vector since the late 1980s because of the dramatic increase of invasions associated with it globally (Fofonoff *et al.* 2003). As long as the quantity of discharged ballast water stays at the present level, it has to be expected that introductions via ballast water will continue unless some kind of effective measure to prevent introductions by ballast water is developed. It is also known that some species introduced via ballast water have caused various economical or ecological impacts. Examples include the zebra mussel (*Dreissena polymorpha*), in the Great Lakes (Morton 1997), Japanese dinoflagellates in Australia (Jones 1981) and the American ctenophore (*Mnemiopsis leidyi*) in the Black Sea (Haribson & Volovick 1994). In response to this threat by ballast water various approaches to cope with ballast water introductions are now in place at the international as well as regional level (Williamson *et al.* 2002). The best known is “The International Convention for the Control and Management of Ships’ Ballast Water and Sediments”, adopted by the plenary conference of the International Maritime Organisation (IMO) in 2004. This convention requires management and control of ballast water, and it is expected that introductions via ballast water will decrease drastically as a result of its implementation.

Hull fouling

For a long time, hull fouling was considered to be the most important vector for introductions, but after World War II, it was considered less important as a vector because: (1) the expanded use of increasingly effective antifouling paints, (2) ships spending less time in port, and (3) the increased speed of ships (Allen 1953, Carlton 1985, Fofonoff *et al.* 2003). This assumption led to a strong focus on ballast water as the primary vector for introduction. However, Lewis (2001) referred recently to the importance of hull fouling, mentioning several reasons: (1) in spite of the development of antifouling paint, most vessels carry fouling organisms in their unprotected niches, (2) the expansion of the inter-docking cycle may bring on significant levels of fouling in poorly protected areas, (3) effective antifouling paint, which includes TBT, will be banned in 2008, (4) even at high speed, some recessed places can provide havens for fouling organisms, (5) the shortening of the sailing time between ports works advantageously for some species, and (6) some kinds of ships are stationary or laid up for long periods of time. In addition to Lewis (2001), other investigations describe the importance of hull fouling (Cranfield *et al.* 1998, Lewis 2001, Gollasch 1999, Gollasch 2002, Coutts *et al.* 2003, Godwin 2003, Minchin and Gollasch 2003, Otani 2004).

INTRODUCED MARINE SPECIES IN JAPAN

A short history of research on introduced marine species in Japan

The first review of introduced marine species in Japan appeared in 1980. This research was carried out by Arakawa (1980) and reported on 13 introduced species (Tab. 1). Subsequently, although there was some research on introduced species in Tokyo Bay (e.g., Asakura 1992, Kajihara 1996, Furota 1997,

2002) and in Osaka Bay (e.g., Nabeshima 2002), no nationwide review of introduced species was published until the review of Otani (2002). Referring to Asakura (1992), Otani (2002) added five new species to the 13 already described by Arakawa (1980) (Table 1). However, no Japanese researchers, including Otani (2002), applied criteria to judge whether their species were introduced. In addition, due to insufficient records of occurrence in the past, recent taxonomic rearrangements, and confusion over some species reported in those papers, it was likely that some of the species reported in the past as introduced might not actually be so. When Iwasaki *et al.* (2004) surveyed these problems, based on their questionnaire survey carried out during 2002-2003, they reported 26 unintentionally introduced species (Tab. 1), 15 intentionally introduced species, and 20 cryptogenic species in Japanese waters. The determination of status, whether native, introduced, or cryptogenic, was based on new criteria modified from Ruiz *et al.* (2000).

Table 1 Marine organisms unintentionally introduced to Japanese waters reported in each paper. *: Newly arranged to Cryptogenic species by Iwasaki.

Species	Arakawa (1980)	Otani (2002)	Iwasaki *et al.* (2004)
Annelida			
Hydroides elegans	X	X	X
Ficopomatus enigmaticus	X	X	X
Tentaculata			
*Zoobotryon pellucidum**	X	X	
*Bugula californica**	X	X	
Mollusca			
Stenothyra sp.			X
Crepidula onyx		X	X
Nassarius sinarus			X
Cuthona perca			X
Mytilus galloprovincialis	X	X	X
Perna viridis	X	X	X
Xenostrobus securis		X	X
Mytilopsis sallei		X	X
Petricola sp. cf. *lithophaga*			X
Phacosoma gibba			X
Mercenaria mercenaria			X
Arthropoda			
Amphibalanus amphitrite	X	X	X
Amphibalanus eburneus	X	X	X
Amphibalanus improvisus	X	X	X
Amphibalanus variegatus	X	X	X
Amphibalanus venustus	X	X	X
Amphibalanus glandula		X	X
Pyromaia tuberculata			X
Carcinus aestuarii			X
Callinectes sapidus			X
Chordata			
*Ciona intestinalis**	X	X	
Polyandrocarpa zorritensis		X	X
Molgula manhattensis	X	X	X
Phaeophyta			
Cutleria multifida			X
Chrolophyta			
Caulerpa taxifolia			X
Total	13	18	26

Vectors and source bioregions

This study of the vectors responsible for introduction was restricted to the 26 unintentionally introduced species that Iwasaki *et al.* (2004) reported. Intentionally introduced ones (whose vectors were obvious) and cryptogenic species (where it was unclear whether they were introduced or not) are not discussed.

The vectors and source bioregions of each species introduced into Japan were decided by reference to the general literature (see Otani 2004), except that source bioregions were rearranged into the new bioregions used in Hewitt *et al.* (1999). Vectors were compiled into five broad categories (see Cranfield *et al.* 1998): hull fouling, hull fouling or ballast water, ballast water, fisheries, and others or unknown (Tab. 2).

When the number of introduced species by each vector was calculated using data from Japan, “hull fouling” accounted for 42.3%. Adding the category of “hull fouling or ballast water”, (which accounts for 23.1%), and the category of “hull fouling or cargo fouling”, (included in “others or unknown”), to this number, a total of 69.2% of all the species have been introduced by shipping (Fig. 1). In the category of “hull fouling or ballast water”, there are three species such as the spider crab *Pyromaia tuberculata*, the Mediterranean green crab *Carcinus aestuarii*, and the blue crab *Callinectes sapidus*, which we consider not to have been introduced via ballast water but to have been introduced via hull fouling (Otani 2004). Other than these species, it is considered that the Northern quahog *Mercenaria mercenaria* was probably introduced by hull fouling (Otani 2004). If these species are included in “hull fouling”, the number of species introduced by hull fouling increases. For other vectors, “others or unknown” accounts for 23.1% and “fisheries” accounts for 11.5% (Fig. 1). There were no species introduced through ballast water only.

Species have been introduced to Japan from all over the world. Most introduced species are from the East Asian Seas and the North East Pacific, each with six introduced species (Fig. 2). There are only three introduced species from the North West Pacific, in spite of the very similar climate and the high frequency of seaborne trade.

Table 2 Marine organisms unintentionally introduced to Japanese waters, their vectors, and their source regions. Vectors followed Otani (2004) except for *Phacosoma gibba*. Source regions follow Hewitt *et al*. (eds.) (1999). * Rearranged from Okoshi (2004). Abbreviations: A, Accidental release; B, Ballast water; C, Cargo fouling; H, Hull fouling; F, Fisheries.

Species	Presumed primary vector	Presumed alternative vector	Presumed source bioregion
Annelida			
Hydroides elegans	H	B	East Asia Sea, Australia and New Zealand
Ficopomatus enigmaticus	H	B	East Asia Sea, Australia and New Zealand
Mollusca			
Stenothyra sp.	F		North West Pacific
Crepidula onyx	H		North East Pacific
Nassarius sinarus	F		North West Pacific
Cuthona perca	H		Unknown
Mytilus galloprovincialis	H		North East Pacific, Mediterranean, North East Atlantic
Perna viridis	H		East Asian Sea, Central Indian Ocean
Xenostrobus securis	H		Australia and New Zealand
Mytilopsis sallei	H, C		East Asian Sea
Petricola sp. cf. *lithophaga*	Unknown		Unknown
Mercenaria mercenaria	Unknown		North West and North East Atlantic, North East Pacific
******Phacosoma gibba*	*F		*North West Pacific
Arthropoda			
Amphibalanus amphitrite	B, H		East Asian Sea
Amphibalanus eburneus	H		North West Atlantic
Amphibalanus improvisus	H		Unknown
Amphibalanus variegatus	Unknown		Unknown
Amphibalanus venustus	Unknown		East Asian Sea
Amphibalanus glandula	H		North East Pacific
Pyromaia tuberculata	H	B	North East Pacific
Carcinus aestuarii	H	B	Mediterranean
Callinectes sapidus	H	B	North West and North East Atlantic
Chordata			
Polyandrocarpa zorritensis	H		Australia and New Zealand
Molgula manhattensis	H		North West Atlantic, Wider Caribbean, North East Pacific
Phaeophyceae			
Cutleria multifida	H		Unknown
Chlorophyceae			
Caulerpa taxifolia	A?		Unknown

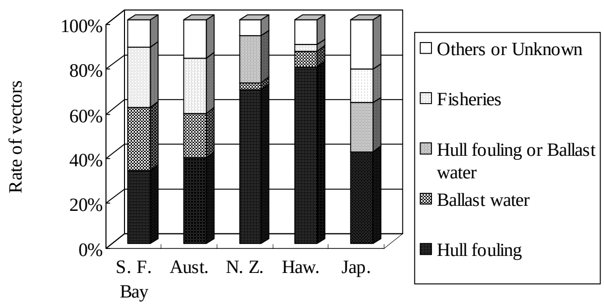

Figure 1 Relative importance of various vectors responsible for the introduction of marine organisms among different regions. (S. F. Bay: San Francisco Bay in 1995; Aust.: Australia in 1995; N. Z.: New Zealand in 1998; Haw.: Hawaii in 1999; Jap.: Japan in 2004) (based on data in Cohen and Carlton 1995, Hewitt and Martin 1996, Cranfield *et al.* 1998, Eldredge and Carlton 2002, Otani 2004)

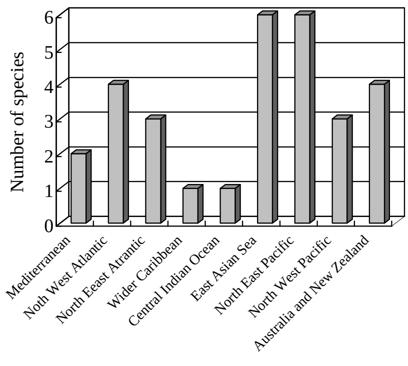

Figure 2 Presumed source bioregions for introduced species in Japanese waters. (In many cases more than one possible vector is considered for a species, so that the total for all bioregions exceeds 26.)

REGIONAL DIFFERENCES IN THE RELATIVE IMPORTANCE OF VECTORS

The number of unintentionally introduced species for each vector was determined to establish the importance of different vectors in Japan. To check whether or not these vectors are specific to Japan, the number of species per vector in some other regions was compared with that in Japan (Fig. 1). These regions are San Francisco Bay, Australia, New Zealand and Hawaii. After removing intentionally introduced species from the results, the number of introduced species per vector was recalculated, except for New Zealand. In the case of New Zealand, the result of Cranfield *et al.* (1998) was used because this rate was calculated based on 148 species unintentionally introduced into New Zealand. In the case of Australia, since Hewitt and Marchin (1996) showed all the possible vectors for the introduction of each species, the number of introduced species per vector was recalculated following the way of Hewitt *et al.* (2004), which gives equal weighting to every vector.

As expected, hull fouling is the most important vector for introduction in all regions, including Japan (Fig. 1). It is tempting to think of this as an inheritance from earlier days, before the use of ballast water was developed, because it is known that many slow-moving wooden sailing ships travelled the world with heavily-fouled hulls (e.g., Carlton 2001b). However, taking account of some recent indications about the importance of hull fouling (Cranfield *et al.* 1998, Lewis 2001, Gollasch 1999, 2002, Coutts *et al.* 2003, Godwin 2003, Minchin and Gollasch 2003, Hewitt *et al.* 2004, Otani 2004) and the fact that hull fouling is the most important vector in all regions, as shown in this study, it cannot be said that hull fouling is only an inheritance. Rather, it should be considered as the most important vector, not only in the past but also at present (see also Cranfield *et al.* 1998).

The importance of introduction by ballast water is different in different regions. For example, the percentage of species that have been introduced by ballast water to San Francisco Bay and Australia is at least 20%. In contrast, this rate is less than 10% in the other three regions. In Japan, notably, there are no species that have been introduced by ballast water alone. The relative importance of introduction by ballast water may be a result of differences in quantity and quality of ballast water discharged in each region.

To check this, I first examined the quantity of discharged ballast water by a single ship in the different regions (Fig. 3). The quantity of discharged ballast water is larger in San Francisco Bay, Australia, and New Zealand than in Japan and Hawaii.

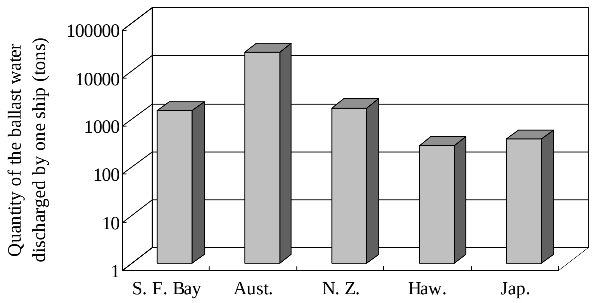

Figure 3 Quantity of ballast water discharged per ship in different regions. This includes all the ships that entered ports of the region, except for Australia. Australia's case includes only bulkers and tankers. (S. F. Bay: San Francisco Bay from 1999 to 2001; Aust.: Australia in 1991; N. Z.: New Zealand from 1996 to 1997; Haw.: Hawaii from 1999 to 2001; Jap.: Japan in 1997) (based on data of Kerr 1994, Hay *et al.* 1997, Ruiz *et al.* 2001, Raaymakers and Gregory 2002)

Comparing these values with the number of introduced species by vector, it is clear that regions with high levels of ballast water discharge correspond to regions that have many species introduced by ballast water (except for New Zealand). This implies that the number of species introduced by ballast water is related to the quantity of ballast water discharged. The quantity of discharged ballast water differs among regions due to differences in their trading patterns, types of ships, etc. Since it is considered that before the 1960s many merchant navies operated without discharging much ballast water, except for some older types of bulk carriers and tankers, it is possible that the difference in the quantity of discharged ballast water caused by trading patterns was smaller than it has been since. Since the 1960s, because of the rapid industry growth in Western Europe and Japan, demand for raw materials such as ore, coal, grain or crude oil increased (Ogawa 1997). In response to this, specialised carriers were developed, including for ore and coal carriers (bulkers), car carrier, and so on. In Japan, where shipbuilding was supported by the Japanese government and by long-term cargo freight guarantees, the number of specialised carriers and enlarged tankers increased in this period (Ogawa 1997). Unlike old merchant navy, bulkers and tankers take up a large quantity of ballast water on the way to the port of loading and discharge almost all of it at the loading port while loading dry bulk or liquid commodities. It is known that the quantity of ballast water taken up by one bulker accounts for 30% to 40% of its DWT (deadweight tons) (Kerr 1994). For example, a 150,000 DWT bulker loads 45,000 tons to 60,000 tons of ballast water (Kerr 1994) and

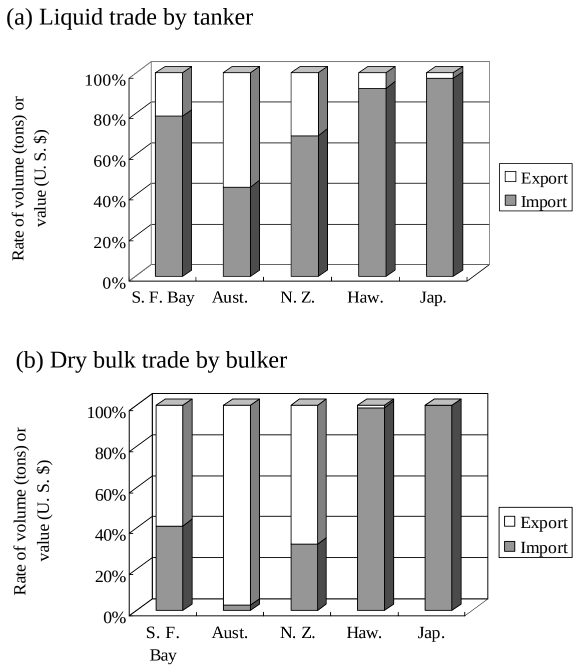

Figure 4 Rate of export and import in volume (tons) transported by dry bulk carrier by region except for New Zealand. For New Zealand import and export are by value (US$). (S. F. Bay: San Francisco Bay in 2000; Aust.: Australia from 2000 to 2001; N. Z.: New Zealand in 1997; Haw.: Hawaii in 2000; Jap.: Japan in 1997) (based on the data of United Nations 2000, U. S. Army Corps of Engineers 2001, The Japanese shipowners' Association 1999, Bureau of Transport and Regional Economics 2003)

discharges almost all of it to allow loading of commodities. Taking into consideration the fact that the sum total of bulkers and tankers accounts for 74.0% by DWT of all the fleets in the world in 2003 (Lloyd's Register Fairplay 2004) and that other parts of the merchant navy do not discharge a lot of ballast water at once (e.g., Kerr 1994, Hay *et al.* 1997), it can be easily recognised that almost all the discharged ballast water in the world is derived from bulkers and tankers. It is therefore assumed that the difference in quantity of discharged ballast between regions is associated with them. Bulkers or tankers call with full ballast water to the region from where dry bulk or liquid commodities are exported. Consequently, regions whose export rate of dry bulk commodities exceeds the import rate are defined as exporters and the reverse ones are defined as importers. The largest quantity of ballast water will be discharged at the exporters and only a small quantity will be discharged at the importers. For liquid trade, which depends on tanker transport, all regions are assigned importer status except for Australia. Meanwhile, for the dry bulk trade, San Francisco Bay, Australia and New Zealand are assigned exporter status and Hawaii and Japan are assigned importer status (Fig. 4). Among exporters, only Australia is an exporter for both. However, it can be considered that the ballast water discharge of Australia depends on bulkers because the volume of dry bulk commodities exported by bulker is more than 13 times that of liquid commodities exported by tanker (Bureau of Transport and Regional Economics 2003).

It is concluded that exporters have a high ballast water discharge by bulkers and that there is a high possibility that many introductions of marine species were caused by ballast water of bulkers. The fact that the rate of introduction via ballast water is higher in San Francisco Bay and Australia than in Hawaii and Japan supports this. New Zealand's case is different. Although this country is considered an exporter of dry bulk commodities (Fig. 4), only a few introduced species are by ballast water, as seen in the fact that the number of species introduced by ballast water itself is only 3% (Cranfield *et al.* 1998). Furthermore, "similar numbers of adventives that arrived in hull fouling have become established in New Zealand over the last 40 years as in the previous 50 years" (Cranfield *et al.* 1998) and so the importance of hull fouling has not changed for a long time. There is also no evidence that the rate of introductions via ballast water has increased. In view of the wide range of invertebrate larvae that were found in the wide variety of vessels and, in particular, in the ballast tanks of bulk carriers (Hay *et al.* 1997), and the fact that New Zealand is an exporter, it would be prudent for them to be prepared for a potential increase of introduction via ballast water in future. In spite of national regulatory measures in New Zealand to control introductions via ballast water, as long as there are living organisms in ballast tanks, the threat of introduction will continue.

In San Francisco Bay and Australia, although hull fouling is the most important vector, it is also thought that ballast water has become a major vector in the past 10 or 20 years (Fofonoff *et al.* 2003, Hewitt *et al.* 2004, Wonham and Carlton 2005). More than 20% of species introduced by ballast water were recorded in these regions. Although national regulatory measures to control introductions via ballast water have been implemented in these regions based on the voluntary guideline adopted at IMO in 1997, the increase of introductions via ballast water needs further attention. It is likely that the rapid increase of transport of dry commodities around the world in the past 20 years has led to a major increase in ballast water discharges in those regions which have been the world's dry commodities exporters (UNCTAD Secretariat 2003) (Fig. 5).

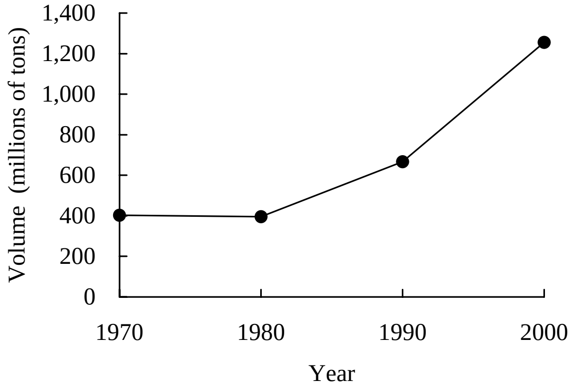

Figure 5 The transition of dry commodities carried by bulker of over 18,000 DWT (dead weight tons) worldwide. (Based on the data from UNCTAD Secretariat 2003)

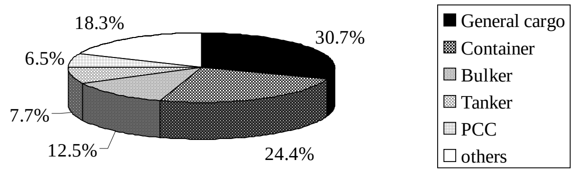

Figure 6 Relative number of ships calling at Japanese ports by ship's type in 1997. (Modified from The Japan Association of Marine Safety 1999)

Introductions via ballast water are lower in importers than in exporters, but importers do not necessarily have zero-discharge ballast water. The reason why introductions via ballast water are low for importers may be due to the "age" of the ballast water in the ballast tanks. As described above, the operation of ballast water is different among ship types or according to the way they are used. More than half of the ocean-going vessels calling at Japanese ports are general cargo ships and container ships (The Japan Association of Marine Safety 1999) (Fig. 6). Although bulkers and tankers still account for about 20% of vessels, they do not discharge a lot of ballast water in Japan because they call with a full load. They only discharge a little ballast water from their after-peak tanks depending on the necessity of adjusting their trim to ensure even keel conditions before entering port.

We estimated the rate of discharge of ballast water by three ship types, such as PCCs (Pure Car Carriers), general cargo ships and container ships in Japan, based on 2002 data from the National Ballast Water Information Clearinghouse (Tab. 3). Although these data are for US ports, considering that the ballast water operations of these three ship types is the same in the US and Japan, it is possible to apply them to Japan.

It is estimated that less than 10% of general cargo ships and PCCs and less than 15% of container ships calling at Japanese ports discharge ballast water. The volume of discharged ballast water into Japanese ports is only about 5% of all the ballast water brought into Japan with these three ship types. This means that the ballast water that is discharged is likely to have been held inboard for a long period. As a general rule, abundance and species diversity of plankton decrease with the length of the confinement of the organisms in the tanks (Gollasch *et al.* 2000). This tendency has been documented (Chu *et al.* 1997, Gollasch *et al.* 2000, Wonham *et al.* 2001), leading to a conclusion that when ballast water is discharged, most of the organisms in it have already died, with the exceptions of diatoms, protozoa and some of the copepoda (e.g., Cohen *et al.* 2000, Chu *et al.* 1997) including the harpacticoida, *Tisbe graciloides*, which are known to increase their number (Gollasch *et al.* 2000). This may suggest the reason why introduction via

Table 3. The number of ships that either discharged or did not discharge ballast water and the amount of ballast water carried and discharged for six US ports by ship type in 2002

		General cargo ship	Container-ship	PCC (Pure Car Carrier)
A	Number of ships that entered with ballast water	282	2 218	126
B	Number of ships that discharged ballast water	18	316	12
	B/A (%)	6.4	14.2	9.5
C	Amount of ballast water carried (metric ton)	908 090	16 933 567	67 505
D	Amount of ballast water discharged (metric ton)	56 327	864 355	25 757
	D/C (%)	6.2	5.1	3.8

1. Followed Packard 1984 to classify ships' type
2. Selected ports are Long beach, Los Angeles, New York, Oakland, San Francisco, and Seattle.
3. Ships for which the type was unknown, or which entered without ballast water were omitted

(Based on data of National Ballast Water Information Clearinghouse) ballast water is of little importance to importers such as Japan.

THE MECHANISM OF INTRODUCTION VIA HULL FOULING

Minchin and Gollasch (2003) described six mechanisms for introduction by hull fouling. They are: (1) spawning and brood release from hulls, (2) detachment of mobile specimens from hulls, (3) colonisation by detachment of organisms from hulls, (4) dropping of fouling organisms by cleaning of hulls, (5) disposition of untreated wastes by the cleaning of hulls at boatyards, and (6) colonisation from hulls of wrecks.

Among these, it is considered that spawning and brood releases from hulls are the most important mechanism for colonisation (Minchin and Gollasch 2003). Lewis (2001) also pointed out that, since a single fertile fouling organism has the potential to release many thousands of eggs, spores or larvae into the water, each with the capacity to found new populations, hull fouling could play an important role for introductions. For example, more than 20 of the Onuphidae (Polychaeta), more than 50 of the European clam *Corbula gibba*, and one male and two ovigerous (= egg bearing) females of the European green crab *Carcinus maenas* were found in a sea chest (Coutts *et al.* 2003) and if these species spawn, considerable numbers of larvae will be released into the water. For example, a 46mm female of *Carcinus maenas* produced 185,000 eggs and released larvae simultaneously into the sea (Yamada 2001). The number of larvae potentially released by the two females in the example above, would amount to 370,000. It is considered that such larvae or eggs can be released through the cooling system of the generator while it is operating during anchorage. When these eggs or larvae are released into the water they may be damaged by the rise in temperature of cooling water or by the intake pump. The temperature of water taken into the cooling system goes up by nearly 10 °C before being discharged into the sea. Research on damage caused by the rise of water temperature for species such as *Acartia tonsa* (Copepoda), zoea of *Sesarma cinereum* (Grapsidae), and the hard clam *Meretrix lusoria* in the cooling system of a power station, shows that the likelihood of death through water temperature increase is low, except during summer when the temperature of the water discharged rises close to 40°C (Suisei Seibutsu to On-Haisui Kenkyu Kyougikai 1973, Dotsu and Kinoshita 1988). This means that almost all the larvae in the cooling system are likely to be released alive. With regard to mechanical damage, it is said that when organisms pass through impellers of the ballast pump, they may get damaged and die within a few days (e.g., Gollasch *et al.* 2000). But given the importance that ballast water plays in introduction in some regions, the mechanical damage caused by the ballast water impeller seems not to be a serious impediment to dispersal. The structure of the pumps used in the cooling system and in the ballast tank is the same, so organisms are likely to live in the cooling system after passing through the pump and likely to be released live into the water.

As in the example of *Carcinus maenas*, the number of larvae discharged from a hull structure like the sea chest is likely to be large. In Japan’s case, hull fouling is a much more significant vector, compared to ballast water.

Among the six mechanisms for introduction via hull fouling, there is no doubt of the importance of spawning and brood release as a mechanism for introduction. However, depending on circumstances, other mechanisms are also likely to be important. A survey to clarify the role of different mechanisms for introductions into Japan via hull fouling is needed.

RISK ASSESSMENT

It has been said that over 15,000 species are moved around the world every week in ballast water (Steneck and Carlton 2001). When we add to this the number of organisms that can be moved around the world attached to hulls, we reach a vast number. The situation would be very serious if all of them succeeded in establishing themselves in a new region. Fortunately, however, only a fraction of them survive in the new conditions (Steneck and Carlton 2001).

There are two important conditions that make introduction possible. The first is similarity of climate and marine conditions, such as salinity. Secondly, the volume of shipping traffic and geographical proximity are important (e.g., Gollasch 2002, Clarke *et al.* 2003). For example, in respect of climate, it is said that introductions tend to occur easily between both sides of the same ocean within the same hemisphere, such as between the Asian region and the west coast of North America facing each other across the Pacific Ocean, or between northern Europe and the east coast of North America facing each other across the Atlantic Ocean (e.g., Carlton 1987, Carlton and Geller 1993). Gollasch (2002) states that the risk of introduction from the same climate zone is the highest and that it reduces with the degree of difference of the climate between the donor and the receiver region. It can be seen also from the example of Japan that climate similarity must have facilitated introductions. Referring to the bioregions described by Hewitt *et al.* (eds.) (1999), there are many source bioregions where species introduced into Japanese waters originated. Among them, the North East Pacific and the East Asian Sea are the largest bioregions that are likely to be sources of introduced species in Japanese waters (Fig. 2). The North East Pacific and the North West Pacific are on nearly the same latitude and have a common climate zone with Japan from the subtropical zone to the boreal zone (Nishimura 1981). It is concluded that the commonality of climate between two regions facilitated success in Japanese waters of introductions resulting from the high frequency of shipping traffic (see Raaymakers & Gregory (eds.) 2002). The North Pacific route for introduction mentioned by Carlton (1987) reflects the climate similarity on both sides of the north Pacific Ocean as well as the high frequency of shipping traffic in this area.

The East Asian Sea, another important bioregion for introductions in Japan, overlaps the Indo-West Pacific Region (Nishimura 1981). Its characteristic climate is subtropical to tropical. There are few subtropical zone in Japan; the range from Inubo Saki in the south up to Kyushu belongs to the warm-temperate zone (Asakura 2003). In other words, the Japanese climate zone is largely to the north of this subtropical source region. Gollasch (2002) suggested that introductions can happen from outside the same climate zone, and this happened in the case of Japan. Probably, such introductions occurred because the climate conditions are still fairly similar, combined with the high frequency of shipping traffic between Japan and the East Asian Sea (see Raaymakers and Gregory (eds.) 2002).

Although there are only three reported introduced species from the North West Pacific (Fig. 2), the possibility of introductions from this zone must be consider as high, due to its proximity and to being in the same climate zone. The report of ballast water risk assessment carried out for Dalian in the Peoples Republic of China, (Clarke *et al.* 2003), considers that the risk of introduction from a nearby sea is the highest. Some Korean ports and Iwakuni port in Japan are ranked as “risky” ports with regards to introductions into Dalian port and other ports in China are considered to be “the most risky” ones for Dalian (Clarke *et al.* 2003). The risk depends on the high frequency of the shipping traffic, the climate similarity and the proximity between these ports and Dalian. The considerations for Dalian can also be applied to Japan. It can be said that the risk of introductions from Korea or China, including Dalian, to Japan is high. Although there are few introductions by shipping from these regions now, it would be best to accept that there will be more introductions from these regions with increased shipping traffic (see Raaymakers and Gregory (eds.) 2002). For Britain, it is known that most introduced species came from mainland Europe and that all of them are secondary introductions (Eno *et al.* 1997). It is hence possible that the introductions between regions that are close to each other, including Japan also include secondary introductions, and possibly such secondary introductions from nearby regions may outnumber introductions of species that are native in the nearby source-regions. Secondary introductions from nearby regions are likely to be a problem in Japan in the future.

MEASURES JAPAN SHOULD TAKE TO PREVENT OR REDUCE INTRODUCTIONS OF MARINE ORGANISMS

Japan is one of the biggest sources of ballast water to foreign countries. The volume of ballast water taken from Japan was about 318 million tons in 1997, accounting for more than 10% of all the ballast water discharged in the world in that year (Otani 2004). This means that Japan has to be concerned about the problem of introductions caused by ballast water and has to address, more than most other countries in the world, the management and control of ballast water. At the 1st East Asia Regional Workshop held in Beijing in 2002, Japan expressed a wish to consider measures, including legal aspects, to cope with the problem of ballast water following the convention adopted by the IMO in 2004 (Raaymakers and Gregory (eds.) 2002). However, considering that introduced species from Japan, like the starfish *Asterias amurensis*, have caused damage to the economy or ecosystems in other countries, such as Australia (e.g. Byrne 1996, Byrne *et al.* 1997), Japan should develop statutes and systems and develop treatment technology as fast as possible.

In this study, it was assumed that Japanese marine introductions were mainly caused by hull fouling, but this is not based on actual research data. So far, there have been no surveys of hull fouling carried out specifically focussed on introductions. Furthermore, there is no data about sea chest fouling. Actual surveys of hull fouling, including surveys of sea chests, are required. In particular, such surveys are urgently needed on ships that move back and forth between Japan and countries of the North East Pacific, the East Asian Sea, and nearby countries in the North West Pacific, because as known from the assessment in this study, countries in these bioregions are considered to be the main source regions for introduced marine species in Japan.

With respect to baseline surveys, there are only a few reports about the distribution of marine species in Japan. In particular, there is still insufficient information about the distribution of marine species in Tokyo Bay, Osaka Bay, and Ise Bay, where there are large ports (and where, hence, a large number of introduced species will be expected). This lack of data is a big obstacle to coping with the problem of marine introduction to Japan. It is therefore necessary to carry out, as soon as possible, a nationwide baseline survey, including these areas, to collect data about the presence and distribution of marine species in our waters. This information would enable us, in the future, to detect newly introduced species at an early stage and to take measures suitable for Japanese conditions to prevent or to reduce further marine introductions.

In addition to this, in order to prevent or reduce introductions by hull fouling, which is a major vector in Japan, it is necessary to: (1) develop effective and non-toxic antifouling paint technology and to recommend its use, (2) develop technology to minimise the translocation of organisms on ships' hulls in areas such as sea chests and other parts that are not painted with antifouling paint (these include DDSS (Dry Docking Support Strips), propeller and propeller shaft), (3) increase the frequency of ship dockings, to inspect and to clean hulls, and (4) regulate by statute or ban the underwater cleaning of ships' hulls. Because introductions are caused by international trade and exchange, it is impossible to address the problem in one country or region on its own. It is necessary to address this problem at an international level, just like problem of ballast water has already been addressed internationally.

ACKNOWLEDGEMENTS

I express sincere thanks to Dr. K. Iwasaki, Nara University, for useful comments and suggestions. Thanks are extended to Dr. C.L. Hewitt, New Zealand Ministry of Fisheries, for valuable suggestions about the way to analyse data, and Dr. F. McEnnulty, CSIRO Marine Laboratory and Mr. J.A. Lewis, Australian Government Department of Defence, for providing useful papers. I also thank the Bureau of Transport and Regional Economics for permitting the use of their data.

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Human-mediated introduction of marine organisms in Japan: a review

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Abstract The present status of human-mediated introductions of marine organisms in Japan is reviewed, based largely on the results of a questionnaire survey conducted in 2002–2003 by the Committee for the Preservation of the Natural Environment of the Japanese Association of Benthology. Taxa were classified according to criteria of known or unknown geographic origin, established invasion history, and presumed dispersal mechanisms associated with human activities. According to these criteria, 42 taxa were designated as introduced alien species, 26 taxa as species introduced from abroad (for fisheries, fishbait, or unintentionally) but where populations that are native to Japan also exist, 20 taxa as cryptogenic species, and 14 taxa as native species that were introduced domestically from an area where they were native in Japan to another area within Japan where they were not native. About half (22 spp.) of the alien species were introduced via shipping, and another half (19 spp.) for fisheries or by unintentional release with imported clams. The introduction rate of the 42 alien species has increased over the past century, with seven or eight species being introduced per decade after 1960. Several alien species have recently become widespread, from the Pacific coasts of central Japan to the Japan Sea coasts or northward, at a rate of 10–26 km per year^{-1}. The sites of the first records of alien species introduced via shipping were concentrated in Tokyo Bay and the eastern part of the Seto Inland Sea, and these were considered to have been the starting points for their dispersal within Japan. Impacts of several introduced species on native ecosystems, fisheries and other industries are also reviewed.

Keywords: alien species; human-mediated introduction; marine organisms; range expansion; rate of spread

INTRODUCTION

Human-mediated introduction of marine organisms beyond their native range has long been of great interest for ecologists and evolutionary biologists. Much information on many invasive marine organisms has been steadily accumulated for the development of risk assessments and management of marine invasions. Since the 1980s, introduced marine animals and plants have been reported in several countries, sea areas or continents (Pacific Ocean: Carlton 1987, Williamson *et al*. 2002, Hong Kong: Morton 1987, Hawaii: Coles *et al*. 1999, Australia: Hutchings *et al*. 1983, New Zealand: Cranfield *et al*. 1998, North America: Ruiz *et al*. 2000, Cohen and Carlton 1995, Europe: Leppäkoski *et al*. 2002).

In Japan, several authors have reported on regional fauna of introduced marine animal species (Tokyo Bay: Asakura 1992, Kajihara 1996, Furota 1997, 2001, 2002, Osaka Bay: Nabeshima 2002), their invasion history and distribution of introduced sessile animals (Arakawa 1980, Otani 2002), and the presumed vectors of 25 introduced marine organisms (Otani 2004). However, all these studies have not applied criteria to judge whether the species were introduced or not. Recent taxonomic rearrangements or confusion over some species reported in these papers have suggested that some of the species reported in the past as introduced might not be so. Such taxonomic problems, and insufficient survey records in the past for introduced marine organisms, have made it difficult to decide if the species are native or introduced (Carlton 1996). The application of standard criteria is essential to judge demonstrably human-mediated introduction of marine organisms. However, there have been no such systematic studies on a nationwide scale in Japan, and very few throughout the world (Chapman and Carlton 1991, 1994, Ruiz *et al*. 2000).

In 2002 and 2003, the Committee for the Preservation of the Natural Environment of the Japanese Association of Benthology (CPNE), carried out a questionnaire survey on the occurrence of introduced marine organisms in the field, including both published and unpublished records (Iwasaki *et al*. 2004a). The results obtained from the survey have been analysed by the committee, and the invasion history, geographic distribution and rate of range extension of introduced species in Japan have been published by Iwasaki *et al*. (2004a, 2004b) and Kimura *et al*. (2004). The present paper reviews these studies and provides an overview of human-mediated introduction of marine organisms in Japan.

Pages 104-112. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

METHODS OF THE QUESTIONNAIRE SURVEY

In 2002 and 2003, the CPNE sent a questionnaire, by e-mail or post to about 150 members of the Japanese Society of Benthology, the Sessile Organisms Society of Japan, the Malacological Society of Japan, and the Plankton Society of Japan, asking for the date and site of records for marine organisms considered to be introduced by human activities. Additionally, we asked the members to detail any documents or publications which have reported the occurrence of an introduced species in Japan. As a result of this survey, 94 respondents reported a total of 102 taxa.

Criteria for assessing the invasion and population status

Iwasaki *et al*. (2004a) assigned the 102 taxa to one of three categories of invasion status: introduced species, cryptogenic species (*sensu* Carlton 1996), and native species, with a set of criteria described below. Additionally, the report further classified the introduced species as one of the following (see Fig. 1):

a) Alien species introduced to Japan from abroad. The species does not have native populations in Japan.
b) Species introduced from abroad, but which also has populations that are native to Japan. The species hence has native populations both in Japan and abroad. The introductions took place for fisheries, fishbait or unintentional.
c) Domestically introduced: a species that is native in Japan but that has been introduced (= human induced movement) to another area in Japan where it is not native.

The criteria used to assign one of the three categories of invasion status were:

(1) Introduced species
  1) The species is not recognised as native in an area, and
  2) The distinction between the native and introduced range is known or inferred, and
  3) Vectors for the species to the area can be confirmed or inferred.
  4) For species that also have native populations in Japan, we assigned the invasion categories “introduced from abroad” or “introduced domestically” when their vectors for introduction to the new area from their native region were confirmed or inferred.

(2) Cryptogenic species (possible introductions: *sensu* Carlton 1996)
  1) The species is not recognised as native in an area, but
  2) The above criteria of (1)-2) and (1)-3) do not apply to the species, or
  3) The species scientific identity cannot be established due to taxonomic problems or confusion.

(3) Native species
  1) The species whose native range is well established and where clear evidence of native status is available.

Iwasaki *et al.* (2004a) classified the population status of each non-indigenous species as either established

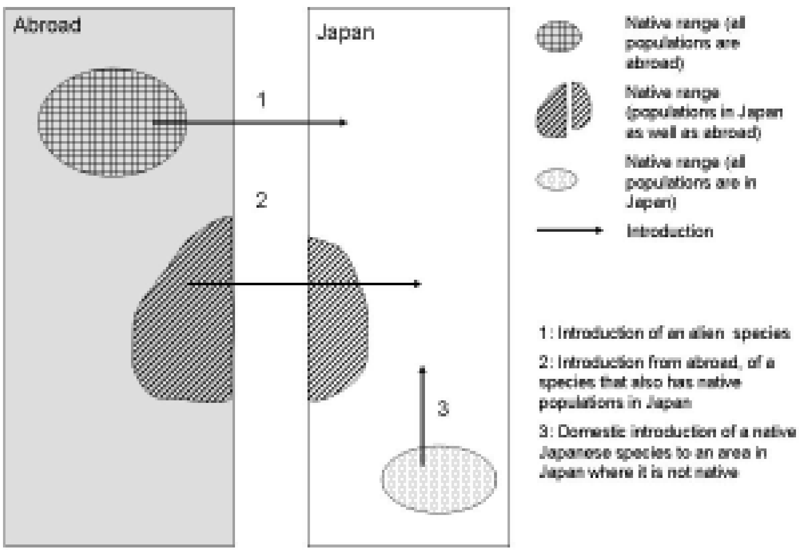

Figure 1 Further classification of introduced species (see text).

or unknown, according to the following criteria:

(1) Established

1) The species' occurrence has been confirmed for two or more years in at least one prefecture since 1990, and

2) Occurrence of breeding individuals has been confirmed in the field since 1970, or the occurrence of two or more cohorts, with very different size classes, has been confirmed for two or more successive years in one area since 1970.

(2) Unknown

1) Species other than (1) above.

RESULTS FROM QUESTIONNAIRE SURVEY

Alien species introduced into Japan

A total of 42 species were designated as alien species by Iwasaki *et al.* (2004a) (Tab. 1 and 2). Twenty-two species were presumably transferred through shipping (Tab. 1), and Otani (2004, 2006) considers that the most plausible vector for most of the species was fouling on ship hulls. Establishment of populations of four species, the nudibranch *Cuthona perca*, the barnacles *Balanus variegatus cirratus* and *Balanus venustus*, and the crab *Callinectes sapidus*, could not be confirmed by Iwasaki *et al.* (2004a) (Tab. 1).

Nineteen taxa were introduced intentionally for fisheries (16 species), or unintentionally with the imported aquatic products (3 spp.) (Tab. 2). The snail *Nassarius* (*Zeuxis*) *sinarus*, which is considered to have been introduced unintentionally with bivalves imported from the Korean Peninsula, established its population in the Ariake Inlet, the largest inlet in Japan. The bivalves, *Corbicula* spp., have been imported abundantly from China and Korea, established their populations in many estuaries and rivers, and now expand their range rapidly in many prefectures. The population status of 16 other species is unknown due to the scarcity of information (Iwasaki *et al.* 2004a) (Tab. 2).

The green alga *Caulerpa taxifolia* is native to subtropical regions of Japan. However, the Mediterranean-adapted clones probably escaped from aquaria were found in temperate regions in 1992,

Table 1 Alien marine organisms introduced probably via shipping, modified from Iwasaki *et al.* (2004a). Population status; E: established, U: unknown. First record; HO: Hokkaido island, JS: Japan Sea, PO: Pacific Ocean, SIS: Seto Inland Sea, ECS: East China Sea including Ariake Inlet, SWI: South West Islands (For locations, see Fig. 3). **F**: year of first record in Japan, +: year unknown. Source region and presumed vector for each species are listed in Otani (2004, 2006).

Species	Population status	First record					
		HO	JS	PO	SIS	ECS	SWI
Gastropoda							
Crepidula onyx	E	2001	2000	**F1968**	1978	1988	
Cuthona perca	U			**F1992**			
Bivalvia							
Mytilus galloprovincialis	E	1995	1941	1935	**F1932**	1950	
Perna viridis	E		1992	1980	**F1967**	2000	1983
Xenostrobus securis	E		1986	1979	**F1972**	2003	
Mytilopsis sallei	E		1984	**F1974**	1990		
Petricola sp. cf. *lithophaga*	E			1989	**F1985**		
Mercenaria mercenaria	U			**F1998**			
Polychaeta							
Ficopomatus enigmaticus	E		1990s	1969	**F1966**		1980
Hydroides elegans	E		1983	**F1936**	1962	1950	1970s
Crustacea							
Balanus amphitrite	E	1963	1963	**F1935**	1938	1937	
Balanus variegates cirratus	U		1937		1963	**F1936**	
Balanus venustus	U		**F1967**				
Balanus eburneus	E		1963	**F1950**	1963	1963	
Balanus improvisus	E		1967	**F1952**	1962	1963	
Balanus glandula	E	2000		**F2000**			
Pyromaia tuberculata	E		1982	**F1970**	1970s		
Carcinus aestuarii	E		1996	**F1984**	1996		
Callinectes sapidus	U			**F1975**	1984		
Ascidiacea							
Polyandrocarpa zorritensis	E			**F1991**	1999	**F1991**	
Molgula manhattensis	E		1992	1975	**F1972**		
Phaeophyta							
Cutleria multifida	E			+	+		F1957

Table 2 Alien marine organisms introduced intentionally for fisheries (Fisheries) or aquarium industry (Aquarium), or unintentionally with the aquatic products (Unintentional), modified from Iwasaki *et al.* (2004a).

Species	Vector	First record	Population status
Gastropoda			
Haliotis rufescens	Fisheries	1966	Unknown
Haliotis kamtschatkana	Fisheries	1980s	Unknown
Haliotis tuberculata	Fisheries	1980s	Unknown
Stenothyra sp.	Unintentional	2000	Unknown
Nassarius sinarus	Unintentional	2000	Established
Bivalvia			
Ostrea edulis	Fisheries	1952	Unknown
Ostrea lurida	Fisheries	1948	Unknown
Crassostrea virginica	Fisheries	1956	Unknown
Corbicula sp.	Fisheries	1987?	Established
Phacosoma gibba	Unintentional	2002	Unknown
Meretrix petechialis	Fisheries	1969	Unknown
Crustacea			
Penaeus chinensis	Fisheries	1965	Unknown
Homarus americanus	Fisheries	1914	Unknown
Homarus gammarus	Fisheries	1978	Unknown
Eriocheir sinensis	Fisheries	1999?	Unknown
Osteichthyes			
Acipenser sinensi	Fisheries	1965	Unknown
Acipenser sturio	Fisheries	1975	Unknown
Salmo gairdneri	Fisheries	1929	Unknown
Salmo salar	Fisheries	1980	Unknown
Chlorophyta			
*Caulerpa taxifolia**	Aquarium	1992	Unknown

*: Mediterranean-adapted clones

1993 and 1994 (Iwasaki *et al.* 2004a). The population status of these is unknown (Tab. 2). We assigned this Mediterranean-adapted clones as "introduced species" because their ecology, physiology, morphology and potential impacts on native ecosystems are quite different from those of native populations.

Rate of introduction and site of first record for alien species

Analysis of the years of the first record for 42 alien species suggests that the rate of introduction has increased over the past century, with seven or eight species being introduced per decade after 1960 (Fig. 2) (Iwasaki *et al.* 2004a).

The sites of the first records for 42 alien species, which were reported by Iwasaki *et al.* 2004a, are shown in Fig. 3. Most species were recorded first along the coast of the Pacific Ocean or the Seto Inland Sea, reflecting the quantity and concentration of foreign trade goods in the regions. More than half of the 22 species introduced via shipping were first found in Tokyo Bay (6 spp.) or in the eastern part of the Seto Inland Sea including Osaka Bay (6 spp.), where the large ports are concentrated. Accordingly,

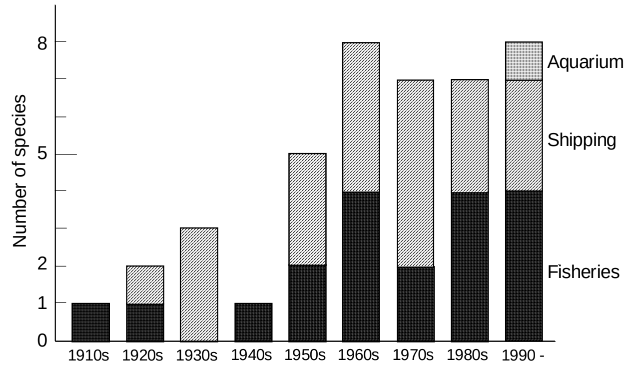

Figure 2 Years of the first records for 42 alien species introduced into Japan, modified from Iwasaki *et al.* (2004a). Probable vectors via which the species were introduced ("fisheries" for release or aquaculture, "shipping" in hull fouling or ballast water transport, and "aquarium" industry) are shown in different shadings.

the establishment of monitoring systems at the large ports in these regions is essential to detect new alien species at the early stage of their introduction via shipping.

The sites of first records for species introduced for fisheries are distributed widely in Japan. However, most intentional introductions were conducted by national or prefectural institutes for fisheries science and the sites of first records are concentrated around these institutes.

Rate of spread of several alien species

Data on temporal change in geographic distributions revealed that many alien species have become widespread recently, from the Pacific coasts of central Japan to the coasts of the Japan Sea or northward (Tab. 1) (Iwasaki *et al.* 2004a). The rate of spread for 8 alien species which had over 50 records for their occurrence in the field was calculated through regression analyses of the farthest distances of the recorded sites from the sites of first records against the time after the year of first record (Iwasaki *et al.* 2004b). All 8 species are considered to have been introduced via shipping. Five of the 8 showed a significant correlation between the greatest distance of spread in each year and time after the first record. Their average rate of spread ranged from 10 to 26km year^{-1} (Tab. 3), 26.4km year^{-1} for the slipper snail *Crepidula onyx*, 10.9km year^{-1} for the Mediterranean mussel *Mytilus galloprovincialis*, 23.9 km year^{-1} for the mytilid mussel *Xenostrobus securis*, 13.9km year^{-1} for the European barnacle *Balanus improvisus*, and 24.7km year^{-1} for the Mediterranean green crab *Carcinus aestuarii*. The distance-versus-time curves for the five species showed no saturation phase during which no

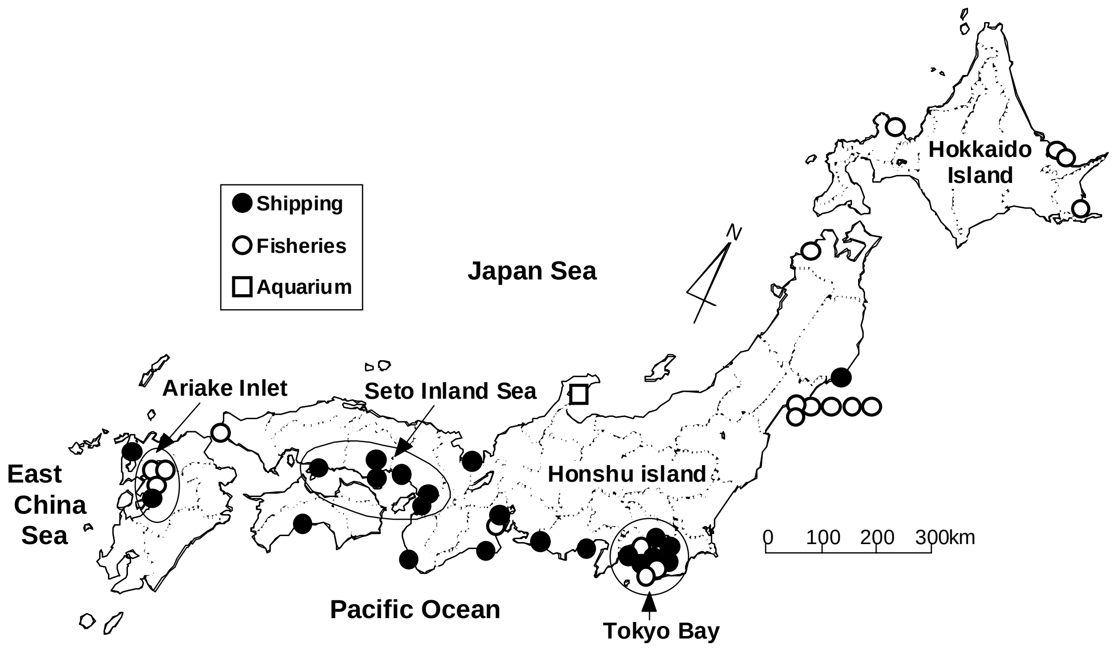

Figure 3 Sites of first records for 42 alien species introduced into Japan through fisheries for release or aquaculture (open circle), shipping (either hull fouling or ballast water) (closed circle), or aquarium industry (open rectangle).

range expansion takes place in the final stage of invasion. Thus their geographic ranges were considered to be expanding still. For the other three species, range expansion for the two barnacles, *Balanus amphitrite* and *B. eburneus*, was considered to have occurred only in the early stage of invasion and to have ceased by 10–15 years after the date of the first record. Information on the geographic distribution for the remaining species, the serpulid polychaete *Hydroides elegans*, was not sufficient to examine the pattern of its range expansion.

The sites of the first records for unintentionally introduced species were considered to have been the starting points for their spread in Japan (Iwasaki *et al.* 2004b). We suggest that eradication or control of the initially established populations as soon as they are discovered is essential to prevent the spread of introduced marine organisms in Japan.

Table 3 Long term trend of the range expansion of 8 non-indigenous species was examined by linear regression (y = a + bx) of the farthest distance from the site of first record (y) against the year (x) after first record (the year of first record = 0). The rate of spread is estimated by the slope (b) of the regression. n: number of samples (If there are multiple records in a year, only the record with the farthest distance from the site of first record was used. So the number of samples is smaller than 50 in several species.), R^2: coefficient of determination, a: Y-intercept, b: slope of regression, P: probability. After Iwasaki *et al.* (2004b).

Species	n	R^2	a	b	P
Gastropoda					
Crepidula onyx Sowerby	35	0.730	125.3	26.4	<0.001
Bivalvia					
Mytilus galloprovincialis Lamarck	58	0.215	497.1	10.9	<0.001
Xenostrobus securis (Lamarck)	32	0.677	14.2	23.9	<0.001
Polychaeta					
Hydroides elegans (Haswel)	31	0.068	271.1	3.9	0.158
Crustacea					
Balanus amphitrite Darwin	53	0.046	835.1	-5.2	0.121
Balanus eburneus Gould	38	0.032	924.5	7.3	0.279
Balanus improvisus Darwin	44	0.295	223.1	13.9	<0.001
Carcinus aestuarii Nardo	14	0.329	-54.7	24.7	0.032

Species introduced from abroad, but for which native Japanese populations also exist

Iwasaki *et al*. (2004a) designated twenty six taxa as species which have been introduced from abroad to Japan for fisheries or as fishbait (Gastropoda: 9 spp., Bivalvia: 10 spp., Brachiopoda: 1sp., Polychaeta: 3spp., Crustacea: 1sp., Osteichthyes: 2 spp.). All introductions were from China or Korea, intentionally for fisheries, or unintentionally with imported aquatic products (Tab. 4).

Yokogawa (1997) reported morphological and genetic differences between the introduced Chinese populations of the red arch shell *Scapharca broughtonii* and the Japanese ones. Introduction into Japan of different populations may disturb the genetic diversity of native populations through inbreeding. Before such introduction from abroad is considered, special caution and scientific research is needed to ensure the

Table 4 Introduced species from abroad but where native Japanese populations also exist: introductions for potential release as human food (Release) or fish bait (Fish bait), and unintentionally with imported aquatic products (Unintentional), modified from Iwasaki *et al.* (2004a).

Species name	Vector
Gastropoda	
Umbonium moniliferum	Unintentional
Batillaria cumingii	Unintentional
Euspira fortunei	Unintentional
Glossaulax didyma	Unintentional
Glossaulax reiniana	Unintentional
Rapana venosa	Unintentional
Reticunassa festiva	Unintentional
Varicinassa varicifera	Unintentional
Bivalvia	
Scapharca broughtonii	Release
Scapharca kagoshimensis	Unintentional
Crassostrea gigas	Unintentional
Mactra chinensis	Unintentional
Mactra veneriformis	Unintentional
Macoma contaculata	Unintentional
Sinonovacula constricta	Release
Ruditapes philippinarum	Release
Cyclina sinensis	Unintentional
Brachiopoda	
Lingula unguis	Unintentional
Polychaeta	
Perinereis nuntia	Fish bait
Perinereis aibuhitensis	Fish bait
Marphysa sanguinea	Fish bait
Crustacea	
Philya pisum	Unintentional
Osteichthyes	
Oncorhynchus kisutch	Release
Oncorhynchus tshawytscha	Release

conservation of native genetic resources (ICES 1995), however, no such measures have ever been taken in Japan.

"Domestic introduction" of Japanese native species

Fourteen taxa were designated as species native to Japan but introduced (= human induced movement) within Japan to regions where they are not native (Gastropoda: 3 spp., Bivalvia: 5 spp., Polychaeta: 1 sp., Crustacea: 3 spp., Echinoidea: 1 sp., Rhodophyta: 1 sp.) (Iwasaki *et al.* 2004a). The vector for about 80% of them is considered to be intentional and unintentional releases for fisheries. However, the list of such species introduced to areas where they are not native through fisheries (Iwasaki *et al.* 2004a) is not exhaustive. It is likely that many such species were introduced to non-native areas in Japan since the late 19th century (Murakami 1999).

Cryptogenic species

Twenty taxa were considered to be cryptogenic species which can not be recognised as either native or introduced (Gastropoda: 3 spp., Bivalvia: 3 spp., Bryozoa: 2 spp., Polychaeta: 2 spp., Crustacea: 3 spp., Ascidiacea: 1 sp., Osteichthyes: 2 spp., Dinophyceae: 2 spp., Rhodophyta: 1 sp., Chlorophyta: 1 sp.) (Iwasaki *et al.* 2004a). This is largely due to taxonomic problems in which current species names are invalid or to the scarcity of information on geographic distribution, invasion history or presumed invasion vectors.

IMPACTS OF INTRODUCED SPECIES

Genetic disturbance through hybridisation

Inoue *et al.* (1997) and Rawson *et al.* (1999) reported that genetic mixing between the Mediterranean mussel *M. galloprovincialis* and the native mussel *M. trossulus* was occurring on Hokkaido Island, the most northern part of Japan, suggesting hybridisation between the two species. The possibility of genetic disturbance through hybridisation or introgression has been pointed out in the case of the Chinese mitten crab *Eriocheir sinensis* and the native mitten crab *E. japonica* (Kobayashi 2003), the Chinese hard clam *Meretrix petechialis* and the native hard clam *M. rusoria* endemic to Japan (Kosuge 2002), alien and native Colbiculid bivalves (Komaru 2002), and the introduced (from abroad) and native populations of species native to Japan (Yokogawa 1997), although there has been no research so far to confirm this.

Exclusion and predation of native species

In the Tohoku District, northern part of Honshu Island, the Mediterranean mussel *M. galloprovincialis* has covered and out-competed native species, such as the barnacle *Chthamalus challengeri*, the oyster *Crassostrea gigas*, the mytilid bivalve *Septifer virgatus*, and the brown alga *Hizikia fusiforme* in the lower intertidal zones (Hoshiai 1958, 1960, 1961, 1964, 1965). It is believed that rocky intertidal communities of sheltered shores in Honshu, Shikoku and Kyusyu Islands have changed drastically after the invasion of this species.

The striped barnacle *Balanus amphitrite*, which was introduced probably in the 1930s, extended its geographical range in 1960s and 1970s over almost all of Honshu Island. Its predominance on the estuarine hard substrata is suggested to have drastically decreased the density of the native barnacle *Balanus reticulatus* (Yamaguchi 1989).

Until the mid-1990s, the native moon snail *Euspira fortunei* occurred only in the Ariake Inlet with very low density and it has been considered endangered in Japan. Since the mid-1990s, however, populations of this species have been introduced from abroad to regions in Japan where the species is not native. These introductions were unintentional, with the edible clam *Ruditapes phillipinarum* imported from China and Korea. Outbreaks of the populations introduced from abroad occurred on several locations on the shores of Honshu Island, and the carnivorous snails predated clam and other native bivalves, presumably causing drastic changes in the species composition of native sand-mud flat communities (Okoshi 2004).

Economic damage to fisheries

An outbreak of the serpulid polychaete *Hydroides elegans* in Hiroshima Bay has caused the heavy economic loss to cultured oyster crops through fouling on their shells, estimated at ¥3 billion in 1969 (equivalent to ca ¥10 billion today) (Arakawa 1971). Removal of the calcareous tubes from the shells of cultured oysters and pearl oysters is recognised as a great nuisance to oyster farmers.

The economic damage caused by *M. galloprovincialis* to the aquaculture of oysters, pearl oysters and scallops has been reported many times. For instance, an outbreak of this species near Hiroshima Bay in 1973 caused serious economic damage to cultured oyster crops, with a loss estimated at ¥500 million (equivalent to ca ¥1.5 billion today) (Arakawa 1974).

The carnivorous snail *Nassarius sinarus*, which was presumably introduced from Korea with imported bivalves, had an outbreak in the Ariake Inlet and predated gobies in the fishing nets (Fukuda 2004).

The above-mentioned introduced moon snail *Euspira fortunei* caused collapse of local clam fisheries in several shores from 2000 to 2004 (Okoshi 2004). Local fisheries cooperative associations tried to exterminate the populations that were introduced from abroad, but failed.

Economic damage to power plants and other factories

Mytilus galloprovincialis and the green mussel *Perna viridis* are considered to be the first and second worst fouling organisms of intakes of power stations and other factories on the Pacific coasts, causing the greatest amounts of serious damage to equipment and resulting in enormous costs of removing the mussel beds (Anon. 2003). Although no economic losses attributable to fouling mussels have been estimated, it is believed that the costs would have been reduced by more than half without the invasion of the mussels (Kajihara 1983). Other alien species such as the Serpulid polychaete *H. elegans*, the striped barnacle *B. amphitrite*, the European barnacle *B. improvisus*, and the ivory barnacle *B. eburneus* are also known to cause fouling damage to power stations (Anon. 2003).

Management and policy against marine invasion in Japan

To date, no Japanese official agencies or private sectors have taken effective measures to control or manage the introduction of marine organisms, or to promote public awareness (Williamson *et al*. 2002). The Invasive Alien Species Act, which was enforced in June 2005 in Japan, designates Invasive Alien Species and prohibits them from being raised, imported or otherwise handled. Surprisingly, however, no marine organisms are designated as Invasive Alien Species in this Act. Scientists should collect the information on ecological impacts of invasive marine species and keep appealing to the authorities to prevent their introduction.

Far East Asia, including Japan, is well known as one of the major donor regions of introduced marine organisms (Carlton 1987). Immediate official initiatives in cooperation with other countries are essential to prevent or reduce human-mediated introductions to Japan as well as from Japan to other countries.

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Short paper

The arrival, establishment and integration of an invasive alien marine mussel into foreign ecosystems

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Biological invasions have rapidly emerged as one of the great threats to the integrity of natural systems, and alien invaders are now found in most of the earth's habitats. These species can bring environmental and economic damage through their direct impacts, but in a broader sense, this mass movement of species is breaking down natural biogeographic barriers and homogenising the earth's biota (Lockwood and McKinney 2001, Crooks and Suarez in press). This is exemplified by examining the coastal marine waters on both sides of the North Pacific. The bays and estuaries of California have been heavily invaded, with over 160 recognised alien species in San Francisco Bay (Cohen and Carlton 1995) and over 80 in San Diego County. Initial investigations on the extent of biological invasion in Japan have revealed over 40 alien marine species (Iwasaki 2006, Kado and Nanba 2006, Otani 2006) . This rampant translocation of species has resulted in a dramatic increase in the species now shared between California and Japan. There are many species native to Japan now found in California (e.g., the mussel *Musculista senhousia*, the fish *Acanthogobius flavimanus*, and the alga *Sargassum muticum*), as well as species native to California now established in Japan (e.g., the crab *Pyromaia tuberculata* and the barnacle *Balanus glandula*). There is also a suite of established alien species not native to either area (e.g., the mussel *Mytilus galloprovincialis* and the barnacle *Balanus amphitrite*).

While examining patterns of invasion such as these are informative, additional mechanistic understanding can be obtained by looking at the actual process of invasion. A biological invasion is a complicated series of events, involving many interacting factors related to the invader, the vector, and the source and recipient ecosystems. However, an invasion can be conveniently broken down into three phases: arrival, establishment, and ecological integration (Vermeij 1996, Crooks 2002c). These phases are characterised by different ecological processes and successful invaders must pass through each phase. Examination of invasion by a model invader, the marine mussel *Musculista senhousia*, identifies some of the important factors operating during biological invasions.

In terms of the arrival phase of invasion, *Musculista* is a well-travelled species, having spread from in its native range in the western Pacific and Indian Oceans to the Mediterranean, the eastern Pacific, and Australasia (Crooks 1996). Part of its success relates to its ability to be transported both as sedentary adults and planktonic larvae, as well as its use of multiple vectors of invasion, including canals, ballast water, ship fouling, and with the movement of commercially important oysters. *Musculista* also possesses traits that allow it to rapidly establish itself, particularly in disturbed or degraded environments. It is a classic invasive species, with a short life-span, high mortality, small size, and large reproductive potential (Crooks 1996). Once it has invaded, the integration of *Musculista* into an ecosystem can affect resident biota in several significant ways. Its most substantial effects result from its ability to modify invaded soft-sediment habitats through the construction of dense mats. These thick carpets of mussels on mudflats and the subtidal seafloor can dramatically inhibit native species such as clams and eelgrass (Reusch and Williams 1998, Crooks 2002b). However, smaller organisms able to live within the mat matrix are facilitated by the mussel's presence (Crooks 1998, Crooks and Khim 1999). In addition to its role in modifying habitats, the mussel also serves as a food source for native species and is host to intermediate life stages of trematode flatworms (Crooks 2000).

As well as highlighting the value of considering the invasion process in relatively discrete stages, examination of the worldwide invasion of *Musculista senhousia* also emphasises several topics of general importance to invasion biology. First, the temporal pattern of *Musculista* invasion indicates that prolonged time lags might exist during invasions. Such lags might occur during any phase of the invasion process (Crooks 2005). For example, invaders might exhibit long lags before taking advantage of a particular invasion vector, exist for long periods in low numbers within invaded systems before sudden population explosions, or have effects that are slow to manifest themselves (Kowarik 1995, Crooks and Soulé 1999). Any such lag will hamper predictive ability.

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The second general topic is that the factors that make ecosystems vulnerable to invasion are typically difficult to identify, and are often confounded with the supply of potential invaders to the system (Lonsdale 1999). For example, it is difficult to know whether areas such as urban lots, roadsides, and ports are more invaded because they are intrinsically more vulnerable, or whether they are subjected to larger numbers of potential invaders (i.e., experience higher propagule pressure). Experimental approaches can offer some ability to distinguish between the ecosystem properties that influence invasibility and the role of invader supply, and recent work in San Francisco Bay indicates that habitat quality alone can influence invader success (Crooks *et al.* unpub. data).

Third, as is evident with *Musculista*, some of the most profound effects of invaders result from the physical modification of habitats. Despite the potentially massive effects of invaders that alter habitats, the role of invasive ecosystem engineers has only recently begun to receive increasing attention (Crooks 2002a, Cuddington and Hastings 2004), reflecting a trend in ecology as a whole (Jones *et al.* 1994, 1997). A first examination of invasive ecosystem engineers as a group indicates that exotics that modify habitat complexity may have some relatively predictable effects on resident species (Crooks 2002a). In general, aliens that increase complexity tend to have "positive" effects on species abundance and/or diversity. For example, invasive trees that increase habitat heterogeneity (Hanowski *et al.* 1997) or seagrasses that form meadows on formerly bare tidal flats (Posey 1988) can in fact facilitate suites of resident fauna. Those aliens that decrease complexity, however, tend to have the reverse effect, as seen with grazing herbivores that destroy vegetative structure on islands (van Vuren and Coblentz 1987). Continued study of invasive engineers, and alien species in general, will increase our knowledge about the processes, consequences, and potential management of invasions, and will also contribute to our general understanding of the role of species in ecosystems.

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Differences in habitat use of the native raccoon dog (*Nyctereutes procyonoides albus*) and the invasive alien raccoon (*Procyon lotor*) in the Nopporo Natural Forest Park, Hokkaido, Japan

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Abstract We investigated differences in home range sizes and in the pattern of habitat use between the raccoon (*Procyon lotor*), an invasive alien species, and the native raccoon dog (*Nyctereutes procyonoides albus*), which lived sympatrically in Nopporo Natural Forest Park, Hokkaido, Japan. The mean home range size of raccoons (±2SD) was 116.2ha (±203.8ha), and that of raccoon dogs was 125.2ha (±71.1ha). The mean size of core areas was 12.1ha (±18.4ha) for raccoons, and 10.6ha (±12.8ha) for raccoon dogs. There were no significant differences between these two species in home range size (U-test, $p=0.571$) or in core areas (U-test, $p=0.571$). The pattern of habitat use, however, differed between these two species. Diurnal resting sites were located in woodland margins for raccoons, but in woodland for raccoon dogs. A comparison of the utilisation rates of farm fields in the home range showed that raccoons used farm fields significantly (U-test, $p<0.05$) more than raccoon dogs. This result supports that in Japan raccoons use farmer's properties for feeding and breeding.

Keywords: raccoon; raccoon dog; pattern of habitat use; invasive alien species; home range; radio-tracking

INTRODUCTION

The irresponsible breeding and release of pet raccoons (*Procyon lotor* Linnaeus, 1758) in Japan, coupled with their rapid naturalisation, has caused this species to pose problems to native habitats and biodiversity (Ikeda 2000). Raccoons naturalised 28 years ago in Hokkaido and dispersed to more than half of its municipalities. Now, their number is estimated to be over three thousand in the central part of the island (Hokkaido Prefecture 2003). Severe damage to crops and predation on endangered species such as the Japanese crayfish (*Cambaroides japonicus*) have been reported (Hori and Matoba 2001, Ikeda *et al.* 2004).

The raccoon dog (*Nyctereutes procyonoides albus* Beard, 1904) is a native carnivore of Family Canidae in Japan, whose body size (Kinoshita and Yamamoto 1996, Kishimoto *et al.* 1998, Zeveloff 2002, Asano 2003b), breeding season (Ikeda 1983, Asano 2003a), seasonal activity patterns (Yamamoto 1993, Kurashima *et al.* 1998) and diet (Hori and Matoba 2001, Furukawa 2001, Kasuya 2001) are similar to those of the feral raccoon Small animals and fruits are major food sources for both species. The impact of raccoons on the native raccoon dogs is important and it is a frequently debated issue in nature conservation in Japan.

Raccoons are known to live sympatrically with raccoon dogs in Hokkaido, but information about the relationship between these two species is still incomplete. In the present study we report on the differences in the pattern of habitat use between these two sympatric nocturnal carnivores, during daytime and night-time, in Nopporo Natural Forest Park, Hokkaido, Japan.

METHODS

Study area

The study was conducted in the Nopporo Natural Forest Park, located 11 - 15km east of the City of Sapporo (43'25"N, 141'32"E), central Hokkaido, Japan. This is a 2,040ha sub-isolated forest (ca. 6×4km), with altitudes between 20 to 100m above sea level. The annual precipitation and the mean temperature are 899mm and 6.6ºC (-25.8ºC in February and up to 29.8ºC in August), respectively (Data from Japan Meteorological Agency). This forest is a mixed conifer-hardwood forest, dominated by *Abies sachalinensis*, *Acer mono*, *Quercus mongolica* var. *grosserrata*, and *Fraxinus mandshurica* var. *japonica* (Ishikawa 1989).

The presence of raccoons was first verified in this forest in 1992 (Kadosaki 1996). Subsequently, they have been suspected of serious agricultural damage and attacks on a colony of herons (*Ardea cinerea*) inside the forest (Ikeda 1999, Hokkaido Prefecture 2003). They are very attracted to crops like sweet corn, melons and strawberries. During the study period, only strawberries were ripening in and around the research fields.

Trapping and radio-collaring

Traps were set for 48 days in total, from 26 March to

Pages 116-121. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

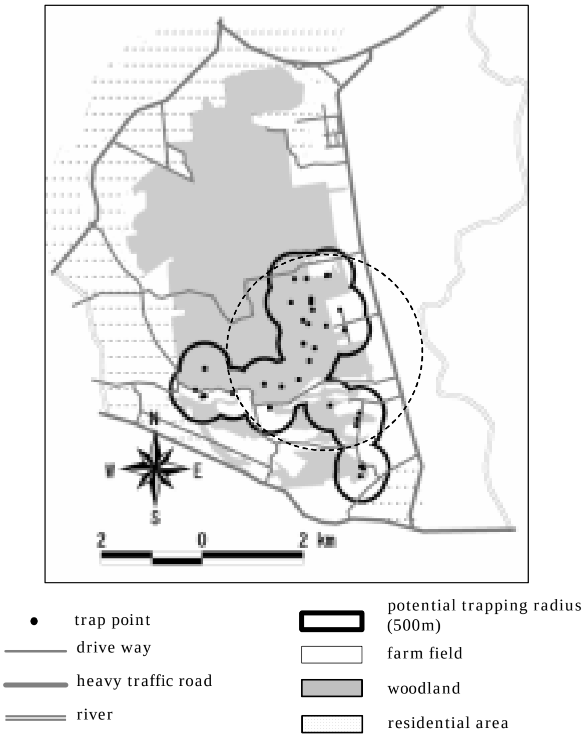

Figure 1 Nopporo Natural Forest Park and the adjacent areas. This park is wedged between the north-western residential area and the south-eastern farm fields. We set a maximum of 34 traps in the southern part of the park and estimated the potential trapping radius of each trap as 500m. The dashed line shows the research target area, where the home ranges of some raccoons and raccoon dogs were thought to overlap.

24 May 2003 (except for 23 - 29 April and 6, 7, and 14 - 16 May), using a maximum of 34 traps (25 days for each trap; 851 Trap-nights; Fig.1) in the southern part of Nopporo Natural Forest Park, where both species had been trapped most frequently during the previous year (Maesaki *et al.* 2002). Live cage traps (model #1089: 82Lx27Wx30H cm; ca 4.0kg; Havahart, Litiz, Pennsylvania, USA) were used, baited with dog food, corn snacks and doughnuts.

We set traps at intervals of about 500m, which was smaller than the raccoon's home range length reported by Kurashima and Niwase (1998). We estimated that this distance is the potential trapping radius for each trap (Fig. 1).

On their first capture, trapped raccoons and raccoon dogs weighing 3.5kg or more were immobilised with an intramuscular injection of ketamine hydorochloride (Ketaral, 1mg/kg), medetomidine hydrochloride (Domitor, 40µg/kg) and midazolam (Dormicam, 0.15mg/kg). Body mass, head-body length and tail length were measured and radio-collars were fitted (ca 120g, ATS, Inc., USA). After finishing these procedures, the reversal agent atipamezole hydrochloride (Antisedan, 40µg/kg) was administered, assisting recovery. Animals were released at the site of capture after approximately two hours, by which time they had fully recovered from the anaesthesia.

Juvenile individuals of less than 3.5kg and individuals suspected to be suffering from sarcoptic mange (*Sarcoptes scabie* Linnaeus, 1758) were not collared, for animal welfare reasons.

Radio-tracking

From June to July in 2003, we recorded telemetry locations for all radio-collared individuals once a day in the daytime, between 0700 - 1800. As both raccoons and raccoon dogs are nocturnal (Mech *et al.* 1966, Yamamoto 1993), we assumed that these location points represented their resting sites. We radio-tracked

Table 1 Radio-collared raccoons and raccoon dogs, both of which were captured in Nopporo Natural Forest Park, Hokkaido in May, 2003. All individuals in the table were radio-tracked during the daytime, and those with asterisks were also radio-tracked at night. The "head body" was larger estimation since we measured the length along the curve of neck.

Species	ID	Sex	Body length Head Body + tail length (cm)	Body mass (kg)	Radio-tracking at night
Raccoons (*Procyon lotor*)	r-f1	female	64.0 + 28.0	6.0	*
	r-f2	female	61.0 + 31.0	8.0	*
	r-f3	female	63.0 + 25.0	4.4	*
	r-f4	female	62.5 + 31.0	5.7	
	r-f5	female	64.0 + 23.5	6.4	
	r-f6	female	62.0 + 28.0	5.6	
	r-f7	female	54.0 + 24.0	4.3	
	r-f8	female	62.0 + 28.0	5.4	
	r-m1	male	60.0 + 26.0	4.2	*
Raccoon dogs (*Nyctereutes procyonoides albus*)	rd-f1	female	56.7 + 17.0	5.6	*
	rd-f2	female	51.0 + 16.0	4.3	*
	rd-f3	female	54.0 + 16.0	3.5	
	rd-f4	female	58.0 + 21.0	3.8	
	rd-m1	male	57.0 + 18.0	3.8	*

particular individuals whose nocturnal home ranges were assumed to overlap with each other (based on the fact that their resting sites overlapped). This radio-tracking at night was done every hour between 1800 - 0700, once per week.

Radio signals were detected with a hand-held four-element Yagi antenna and receiver (FT290, YAESU, Japan). Azimuths were triangulated from at least three positions to fix a location.

Data analysis

To examine only the nocturnal activities of raccoons and raccoon dogs, which live sympatrically, we compared location data of individuals monitored for more than three nights. Home ranges as well as core areas that each animal used frequently were calculated based on the whole nocturnal activity pattern for each individual. Their home ranges were estimated with the 100% Minimum convex polygon method and the core areas were estimated with the 50% Fixed Kernel method (Worton 1989), which calculated the utilisation distribution as a grid coverage using ad hoc calculation of a smoothing parameter. For these analyses, Animal Movement ver. 2.0 (Hooge and Eichenlaub 1997), and the add-in software of ArcView (Esri Inc., Redlands, California, USA) were used.

We examined the statistical significance of differences in size of home range, core area, and utilisation rates of farm fields, using Mann-Whitney's U-test ($p<0.05$).

RESULTS

Trapping and radio-tracking

Nine raccoons (one male and eight females) and nine raccoon dogs (one male, five females, and three of unknown sex) were trapped and all nine raccoons and five raccoon dogs (one male and four females) were radio-collared. One female raccoon dog weighing less than 3.5kg and three other raccoon dogs suspected to be suffering from sarcoptic mange were not collared.

Based on the telemetry location data for resting sites, we selected four raccoons (one male and three females) and three raccoon dogs (one male and two females), because they were assumed to have home ranges overlapping with the other species. The home ranges of other radio-collared individuals (five raccoons and one raccoon dog) did not have such overlapping at all (see details on Tab. 1 and Figs. 2 and 3).

We obtained 355 locations for the four raccoons and 247 locations for the three raccoon dogs for nocturnal activities. Radio transmitter signals stopped for one female raccoon and one female raccoon dog in July. Due to insufficiency of data, we therefore omitted the July data from our analyses.

Home range size

Fig. 4 shows the nocturnal activity patterns of four raccoons (one male and three females) in the early

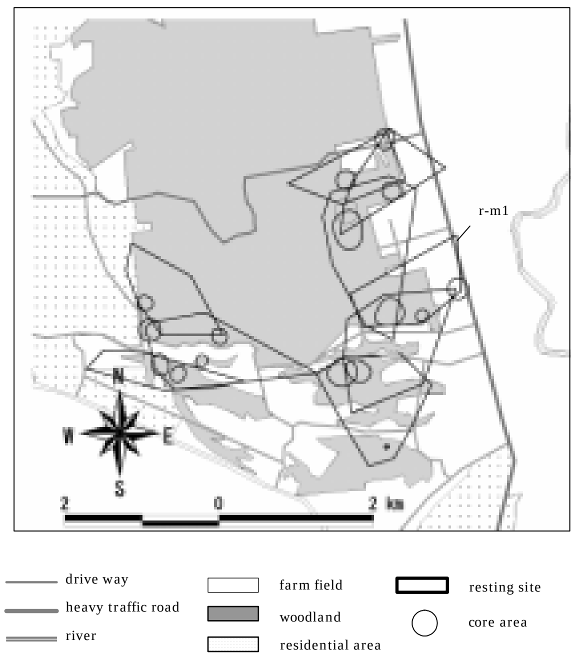

Figure 2 Resting sites and core areas of nine raccoons obtained from diurnal radio-tracking in Nopporo Natural Forest Park, in June and July 2003. r-m1 represents a male raccoon.

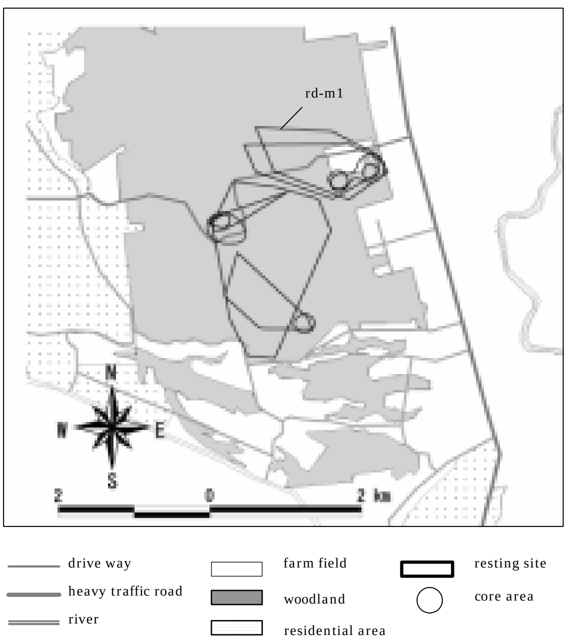

Figure 3 Resting sites and core areas of five raccoon-dogs obtained from diurnal radio-tracking in Nopporo Natural Forest Park, in June and July 2003. rd-m1 represents a male raccoon-dog.

summer of 2003, while Fig. 5 shows those of three raccoon dogs (one male and two females) for the same duration. The mean nocturnal home range size of the raccoons (±2SD) was 116.2ha (±203.8ha), whereas that of the raccoon dogs was 125.2ha (±71.1ha). The mean size of nocturnal core areas was 12.1ha (±18.4ha) for the raccoons and 10.6ha (±12.8ha) for the raccoon dogs. There were no significant differences between the two species in either home range size (U-test, $p=0.571$) and in the size of core areas (U-test, $p=0.571$).

The diurnal resting sites and core areas indicated den sites and breeding nests. Both species had plural resting sites, but the number of core areas was different between raccoons and raccoon dogs. While seven out of eight raccoons had several core areas, three out of four raccoon dogs had a single diurnal core area during the study periods (Figs. 2 and 3).

Habitat use

The pattern of habitat use differed between these two species. Resting sites and core areas of raccoons in the daytime were mostly located on woodland margins, except for one male raccoon (r-m1) occupying a farmer's barn in July of 2003 (Fig. 2). By contrast, all of the resting sites of raccoon dogs were located within woodland areas (Fig. 3).

In nocturnal activities, raccoons went out of the forest frequently and sometimes crossed a road with heavy traffic (Fig. 4). Three of them had multiple core areas both inside and outside of the woodland. Raccoon dogs, however, stayed inside the forest and rarely used farm fields during the study periods (Fig. 5). Raccoons used the farm fields significantly (U-test, $p<0.05$) more than raccoon dogs (see details in Tab. 2).

DISCUSSION

This study revealed different habitat use between raccoons and raccoon dogs despite their sympatric co-existence. Both raccoons and raccoon dogs had several daytime resting sites, but raccoons seemed to be less fixed to a particular den site than raccoon dogs. It has previously been reported that raccoons frequently shift diurnal resting sites, particularly when they are using dens on the ground during summer and autumn (Mech *et al.* 1966, Shirer and Fitch 1970). Fritzell (1978) observed raccoons in North Dakota rarely reusing daytime resting sites on successive days. Gehrt (2003) supposed that this might partially be because raccoons had a large home range, as this makes it more difficult to return to a previous resting site. It was not clear whether raccoons in the present study used dens on the ground, but the size of their home ranges was as large as reported elsewhere, where sizes from 50 to 300ha were given (reviewed by Gehrt (2003)).

Raccoon dogs in Honshu, the main island of Japan, have several daytime resting sites in their home ranges (Yamamoto. 1993). Although raccoon dogs in the present study also had some daytime resting sites,

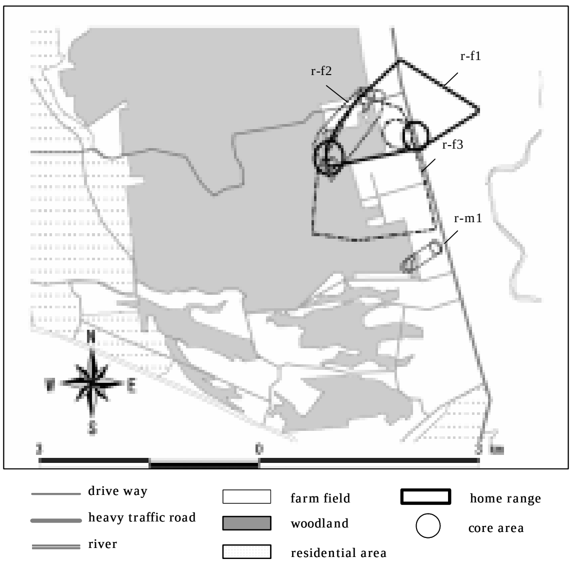

Figure 4 Nocturnal home ranges and core areas of four raccoons radio-tracked in Nopporo Natural Forest Park, for two months from June to July 2003. r-m1 represents a male raccoon. Home range and core area of each individual are shown with specific line pattern.

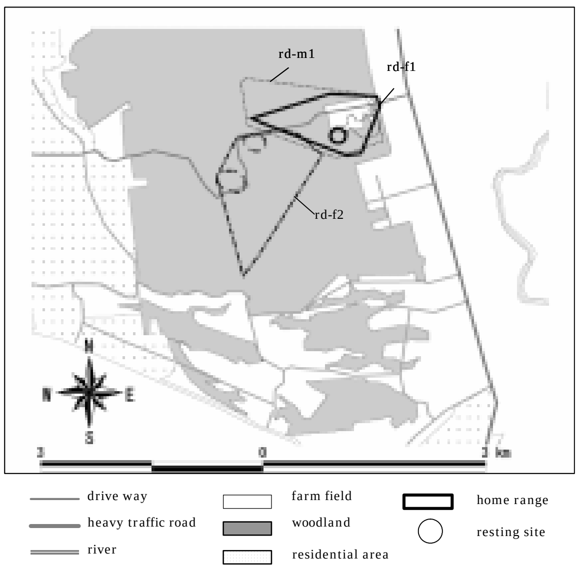

Figure 5 The home ranges and the core areas of three raccoon-dogs radio-tracked at night in Nopporo Natural Forest Park, June and July 2003. rd-m1 represents a male raccoon-dog. Home range and core area of each individual are shown with specific line pattern.

Table 2 Sizes of nocturnal home ranges and core areas of raccoons and raccoon dogs radio-tracked in Nopporo Natural Forest Park, Hokkaido, June to July 2003. Percentage area of farm field areas in home range and core area are shown.

Species	ID	Home range, 100% MCP[1]		Core areas, 50% FK[2]	
		Area (ha)	Farm-field (%)	Area (ha)	Farm field (%)
Racoons (*Procyon lotor*)	r-f1	161	74	25	43
	r-f2	59	35	6	37
	r-f3	236	52	14	88
	r-m1	9	63	5	52
Raccoon dogs (*Nyctereutes procyonoides albus*)	rd-f1	85	27	3	11
	rd-f2	153	0	14	0
	rd-m1	138	15	14	36

100% MCP[1] : 100% Minimum Convex Polygon method, 50% FK[2] : 50% Fixed Kernel method.

they were also fixed on the use of a particular breeding site. In raccoon dogs, both the male and female participate in pup rearing (Ikeda 1983, Yamamoto 1987, Kauhala and Saeki 2003), and take turns to attend the den for 30 to 50 days after the breeding. (Fukue 1991, Saeki 2001). In Japan, the breeding season of raccoon dogs is reported to be from April to June. Our study period, June to July, is supposed to be part of the pup rearing season, thus individuals might have been fixed on particular dens.

Raccoons used areas of farm fields more than raccoon dogs. It is well-known that raccoons utilised farmer's barns for feeding and breeding in Japan (Ando and Kajiura 1985, Maesaki *et al.* 2002, Ikeda *et al.* 2004). Ikeda (1999) reported that these buildings could have helped the introduced raccoons to naturalise in Hokkaido through the cold winter.

In their native range in North America, raccoons prefer to live near the water or moderately moist environments such as hardwood and mangrove swamps, marshes, and bottomland forests (Stuewer 1943, Zeveloff 2002), because much of their diet is sourced from aquatic areas (Stuewer 1943, Lotze and Anderson 1979, Sanderson 1987). In our study area there are many streams running through the woodland to the farm fields, although small streams are not shown in our map (Fig. 1). It is also known that raccoons can thrive in a wide variety of habitats in their native range. They commonly live in suburban residential areas and on cultivated and abandoned farm fields (Zeveloff 2002).

By contrast, the raccoon dogs in our study area tended to stay inside the forest during the daytime as well as the night-time, even if their core areas were in marginal woodland areas. This result is consistent with the fact that local people are often unaware of raccoon dogs being in the adjacent forests.

In Honshu, not much information on the feral raccoon is available, compared with information on the raccoon dog, which is one of the most popular mammals due to its frequent appearances in residential areas (Yamamoto *et al.* 1995, Sonoda and Kuramoto, 2001), rice fields, croplands and abandoned fields (Teduka *et al.* 1999, Sonoda 2001, Kauhala and Saeki 2004). Thus, if the tendency of the raccoon dogs in this study is representative for raccoon dog behaviour in Hokkaido, this would be clearly different from their behaviour in Honshu.

The difference in habitat use between these two species also evokes the possibility of direct and/or indirect inter-specific competition. Although the present study, however, does not provide any evidence of competition, ongoing monitored video camera recordings could clarify this possibility, as raccoons visited the dens and latrine sites of raccoon dogs. Unfortunately, small sample sizes limited the scope for detailed analysis of dynamic interactions between raccoons and raccoon dogs, but we showed that the habitat use between these two carnivores was different, even though they live sympatrically.

ACKNOWLEDGEMENTS

We wish to thank all the students who assisted our trapping, anesthetising and radio-tracking in this study. We are also grateful to Nopporo Natural Forest Park Office and the Nature Preservation Division of the Hokkaido Government which supported our study. Especial thanks to Dr. Asano (United Graduate School of Veterinary Sciences, Gifu University), who gave us precise advice on anaesthesia. This study was supported by the Ministry of the Environment as a Global Environment Research Programme (F3: 2002 - 2004).

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Short paper

Direct and indirect effects of an alien mongoose (*Herpestes javanicus*) on the native animal community on Amami-Oshima Island, southern Japan, as inferred from distribution patterns of animals

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INTRODUCTION

The small Indian mongoose (*Herpestes javanicus*) is native to the area from the Middle East to the Malay Peninsula. It has been deliberately introduced to many tropical areas, mainly as an attempted biological control agent for rats or pit vipers. For example, it was introduced to the Hawaiian Islands, West Indies and Fijian Islands. The introduced mongoose preys on non-target native vertebrates, and it is now largely blamed for the historical declines and extirpations of many native species on islands (Gorman, 1975, Roots, 1976, Honegger, 1981, Nellis and Small, 1983, Nellis *et al.*, 1984, Cheke, 1987, Case and Bolger, 1991, Henderson, 1992). The mongoose has also been introduced to Japan: to the southwestern islands of Amami-Oshima Island and Okinawa Island.

Amami-Oshima Island (712km^2) is the second largest of the southwestern islands. It has a subtropical wet climate and 85% of the land area is forest. The forest of Amami-Oshima Island is home to a large number of endemic species of the southwestern islands. In 1979, 30 mongooses were released to control the poisonous snake; habu (*Trimeresurus flavoviridis*). Since then, the mongoose has rapidly expanded its distribution, causing impacts on native animals. For example, the Amami rabbit (*Pentalagus furnessi*), a natural monument in Japan and regarded as a flagship species, inhabited almost all of the forested area in 1976, before the introduction of the mongoose. After the introduction, the area of distribution of the Amami rabbit has gradually decreased, corresponding to the range expansion of the mongoose (Sugimura *et al.* 2000, 2003, Yamada *et al.* 2000, 2002).

It is thought that, in addition to the Amami rabbit, other native species have also been reduced in abundance in areas where the mongoose has invaded. Such responses, however, are rarely quantified. The objective of our study was to determine the present distribution patterns of native animals along an historical gradient of mongoose establishment, to detect effects of the mongoose on a wide range of animals, including insects and vertebrates.

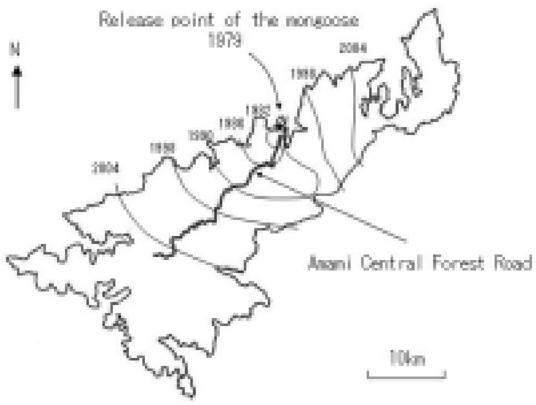

Figure 1 Expansion of the mongoose distribution and the location of the study sites

Pages122-123. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

METHODS

Our survey was conducted along the Amami central forest road (Total 41.1km). This forest road starts near the original release point of the mongoose and leads to areas where it is still absent (Fig. 1).

To assess the present distributions of native animals, we combined the following two methods: night-time driving censuses for native vertebrates and adhesive traps for insects. Night-time driving censuses were started more than 1 hour after sunset. We searched for vertebrates occurring on or around the road from a car at a constant speed of about 10km h^{-1}. We recorded species and location when we encountered vertebrates. We also recorded the call of the Amami rabbit, which is distinguishable from those of other animals. To assess relative insect abundance, two adhesive traps (Earth Chemical Co., Ltd., Gokiburi-Hoihoi) were placed on the ground at each of the 27 plots that were established on the forest floor 20-80m from ACFRoad. The distance between adjacent plots was 1.5km.

RESULTS

Most of the native mammal, bird, amphibians, reptile vertebrates observed in this survey showed an inverse distribution pattern to that of the mongoose: Amami rabbit, Amami woodcock (*Scolopax mira*), Amami tip-nose frog (*Rana amaminensis*), Otton frog (*Rana (Babina) subaspera*), Ishikawa frog (*Rana ishikawae*) and the Akamata snake (*Dinodon semicarinatus)*, were all scarce in areas where the mongoose invaded long ago.

On the other hand, two insect species that were captured in sufficient numbers, showed positive density patterns in relation to the mongoose distribution. For example, higher densities of the forest cricket (*Cardiodactylus novaeguineae*) and the small cockroach (*Margattea satsumana*) were found in areas where the mongoose invaded long ago.

DISCUSSION

Our results showed that the mongoose appears to cause a reduction in, or even local extinction of, many native vertebrates through a strong top-down effect. Forest crickets and small cockroaches are prey of the Amami tip-nose frog and the Otton frog (Watari, unpublished data). It is therefore likely that the increase in these insects is due to indirect effects of increased mongoose predation on the native predators. This trophic cascade may only be one of many wider and unpredicted community effects. It is, therefore, important to carefully monitor the dynamics of these interactions and to consider not only the direct effects but also the indirect effects of mongoose predation.

It seems clear that many native vertebrate species will continue to decline if the mongoose is allowed to spread and establish over the whole island. Thus, to protect the remaining native animals, it is essential to prevent further expansion of the mongoose's distribution.

ACKNOWLEDGEMENTS

We thank Shintaro Abe, Tadashi Miyashita and Richard Walters for constructive comments. Our study was funded by WWF-Nikko Green Investors Fund and WWF Japan Fund Grant Program.

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Part 3

Eradication and Control

The eradication of mammals from New Zealand islands

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Abstract Data on eradication operations against alien mammals on New Zealand islands show that there was a substantial increase in the number of successful eradications in the 1980s and 1990s. The most significant change has been in the ability to eradicate rodents from increasingly large islands (to over 11,000ha), using aerial poisoning techniques. Based on the New Zealand experience, there are good prospects for further eradications of alien mammals from islands around the world, facilitating ecological restoration and the recovery of threatened species. However, instances of reinvasion of rats (*Rattus* spp.) and stoats (*Mustela erminea*) onto previously cleared islands illustrate the importance of prevention, effective monitoring and a fuller understanding of invasion risks.

Keywords: invasive alien species; invasive mammal eradication; eradication on islands; island conservation; New Zealand

INTRODUCTION

Prior to the arrival of humans, the New Zealand terrestrial fauna contained no land mammals, apart from some small bats. The fauna had evolved without mammalian predation and included many endemic species of flightless birds, unusual reptiles, primitive frogs, and large insects. The diverse endemic flora had similarly evolved without mammalian browsing pressure.

The first people to settle in New Zealand, probably less than 800 years ago (Wilmshurst and Higham 2004), were from Polynesia. Associated with this settlement, Pacific rats (*Rattus exulans*) and domestic dogs (*Canis familiaris*) were introduced. Rat predation probably caused the extinction of several small animals, including frogs, small flightless birds, and large flightless insects (Atkinson and Moller 1990; Atkinson and Millener 1991). Overall, at least 40 species of vertebrates (mainly birds) are known to have become extinct, through predation by humans and introduced mammals, in the wake of this first wave of human settlement (Worthy and Holdaway 2002).

The second wave of human colonisation was by Europeans, beginning about 200 years ago. This resulted in the extinction of at least 12 more vertebrates (mostly birds), and the severe decline and range contraction of many other animal and plant species. Among the many alien species introduced to New Zealand by Europeans were a further 52 mammal species, of which 30 became established and 14 are widespread (King 2005).

Several alien mammals have been deliberately introduced to islands around New Zealand (e.g. possums, wallabies, rabbits, pigs, cattle, goats, sheep,

Figure 1 Map of New Zealand, with islands referred to in text.

Pages 127-141. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds.), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

deer, cats), or have colonised them by accidental transport or natural dispersal (e.g. rodents, stoats) (Russell *et al.* 2004). The spread of some predatory mammals throughout 'mainland' New Zealand has been relatively recent: black rats (*Rattus rattus*) first colonised the North Island in the 1860s and the South Island in the 1890s (Atkinson and Moller 1990), and mustelids were introduced in the late 1880s (King 1990). The subsequent tide of alien predation has exiled many vulnerable native birds and other animals to isolated islands, which have not been colonised by these mammals.

New Zealand has now lost over 40% of its pre-human land-bird fauna (Atkinson and Millener 1991; Worthy and Holdaway 2002) and no other country has a higher proportion of its surviving avifauna classed as threatened (Clout 2001). Of the surviving 287 New Zealand bird species (150 of them endemic), 70 are classed as threatened in the latest IUCN Red List (Baillie *et al.* 2004). Forty-three of these threatened species are endemics, and several of them now exist only on mammal-free islands or in dwindling mainland populations. Islands free of alien mammals are therefore invaluable as wildlife refuges. New Zealand has a large number of islands that are of conservation interest, either because they are still free of invasive mammals, or have the potential for eradication of such animals (Fig. 1). As awareness of the adverse effects of introduced mammals on native wildlife and ecosystems has risen, attempts have increasingly been made to eradicate mammals from such islands, permitting ecological restoration and the recovery or translocation of native biota.

In this paper we collate information on attempts to eradicate alien mammals from New Zealand islands, review these data, analyse trends and success rates, and draw conclusions on the lessons learned. One of our purposes is to bring together all of the information that is currently available, in the hope that its publication will stimulate others to add to the dataset and to update it as further eradications occur in future.

DEFINITIONS AND DATA PRESENTATION

For the purposes of this paper we define an eradication attempt as a project in which the defined goal was complete eradication of an existing population of a particular mammal species from a New Zealand island. Under this definition, we do not include instances of the removal of a single individual of a species, or cases where populations either died out naturally or some time after a 'control' (as opposed to eradication) campaign of some kind. We collated all available information on attempted eradications from a range of sources, including publications, file reports and personal accounts. Successful eradications are generally deemed to be those where there was no evidence of presence of the target species after the eradication programme was completed. All data on successful eradications are summarised in Tab. 1 of the Appendix. The format is based on the original database of eradications, compiled by Veitch and Bell (1990). The table is organised by species and gives the name of the island concerned, its area, the approximate date when the alien mammal species became established on the island, the eradication team leader, date when the eradication commenced, methods used, date when it was deemed complete, and key references. Data on incomplete, stopped and unsuccessful eradications are provided in Tab. 2 in the Appendix, under similar headings. Data on known re-invasions (i.e. where it is certain that the original population was eradicated and another established subsequently) are given in Tab. 3 of the Appendix.

HISTORY AND TRENDS OF MAMMAL ERADICATIONS

Eradication programmes against mammals on New Zealand islands started in the early 1900s. Rabbits (*Oryctolagus cuniculus*) were eradicated from Ngawhiti Island (5ha) in the Marlborough Sounds in 1912, goats (*Capra hircus*) from South East Island (219ha) in the Chathams group in 1915, and cattle (*Bos taurus*) from Kapiti Island (1965ha) in 1916. Other early eradications include the removal of rabbits from Tiritiri Matangi (196ha) around 1920, cats (*Felis catus*) from Stephens Island (150ha) around 1924, goats from Kapiti Island in 1928, and pigs (*Sus scrofa*) from Aorangi (110ha) in the Poor Knights group in 1936 (Tab. 1 in Appendix). In the 1940s to 1970s eradications were mainly of large or medium-sized mammal species (>10kg), mostly using standard techniques of shooting and trapping. Over this period there were almost no successful planned eradications of small mammals (<1kg), such as rodents (Towns and Broome 2003). This pattern changed dramatically in the 1980s and 1990s, when more eradications occurred and many of them were of rodent populations (Fig. 2).

Prior to the 1980s it was generally thought impossible to eradicate rodents even from small islands. In 1976, Yaldwyn (1978) summed up the conclusions of a conference of experts on the ecology and control of rodents in New Zealand by stating that the prospects

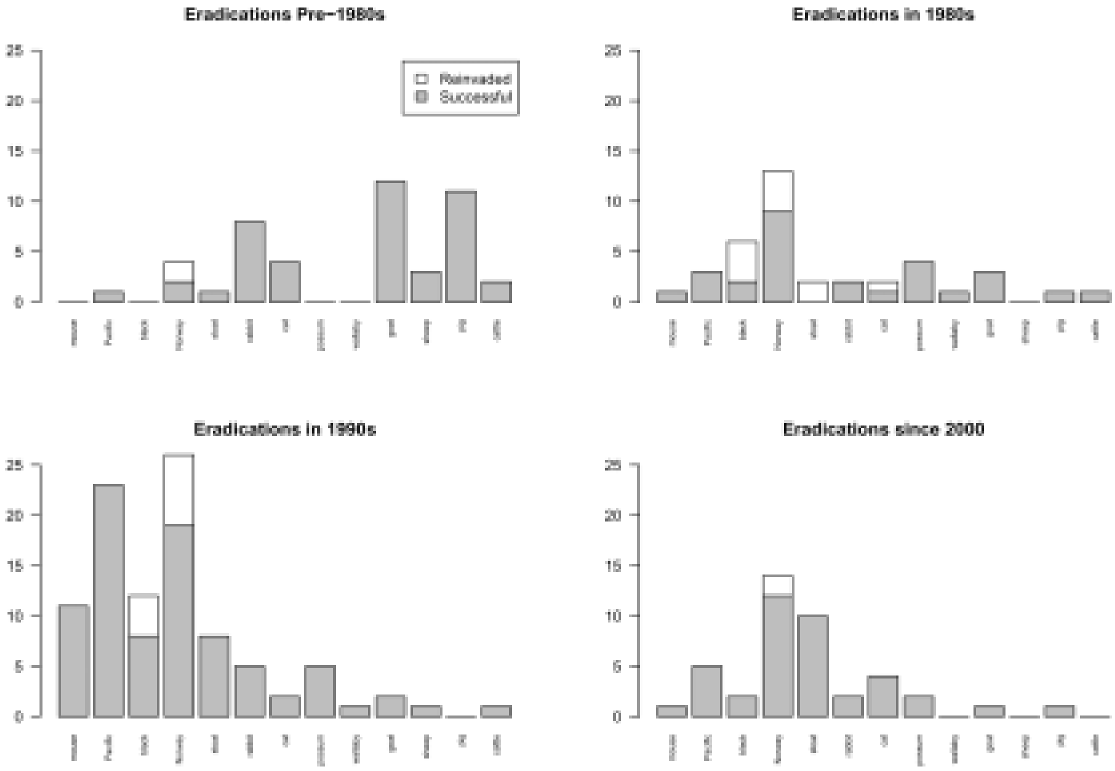

Figure 2 Eradications of alien mammals from New Zealand islands before 1980, during the 1980s, during the 1990s, and since 2000.

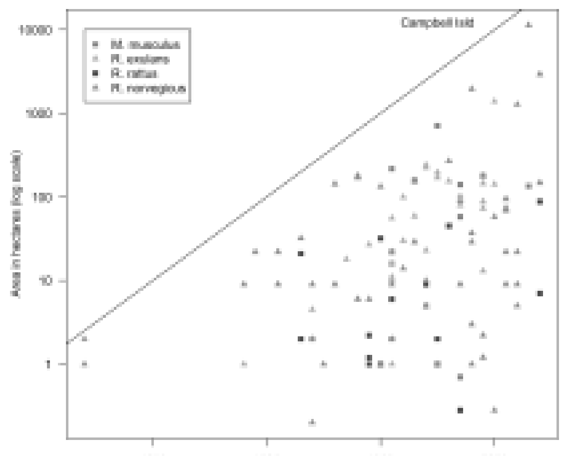

Figure 3 Eradications of alien rodents from New Zealand islands, 1964-2004, showing size of island (ha) and the rodent species eradicated.

for eradication of populations of these mammals from islands were 'remote'. Key improvements in the potential to eradicate rodents have been the availability from the 1980s of new 'second generation' anticoagulant baits, the development of rigorous eradication planning (Cromarty *et al.* 2002), and improved methods of aerial bait distribution. With this enhanced capacity, house mice (*Mus musculus*) have now been eradicated from 13 islands up to 710ha, Pacific rats (*Rattus exulans*) from 32 islands up to 2938ha, black rats (*R. rattus*) from 16 islands up to 157ha (although this includes 4 islands which were subsequently reinvaded by Norway rats), and Norway rats (*R. norvegicus*) from 44 islands up to 11,300ha (Campbell Island, in the New Zealand sub-Antarctic) (Fig. 3).

Some recent rodent eradications have successfully removed two species of rat in the same operation (e.g. on Kapiti, Mayor and Raoul islands). On the other hand, some islands successfully cleared of rats and stoats have subsequently been reinvaded; sometimes more than once (Tab. 3 in Appendix). The number of successful rodent eradication operations (Tab. 1 in Appendix) does not therefore equal the number of islands cleared of these mammals.

Successful eradications of stoats (*Mustela erminea*), especially in Fiordland, have been another feature of the past decade, due to improved trapping technology, careful planning and commitment to preventing reinvasions. Rates of eradication of medium sized mammals, such as possums, cats and rabbits have also increased since the 1980s, but less dramatically than those for rodents and stoats (Fig 2.). There is still a general trend for larger mammals in particular to have been eradicated from islands of larger maximum area (Fig. 4), but recent eradications of species such as Norway rats from large islands have resulted in this trend being less evident than it was in the past.

SUCCESS AND FAILURE RATES

Unsuccessful or stopped eradication attempts are probably recorded less thoroughly than successes. However, since important lessons can sometimes be learned from failures, some analysis is worthwhile. For example, considering only operations against rodents in the 1990s, there were four stopped eradications and seven unsuccessful ones. Five of the latter were against house mice, including two failures on the same island in successive years (Tab. 2 in Appendix). Comparing these stopped and unsuccessful eradications with successful ones against rodents in the same decade, eradication success rates in the 1990s were 100% for Pacific rats (n=23) and black rats (n=8), 86% for Norway rats (n=22), and 68% for house mice (n=19). Closer examination of the data for Norway rats reveals that one of the stopped eradications and one of the failed ones in the 1990s were on the same island (Rakino in the Hauraki Gulf), from which this species was successfully eradicated in 2002.

REINVASIONS

The accepted set of conditions for successful eradication (Bomford and O'Brien 1995) are proper planning, a commitment to complete the operation, putting the entire population of the target species at risk, removing them faster than they reproduce, and

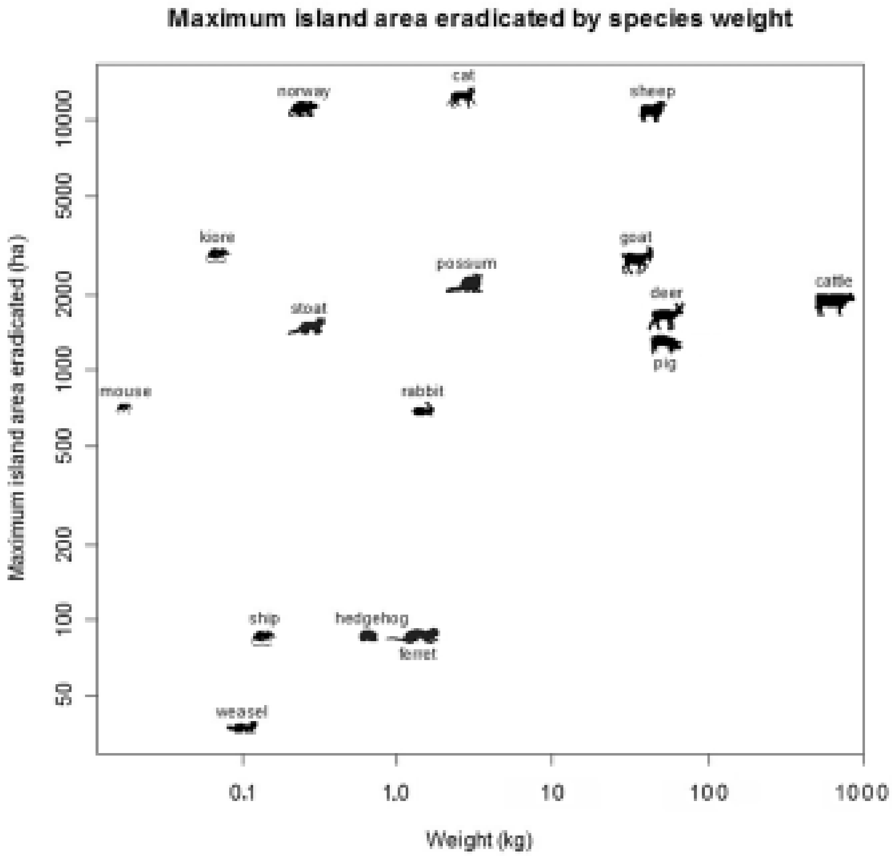

Figure 4 Maximum areas (ha) of New Zealand islands from which particular mammal species have been eradicated, plotted against mean weight (kg) of each species.

preventing reinvasion. For species that can be accidentally reintroduced or can swim across water gaps, reinvasion is a particular concern (Russell et al. 2005). Data on reinvasions of New Zealand islands (Tab. 3 in Appendix) reveals particularly high risks of reinvasion by Norway rats. Since the 1980s there have been four reinvasions of black rats onto islands from which they had previously been eradicated, but at least 19 reinvasions of Norway rats. Six of the latter reinvasions were successive ones on Motuhoropapa (9ha) and four of them on neighbouring Otata (22ha), both in the Noises group, Hauraki Gulf. Reinvasion was most probably occurring from Rakino island, over 2km away, which was completely cleared of all Norway rats in 2002.

Among the other species that have been recorded as reinvading islands (Tab. 3 in Appendix), stoats are a particular conservation concern in New Zealand because of their impacts as predators, combined with the virtual certainty that any female stoat will be pregnant (King 1990) and hence capable of establishing a population. Stoats are capable of swimming at least 1.2km in New Zealand waters (Taylor and Tilley 1984), so any islands less than this distance from a source population (especially in sheltered waters) must be considered to be at risk. The treatment and subsequent careful monitoring of entire local archipelagos, as is being attempted in SW Fiordland (M. Willans pers. comm.) and South Island lakes (S. Thorne pers. comm.), seems the best eradication strategy for these highly mobile and dispersive mammals.

CONCLUSIONS

The rapid rise in the 1980s and 1990s in the number of successful eradications of alien mammals fuelled yet more eradication attempts on larger and larger islands around New Zealand. There was a particularly notable increase in the 1980s in the rate of eradication of rodents and in the increased size of islands from which they were cleared. With careful planning, and the use of modern techniques such as global positioning systems to ensure the even aerial distribution of toxic baits, it is now possible to remove rats from large islands over 10,000ha in area (Towns and Broome 2003). By the end of 2004 there had been 218 successful eradications of 17 different alien mammal species from New Zealand islands. Several of these eradications were of different mammals on the same islands, resulting in many islands that are now free of all alien mammals and therefore have greatly enhanced potential for further ecological restoration.

Along with an improved capacity to eradicate mammals of all species from increasingly large islands, there has been a trend towards targeting two or more species of alien mammals in a single operation. Recent examples of this are the successful joint eradications of Pacific and Norway rats from Kapiti Island; Pacific rats, Norway rats and cats from Mayor and Raoul Islands; and rabbits, black rats and stoats from Quail Island.

Prevention of reinvasion is emerging as another key focus. The reappearance of rat populations on several islands, along with other recent instances of individual rats being seen or caught on supposedly rat-free islands in New Zealand (e.g. Matakohe, Moturoa, and Rotoroa islands), stresses the importance of prevention measures, regular monitoring of supposedly rat-free islands, and a better understanding of the behaviour of invading rodents. There are many islands that have had rodent populations eradicated in the past decade, including several that are close inshore or otherwise vulnerable, so more reinvasions should be anticipated in the future.

New Zealand experience with eradication of alien mammals from islands has obvious relevance for other parts of the world, although the absence of native mammals and the fact that most islands are uninhabited by people makes the situation in New Zealand relatively simple. A major challenge for the future is therefore to improve capacity to conduct multiple eradications of alien species on inhabited islands, and to keep such islands free from reinvasion, so that local people and native biodiversity can both prosper.

ACKNOWLEDGEMENTS

We thank the many people who have provided information on eradications of alien mammals and other animals from New Zealand islands and who have assisted in compilation of the database summarised in the Appendix to this paper. Sources of information, and leaders of eradication operations, are acknowledged in the tables and supporting references in the Appendix. Any omissions or errors are inadvertent. We would be pleased to receive corrections or additional data, so that this information source can be updated in the future.

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APPENDIX

ERADICATIONS OF MAMMALS FROM NEW ZEALAND ISLANDS (TO 2004)

The authors would be grateful to be made aware of any omissions or errors in this compilation.

Table 1. Operations which have resulted in the successful eradication of alien mammals from islands around New Zealand. Methods listed are: P=poison; T=trap; S=shooting; D=dogs. * = date confirmed after a 2 year confirmation process, † = ongoing programme due to high reinvasion risk.

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Date Completed	Reference
HOUSE MOUSE *Mus musculus*							
Allports (Marlborough)	16	c1900	D. Brown	1989	P	1991*	Brown 1993a
Browns (Hauraki Gulf)	58	?	C. R. Veitch	1995	P	1997*	Veitch 2002a
Enderby (Auckland)	710	1820s	N. Torr	1993	P	1995*	Torr 2002
Mana	217	1800s	P. Todd, T. Hook	1989	P	1991*	Hook & Todd 1992
Mokoia (Lake Rotorua)	135	c1965	B. Mossman	2001	P	2003*	Armstrong et al. 2001
Motuihe (Hauraki Gulf)	179	>1987	C. R. Veitch	1997	P	1999*	Veitch 2002b
Moturemu (Kaipara)	5	?	I. McFadden	1992	P	1994*	I. McFadden pers. comm.
Motutapere (West Coromandel)	45	?	P. Thomson	1994	P	1996*	P. Thomson pers. comm.
Motutapu (Marlborough)	2	?	D. Brown	1989	P	1991*	Brown 1993a
Mou Waho (Lake Wanaka)	140	1995	B. McKinlay	1995	P, T	1997*	McKinlay 1999
Papakohatu (Crusoe)	0.7	?	M. Lee	1996	P, T	1997	Lee 1999
Rimariki	22	?	C. Smuts-Kennedy	1989	P	1991	Veitch & Bell 1990;
Whenuakura (Whangamata)	2	c1980	I. McFadden	1983	P	1984	Newman 1985
PACIFIC RAT *Rattus exulans*							
Arch (Mokohinau)	1	?	I. McFadden	1990	P	1991	McFadden 1992b
Burgess (Mokohinau)	56	?	I. McFadden	1990	P	1991	McFadden 1994
Codfish (Whenua Hou)	1396	?	P. McClelland	1998	P	2000*	McClelland 2002
Cuvier	170	<1827	P. Thomson	1993	P	1995*	Towns et al. 1995
Double (Mercury)	27	c1900	I. McFadden	1989	P	1989	McFadden 1992a
Fanal (Mokohinau)	73	?	C. R. Veitch	1997	P	1999*	Veitch 2002c
Flax (Mokohinau)	1	?	I. McFadden	1990	P	1991	McFadden 1994

(APPENDIX Table 1 continued)

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Date Completed	Reference
Inner Chetwode (Nukuwaiata)	242	?	D. Brown	1993	P, T, D, S	1994	Brown 1997
Kapiti	1965	<1850	R. Empson	1996	P	1998*	Empson & Miskelly 1999
Korapuki (Mercury)	18	c1900	I. McFadden	1986	P	1987	McFadden & Towns 1991
Lizard (Mokohinau)	1	1977	C. R. Veitch	1978	P	1978	McCallum 1986
Long	142	?	B. Cash	1997	P	2000*	B. Cash pers. comm.
Maori Bay (Mokohinau)	11	?	I. McFadden	1990	P	1991	McFadden 1994
Mauimua (Lady Alice)	155	c1800	K. Hawkins	1994	P	1996*	Towns & Parrish 2003
Mauipae (Coppermine)	80	?	K. Hawkins	1995	P	1997*	Towns & Parrish 2003
Mauiroto (Whatupuke)	102	?	K. Hawkins	1993	P	1997	Towns & Parrish 2003
Mayor (Tuhua)	1277	?	A. Jones	2000	P	2002*	Williams et al. 2000
Middle Chain (Aldermen)	23	?	R. Thorpe	1992	P	1994*	Thorpe 1997
Motuara (Marlborough)	59	?	W. Cash	1991	P	1993*	Cash & Gaze 2000
Motuopao (Far North)	30	?	D. McKenzie	1989	P	1992*	McKenzie 1993
Motupapa 'Stack C' (Mokohinau)	2	?	I. McFadden	1990	P	1991	McFadden 1994
Putauhinu (Stewart)	145	?	P. McClelland	1997	P	1999*	McClelland 2002
Rarotoka (Centre)	88	?	P. McClelland	1997	P	1999*	McClelland 2002
Raoul (Kermadecs)	2938	c1300s	M. Ambrose	2002	P	2004*	M. Ambrose pers. comm.
Red Mercury (Mercury)	225	?	P. Thomson	1992	P	1994*	Towns et al. 1994
Rurima (BOP)	4.5	c1900	I. McFadden	1983	P	1984	McFadden & Towns 1991
Stacks B-G,I,J (Mokohinau)	10	?	I. McFadden	1990	P	1991	McFadden 1994
Stanley (Mercury)	100	<1900	I. McFadden	1991	P	1992	Towns et al 1993
Tiritiri Matangi (Hauraki Gulf)	196	?	C. R. Veitch	1993	P	1995*	Veitch 2002d
Trig (Mokohinau)	16	?	I. McFadden	1990	P	1991	McFadden 1994
Whakaterepapanui (Rangitoto)	74	?	P. Gaze	1999	P	2001*	P. Gaze pers. comm.
Whangaokena (East)	13	?	D. Peters	1997	P	1999*	Bassett 1999
BLACK RAT *Rattus rattus*							
Awaiti	2	?	D. Taylor	1982	P	1983	Taylor 1984
Black Rocks (BOI)	1	?	T. Shaw	1992	P	1995*	Shaw 1997
Haulashore (Nelson)	6	?	R Taylor, B Thomas	1991	P	1991	Thomas & Taylor 2002
Iona (Stewart)	7	?	M. Wylie	2004	P	2004	B. Beaven pers. comm.
Little Rat (BOI)	1	?	D. Taylor	1992	P	1995*	Shaw 1977
Mokopuna (Leper)	1	c1961	S. Butcher	1988	P	1990*	I. McFadden pers. comm.
Mouse (BOI)	1	?	D. Taylor	1992	P	1995*	Shaw 1997
Phil's Hat (BOI)	1	?	D. Taylor	1992	P	1995*	Shaw 1997
Quail (Lyttleton)	88	?	QI Trust, M. Bowie	2002	P	2004*†	Kavermann et al. 2003
Rat (BOI)	2	?	D. Taylor	1992	P	1995*	Shaw 1997
Somes (Wellington)	32	c1961	S. Butcher	1988	P	1990*	McFadden pers. comm.
Tawhitinui	21	?	D. Taylor	1982	P	1983	Taylor 1984
NORWAY RAT *Rattus norvegicus*							
Black Rocks (BOI)	1	?	T. Shaw	1992	P	1995*	Shaw 1997
Breaksea (Fiordland)	170	1800s	R Taylor, B Thomas	1988	P	1988	Taylor & Thomas 1993
Browns (Hauraki Gulf)	58	1980s	C. R. Veitch	1995	P	2000	Veitch 2002a
Campbell	11330	c1810	P. McClelland	2001	P	2003*	McClelland & Tyree 2002
David Rocks (Hauraki Gulf)	1	1960	D. Merton	1960	P	1964	Moors 1985a
East & West Atoll (BOI)	1	?	D. Taylor	1992	P	1995*	Shaw 1997
Hauturu (Whangamata)	10	c1972	P. Thomson	1992	P	1994*	Glassey 2004
Hawea (Fiordland)	9	1800s	R Taylor, B Thomas	1986	P	1986	Taylor & Thomas 1989
Kapiti	1965	<1850	R. Empson	1996	P	1998*	Empson & Miskelly 1999
Koi (Hauraki Gulf)	0.28	2000	M. Lee	2000	P	2000	M. Lee pers. comm.
Maria (Hauraki Gulf)	2	1959	D. Merton	1960	P	1964	Moors 1985a
Matakohe (Limestone)	37	?	J. Craw	1996	P	1998*†	Ritchie 2000
Maukaha Rocks (Whangamata)	0.2	1984	McFadden, Wilke	1984	P	1984	Newman 1985
Mayor (Tuhua)	1277	?	A. Jones	2000	P	2002*	Williams et al. 2000
Mokoia (Lake Rotorua)	133	<1839	P. Jansen	1989	P	1990†	Owen 1997, pers. comm.
Motiti	1	?	A. Walker	1990	P	1990	McKenzie pers. comm.
Motu-O-Kura (Hawkes Bay)	14	c1936	J. Adams	1990	P	1992*	Adams 1997
Motuhoropapa (Hauraki Gulf)	9	2002	G. Wilson	2002	P	2002	Wilson 2003
Motuihe (Hauraki Gulf)	179	1997	C. R. Veitch	1997	P	1999*	Veitch 2002b
Motungara (Kapiti)	3	<1850	R. Empson	1996	P	1998*	Empson & Miskelly 1999
Motutapu (BOI)	1	?	D. McKenzie	1990	P	1990	D. McKenzie pers. comm.
Motuterakihi	1	?	D. Taylor	1985	P	1985	D. Taylor pers. comm.
Moutohora (Whale)	143	c1920	P. Jansen	1985	P	1986	Jansen 1993
Otata (Hauraki Gulf)	22	2002	G. Wilson	2002	P	2002	Wilson 2003
Pakatoa (Hauraki Gulf)	29	1997	M. Lee	1998	P	1998*†	M. Lee pers. comm.
Puangiangi (Rangitoto)	69	?	P. Gaze	1999	P	2001*	D. Brown pers. comm.

(APPENDIX Table 1 continued)

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Date Completed	Reference
Rakino (Hauraki Gulf)	148	1920-72	Hix, Waters, Wilson	2002	P	2004*†	J. MacKenzie pers.comm.
Raoul (Kermadecs)	2938	1921	M. Ambrose	2002	P	2004*	M. Ambrose pers. comm.
SW Crater Rim (BOI)	1	?	D. Taylor	1992	P	1995*	Shaw 1997
Tahoramaurea (Kapiti)	1	<1850	R. Empson	1996	P	1998*	Empson & Miskelly 1999
Takangaroa	6	1987	T. Clarkson	1988	P	1988	Taylor 1989
Tarahiki	5	c1998	G. Wilson	2000	P	2002*	G. Wilson pers. comm.
Taranaki (BOI)	1	?	D. McKenzie	1990	P	1990	D. McKenzie pers. comm.
Te Haupa (Saddle)	6	<1981	R. Gilfillan	1989	P	1989	Tennyson & Taylor 1999
Tinui (Rangitoto)	95	?	P. Gaze	1999	P	2001*	D. Brown pers. comm.
Titi (Marlborough)	32	<1955	B. Bell, D. Merton	1970	P	1983*	Gaze 1983
Unnamed Cape Wiwiki A ('Snail Rock')	2.2	1989-99	A. Walker	1999	P	1999	R. Parrish pers. comm.
Unnamed Cape Wiwiki B ('Gorse Island')	1.2	1989-99	A. Walker	1999	P	1999	R. Parrish pers. comm.
Ulva (Stewart)	270	?	L. Chadderton	1992	P	1996†	Thomas & Taylor 2002
Wainui (BOI)	2	?	D. McKenzie	1991	P	1991	D. McKenzie pers. comm.
Whakaterepapanui (Rangitoto)	74	?	P. Gaze	1999	P	2001*	D. Brown pers. comm.
Whenuakura (Whangamata)	2	c1982	McFadden, Wilke	1983	P	1984	Newman 1985
WEASEL *Mustela nivalis vulgaris*							
Matakohe (Limestone)	37	?	J. Craw	1996	T	1996†	Ritchie 2000
STOAT *Mustela erminea*							
Adele (Nelson)	88	1982	T. Shaw	2003	T	2004†	T. Shaw pers. comm.
Anchor (Fiordland)	1280	?	Willans,Munn,Elliott	2001	T	2001†	Willans 2002
Bauza (Fiordland)	475	?	Willans, Elliott	2002	T	2003†	Willans 2002
Te Kakahu (Chalky)	511	?	Willans, Munn	1999	T	1999	Willans 2000
Doubtful east (Lake Te Anau)	120	?	M. Willans	2000	T	2002†	Willans 2002
Doubtful centre (Lake Te Anau)	40	?	M. Willans	2000	T	2002†	Willans 2002
Doubtful west (Lake Te Anau)	120	?	M. Willans	2000	T	2002†	Willans 2002
Erin (Lake Te Anau)	75	?	M. Willans	2000	T	2002†	Willans 2002
Matakohe (Limestone)	37	?	J. Craw	1996	T	1996†	Ritchie 2000
Maud (Marlborough)	309	c1989	D. Crouchley	1990	T, S	1993†	Crouchley 1993
Moturoa (BOI)	157	?	P. Asquith	1993	T	1993†	Asquith 2004
Mou Waho (Lake Wanaka)	140	?	S Thorne, J Flemmin	1997	T	1997†	S. Thorne pers. comm.
Mou Tapu (Lake Wanaka)	120	?	S Thorne, J Flemmin	1997	T	1997†	S. Thorne pers. comm.
North Passage (Fiordland)	10	?	M. Willans	1998	T, D	2000	Willans 2000
Otata (Hauraki Gulf)	22	1948	Captain Wainhouse	?	S	1955	B. Neureuter pers. comm.
Quail (Lyttleton)	88	?	Quail Isld Trust	2002	T	2004*†	Kavermann et al. 2003
Silver (Lake Hawea)	25	?	S Thorne, J Flemmin	1997	T	1997†	S. Thorne pers. comm.
South Passage (Fiordand)	176	?	M. Willans	1998	T, D	2000	Willans 2000
Stevensons (Lake Wanaka)	65	?	S Thorne, J Flemmin	1997	T	1997†	S. Thorne pers. comm.
FERRET *Mustela furo*							
Quail (Lyttleton)	88	?	Quail Isld Trust	2002	T	2002†	Kavermann et al. 2003
HEDGEHOG *Erinaceus europaeus*							
Quail (Lyttleton)	88	?	QI Trust, M. Bowie	1999	P, T	2003	Thomsen et al. 2000
RABBIT *Oryctolagus cuniculus*							
Browns (Hauraki Gulf)	58	c1975	F. David	1985	P, T, D, S	1991	Veitch 2002a
Enderby (Auckland)	710	1865	N. Torr	1993	P, T, D, S	1994	Torr 2002
Korapuki (Mercury)	18	c1900	I. McFadden	1986	P, S	1988	Towns 2002
Mokopuna (Leper)	1	c1946	L. Bell	1947	?	1950	Anon 1951
Motuihe (Hauraki Gulf)	179	?	S. Mowbray	2002	P, S	2004	D. Thompson pers. comm
Motunau (Canterbury)	3	c1850	Motunau Rabbit Brd	1958	P, S	1962	Cox et al. 1967
Moutohora (Whale)	143	1968	P. Jansen	1985	P, T	1987	Jansen 1993
Native (Stewart)	66	c1942	S. Corboy	c1949	T, S	1950	R. Taylor pers. comm.
Ngawhiti (Marlborough)	5	c1910	?	c1912	?	c1912	Gibb & Williams 1990
Otata (Hauraki Gulf)	15	?	Captain Wainhouse	?	S	1945	B. Neureuter pers. comm.
Quail (Lyttleton)	88	c1855	J. Trotter	1997	P, T, D, S	2004	D. Brown pers. comm.
Rose (Auckland)	75	1850	N. Torr	1993	P, T, D, S	1994	Torr 2002
Stanley (Mercury)	100	c1900	I. McFadden	1991	P	1992	Towns et al. 1993
Stewart [part]	?	1942	Dept Agriculture	c1948	T, S	1950	R. Taylor pers. comm.
Taieri (Dunedin)	7	?	J. Pearce	1987	P	1996	J. Pearce pers. comm.
Takangaroa	6	<1930	T. Clarkson	?	S	<1950	Taylor 1989
Tiritiri Matangi (Hauraki Gulf)	196	<1894	E. Hobbs	c1900	?	c1920	Veitch 2002d

(APPENDIX Table 1 continued)

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Date Completed	Reference
CAT *Felis catus*							
Cuvier	170	c1889	D. Merton	1960	T, S	1964	Merton 1970
Herekopare (Stewart)	28	c1925	C. R. Veitch	1970	T, D	1970	Fitzgerald & Veitch 1985
Kapiti	1965	c1900	R. Fletcher	1923	S	1934	Fuller 2004
Hauturu (Little Barrier)	3083	<1870	C. R. Veitch	1977	P, T, D	1980	Veitch 2001
Macquarie	13182	1820	Tasmanian Parks & Wildlife	1997	P, T, D, S	2003	Copson & Whinam 2001
Matakohe (Limestone)	37	?	Clapperton, Pierce	1991	P	1991	Clapperton et al. 1992
Mayor (Tuhua)	1277	?	A. Jones	2000	P	2002	Williams et al. 2000
Motuihe (Hauraki Gulf)	160	c1984	S. Mowbray	2002	T, S	2004	D. Thompson pers. comm
Quail (Lyttleton)	88	<1983	D. Brown	1998	T, S	1998	D. Brown pers. comm.
Raoul (Kermadecs)	2938	1836-72	M. Ambrose	2002	P, T, D	2004	M. Ambrose pers. comm.
Stephens	150	<1895	Lighthouse keepers	1895	S	c1924	Brown 2000
BRUSHTAIL POSSUM *Trichosurus vulpecula*							
Allports (Marlborough)	16	<1980	D. Brown	1989	P, T	1990	Brown 1993b
Codfish (Whenua Hou)	1396	<1925	A. Cox, G. Aburn	1984	P, T, D	1987	Brown unpubl.Brown 2002
Iona (Stewart)	7	c1984	M. Wylie	2004	T	2004	B. Beaven pers. comm.
Kapiti	1965	1893	Alexander, Cairns	1980	P, T, D, S	1986	Cowan 1992
Matakohe (Limestone)	37	?	C. Cooper	1991	S	1991	Clapperton et al. 1992
Motutapere (West Coromandel)	45	<1970	P. Thomson	1994	P	1996	P. Thomson pers. comm
Motutapu (Hauraki Gulf)	1560	1868	S. Mowbray	1990	P, T, D	1996	Mowbray 2002
Rangitoto (Hauraki Gulf)	2321	1868	S. Mowbray	1990	P, T, D	1997	Mowbray 2002
Unnamed Cape Wiwiki A ('Snail Rock')	2.2	?	D. Taylor, T. Shaw	1989	P	1989	Parrish et al. 1995
Unnamed Cape Wiwiki B ('Gorse Island')	1.2	?	D. Taylor, T. Shaw	1989	P	1989	Parrish et al. 1995
BRUSH-TAILED ROCK WALLABY *Petrogale penicillata*							
Motutapu (Hauraki Gulf)	1560	1868	S. Mowbray	1990	P, T, D	1996	Mowbray 2002
Rangitoto (Hauraki Gulf)	2321	1868	S. Mowbray	1990	P, T, D	2000	Mowbray 2002
Great Barrier [part-Reef Point]	?	1980	A. Leigh, K. Purdon	1981	T	1981	Eadie et al. 1990
GOAT *Capra hircus*							
Auckland [part]	4000	1865	K. Timpson, Willense, Cox	1989	P, S	1991	Brown unpubl.
Burgess (Mokohinau)	56	?	C. R. Veitch	1973	S	1973	Veitch 1973
Cuvier	170	1890s	B. Bell	1959	S	1961	Merton 1970
Ernest (Stewart)	25	<1909	Muttonbirders	1980s	?	1980s	Parkes 1990
Great (Three Kings)	407	1889	L. Bell	1946	S	1946	Turbott 1948
Herekopare (Stewart)	28	1973	Muttonbirders	1975	S	1976	R. Tindall pers. comm.
Nukuwaiata	242	?	C. Smuts-Kennedy	1993	S	1993	C. Smuts-Kennedy pers. comm.
Kapiti	1965	c1830	A. S. Wilkinson	1924	S	1928	Wilkinson 1952
Macauley (Kermadecs)	236	<1836	B. Bell	1966	S	1966	Williams & Rudge 1969
Mahurangi	23	c1900	?	?	?	1915	Atkinson 1972
Moturekareka	19	?	C. Hansen	?	S	?	Tennyson et al. 1997
Moutohora (Whale)	143	c1890	NZ Wildlife Service	1971	S	1977	Ogle 1990
Nukutaunga (Cavalli)	13	?	C. Smuts-Kennedy	1972	S	1972	Miller 1976
Ocean (Auckland)	3	1865	CAPE Expedition	1941	S	1943	Rudge & Campbell 1977; Brown unpubl.
Rakitu (Arid)	328	?	D. Agnew	2002	S	2002	Agnew 2002
Raoul (Kermadecs)	2938	<1836	NZ Forest Service	1972	D, S	1984	Parkes 1990
South East (Chathams)	219	1900	Mr McLurg	1914	?	1915	Ritchie 1970
Pouawa	6	?	Andy Bassett	1987	S	1987	A. Bassett pers. comm.
Whangaokena (East)	13	1906	G. Goldsmith	1960	S	1960	H. Hovell pers. comm.
SHEEP *Ovis aries*							
Campbell	11330	1895	Bell, Torr, Cox	1970	S	1991	Brown unpubl.
Kapiti	1965	1850s	P. Rodda	1928	S	1930	Fuller 2004
Mangere (Chathams)	113	c1900	B. Bell	1968	S	1968	B. Bell pers. comm..
South East (Chathams)	219	1914	B. Bell	1957	S	1961	Ritchie 1970
RED DEER *Cervus elaphus scoticus*							
Anchor (Fiordland)	1525	?	Willans, Munn	2002	D, S	2004†	M. Willans pers. comm.
PIG *Sus scrofa*							
Aorangi (Poor Knights)	110	c1820	Major Yerex	1936	S	1936	Challies 1976

(APPENDIX Table 1 continued)

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Date Completed	Reference
Blumine (Oruawairua)	377	c1840	M. Finch	1989	D, S	1989	Clarke et al. 1991
Inner Chetwode (Nukuwaiata)	242	c1900	?	?	S	1926	Internal Affairs Files
Inner Chetwode (Nukuwaiata)	242	c1954	D. Cummings	1959	D, S	1963	Internal Affairs Files
Mayor (Tuhua)	1277	?	A. Jones	2000	D, S	2002	A. Jones pers. comm.
Motuara (Marlborough)	59	c1840	?	1920s	P, D, S	1950s	Clarke et al. 1991
Motuoruhi (West Coromandel)	57	?	?	?	?	<1970	McIlroy 1990
Outer Chetwode (Te Kakaho)	81	c1948	?	1953	S	1953	Internal Affairs Files
Outer Chetwode (Te Kakaho)	81	c1955	D. Cummings	1964	S	1964	Internal Affairs Files
Pickersgill	103	c1840	?	1920s	P, D, S	1950s	Clarke et al. 1991
Rakitu (Arid)	328	?	?	?	?	1960s	McIlroy 1990
Stewart [part]	1040	?	Purdon, Vipond	1948	D, S	1948	Holden 1982
Tuputupungahau (Whale)	13	1950s	Owners	?	?	c1966	Wright 1977
CATTLE *Bos taurus*							
Campbell [part]	2000	1902	Ron Peacock	1984	S	1984	Brown unpubl.
Enderby (Auckland)	710	1894	Nick Torr	1991	S	1993	Torr 2002; Brown unpubl.
Kapiti	1965	c1837	J. L. Bennett	1916	S	1916	Wilkinson 1952
Stewart [part]	?	?	D. Internal Affairs	1940s	S	1940s	Taylor 1976

Table 2 Operations which have not resulted in the eradication of an alien mammal species from an island. These operations are listed as: "incomplete" where the work is continuing or confirmation of the eradication has not been obtained; "stopped" where the work was stopped due to a management decision before the planned work was completed; "failed" where the planned programme was completed and eradication was not successful. Methods listed are: P=poison; T=trap; S=shooting; D=dogs.

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Reference
INCOMPLETE						
HOUSE MOUSE *Mus musculus*						
Mou Tapu (Lake Wanaka)	120	?	S Thorne, J Flemmin	1997	P	S. Thorne pers. comm.
Quail (Lyttleton)	88	?	QI Trust, M. Bowie	2002	P	M. Bowie pers. comm.
PACIFIC RAT *Rattus exulans*						
Bench (Stewart)	121	?	M. Wylie	2005	P	B. Beaven pers. comm.
Hauturu (Little Barrier)	3083	?	R. Griffiths	2004	P	R. Griffiths pers. comm.
Macauley (Kermadecs)	236	c1300s	M. Ambrose	2005	P	M. Ambrose pers. comm.
Mokinui (Big Moggy)	86	?	P. McClelland	2005	P	D. Agnew pers. comm.
Ohinau (East Coromandel)	143	?	J. Roxburgh	2005	P	J. Roxburgh pers. comm.
Pearl (Stewart)	512	?	M. Wylie	2005	P	B. Beaven pers. comm.
BLACK RAT *Rattus rattus*						
Fortyseven (BOI)	1	?	D. Taylor	1990	P	D. Taylor pers. comm.
Harakeke (BOI)	12	?	D. Taylor	1992	P	D. Taylor pers. comm.
Horomamae (Owen)	36	?	J. Ryan	2004	P	D. Agnew pers. comm.
Motukaha (Hauraki Gulf)	0.4	?	I. McFadden, M. Lee	1996	P	Lee 1999
Moturako (GBI)	1	?	G. Taylor	1990	P	G. Taylor pers. comm.
Motutapere (West Coromandel)	45	2002	R. Chappell	2005	P	R. Chappell pers. comm.
Opakau (GBI)	4	?	G. Taylor	1990	P	G. Taylor pers. comm.
Oyster (GBI)	1	?	G. Taylor	1990	P	G. Taylor pers. comm.
Pearl (Stewart)	512	?	M. Wylie	2005	P	B. Beaven pers. comm.
Pukeweka (Stewart)	3	1960s	P. McClelland	2005	P	D. Agnew pers. comm.
Saddle (GBI)	2	?	G. Taylor	1990	P	G. Taylor pers. comm.
Taukihepa (Big South Cape)	939	1960s	P. McClelland	2005	P	D. Agnew pers. comm.
Whakarereupoko (Rerewhakaupoko)	26	1960s	P. McClelland	2005	P	D. Agnew pers. comm.
Wood (GBI)	1	?	G. Taylor	1990	P	G. Taylor pers. comm.
Wood Stack A (GBI)	1	?	G. Taylor	1990	P	G. Taylor pers. comm.
NORWAY RAT *Rattus norvegicus*						
Moturemu (Kaipara)	5	2002	T. Wilson	2004	P	Russell et al. unpubl.

(APPENDIX Table 2 continued)

Location	Area	Date Introduced	Eradication Leader	Date Started	Methods	Reference
Moturoa (BOI)	157	1997	P. Asquith	2003	P	Asquith 2004
Pearl (Stewart)	512	?	M. Wylie	2005	P	B. Beaven pers. comm.
RABBIT *Oryctolagus cuniculus*						
Ohinau (East Coromandel)	143	?	J. Roxburgh	2005	P	J. Roxburgh pers. comm.
BRUSHTAIL POSSUM *Trichosurus vulpecula*						
Fortyseven (BOI)	1	?	D. Taylor	1990	P	D. Taylor pers. comm.
Harakeke (BOI)	12	c1990	D. Taylor	1992	P	D. Taylor pers. comm.
Peach (Whangaroa)	11	?	D. Taylor	1990	P, T	D. Taylor pers. comm.
Pig (Lake Whakatipu)	110	c1975	R Hoetjes, B Barron	1990	P, T	B. Barron pers. comm.
Pigeon (Lake Whakatipu)	168	c1975	R Hoetjes, B Barron	1990	P, T	B. Barron pers. comm.
Tarakaipa	35	?	P. Brady	1991	P, T, D	B. Brown pers. comm.
GOAT *Capra hircus*						
Great Barrier [part]	4030	?	K. Broome	1986	S, D	Parks 1990
Great Barrier	27761	?	?	?	S	D. Agnew pers. comm.
Pourewa (East Coast)	42	<1950	M. Hockey	1992	S	?. Whiting pers. comm.
<u>STOPPED</u>						
HOUSE MOUSE *Mus musculus*						
Silver (Lake Hawea)	25	?	S Thorne, J Flemmin	1997	P	S. Thorne pers. comm.
Stevensons (Lake Wanaka)	65	?	S Thorne, J Flemmin	1997	P	S. Thorne pers. comm.
PACIFIC RAT *Rattus exulans*						
Mauipae (Coppermine)	80	?	D. McKenzie	1992	P	Tyrrell et al. 2000; R. Parrish pers. comm.
NORWAY RAT *Rattus norvegicus*						
Rakino (Hauraki Gulf)	148	1920-72	M. Lee	1997	P	M. Lee pers. comm.
RABBIT *Oryctolagus cuniculus*						
Motuihe (Hauraki Gulf)	179	?	C. R. Veitch	1997	P, T, D	Veitch 2000b
Quail (Lyttleton)	88	c1855	J. Trotter	1988	P	J. Trotter pers. comm.
CAT *Felis catus*						
Motuihe (Hauraki Gulf)	179	?	C. R. Veitch	1997	P	Veitch 2000b
Raoul (Kermadecs)	2938	c1850	C. R. Veitch	1972	T	Fitzgerald et al. 1991
RED DEER *Cervus elaphus scoticus*						
Secretary (Fiordland)	8140	<1965	J. von Tunzelman	1974	P, S	Mark & Baylis 1982
<u>UNSUCCESSFUL</u>						
HOUSE MOUSE *Mus musculus*						
Hauturu (Whangamata)	10	c1980	P. Thomson	1994	P	Glassey 2004
Matakohe (Limestone)	37	?	J. Craw	1997	P	Ritchie 2000
Matakohe (Limestone)	37	?	J. Craw	1998	P	Ritchie 2000
Matakohe (Limestone)	37	?	D. Kokich	2001	P	Brackenbury 2001
Mokoia (Lake Rotorua)	135	c1965	M. Wilke	1996	P	Owen 1997, 1998
Te Haupa (Saddle)	6	?	B. Green	1993	P	T. Wilson pers. comm.
NORWAY RAT *Rattus norvegicus*						
Rakino (Hauraki Gulf)	148	1920-72	McCrae, Ellis, Lee	1992	P	M. Lee pers. comm.
Rotoroa (Hauraki Gulf)	90	?	N. Stark	1991	P	M. Lee pers. comm.
RABBIT *Oryctolagus cuniculus*						
Quail (Lyttleton)	88	c1855	Banks Peninsula Pest Destruction Board	1982	P, S	D. Brown pers. comm.
BRUSHTAIL POSSUM *Trichosurus vulpecula*						
Allports (Marlborough)	16	<1980	T. Neal	1982	P, T	D. Brown pers. comm.

Table 3 Operations which resulted in the successful eradication of an alien mammal species from an island, but where the species re-invaded at a later date. Methods listed are: P=poison; T=trap; S=shooting; D=dogs. † = ongoing programme due to high reinvasion risk. § Reinvaded by Norway rats (*Rattus norvegicus*).

Location	Area	Date Introduced	Eradication Leader	Date Completed	Methods	Date re-invaded	Reference
RE-INVASION							
BLACK RAT *Rattus rattus*							
Duffers Reef (Marlborough)	2	<1983	D. Taylor	1983	P, T	c1990	D. Brown pers. comm.
Goat (Leigh)	9	<1977	T. Wilson	1994	P	1996	T. Wilson pers. comm.
Kauwahaia	1	?	G. Taylor	1989	P	1990s	Taylor & Cameron 1990
Koi (Hauraki Gulf)	0.28	?	M. Lee	1997	P, T	2000§	Lee 1999
Moturoa (BOI)	157	c1800	P. Asquith	1993	P	1997§†	Asquith 2004
Motutapere (West Coromandel)	45	?	P. Thomson	1996	P	2002	R. Chappell pers. comm.
Unnamed Cape Wiwiki A ('Snail Rock')	2.2	?	D Taylor, T. Shaw	1989	P	<1999§	Parrish et al. 1995
Unnamed Cape Wiwiki B ('Gorse Island')	1.2	?	D Taylor, T. Shaw	1989	P	<1999§	Parrish et al. 1995
NORWAY RAT *Rattus norvegicus*							
Motuhoropapa (Hauraki Gulf)	9	2001	G. Wilson	2001	P	2002	Wilson 2003
Motuhoropapa (Hauraki Gulf)	9	1996	I. McFadden	1997	P	2001	Cameron 1998
Motuhoropapa (Hauraki Gulf)	9	1987	I. McFadden	1991	P	1996	Cameron 1998
Motuhoropapa (Hauraki Gulf)	9	1983	P. Moors	1984	P	1987	Moors 1985b
Motuhoropapa (Hauraki Gulf)	9	1981	P. Moors	1981	P	1983	Moors 1985b, 1987
Motuhoropapa (Hauraki Gulf)	9	<1962	P. Moors	1978	T	1981	Moors 1981, 1985b
Motuihe (Hauraki Gulf)	179	<1987	?	1988	P	1997	Veitch 2002b
Moturemu (Kaipara)	5	?	I. McFadden	1994	P	2002	J. Russell pers. comm.
Moturoa (BOI)	157	c1800	P. Asquith	1993	P	1997§†	Asquith 2004
Otata (Hauraki Gulf)	22	<2001	G. Wilson	2001	P	2002	Wilson 2003
Otata (Hauraki Gulf)	22	1991	I. McFadden	1991	P	2001	Cameron 1998
Otata (Hauraki Gulf)	22	1980	P. Moors	1981	P	1991	Moors 1985b, 1987
Otata (Hauraki Gulf)	22	1956-57	P. Moors	1979	P, T	1980	Moors 1981, 1985b
Pakatoa (Hauraki Gulf)	29	?	M. Lee	1993	P	1997	M. Lee pers. comm.
Rotoroa (Hauraki Gulf)	90	?	M. Lee	1997	P	c1999†	M. Lee pers. comm.
STOAT *Mustela erminea*							
Adele (Nelson)	88	<1977	R. Taylor	1981	T	1982	Taylor & Tilley 1984
Maud (Marlborough)	309	c1980	W. Cash	1983	T	1989	Crouchley 1992
CAT *Felis catus*							
Motuihe (Hauraki Gulf)	160	c1800	Steve Boyle	1981	S	c1984	Veitch 1985, 2002b

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The eradication of coypus (*Myocastor coypus*) from Britain: the elements required for a successful campaign

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Abstract A population of coypus, *Myocastor coypus*, became established in Britain following escapes from fur farms. The population peaked in the early 1960's at possibly 200,000 animals. Numbers then fell sharply following the start of a co-ordinated trapping campaign and an exceptionally cold winter. This campaign was unsuccessful in eradicating coypus and the population began to rise again during the 1970's. As numbers rose, coypus caused significant damage to native vegetation, damaged flood defences and agricultural crops.

A second campaign was started in 1981 with the objective of eradicating coypus from Britain within 10 years. This campaign was successful and coypus were eradicated by 1989. One of the key features in the success of the campaign was the close linkage between the research and eradication programmes. The research provided estimates of population size to monitor the progress of the campaign and estimates of the variables necessary to model the population. The research also allowed improvements to be made in management techniques and trapping strategies.

The paper summarises the successful eradication campaign and identifies the elements that were key in getting the eradication campaign started and in it reaching a successful outcome.

Keywords: coypu eradication; *Myocastor coypus*; IAS eradication; Britain

HISTORY OF THE UK POPULATION

Coypus are native to South America and have been widely farmed for their fur. There have been numerous escapes from captivity and in some countries animals have been deliberately released to try and establish feral populations that could be cropped (Lever 1985). Feral coypus are now found in North America, the Middle East, Africa, Japan, and the Asiatic part of the former Soviet Union. In Europe there are widespread populations and they are particularly common in France, Germany, and Italy (Mitchell-Jones *et al.* 1999).

Coypus were originally introduced into Britain in 1929 for fur farming. The farms were not, however, profitable and by 1945 the farming of coypus had ceased in Britain. Farms were often little more than poorly fenced off ponds and streams and, as a consequence, escapes were reported from more than 50% of them (Laurie 1946).

From the original escapes coypus became established in two centres. One, based on a sewage works near Slough, disappeared without any known control in 1956. A second group probably originated from three farms near Norwich, close to the rivers Yare and Wensum, in East Anglia (Laurie 1946). This population eventually expanded to cover virtually the whole of East Anglia (Gosling and Baker 1989); a distribution, at its extremes, of approximately 190km from north to south and 150km east to west.

EARLY POPULATION CHANGES

Quantitative estimates of population size using population reconstruction techniques are only available after 1962 (Gosling *et al.* 1981). Before this, the information is anecdotal, but the population probably started in the mid-1930s and grew progressively with two major checks in the severe winters of 1946/7 and 1962/3 (Norris 1967). Centrally organised control started in 1962 and continued at various levels until the start of the eradication campaign in 1981. Numbers of coypus probably reached a peak in the late 1950s. The population was then believed to number 200,000 (Norris 1967) but this may have been an over-estimate (Gosling and Baker 1989).

DAMAGE

Coypus are generalist herbivores and feed on a wide range of native plants and crops. They generally select the parts of plants which contain the highest nutrient concentrations and, where these include basal meristems, the plant is often destroyed. As a result of such feeding, large areas of reed swamp were eliminated in the Norfolk Broads during the 1950s (Boorman and Fuller 1981). Coypus also favour

Pages 142-147. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

particular species, including the great water dock *Rumex hydrolapathum* and cowbane *Cicuta virosa*, which almost disappeared from large areas when coypu were abundant (Ellis 1965). They also damaged a wide variety of crops including cereals, brassicas, sugar beet and other root crops.

The most important damage in purely economic terms was caused by burrowing. Coypus dug extensive burrow systems into the banks of ditches and rivers which disrupted drainage systems and posed the risk of flooding in low-lying East Anglia. As damage by coypus began to increase alarmingly in the late 1950s, there was a widely based call for an official control campaign.

THE FIRST CONTROL CAMPAIGN 1962-1965

The damage caused by coypus led to two initiatives in 1962. The first was to establish the Coypu Research Laboratory in Norwich, and the second to launch a trapping campaign which was to run until 1965 (Norris 1967). Complete eradication was believed to be impossible, and the aim of the campaign was to reduce coypu numbers and confine the remainder to the Norfolk Broads in eastern England. By necessity this campaign was organised in advance of any results from the Laboratory. The area containing most coypus was divided into nine sectors which were trapped successively by a team of up to 14 specially employed trappers. They started at the outside of the control area and worked inwards towards the area where the density of coypus was highest, in the Norfolk Broads. There was also a large amount of trapping ahead of the campaign carried out by the employees of rabbit clearance societies and by some landowners. Outside the main control area, government pest control staff attempted to clear what were regarded as outlying colonies in co-operation with landowners.

It is possible now to see a number of flaws in the strategy; notably that the main trapper force spent much of its time in clearing relatively low density areas rather than attempting to maximise capture rates. Also, although the effect of immigration into cleared areas was considered, it was not given sufficient weight (Gosling and Baker 1989).

Events were also complicated by the winter of 1962/3; the coldest winter in Britain for over 200 years. The fall in numbers trapped over this winter suggested that 80-90% of all coypus were killed by the cold (Norris 1967). By the end of the campaign in 1965, over 40,000 coypus had been trapped and the main objective had been achieved.

In the absence of a contemporary demographic analysis, it was not clear to what extent trapping was responsible and in retrospect perhaps the main achievement of the trappers was to keep the numbers down to the low levels caused by the cold winter. In ignorance of the quantitative relationship between trapping effort and the population's response, the trapping force that remained was not sufficiently large to prevent an eruption in numbers when a run of mild winters occurred in the early 1970s.

THE COYPU ERADICATION CAMPAIGN 1981-1989.

In 1977 the government set up a committee, The Coypu Strategy Group, to advise on future policy relating to the control of coypus (Anon 1978). In contrast to the earlier campaign, information was available to the Group from the results of a long-term investigation of coypu population ecology, and this was used to plan the 1981 campaign. Over 30,000 coypus were dissected to get information about reproductive biology, age structures and the other information needed to reconstruct past populations and to try to understand why coypu numbers varied.

Results indicated that trapping explains more of the variation in adult populations than winter severity, although the two combined variables accounted for 80% of the variation in the change of coypu numbers (Gosling and Baker 1987). Not all this information was available in the late 1970s, but enough was known to provide an analytical background for simulation models of the population. Simulations were used to assess the effect of employing different numbers of trappers on the population under various climatic circumstances.

A range of these simulations (Gosling *et al.*1983) were available to the Coypu Strategy Group and the option recommended was an attempt to eradicate coypus with a force of 24 trappers. Before the recommendations were accepted by the Government, one more important feature had been demonstrated: that it was possible to eradicate coypus by cage trapping. This was achieved in an exercise carried out on 30km of the river Yare, to the west of Norwich, which included Surlingham Broad. It was possible to demonstrate that coypus could be eradicated by cage trapping on a realistic scale across a range of wetland habitats used by coypus (Gosling *et al.* 1988).

The eradication campaign started in April 1981. Taking into account the reasonable expectation of improvements in trapping techniques and other equipment, it was decided to attempt eradication within ten years. The management and funding of control was also changed. A reconstituted Coypu Control Organisation employed 24 trappers and was funded by the Ministry of Agriculture, Fisheries and Food (50%), Anglian Water Authority (40%) and Association of Drainage Authorities (10%). The

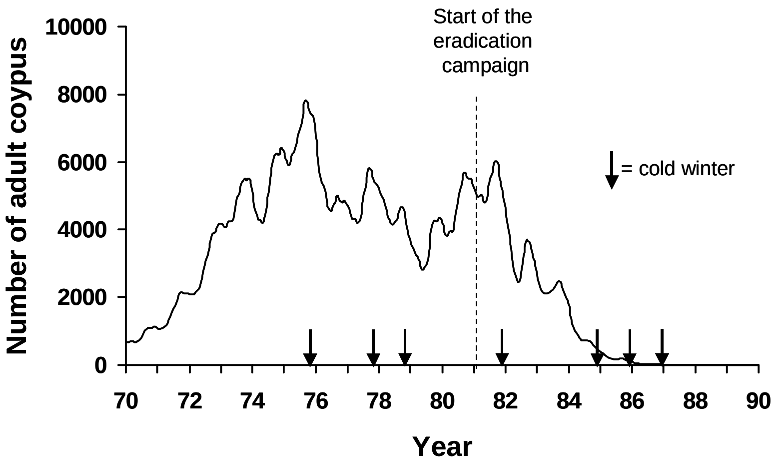

Figure 1 Number of adult coypus present in Britain between 1970 and 1990, reconstructed using the continuous retrospective census technique.

direction of the trapping campaign was also placed under the control of a small management committee that included a representative from the Coypu Research Laboratory. Wider stakeholder interests were represented on a Co-ordinating and Advisory committee, which met periodically to review progress with the campaign and offer guidance.

Staff from the Coypu Research Laboratory gave technical guidance to the control organisation throughout the campaign. An example is the scheme used to deploy trapping effort. Deployment was adjusted every three months using recent capture/trapping effort ratios in each of eight strategic regions. This ratio was weighted to different extents so that effort could be concentrated on high density areas early in the campaign and deployed more widely later on. However, some control was carried out throughout the population, in contrast to the sweep approach adopted in the first campaign against coypus (Gosling and Baker 1989).

The technique used was cage trapping. Traps were inspected every day and any trapped coypus were shot. This technique had the advantage that any non-target animals could be released unharmed. Because of this it was possible to get the co-operation of all landowners, including those with conservation and game interests; this is essential where the objective is whole population removal. During the campaign the trappers achieved an average annual trapping effort of 216,000 trap nights (a trap night is one trap set for one night) (Gosling and Baker 1987) and about 34,900 coypus were caught or otherwise accounted for (Gosling and Baker 1991).

Various improvements were introduced by the Laboratory, including the use of traps on baited rafts. Field trials showed that these were at least 50% more effective than traps set on land and non-target captures were also significantly reduced (Baker and Clarke 1988). Following this work, over 600 rafts were deployed.

The Laboratory also monitored the progress of the campaign both by field checks and by reconstructing the population. The process used to reconstruct the population was a continuous retrospective census, described in detail by Gosling *et al.* (1981). The technique was based on knowing the majority of the adults that were killed and being able to age a sample of these every month using the weight of the dried eye lens to determine age (Gosling, *et al.*1980). A reconstruction of the population between 1970, and the capture of the last coypu known to be caught from the wild (December 1989) is shown in Fig. 1.

There were around 6,000 adult coypus in 1981 at the start of the eradication campaign, but they had effectively been eradicated by the end of 1989. The campaign was helped by an above average number of cold winters. However, it is important to appreciate that cold weather itself would never eradicate all the coypus from Britain; as shown by the recovery after the exceptionally cold winter of 1962/3.

Even when the main technical problems in the operation have been solved, why should the trappers attempt to succeed in an eradication exercise when doing so would also eradicate their jobs? The scheme devised was to restrict funding to a maximum of ten years, and promise the trappers a bonus of up to three times their annual salaries if they succeeded in eradicating the coypu population. The bonus declined progressively after six years to encourage an early end to the campaign. It is impossible to judge the precise effect of this scheme, but we believe it was an essential element. In the end the trappers gained an almost maximum bonus.

It was also necessary to have an independent check on whether or not coypus had been eradicated as the incentive scheme had the risk of 'encouraging' trappers not to report kills which would potentially reduce their bonus. Laboratory staff carried out an independent check throughout East Anglia for coypus for the last four years of the campaign. The technique used was to put out rafts baited with carrots and check these for signs of coypus, such as droppings and teeth marks. The advantage of this technique was that, unlike using cage traps which must be checked every day they are set, a raft need only be checked once every week to ten days. This allowed a much greater area to be surveyed than would have been possible using additional trapping.

A number of automatic camera rafts were also used to confirm the presence of coypus (Gosling 1989). These were rafts that had an infra-red beam running along each side of the raft, breaking the beam triggered a camera which then took a photograph of the animal that had climbed onto the raft. This additional technique was helpful to provide additional 'proof' of any coypus that might remain in the wild as such animals could potentially have resulted in the trapping force having a significantly reduced bonus.

As you cannot prove a negative, the success of an eradication campaign will only be confirmed some long time after it has actually been achieved. For management purposes success criteria need to be established at an early stage. In this campaign 21 months (a 12 month period plus a nine month period) without any coypus being caught or found was deemed to provide sufficient evidence to disband the trapping force. It was also recognised that it would be unreasonable to start this period again if, for example, a single coypu was caught after a year. In this case trapping would continue throughout the 9 month block and the campaign would finish after there had been a further 6 month period without any coypu being found. The size of the bonus earned by the trappers would be calculated from the date that the 'last' coypu was caught, that determined the end of the control campaign.

In January 1989, 21 months had passed without any coypus being trapped (although two elderly male animals were killed by cars) and the Eradication Campaign officially ended (Gosling 1989). The Coypu Control Organisation was disbanded and the trappers were paid their bonus, however, it was recognised that it was likely that a few coypus would still remain. To help find any remaining animals, three field staff were retained by the Coypu Research Laboratory to search for them. In December 1989 this team confirmed the presence of, and subsequently trapped, what was to be the last coypu found in the wild in Britain. The systematic field effort by the Coypu Research Laboratory ceased in March 1992; eradication had been achieved.

DISCUSSION

The successful coypu eradication campaign would not have been undertaken without detailed technical assessments of the effort, costs and likely chances of success. These could only have been achieved by a long term study of population ecology, targeted to a particular control application. The research also allowed operational experience to be gained and it is significant that the arguments for such practical details as the incentive bonus scheme came from biologists. Population trends and the results of field checks were passed back to the control organisation and helped to direct the campaign and to stimulate the efforts of the trappers.

It is possible to identify at least 7 features that were key to being able to set up an effective eradication campaign and to bring it to a successful conclusion.

1) A clear case for eradication could be made. In this case there was clear damage to native flora, crops and drainage interests. It was considered that this threat would remain and that in the longer term successful eradication would cost less than continuous control (Anon 1978).

2) A viable and costed strategy existed. Research into coypu biology and population dynamics and the computer simulations that resulted, allowed the size of the trapping force necessary to achieve eradication to be estimated. The possible time frame within which this might be achieved could also be assessed, although it was recognised that the speed of eradication would depend on the severity of future winters (clearly unpredictable) so accurate prediction would be impossible. Knowing these parameters and the equipment needed to support a trapper allowed realistic estimates of likely future costs to be made. A successful trial eradication exercise, at a realistic scale, gave confidence to those recommending a way forward and those funding the exercise that it could achieve its objectives. There was also the precedent of a successful campaign to eradicate muskrats, *Ondatra zibethicus*, from Britain in the 1930's (Warwick 1934; 1940; Munro 1935; Sheail 1988).

3) An acceptable control technique was used. Cage trapping was demonstrated to be a viable technique for the eradication campaign. Coypus were caught alive and then humanely killed by a single shot to the head from a 0.22 calibre pistol. The technique allowed non-target species to be released unharmed and was one that was generally acceptable to the public at the time. There was very little interference with the trapping campaign although it is likely that protests from 'animal rights' campaigners would be

more significant if a similar campaign was to be repeated now.

4) Existence of sound management structure and finances. Eradication is a long term project and a ten year project plan was put into place to support the campaign. This had the backing of central government and key local interests. It was coupled with local management that had direct input from the Coypu Research Laboratory where research was undertaken into control strategy and techniques.

5) The progress of the campaign could be monitored and there was a process for continual improvement. The continuous retrospective census allowed the progress of the campaign to be monitored. It is essential for the maintenance of a long term campaign that funding bodies have clarity about progress. They would clearly wish to have the opportunity to reconsider their position if it seemed that a campaign was not achieving its objectives. The response of the population to changes in trapping strategy or improved trapping practice can be monitored and the deployment of the trapping force can be altered appropriately.

6) There was an incentive for the trappers to achieve their objective. Those carrying out an eradication exercise will potentially be unemployed if they are successful, this will tend to mitigate against the campaign achieving its objective. Many of the trappers will also spend a long time working hard but not catching anything during the last part of a successful eradication campaign. As catching the target animal is a significant source of positive feedback for a trapper and this may be absent for many months or possibly years, the motivation of the trapping force during the final stages of the campaign is very important. In the coypu eradication campaign an incentive bonus was offered at the start of the campaign to be paid in the event of successful eradication within a ten year period. This appeared to be successful in motivating the trapper force. Other management options may be appropriate in other circumstances but the strategy that will be adopted towards the end of a campaign needs to be considered at an early stage in planning an eradication campaign.

7) It was possible to define the successful end to a campaign. A practical definition was adopted to determine when the eradication campaign was to finish. It is important that trapping effort is not reduced too early or eradication may be jeopardised.

The constructive interaction between applied biology and a centrally organized control operation that characterised the coypu eradication campaign has the potential for wide application in any extensive pest control operation and may be essential for the successful removal of a well established introduced mammal. The success of the campaign should also provide encouragement to future campaigns by confirming that eradication of an introduced mammal is possible even for those with a widespread population.

ACKNOWLEDGEMENTS

The Coypu Research Laboratory was funded by the then UK Ministry of Agriculture, Fisheries and Food. The work of the Laboratory was directed by Professor Morris Gosling throughout the successful eradication campaign.

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Prediction of range expansion and optimum strategy for spatial control of feral raccoon using a metapopulation model

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Abstract A meta-population model is often used to predict spatial distribution of the target species in a regional landscape. Parameters necessary to build a metapopulation model can be obtained from a set of two distribution maps of the same region surveyed in two different periods. Such an approach is suitable to predict range expansion based on the information obtained in the early stage of introduction, where detailed information is lacking.

Range expansion of the feral raccoon (*Procyon lotor*) in Kanagawa Prefecture, Japan was predicted by a meta-population model. Habitat suitability was evaluated from the relation between vegetation and raccoon distribution. Colonisation kernels were estimated from two distribution maps based on surveys conducted in 2001 and 2004. The reliability of the model was evaluated by comparing actual history and simulation.

The optimum strategy for spatial control was identified using this simulation model. The current raccoon population was divided into isolated subpopulations by cluster analysis. One or several subpopulations were hypothetically removed in the simulation of range expansion, and the future reduction of the whole distribution range was calculated. All combinations of subpopulation removal patterns were examined and the most cost-effective removal pattern was identified.

Habitat analysis showed that raccoons preferred landscape with forest patches. Raccoons colonised up to a distance of 2.8km at 50% probability per three years. Results of simulation corresponded to actual history. It was predicted that raccoons will spread over the whole Kanagawa Prefecture by the year 2019, and will reach central Honshu Island in about 2064.

There were five isolated populations in 2004. The benefit / cost ratio changed according to time after subpopulation removal (time horizon). The most cost effective strategy short term was to eradicate isolated small marginal populations. The most cost effective strategy, mid term, was to eradicate all populations located in the marginal front and the most cost effective strategy long term, was to completely eradicate the whole population. For all time horizons, eradication of the central subpopulation was the worst strategy.

Keywords: Kanagawa Prefecture; Japan; metapopulation model; raccoon; *Procyon lotor*

INTRODUCTION

In the early stage of introduction of alien species, we usually have to predict range expansion based on limited information (Fig. 1). The most easily available information is the presence of the focal alien species in a region. The next step will be a distribution map of the species. Information on habitat preferences can also be obtained in the early stage of research. Estimation of population density and other population parameters, such as population growth rate and mortality, on the other hand, require much effort.

A distribution map describes presence or absence of the target species. There is a metapopulation model dealing with presence and absence of local populations in a fragmented landscape (Hanski 1998). Distance-dependent colonisation and extinction probability in a habitat patch are essential parameters, but detailed information such as population density, fecundity, mortality and longevity of individuals is not required

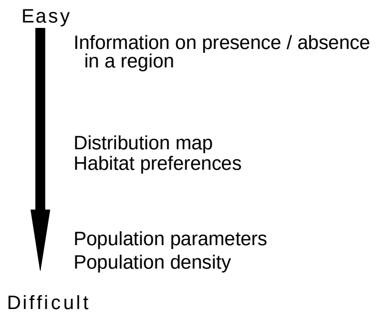

Figure 1 Availability of information as research progresses

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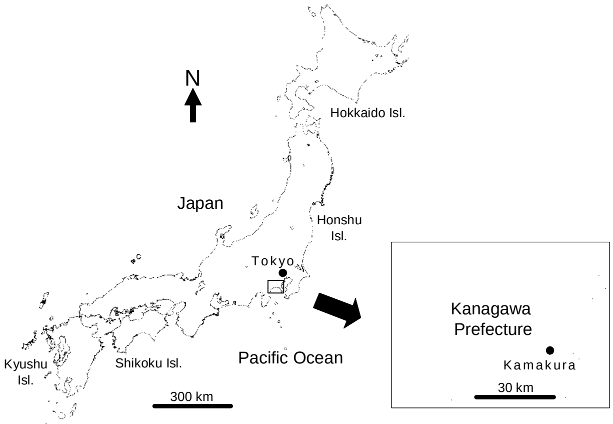

Figure 2 Location of study region.

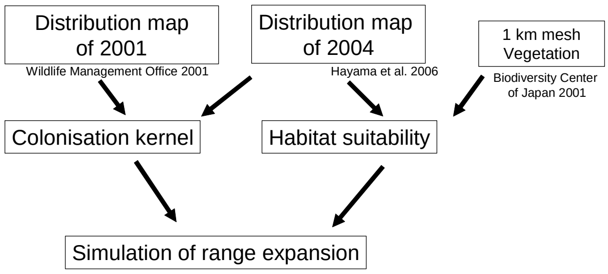

Figure 3 The procedure for simulation of range expansion based on two snapshots of raccoon distribution map and environmental data.

in the metapopulation model. One set of distribution maps surveyed in two different periods provides enough information for the modelling. Habitat suitability is identified using a land cover map and a distribution map of the target species. The colonisation kernel and the probability of extinction from a habitat patch are determined from two distribution maps obtained at two different periods.

The optimum spatial control strategy for an actively spreading alien species needs to be determined in the early stage of introduction and it can be determined using the metapopulation model. The site of the first introduction is usually at the centre of the distribution range, where population density is usually high and damage to economic activities and native biodiversity are also obvious. Although a management programme is often conducted in such high density area, the alien species will continue to spread. In order to reduce future damage, we need to know what the priority areas for eradication are.

Raccoons (*Procyon lotor*) are native to North America, and were brought to Japan as pets. They are now naturalised in several locations in Japan (Japan Wildlife Research Centre 1998). The raccoon is a mesopredator and the major predator of bird eggs in fragmented landscapes in its native range (Schmidt 2003). In Japan, naturalised raccoons become the largest tree climbing predators and the largest predators in suburban and urban area, and can cause considerable ecological impacts (Ikeda 2002).

A precise model to predict raccoon population dynamics in fragmented habitat of its native range is available (Broadfoot *et al.* 2001). However, it requires several population parameters and the initial population size in each habitat patch. These values are currently hard to obtain for populations in Japan, and this approach was hence not taken in this study.

In this study, the range expansion of feral raccoon was predicted based on two distribution maps (Kanagawa Prefecture 2001, records of 1999 to 2001, an early version of Hayama *et al.* 2006, record of 2003 to 2004) using a metapopulation model and the optimum strategy for spatial control was identified using this model.

METHODS

History of feral population in Kanagawa Prefecture

A TV animation programme with a raccoon character was broadcasted in Japan in the 1970s, and many raccoons were imported from North America as pets. Since then naturalised raccoons have been found in several regions in Japan where land cover is a mixture of urban and forest areas. The raccoon was naturalised in Kamakura City, Kanagawa Prefecture, around 1988 and several female raccoons with young were found in 1990 (Fig. 2, Nakamura 1991).

Raccoon resource use and habitat preferences in Kanagawa

The raccoon is a middle size omnivorous mammal. In its native range of North America, it needs cavities in trees, snags, logs, rock piles, underground burrows for shelter and dens, and thus it usually requires forest and shrub lands and avoids industrial areas (Anon 1995, Broadfoot *et al.* 2001). The mixture of forest and other land covers (residential areas and farmlands) results in high population density due to high food availability in residential areas and farmland, and den site availability in forest (Hoffman and Gottschang 1977, Oehler and Litvaitis 1996).

As a preliminary analysis, habitat suitability for raccoons in Kanagawa Prefecture was assessed using the 2.85 x 2.3km grid map of raccoon distribution in 2004 (an early version of Hayama *et al.* 2006, Fig. 3). Percentages of forest cover, farmland and town were calculated based on 1km mesh vegetation data (Biodiversity Centre of Japan 2001) using free GIS

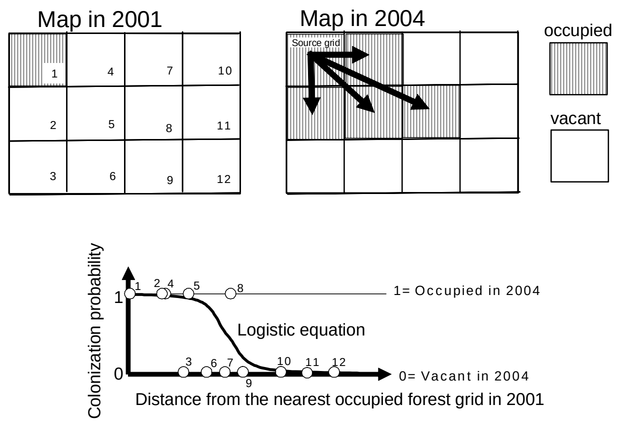

Figure 4 Colonisation kernel was determined using two distribution maps surveyed in 2001 and 2004 (above). Logistic regression was used for colonisation kernel (below). We call a grid "occupied" if raccoon is present and "vacant" if raccoon is absent.

software (Koike 2004). Land cover of the grid was divided in two categories 1) without forest (farmland, residential or industrial areas only), and 2) with at least 10% forest cover. Since the population started to spread from Kamakura City, and has not yet expanded to the entire prefecture, only the area within 20km from Kamakura City was analysed. We call a grid "occupied" if raccoon is present and "vacant" if raccoon is absent. The fraction of grids occupied by raccoon was 82% for the grids with forest, and 60% for the grids without forest. Raccoons significantly preferred the grids with forest ($P<0.05$, Fisher's exact test).

Based on this information, grids with at least some forest (forest cover >10%) were considered as raccoon breeding habitat. Other areas (towns or farmlands without forest) were considered as habitats where raccoons may stay temporarily and feed, but where they do not reproduce.

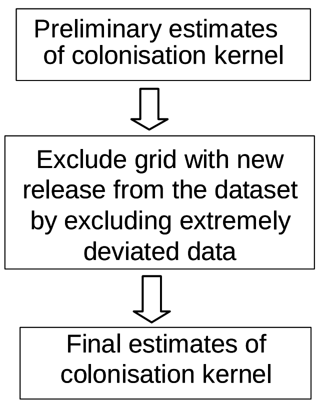

Figure 5 Removing new releases from the analysis is necessary to identify the colonisation kernel accurately.

Colonisation kernel

Each grid (2.85 x 2.3km) was considered as a habitat patch for the metapopulation model (Fig. 4). Raccoons colonise from an "occupied" grid where a raccoon population is present to a "vacant" grid where raccoon is absent. Among occupied grids, grids with at least some forest were considered as source grids where raccoons breed and where new individuals disperse from. Grids without any forest were not considered as source grids. The probability that raccoons colonise a vacant grid, $P(x)$, decreases with the distance from the nearest source grid, x. A logistic function is often used to approximate this (Komuro and Koike 2005),

$$P(x) = \frac{e^{kx+l}}{1+e^{kx+l}} \qquad (1)$$

where k and l are regression coefficients. This function defines a colonisation "kernel" around a given source grid.

In this study, the occupied forest grids in 2001 were considered as source grids, and success or failure of colonisation was judged from the distribution map in 2004. The centre-to-centre distance from the nearest source grid (occupied in 2001) to a given grid (destination grid) was calculated for all grids. One record in the dataset corresponded to one destination grid. Variables in the dataset were presence (value=1) or absence (value=0) of raccoons in 2004, and the distance from the nearest source grid. Logistic regression (SPSS 10.0J) was used to obtain the colonisation kernel (Fig. 4), assuming presence or

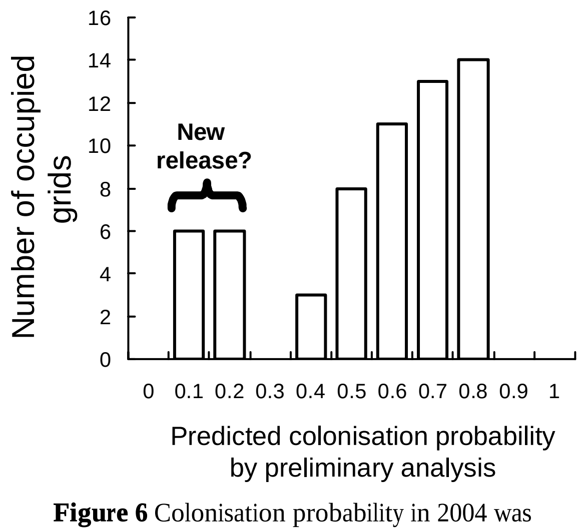

Figure 6 Colonisation probability in 2004 was predicted based on the distribution map in 2001, using the preliminary "colonisation kernel". Several grids had a low colonisation probability, although they were occupied in 2004. This presence of raccoons was considered to be the result of new releases, and such grids were excluded from the final estimation of colonisation kernel.

absence in 2004 as dependent variable, and distance from the source grid as independent variable. The whole dataset was divided into two datasets according to vegetation; destination grids having at least some forest, and those that did not have any forest. Two colonisation kernels were obtained: colonisation with breeding between forest grids (from forest grid to forest grid), and temporal dispersal to non forest areas (from forest grid to other areas).

In some forest grids, raccoon was observed in 2001, but not in 2004. Extinction of a non viable population (e.g. release of only one individual), or failure to detect the presence of raccoons might cause such phenomenon. We included such data in the analysis, so that extinction and failure of detection were considered in the colonisation model to some extent.

A new release of one or more can cause a new colonisation into a highly isolated grid. If we include such data, the resulting colonisation kernel will have a very long tail, and indicate a very long dispersal distance. In Kanagawa Prefecture, raccoons caught in Kamakura City as a pest were sometimes released in forests in the northwest of the prefecture, because people refrained from killing raccoon (Kaneda, personal communication). In order to exclude such additional releases, we analysed the relation between predicted colonisation probability and actual colonisation (Fig. 5). The number of occupied grids should decrease where predicted colonisation probability was small (i.e. distant from source grid), however, there were several grids colonised in 2004 that had no source grid close to them (Fig. 6). Such grids were considered as new releases, and excluded from the final analysis.

Identifying subpopulations

Eradication of a small part of a large population will soon be replenished by individuals from surrounding areas. Thus we divided the whole population in the prefecture into several subpopulations based on isolation distance. Grids occupied by raccoons in 2004 were extracted and geographical distances between these grids were calculated for all combinations of grids. The resulting distance matrix was analysed by the cluster analysis of Minna de GIS (Koike 2004). The clusters of grids that were separated from each other by more than the calculated maximum distance of dispersal ability were defined as subpopulations.

Each subpopulation is expected to have an independent origin of released raccoons. However, since raccoons might be released several times at easily accessible sites (e.g. a forest edge close to an urban area), and feral raccoons caught in towns might also be released in such sites, individuals of a given subpopulation might have various genetic origins. Genetic analysis was hence not used to divide subpopulations.

Simulation model

All grid cells were divided into two categories: with forest, and without forest. The time step used was three years. In each step, the distance from the nearest source grid, x, was calculated for all grids, and the probability to colonise the focal grid, $P(x)$, was evaluated. Presence or absence of raccoons during the next step was then determined stochastically according to $P(x)$. Since it was a Monte Carlo simulation, the results of the simulation differed from trial to trial. An already occupied grid could become empty depending on the probability of 1 - $P(0)$ (representing extinction from the grid). Minna de GIS (Koike 2004) was used for simulation.

Evaluating the reliability of the range expansion simulation

A simulation of past range expansion was made to evaluate reliability of the model using the same grid (2.85 x 2.3km) as the actual distribution data. In this simulation, the initial population started from Kamakura City, where the first naturalisation was found, and spatial spread after 30 years was simulated. The simulated raccoon distribution was compared with the actual distribution of a subpopulation around Kamakura City. Reliability of the simulation was evaluated based on a similarity index, S_t, between the actual raccoon distribution in 2004 and the simulated distribution:

$$S_t = \frac{a_t}{a_t + b_t + c_t} \qquad (2)$$

where a_t was the number of grids where raccoon was present both in the actual data and in the simulation, b_t was the number of grids where raccoon was not observed in the actual situation but was present in the simulation, c_t was the number of grids where raccoon was observed in the actual situation but was not present in the simulation, and t was the year after initial release in the simulation.

Prediction of distant future

Raccoon distribution in the distant future was simulated taking the raccoon map of 2004 as the initial state. In surrounding areas outside Kanagawa Prefecture, there were several areas where raccoons

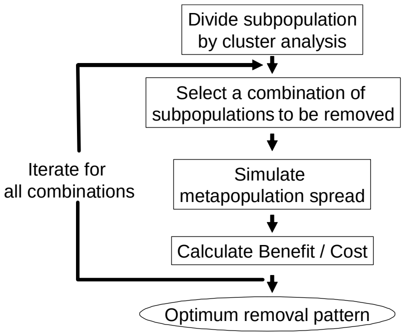

Figure 7 The procedure to identify the optimum pattern for eradication.

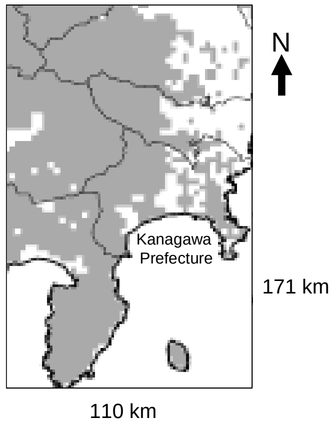

Figure 8 The area (171 km in south-north direction, 110 km in east-west direction) used to determine optimum spatial control strategy. Grey area represent grid with at least some forest, where raccoon can breed.

also naturalised; however, because reliable information was not available for them, we only considered the population in Kanagawa Prefecture.

The 2.5 x 2.5km grid on a transversal Mercator projection was used for this simulation, because the 2.85 x 2.3km grid from the distribution survey is an approximation of the grid defined by latitude and longitude, and will cause an accumulation of errors if the analysis is done for a large area.

Strategy for optimum spatial control

In the simulation, one or several subpopulations were removed completely at one time, and the resulting distribution change was calculated (Fig. 7). We considered a 110 x 171km area including Kanagawa Prefecture (Fig. 8), with 2.5km x 2.5km grids. The simulated removal was carried out for all combinations of subpopulations. Contribution of the removal after t years from the treatment, B_t, was calculated as:

$$B_t = N_t - R_t \tag{3}$$

where N_t was the number of grids occupied by raccoons after t years without any removal treatment, and R_t was the number of grids occupied after t years with removal treatment (Knowler and Barbier 2000).

The cost of eradication for one grid is not known, but the total cost of treatment should be proportional to the number of grids eradicated. It is difficult to calculate an absolute amount of money, but we can compare the cost of various eradication strategies using the number of eradicated grids instead of an amount of money. We optimised the benefit per cost ratio,

$$\text{Benefit / Cost} = \frac{B_t}{E} \tag{4}$$

where E is the number of eradicated grids.

The benefit/cost ratio (equation 4) means the reduced number of occupied grids in future, caused by the eradication of one grid. Eradication of one grid is an investment that will reduce the eradication effort necessary in the future. Compound interest for t years is expressed as

$$(1+r)^t = \frac{B_t}{E} \tag{5}$$

Annual yield (interest), r, was calculated as

$$r = \left(\frac{B_t}{E}\right)^{\frac{1}{t}} - 1 \tag{6}$$

For the calculation, the raccoon population of 2004 was divided into subpopulations. For all combinations of eradicated subpopulations, the benefit/cost ratio was calculated and the most cost effective eradication pattern was identified. Since the simulation was stochastic, it was iterated 50 times, and the average value of the cost/benefit ration was used. Minna de GIS (Koike 2004) was used for optimisation.

RESULTS

Colonisation kernel

Two colonisation kernels of forest-to-forest, and forest-to-other movements were similar (Fig. 9). Colonisation between grids with forest determines the ability for spatial spread of a raccoon population. The colonisation probability decreased to 50% at a distance of 2.8km from the source grid, and the colonisation probability was close to zero at 10km distance.

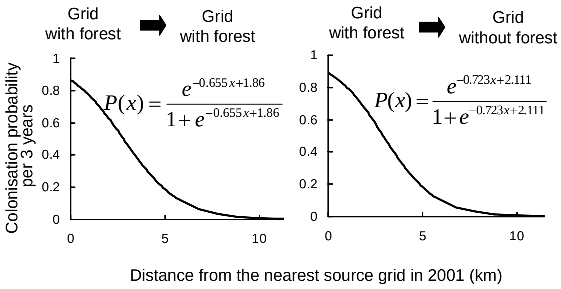

Figure 9 Colonisation kernels. Forest-to-forest colonisation represents true colonisation through breeding, and forest-to-other represents temporal use.

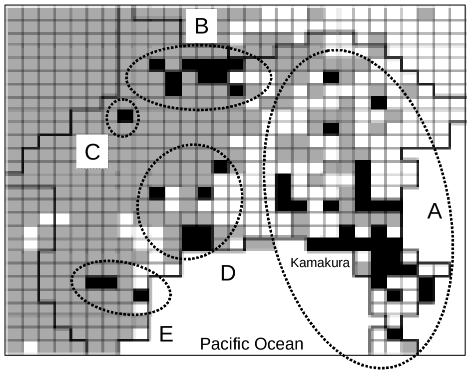

Figure 10 Subpopulation of Kanagawa Prefecture in 2004. Grey grids are habitat where raccoon can breed (forest), and black grids were occupied by raccoon as of 2004.

Subpopulation

Raccoons scarcely moved over a habitat-to-habitat distance longer than 10km in three years and an isolation distance of 10km was hence used to identify subpopulations. Five isolated subpopulations were found in Kanagawa Prefecture (Fig. 10). Kamakura City was at the centre of the largest subpopulation (**A**). The smallest subpopulation was in an isolated single grid (**C**).

Reliability of simulation

In the simulation that started from an original population at Kamakura City, the raccoon population spread continuously and reached the northern border of the prefecture after 30 years (Fig. 11). The similarity between simulated distribution and actual distribution of the subpopulation **A** was highest at 18 years after initial release (Fig. 12). It suggests that in this simulation the raccoon population should start to spread in 1986. This agreed with the actual history where naturalised raccoons were recognised in 1988 (Nakamura 1991), and the simulation model was considered to be reliable.

Future distribution of raccoons

There is a hilly suburban area between Kamakura and the northern border of the prefecture and a few isolated forests exist on steep hill-sides. Such suburban landscape may provide a corridor for raccoons. Most habitats in Kanagawa Prefecture are predicted to be occupied by 2019. Raccoons are predicted to reach central Honshu Island around 2064, and a large part of this island will be occupied after 200 years (Fig. 13).

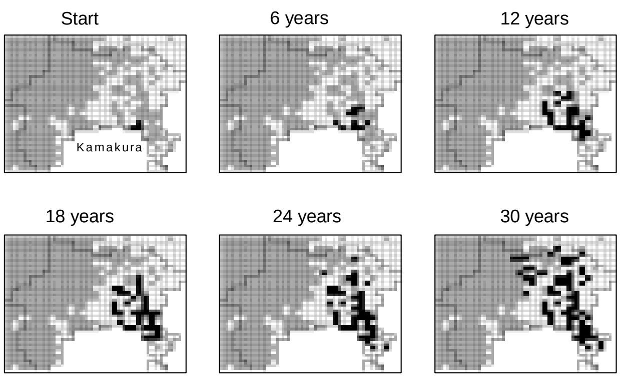

Figure 11 A result of simulation to reproduce past range expansion. Initial population was situated at the Kamakura City.

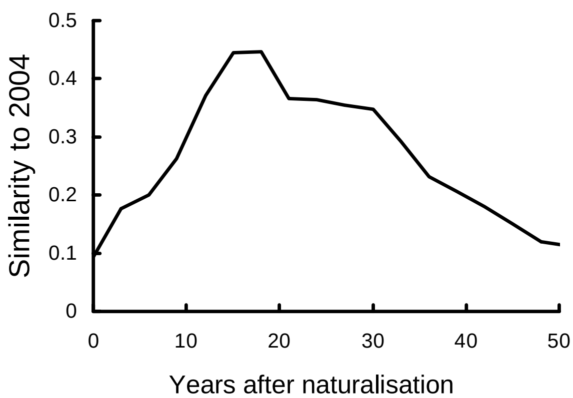

Figure 12 Similarity index (equation 2) between actual raccoon distribution (Hayama et al. 2006) and the result of a simulation of past range expansion (Fig. 11). Since an already occupied patch will go extinct stochastically with the probability of $P(0)=0.87$, the theoretical maximum similarity for the same distribution range is $P(0)^2=0.75$.

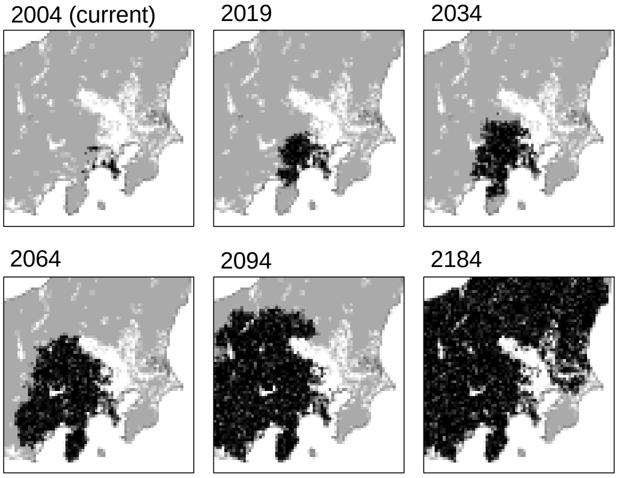

Figure 13 Long term prediction of raccoon range expansion.

Control strategy

The benefit/cost ratio was greatly different depending on the number of years since treatment (time horizon). Although all combinations of subpopulations were evaluated we illustrated typical four cases (Fig. 14); 1) eradication of a marginal isolated subpopulation (subpopulation C), 2) eradication of subpopulations at the front of spread (B, C, D and E), 3) eradication of central subpopulation (A), and 4) eradication of whole population (A, B, C, D, E). Other combinations produced intermediate results and did not produce an optimum benefit/cost for any time horizons.

The eradication of the marginal isolated subpopulation was the most cost effective treatment in the short term (less than 20 years), although the results fluctuated due to stochastic simulation. Eradication of subpopulations at the front of spread was the most cost effective strategy in the middle term (20-60 years). Eradication of all subpopulations was not cost effective in the short term, but, it was the most cost effective treatment in the long term (longer than 60 years). Eradication of the central subpopulation only was the worst strategy for all time horizons.

DISCUSSION

Prediction of range expansion

Prediction of the future distribution of invasive alien species is key information for decision making. Impact assessment should be based on future distribution ranges. If an invasive alien species is spreading, people living in the regions where the species will appear in the future should take countermeasure as soon as possible, because they are potential victims. Simple simulation, using a metapopulation model, was reliable enough to predict the range expansion for an alien species.

There are many small naturalised populations of raccoons in various locations in Japan (Japan Wildlife Research Centre 1998). We did not include such populations in our simulation, and this means that the raccoons will spread sooner than in our simulation (Fig. 13). A nationwide distribution survey would be necessary for a more precise prediction.

Strategy for spatial control

According to the result in this study, isolated small populations (probably newly established) should be removed as the highest priority. It is possible to reduce the rate of spread efficiently with such treatment. The traditional control strategy of removing many individuals from the central population was the worst strategy, because such habitats will be re-populated with individuals from surrounding subpopulations. Contrary to this, in the case of coypus in the United Kingdom, removing individuals from the densest part of the population caused successful eradication (Baker 2006). However, coypus in the United Kingdom had a limited distribution range, and the dense part of the population might have worked as the source population for the marginal area. In the case of raccoons in Kanagawa Prefecture, on the other hand, the population is spreading rapidly, and eradication from the centre is not effective.

In reality, complete removal may be difficult for very large subpopulation. However, isolated small

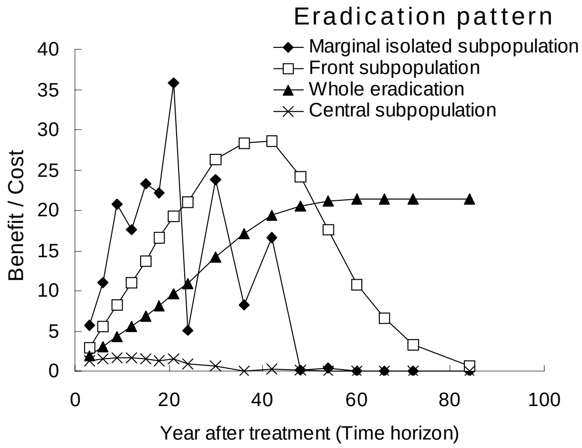

Figure 14 Benefit / cost ration of four patterns of eradication. Benefit / cost ration represents reduced number of occupied grids caused by eradication of one grid in 2004.

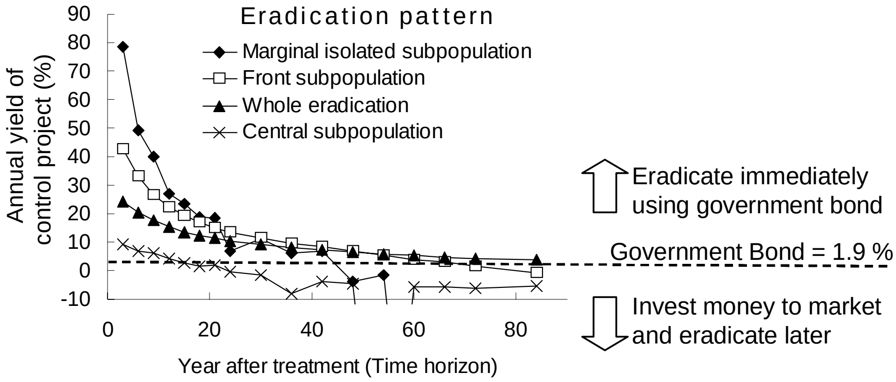

Figure 15 Annual yield of eradication project calculated from range expansion simulation.

subpopulations with one or several reproductive females may be removed easily. As shown in this study, such eradication programmes are very effective and they are highly recommended.

The annual yield of the raccoon control project suggested was higher than that for current government bonds of Japan (as of 2004) in many cases (Fig. 15). If the yield is larger than the interest of government bonds, eradication projects should be carried out immediately using funds from government bonds, to reduce government expenses in future. However in reality, the Japanese government can not issue a new bond to finance such eradication projects, because they already issued too many bonds as of 2004.

Since the geographical range used in the simulation was finite (Fig. 8), occupation of all habitats by raccoons reduced the effect of the initial removal treatment, and the benefit/cost ratio showed a humped shape along the time horizon (Fig. 14). The ratio increased with time, and then decreased in the distant future. If the habitat were to be infinite, the benefit/cost ratio might never decrease; however, the geographical area is always finite in the case of all islands and even for all continents, thus the humped shape in the graph has general relevance.

If an island, already occupied by an alien species, is considered, eradication of the alien species may be postponed from the financial point of view, because the annual yield of the eradication project is lower than that of government bonds. However, if we consider the extinction risk of native species caused by the alien species, the eradication project should be started immediately, given that extinction causes irreversible loss. Benefit should be evaluated not only by the reduction of area where an alien species is present, but by the reduction of the extinction risk of native species.

In this study removal treatment was assumed to occur only once, however, simulation with iterative removal treatments may provide a more realistic tool to plan a control strategy. In addition, models based on precise population dynamic processes will be required to estimate the absolute eradication cost and to evaluate the best eradication technique. Nevertheless a basic spatial control strategy can be constructed using the meta-population model.

ACKNOWLEDGEMENTS

The author thanks N. Ishii and T-H. Chen for valuable comments on the manuscript, H. Hayama and members of Kanagawa wildlife support network Raccoon project for use of unpublished data. Contribution of this NGO has been significant in raccoon issue in Japan.

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Reproductive responses of the mongoose (*Herpestes javanicus*), to control operations on Amami-oshima Island, Japan.

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Abstract The small Indian mongoose (*Herpestes javanicus*) was introduced to Amami-oshima Island in 1979. A control operation began in 2000 and is ongoing. In order to elucidate the effect of the control operation on mongoose populations, we examined reproductive characteristics, such as litter size, duration of the annual breeding period, and body mass of mature females during two different phases of population growth: phase I, during 1990 – 1999, before the control operation, and phase II, during 2001 – 2004, after the control operation. Data from phase II was further divided into two categories: phase IIa, representative of individuals living at the centre of the species' distribution where population density is highest, and phase IIb, representative of individuals living at the periphery of the species' distribution where population density is lowest. The duration of the breeding period in a given year for phase II (12 months) was longer than that for phase I (seven months, from March to September). The mean litter size, which was estimated from the number of placental scars and embryos, for phase IIb (2.93 ± 0.99) was significantly higher than that for phase IIa (2.55 ± 0.90) and phase I (2.26 ± 0.62) ($p<0.05$). The body mass of mature females, which is an index of physical condition, was significantly higher during phase II (491 ± 65.9) than that during phase I (462 ± 57.8) ($p<0.001$). The positive responses in reproductive traits observed after the inception of the control operation are thought to relate to better nutritional conditions brought about by decreased competition among individuals when population density was reduced. This study demonstrates the importance of considering individual responses to the decrease in population density that result from control operations.

Keywords: density dependence; *Herpestes javanicus*; litter size; pest control; small Indian mongoose

INTRODUCTION

The Ryukyu Archipelago in the south-western part of Japan, which includes Amami-oshima Island, became isolated from the Eurasian Continent about 1.5 million years ago, long before Taiwan and the main islands of Japan were separated from the Asian continent (Kizaki and Oshiro 1980) (Fig. 1). Consequently, Amami-oshima Island has a high level of species endemism and is considered to be one of the world's biodiversity hot spots (Sugimura 1988). However, much of the unique fauna on the island is now threatened with extinction as a result of predation by the introduced small Indian mongoose, *Herpestes javanicus*.

The native distribution of the small Indian mongoose ranges from Pakistan and northern India to southern China and the Malay Peninsula, as well as Hainan and Java Islands; in the west it extends to Iran and Iraq (Corbet and Hill 1992) (Fig. 1). This mongoose was deliberately introduced to tropical and sub-tropical regions as a control measure for rats or snakes in sugarcane fields. As a result, the species is now established in Jamaica (Espeut 1882) and numerous other islands of the West Indies (Hoagland *et al.* 1989), the Hawaiian Islands (Bryan 1938), Mauritius (Cheke 1987), Fiji (Gorman 1976), Ngazidja in the Comoro Islands (Louette 1987), and in the Adriatic Islands (Tyrtkovic and Krysufek 1990). These mongoose introductions failed to control rats and snakes, but the mongoose has been implicated in the devastation of the native fauna in these places (Baldwin *et al.* 1952, Seaman and Randall 1962, Nellis and Everard 1983, Coblentz and Coblentz 1985a). Today, The World Conservation Union (IUCN) lists the small Indian mongoose as one of the world's 100 worst invasive alien species (Lowe *et al.* 2000).

In Japan, the mongoose was introduced to the Okinawa Islands in 1910 (Fig. 1), and from there to Amami-oshima Island (Sekiguchi *et al.* 2001) in 1979, in order to control the native poisonous habu snake (*Protobothrops flavoviridis*) and the black rat (*Rattus*

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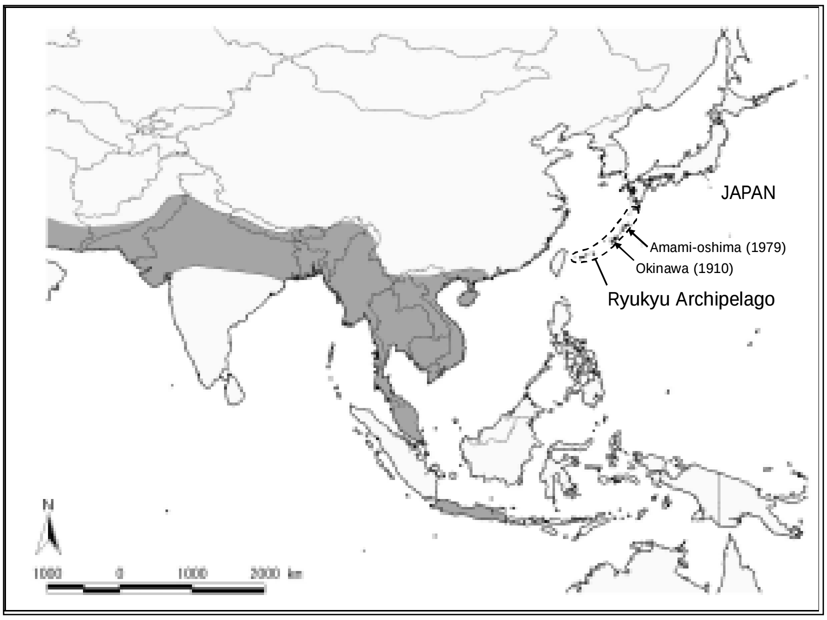

Figure 1 Native distribution (modified from Corbet and Hill 1992, grey area) of *Herpestes javanicus* and the location of two islands, Amami-oshima and Okinawa, in the Ryukyu Archipelago of Japan where *Herpestes javanicus* has been introduced (arrow). Parentheses indicate the year of introduction.

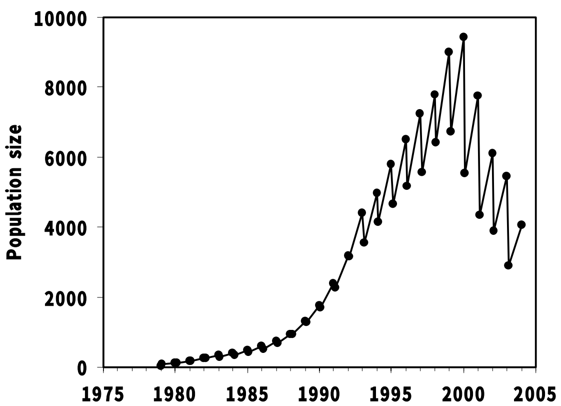

Figure 2 Estimated population size of *Herpestes javanicus* on Amami-oshima Island (Amami Wildlife Conservation Centre, Unpublished). Initial population in 1979 was 30, and continuous natural population growth rate of 40% per annum, (Ministry of the Environment 2003) from 1979 to 2004 was assumed. The number of trapped individuals was subtracted from the estimated population.

rattus) (Abe *et al.* 1991). Since 1979 the mongoose population on Amami-oshima has expanded in both number and distribution (Abe *et al.* 1991, Environmental Agency 2000). Concurrently, the distribution and abundance of the Amami rabbit (*Pentalagus furnessi*), Amami woodcock (*Scolopax mira*), and many other vertebrates and some invertebrates have been reduced or locally exterminated in areas where the mongoose has established (Abe *et al.* 1999, Environmental Agency 2000, Sugimura 2002, Yamada *et al.* 2000, Watari *et al.* 2006, Amami Wildlife Conservation Centre, unpublished data).

In 1996, the Environmental Agency and Kagoshima Prefecture conducted a preliminary investigation to collect baseline biological and ecological data in order to develop effective control strategies (Environment Agency 2000). The Ministry of the Environment began a control operation in 2000, which is ongoing.

In the planning of a control programme, reproductive rate of the target species is one of the most important ecological parameters to be considered to ensure successful eradication (Genovesi 2001). Density dependence of reproduction has generally been observed in many mammal and bird populations (Rödel *et al.* 2004). The female reproductive traits of the small Indian mongoose have been studied on several other islands where this species has also been introduced (Gorman 1976, Hoffman *et al.* 1984, Nellis and Everard 1983, Ogura *et al.* 2001, Pimentel 1955, Soares and Hoffman 1981), but none of these studies specifically considered the effects of population density. The objective of this study was to quantify the reproductive characteristics of the small Indian mongoose, especially litter size and the duration of the breeding season, and relate them to changes in its population density resulting from the control operation. The indirect effects of such control operations on mongoose reproduction are discussed.

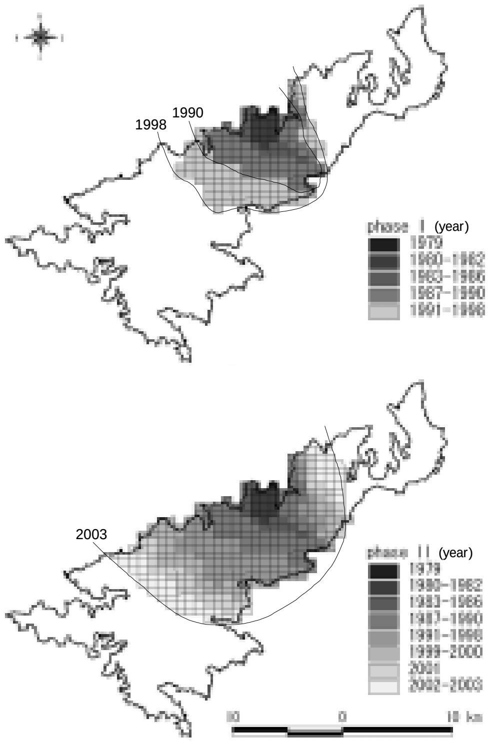

Figure 3 Distribution of *Herpestes javanicus* in phase I (upper) and phase II (lower) on Amami-oshima Island from introduction in 1979 until the end of each phase (modified from Abe *et al.* 1991, Environment Agency 2000, Ministry of the Environment 2004).

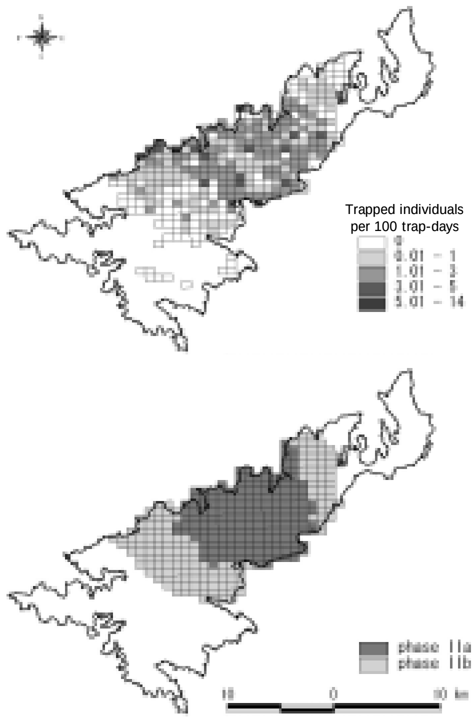

Figure 4 Relative population density (C100TD^{-1}) of *Herpestes javanicus* on Amami-oshima Island under the control operation conducted in fiscal 2003 (upper) (modified from Ministry of the Environment 2004). Based on relative density, the distribution area of phase II was divided into two areas, the centre as phase IIa and the front as phase IIb (lower).

MATERIALS AND METHODS

Control operation and changes of population density

Amami-oshima is a mountainous island (129°19'E, 28°17'N) that has an area of 710km^2 and a maximum elevation of 694m. 70% of the island is covered with sub-tropical forest.

Fig. 2 shows changes in the mongoose population size from its initial introduction to the island in 1979 until the present, based on a 40% annual growth rate and subtraction of trapped animals (Amami Wildlife Conservation Centre, unpublished data). The local government began to control the mongoose in 1993 to reduce its impact on crops and poultry in farmland. As many as 1,000 to 1,500 animals were caught annually until 1999. During those seven years around 9,400 animals were caught in total, but these efforts failed to induce a decrease in population size. To prevent adverse effects on biological diversity, the Ministry of the Environment began a second control operation in 2000 (Yamada 2001). Under the local and national funded control programmes 12,034 mongooses were captured over four subsequent years from 2000 to 2003. As a result, the population was reduced to a third of its maximum size by the beginning of 2000 and it continues to decrease (Fig. 2) but its distribution is expanding gradually because of inadequate trapping to reduce its distribution (Fig. 3). The authors have been catching mongooses since 1989, both before and after the population crash brought about by the second control programme. We shall refer to the catch period between 1989 and 1999 as phase I, and those animals caught from 2001 to 2004 as phase II.

The distribution map for phase I was primarily based on direct observation and that for phase II was mainly on trapping results (Fig. 3). The extent of the revealed distribution in phase I was therefore likely to be less accurate than that for phase II; the true distribution during phase I was likely to be larger than that actually depicted. The distribution of the mongoose population at the 'invasion front' extended by 0 - 2km / year (Abe *et al.* 1991).

An index of mongoose abundance, relative population density, was calculated and expressed as the number of captures per 100 trap-days, corrected for live traps (C100TD^{-1}). In phase I, most trapping was conducted in the centre of the known mongoose population distribution of 1990 and the relative density was very high (8.5 C100TD^{-1} in 1990 - 1992, 4.8 C100TD^{-1} in 1997 - 1999, 6.0 C100TD^{-1} in 2000). The relative density in phase II, decreased to 20 – 25% of what it had been before the start of the second control programme (2.0 C100TD^{-1} in 2001, 1.5 C100TD^{-1} in 2002, 1.2 C100TD^{-1} in 2003) (Abe *et al.* 1991, Abe *et al.* unpublished data, Ishii *et al.* 2006, Ministry of the Environment 2002, 2003, 2004).

In the fiscal year 2003, 2,565 animals were captured over 224,000 trap-days (Ministry of the Environment 2004) (Fig. 4). Relative population density was greater at the centre than at the edges of the mongoose's distribution. We therefore divided the distribution area of phase II into two areas according to population density. Phase IIa represents the centre of the mongoose's distribution where the density was highest and Phase IIb represents the periphery of the mongoose's distribution where the density was lowest (Fig. 4).

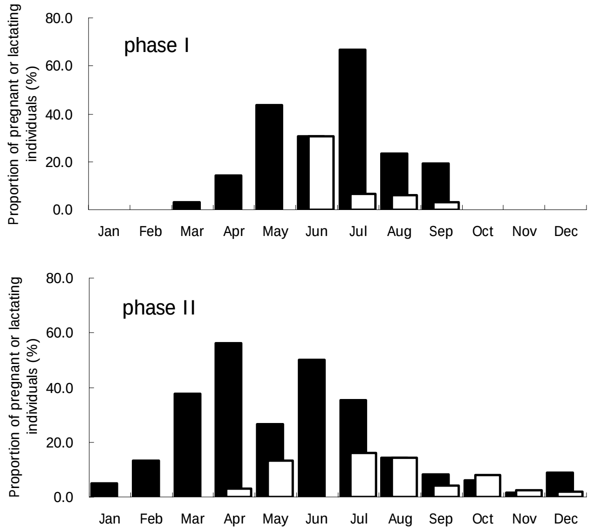

Figure 5 Proportions of *Herpestes javanicus* that were pregnant or lactating in phase I (upper) and phase II (lower). Closed bars indicate pregnant females (sum total percent of OV, CL, and PR), and open bars indicate lactating females (percent of Lac).

Animals and necropsy

Some of the mongooses that were caught by the authors or by mongoose trappers during the control operation were subjected to necropsy. A total of 2,504 animals were killed and measured. A total of 328 females in phase I and 633 females in phase II were available for use in this study. Necropsies and measurements (including body mass) were constructed on all animals.

Animals were sexed and females were divided into three developmental stages, juvenile (permanent teeth not fully erupted), immature (non-parous: no oestrus nor parous sign detected), and mature (parous) based on macroscopic observations of body size, dental eruption, size of nipples, and size of reproductive organs. Females were classed as mature if posterior nipples were distinctive or if a mature ovarian follicle was detected. Mature females were further subdivided into six categories according to their reproductive condition: oestrus with mature ovarian follicle (OV), corpus luteum in ovary without visible foetus (CL), pregnant with obvious swelling of uterine wall (PR), lactating - enlarged teats and active mammary glands (Lac), parous and placental scars countable (PS), and parous but no placental scars detected (No-PS).

Litter size was determined from either the number of placental scars, visible embryos or foetuses in mature females classed other than No-PS. There were two cases in which newborns were brought in from a den. In these special cases, the pups were treated as the litter size of a lactating mother (Lac). All statistical differences between phase I and phase II, a and b were analysed using Student's t-test.

Litter size should ideally be determined from a foetus count. It is thought that using placental scars generally overestimates litter size, while counting live pups often underestimates it. However, in most cases, the total number of corpus luteum was found to equal the total number of embryos. As there was no decrease in number of foetuses with development, an overestimation of litter size for individuals classed as OV, CL, PS, Lac or PR was not apparent. There were only four exceptions to this; all were from individuals captured during phase II. In two cases one of the fertilised eggs failed to implant, in the other two cases there was evidence of embryo absorption. As errors associated with estimating litter size were found to be negligible amongst reproductive stages, the litter sizes of the different reproductive stages were pooled.

RESULTS

Breeding season and fecundity

Of the 328 female mongooses caught in phase I, 70 (21%) were pregnant, and a further 14 (4%) were lactating. Of the 633 female mongooses caught in phase II, 91 (14%) were pregnant, and 29 (5%) were lactating. The duration of the breeding period in a year for phase II (12 months) was longer than that for phase I (seven months, from March to September). Due to small sample sizes (< 20), data collected during the periods May - August for phase I and May - June for phase II may be less reliable. The

Table 1 Litter size (mean ± SD) of *Herpestes javanicus* in each reproductive category on Amami-oshima Island.

	phase I								phase II a								phase II b							
	N	range			Litter size				N	range			Litter size				N	range			Litter size			
OV	1	2			2.00				4	2	-	3	2.25	±	0.50		1				3.00			
CL	4	2	-	3	2.25	±	0.50		22	1	-	4	2.32	±	0.72		13	2	-	5	3.00	±	1.08	
PR	24	1	-	3	2.08	±	0.50		24	2	-	3	2.42	±	0.50		19	2	-	4	3.05	±	0.71	
Lac	4	2	-	3	2.50	±	0.58		17	2	-	7	3.24	±	1.71		9	2	-	6	2.89	±	1.36	
PS	73	1	-	5	2.32	±	0.66		89	1	-	5	2.53	±	0.77		31	2	-	6	2.84	±	1.04	
Total	106	1	-	5	2.26	±	0.62		156	1	-	7	2.55	±	0.90		73	2	-	6	2.93	±	0.99	

OV: oestrus with mature ovarian follicle, CL: corpus luteum in ovary without visible foetus, PR: pregnant with obvious swelling of uterine wall, Lac: lactating, enlarged teats and active mammary glands, PS: parous and placental scars countable.

percentage of captured pregnant females shows peaks in May and July in phase I , and April and June in phase II (Fig. 5).

A total of 335 litters were countable (Tab. 1). The mean litter size during phase I was 2.26 ± 0.62 (n=106) versus 2.67 ± 0.95 (n=229) during phase II ($p>0.05$). The mean litter size in phase IIb (2.93 ± 0.99) was significantly greater than that in phase IIa (2.55 ± 0.90) and phase I ($p<0.05$).

Body mass

The mean body mass of healthy mature females was 462 ± 57.8g (n=123) in phase I, 487 ± 64.9g (n=121) in phase IIa, and 500 ± 68.3g (n=49) in phase IIb (PR individuals excluded from analyses) (Fig. 6). The body mass of mature females, which is considered to be an index of physical condition, was significantly greater during phase II (491 ± 65.9) than during phase I ($p<0.001$); but there was no significant difference between phases IIa and IIb ($p>0.05$).

DISCUSSION

Nutritional condition and reproductive activity

Body mass; an index of nutritional condition, was significantly heavier in phase II than in phase I. This indicates that food availability or quality for the mongoose may have differed between the two phases. Changes in litter size, breeding season and fecundity are thought to be related to changes in nutritional conditions associated with the decrease of population density. We suggest that the control operation has indirectly influenced individual food availability through changes in population density.

Density dependence is a common feature in the dynamics of animal populations (Kirkpatrick 1988). Otali and Gilchrist (2004) reported a relationship between additional feeding and reproduction in another mongoose species, the banded mongoose (*Mungos mungo*). In that study refuse-feeding adults were heavier and in better physical condition than non-refuse-feeding adults, and refuse-feeding females carried more foetuses than non-refuse feeding females. However, due to higher predator density in refuse areas, Otali and Gilchrist (2004) found no difference in the number of independent young per female.

Since there is no species that preys on mongoose on Amami-oshima Island, a larger litter size is likely to lead to greater population growth. As their population density decreases, the effect of intraspecific competition for food resource is likely to decrease. We conclude that nutritional condition is a major factor influencing growth, reproduction, and pup survival in mongoose populations like many other animals.

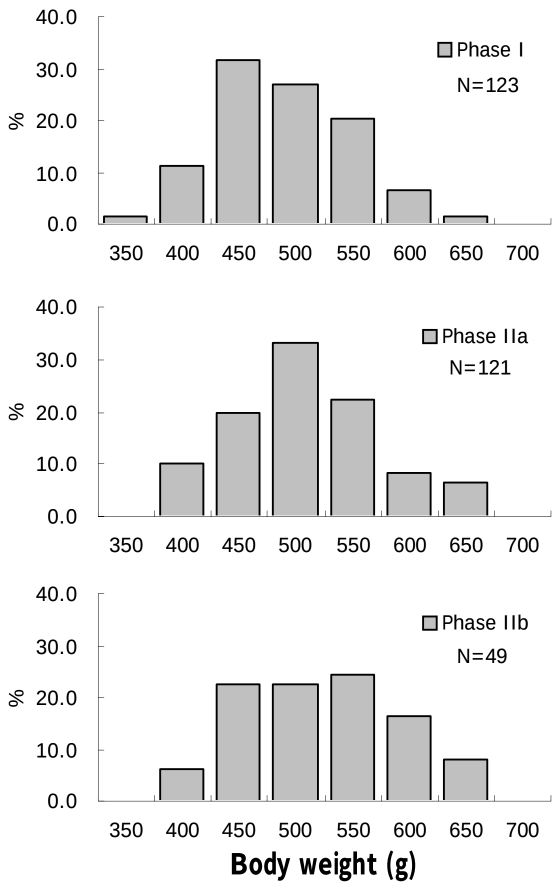

Figure 6 Body mass of mature, non-pregnant female *Herpestes javanicus* on Amami-oshima Island in phase I (top), phase IIa (middle), and phase IIb (bottom).

Oestrus cycle

The number of oestrus cycles within a year contributes to the population growth rate. The small Indian mongoose is an induced ovulator (Hoffman and Sehgal 1976, Nellis and Everard 1983, Pearson and Baldwin 1953), requiring a copulatory stimulus for ovulation and corpus luteum formation; ovulation never occurs without copulation. Most copulation events result in pregnancy, and the gestation period is approximately seven weeks (49 days). The young undertake their first activity outside the den at about four weeks of age and follow their mother on their first hunting trip at about six weeks of age (Nellis 1989). Since seasonal oestrus in the year is post-lactational (Gorman 1976), it takes three to four months from copulation to the next oestrus. At least six months are therefore needed for a female to successfully rear two consecutive litters. It would be expected that captured pregnant females would show two reproductive peaks separated by an interval of three months.

Studies on captured female mongoose on Viti Levu, Fiji (Gorman 1976) and Okinawa Island (Ogura *et al.* 2001) found a single reproductive peak. Conversely, two distinct peaks were observed on Oahu, Hawaii (Pearson and Baldwin 1953), Puerto Rico (Pimentel 1955), and St. Croix (Nellis and Everard 1983), and three distinct peaks on Grenada (Nellis and Everard 1983). In the latter areas, there is an interval of three to four months between reproductive peaks and females might produce two or three litters per year.

On Amami-oshima Island, the reproductive season was found to be longer under conditions of low population density. The percentage of captured pregnant females showed two peaks in phase I as well as phase II, but the time interval between these two peaks was found to be only two months. This indicates that not all females have two litters in a season, although the frequency of mongooses producing two litters a year does appear to be higher when the breeding season is longer, such as in phase II.

It is possible that a longer breeding season at low population densities reflects a reduced ability of females to time copulation with their oestrus cycle. At low population densities the probability that a male can encounter and copulate with a female during the time of oestrus is likely to be lower. This may have extended the female's oestrus cycle.

To evaluate true female productivity, it is critical to know how many litters they can produce per year. It is possible by careful examination of the uterus wall to sometimes distinguish two sets of scars, based upon differences in size and intensity. In such cases it is possible to ascertain that the mongoose has had two litters that year. However, we found two sets of scars to be exceptionally rare. It is thought that placental scars of the mongoose become smaller and less distinct within a few months after parturition. It is therefore very difficult to evaluate a female's annual fecundity. In one case in phase II, a mother in early pregnancy was trapped together with her pup. In that particular case it was possible to show a second pregnancy within the same breeding season.

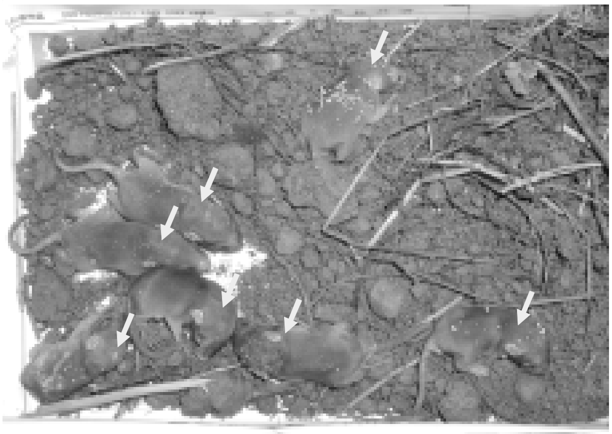

Figure 7 Seven pups of *Herpestes javanicus* from one den on Amami-oshima Island.

Maximum litter size

In phase I, the largest litter size recorded was 5 placental scars, of which there was only one case. This is consistent with previous studies on the introduced mongoose on other invaded islands in the Caribbean, Hawaii and elsewhere (Coblentz and Coblentz 1985b, Nellis and Everard 1983, Gorman 1976, Pimentel 1955, Pearson and Baldwin, 1953), where litters of five pups were also recorded to be the maximum litter size. On most islands where the mongoose has been introduced, the maximum litter size is usually three or four, rarely five. Elsewhere in Japan, on Okinawa Island, the maximum number of PS has been reported as five. It has suggested that in cases where 5 PS or more occur that several embryos might be lost after implantation (Ogura *et al.* 2001). Litter sizes of six and seven have not been previously reported . But on Amami-oshima Island, during phase II, there were 12 cases in which litters of five or more were found. These comprised two cases of 5 corpus luteum, three cases of 6 placental scars with lactating condition, four cases of 5 placental scars and one case of 6 placental scars. In addition, one litter of five and one of seven newborns were found (Fig. 7). There is no doubt that exceptionally large litter sizes can occur under good nutritional conditions, during control operation and at the edge of the distribution and that they do not necessarily result in foetus absorption or stillbirths.

Implications for control strategy

Demographic changes, such as changes to age structure, survival rate or growth rate, are currently unknown for the small Indian mongoose on Amami-oshima Island and require further study. Annual growth rate of the mongoose on Amami-oshima Island has been estimated to be 40% (Ministry of the Environment 2003), but this study indicates that the population growth rate may be higher. Increased productivity rates at low population density are expected to increase annual growth rates where there is an absence of a top-predator. Our study highlights the importance of taking into account density-dependent effects in control operations. Eradication of the mongoose will require extensive pressure from a well organised trapping team. It may also be necessary to use other methods and techniques such as reducing fertility, poisoning, and searching with dogs to rise above the higher population growth rate and achieve eradication of this invasive alien predator.

ACKNOWLEDGEMENTS

We wish to thank the Ministry of the Environment, the Japan Wildlife Research Centre, and the Amami Mammalogical Society for trapping mongooses. We also thank Dr. Richard J. Walters (Tokyo University) for his cooperation and helpful suggestions.

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Short paper

Population estimation and control of the introduced mongoose on Amami-oshima Island

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INTRODUCTION

The small Asian mongoose (*Herpestes javanicus*) was introduced to Amami-oshima Island (712km²) of Kagoshima Prefecture, Kyushu around 1979 with the aim of controlling the venomous snakes (*Trimeresurus flavoviridis*) and harmful rats (*Rattus rattus*). The mongoose population began spreading from the release site in northern Naze City, and in the fiscal year (FY, from April to the next March) 1993 trapping was started to prevent poultry predation and crop damage.

With growing concern about the impact on native fauna including endangered species, in FY1996 the Environment Agency (now the Ministry of the Environment) directed the Japan Wildlife Research Center to conduct a survey on the population status and other ecological aspects of the mongoose (Environment Agency *et al.* 2000). Based on the survey results the Environment Agency started an intensive trapping campaign to eradicate mongooses in FY2000 (Ishii 2003). This paper describes and examines the results of the status survey and outcomes of the control campaign.

STATUS SURVEY

Methods

The mongoose population survey was carried out from FY1996 to FY1999. To estimate population size, 18 trap-lines were established throughout the island. On each line 50 cage-traps baited with fish sausage were placed 100m apart and set for 14 days. The population size in the effective trapping area was calculated by the Leslie model (e.g. Krebs 1999) using the relation between the numbers of captured animals per day and the accumulated number of animals captured by the previous day. The effective trapping area was defined as the area within the radius of a circle equal to the average home range size, from the trap-line.

Results

It was revealed that the mongoose population occupied about 30% of the island in FY1999. The 100TN index (number of animals captured per one-hundred trap-nights) varied depending on the trap-line location. The distribution of this index showed a clear pattern with the highest value in the central part of the island and declining values towards the peripheral areas. We stratified the mongoose range into three zones according to this index value and calculated the population using the area of each zone and the average density. The total population size was estimated at about 5000 in FY1999.

We also estimated the annual rate of population increase using the number of mongooses removed during each fiscal year, with the equation: $N_t = (1+a) N_{t-1} - C_{t-1}$, where a is the annual increase rate; N_t, the population at the fiscal year t; C_t, the number of mongooses removed during the fiscal year t. We assumed that the population had started from 30 individuals released in 1979 and reached 5000 after 20 years, and we estimated the annual increase rate at about 40%.

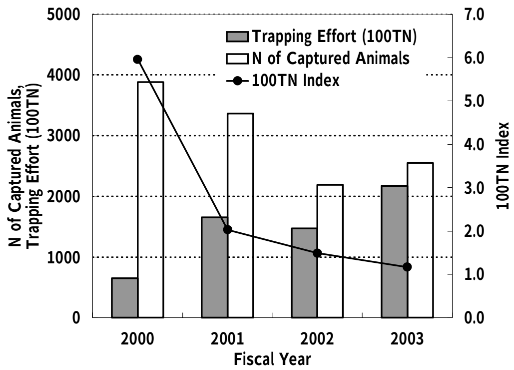

Figure 1 Trapping efforts and results of the mongoose control on Amami-oshima Island.

Pages 165-167. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

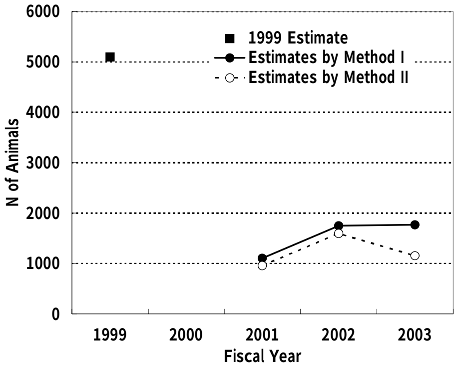

Figure 2 Population estimates of the mongoose on Amami-oshima Island.

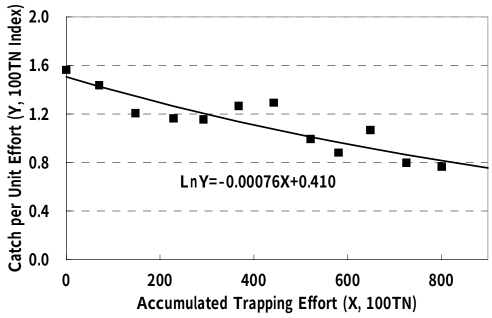

Figure 3 Relationship between trapping effort and catch per unit effort in FY2003.

CONTROL CAMPAIGN

Methods

The survey results indicated that serious action was urgently required, so the Environment Agency launched an intensive control program in FY2000. The mongooses were trapped by local residents, mainly on a bounty system, and from the start of FY2000 to the end of FY2003 approximately 12,000 individuals were captured (Fig. 1).

The Japan Wildlife Research Centre continuously investigated the changes in population status due to the control campaign. From FY2001, the population size was estimated by two methods. Method I was almost the same method used for the previous estimation. Nine to 15 trap-lines were operated throughout the island, and traps were set for 7 days around the end of each fiscal year (the end of March). Population density was considered as the same throughout the range, since density had significantly decreased from the peak observed in FY1999 and the geographical pattern was not as pronounced as before the control campaign.

Method II used data obtained through the trapping campaign. Number, date and location of traps, and number of mongooses caught were recorded for the whole island. The population was calculated using the Leslie model based on the relation between 100TN index for a week and the accumulated number of animals captured by the end of the previous week. Data used was from the start of January to the end of March, to avoid the effect of population increase due to breeding (breeding ceased before January). The population at the end of FY was calculated by deducting the number of animals actually captured by the end of March, from the estimated population at the start of January.

Results

The population estimations by the two methods produced similar results and showed a marked reduction from FY1999 to FY2001 (Fig. 2). However, after FY2001, the population was not being reduced and stabilised at about 1000-2000. These estimates agreed with the changes in 100TN index of each fiscal year (Fig. 1). Seasonal changes in the index suggested that after FY2001 the population increased from August due to the breeding and decreased after December by the trapping, and had been rather stable under the current trapping pressure. The distribution of the population was spreading and had occupied over half of the island by the end of FY2003.

CONCLUSIONS

Population estimation in FY1999 was fairly accurate, if not precise, in light of the marked decline caused by the trapping of nearly 4000 animals during FY2000 (Fig. 1). However, if we assume an annual rate of population increase of 40% and a population size of 1000-2000 after FY2001, and consider the actual number of animals removed during each fiscal year, the slowdown in population reduction cannot be explained. Therefore, the rate of population increase is postulated to have changed after FY2000 (Abe *et al.* 2006), although an underestimation of population size may also be responsible to some degree.

Whatever the case, considering the current lack of alternative measures to reduce the mongoose population, the control project should make more intensive trapping efforts. For example, according to the relation between 100TN index and accumulated TN by the end of the previous week from January to March 2004, the trapping efficiency is 0.00076 per 100TN for the whole population (Fig. 3). This means that to decrease the population to one tenth (or by

90%), 300,000TN would have been required instead of actual 86,100TN from January to March 2004. Also planned trapping allocation is necessary to diminish the range.

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Short paper

Population parameters of an alien turtle (*Chelydra serpentina*) in the Inbanuma basin, Chiba Prefecture, Japan

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INTRODUCTION

The snapping turtle (*Chelydra serpentina*) is widely distributed from southern Canada to Ecuador. This turtle is harvested or farmed in the USA for food and pet trade, and shipped to various countries. Some individuals are released or have escaped into the wild outside the native range, including the western states of the USA and in the UK (Beebee and Griffith 2000, Stebbins 2003). In Japan, the snapping turtle was discovered in the Inbanuma basin in 1978, and it has frequently been found since the mid-1990s (Kobayashi 2000). Evidence that the species is reproducing in this area was obtained in 2002 (Kobayashi *et al*. 2006).

The purpose of our project is to identify an effective strategy for the management of the feral snapping turtle by developing a population dynamics model. In order to predict the whole population dynamics with a matrix based model, we need information on the current size and structure of the feral population, and on population parameters, including reproduction and mortality rates in the different stages of the life cycle.

This paper describes the estimation of population size, fecundity, body size at female maturity, and mortality in the egg stage.

MATERIALS AND METHODS

Research site and the turtle

Lake Inbanuma is an eutrophic water system. The water area and average water depth are 1,115ha and 1.7m respectively. The lake is surrounded with paddy fields. Most of the snapping turtles were found in Kashima River, the largest river running into the lake, and Takasaki Stream. According to the database of the Japan Meteorological Agency, the annual average temperature of the area in 2003 was 14.1˚C

The snapping turtle is a large omnivorous species. The life cycle was divided into five stages: egg, hatchling, juvenile, subadult, and adult. In the study area, nesting was observed from early to mid June, and emergence from the nest from early September to early November. All adult turtles captured were identified as the northern subspecies *Chelydra serpentina serpentina*, except for those individuals having abnormal or injured carapace (which meant that subspecies could not be determined).

Estimation of population size

The whole area of Inbanuma basin was surveyed to estimate the distribution of the turtle. We collected turtles, using funnel traps, at eight sites in Kashima River and Takasaki Stream from April 2001 to October 2003. Turtles were marked on their carapaces with individual hole-patterns, using a drill, and were released at the point of capture. The population density of the turtle was estimated by two mark-recapture methods: 1) Petersen and 2) Jolly-Seber (Krebs 1998). Population size was calculated from the population density multiplied by the distribution area.

Since the traps caught adults and subadults only, the estimated population size was of these stages, and the total population size including juveniles should be larger than this estimate.

Estimation of population parameters

Information on population parameters was collected in a segment of the Takasaki Stream in 2003 and 2004. Fecundity was calculated from the number of eggs per clutch. Mortality by nest predation was inferred from video camera records. Emergence success from the nest chamber was calculated from the number of hatched turtles. The body size at first reproduction was estimated from the minimum carapace length of egg-laying females.

RESULTS

A total of 328 turtles (217 individuals) were captured during the study period from 2001 to 2003. Two mark-recapture methods produced a similar population density estimate: about 180 individuals per kilometre. Most of the turtles were recorded from Kashima River and Takasaki Stream with the core

Pages 168-169. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

distribution being a segment of about 16km in length. Population size for the whole catchment was estimated roughly as several thousand individuals.

The minimum carapace length for mature females was 183.7mm. There was no nest predation in any of the 25 nests observed. Average clutch size (±SE) was 30.1 (±12.0), and 51.6% of eggs resulted in hatching and emergence from the nests.

DISCUSSION

The snapping turtle population in this area showed a smaller size at female maturity, similar (or slightly lower) clutch size, lower nest predation, and higher emergence success than populations in the native range (Iverson *et al.* 1997). In the native range this species has a high mortality rate in the egg and hatchling stages through predation (Hammer 1969, Congdon *et al.* 1987). Our results suggest that this introduced population has a higher potential for population growth than the population in the native range, due to the lack of nest predation. Early control action is required to prevent further spread.

Information is lacking on the parameters of growth and mortality from juvenile to adult stage, and these should be estimated through further study. We will evaluate the impact of trapping (adults and subadults only) and nest removal on the population viability using a matrix based population model based on these parameters.

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Part 4
Database

Alien earthworms in the Asia/Pacific region with a checklist of species and the first records of *Eukerria saltensis* (Oligochaeta : Ocnerodrilidae) and *Eiseniella tetraedra* (Lumbricidae) from Japan, and *Pontoscolex corethrurus* (Glossoscolecidae) from Okinawa

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Abstract Of the 80 or so earthworms in Japan (including Ryukyus) approximately 50% are alien species that were most likely introduced inadvertently by human activities. This diversity compares with about 93 species found in Korea (25% alien), 71 species in Taiwan (38% alien), and 152 from mainland China (26% alien). In comparison, 715 species are known from Australia (ca. 10% alien) including 230 from Tasmania (ca. 11% alien), and New Zealand has 199 species (ca. 14% alien). Australia, Tasmania and New Zealand have relatively low numbers of aliens; the latter two regions are especially similar with 60-67% of their aliens being Lumbricidae of direct or indirect European/Middle-eastern origin. Mainland Japan and the Ryukyus, Taiwan, Southeast Asia and southern China are similar with most of the ca. 26-65% aliens due to exchanges of Oriental pheretimoids (species ex *Pheretima* auct.). North America is intermediate yet more varied: one third of its 180 species are non-native of which about half are Lumbricidae. Biodiversity differences for natives are accounted for by geological histories (e.g. plate tectonics, volcanism, glaciation) and by topographic and climatic factors, whereas distribution of the aliens echo patterns of human migration and trade and, overall, appreciation of both groups is determined by the intensity of taxonomic treatment. Newly reported from Japan are the semi-aquatic *Eiseniella tetraedra* (Savigny, 1826) and *Eukerria saltensis* (Beddard, 1895), and from Okinawa the circum-tropical *Pontoscolex corethrurus* (Müller, 1856). Both latter species originate from South America. *Eukerria saltensis* is considered a pest in Australian rice paddies but despite discovery in drains at Kamakura and a river in Tokyo, it is not known under Japanese rice and its risk here is as yet unquantified.

Keywords: Australasia; America; exotic species diversity; *Pheretima*; lumbricids.

SPECIES COMPOSITIONS

Earthworms are an ancient group with generally weak means of dispersal, thus Family origins are partly determined by plate tectonics (e.g. Michaelsen 1922, from Lee 1994) (see Tab. 1 and Appendix 1). The degree of endemism depends on the geological history and current climate of the region as well as the intensity of glaciations and/or volcanism, whereas the present-day distribution of aliens is strongly influenced by recent, historical and pre-historical human trade and migrations before quarantine barriers were implemented (Stephenson 1930, Gates 1972, Easton 1981, Lee 1987, Blakemore 1999). The adult worms, or their cocoons, can be easily transported in soil of potted-plants (Gates 1972). Australia and New Zealand, due to their remoteness and isolation from major world trade except in the last 200 years of recorded history, provide useful information on the capability and speed of spread of alien species. Reflecting their European settlement this region tends to have more Lumbricidae compared to the Asian countries where both native and alien species are more often Megascolecidae or Moniligastridae. Korea and northern China appear exceptions as cooler climates allow a relatively greater (natural?) abundance of Lumbricidae. However, several Oriental species are now widely distributed, for example, there are reports of components of the *Metaphire hilgendorfi* species-complex that possibly originated in Japan, viz. *Metaphire agrestis*, *M. hilgendorfi* and *Amynthas tokioensis*, from North America (Hendrix and Bohlen 2002, Blakemore 2003, 2005). A summary of the relative proportions of Lumbricidae - including *Eisenia japonica* supposedly native to Korea and Japan, and Megascolecidae - mainly pheretimoids, is shown in Tab. 2.

From a total earthworm fauna of ca. 5,500 described species, the Holarctic lumbricids comprise about 600, whereas roughly 900 Oriental pheretimoids

Pages 173-181. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

Table 1 Alien earthwarms in Australis (Aust.), Tasmania (Tas.), New Zealand, Japan, Ryu-kyu Islands, Korea, Taiwan, China, South East Asia, U.S.A. and Canada (based on Appendix 1).

	Aust. (excl. Tas.)	**Tas.**	**N.Z.**	**Japan (excl. Ryuku)**	**Ryu-kyu Isls.**	**Korea (inc. Cheju)**	**Taiwan**	**China (incl. Hainan)**	**SE Asia**	**U.S.A. and Canada**
Approx. No. of aliens in region (A)	63	27	27	33	18	23	27	40	50+	60
Approx. No. of natives in region (N)	450	203	172	38	10	70	44	112	?	120
Approx. TOTAL spp. (A+N)	513	230	199	71	28	93	71	152	?	180
Aliens [A/(A+N)] x 100 %	12.3	11.7	13.6	46.5	64.3	24.7	38.0	26.3	?	30.0

are known (Easton 1983, Sims, 1983, Sims and Easton 1972, Blakemore 2004b, 2004c, 2005), and these two groups each contribute about a third to the 110 most common alien species now found around the world (Blakemore 2002). The remaining third of 'cosmopolitan species' have diverse origins (Tab. 1 and Appendix 1). Although wide environmental tolerance is often characteristic of aliens, their ability to survive in a new region once introduced is influenced by the local climate and ecology (Lee 1985, 1987).

Apart from determining new natives, one of the challenges in ecological taxonomy is distinguishing the aliens from the natives and assessing the diversity and distribution of both. Regional comparisons help us appreciate mechanisms of initial introduction and to chart the relative rates of dispersal and differentiation.

BIOGEOGRAPHY AND COMPARATIVE SPECIES DIVERSITY

The 3,000 islands of Japan extend almost 3,000 km from subarctic Hokkaido in the northeast to subtropical Okinawa to the southwest, occupying 378,000km^2 similar in total area to Britain and Ireland, just smaller than California, but just larger than New Zealand or the Korean peninsula. Between 56,000 to 10,000 years ago during the glacial Pleistocene, Japan was connected to Korea, and the southernmost Ryukyu chain of islands were united with Taiwan which itself was periodically connected to China (Tsai *et al.* 2000a) facilitating natural exchange of fauna. The Ryukyu archipelago stretches across nearly 1,000km of ocean and includes the main islands of Okinawa, Miyako, Ishigaki and Iriomote. Ten Okinawan native worms are known, with all but two species also recorded from the main islands of Japan, although none are in common with the Taiwan fauna.

A checklist of Japanese earthworms by Easton (1981) reported 70 species, the majority pheretimoids (i.e., species ex *Pheretima* auct.) that frequently have parthenogenetic morphs. About 60 new names were proposed in Ishizuka (2001) but most were polymorphic synonyms, giving a new total of just 80 valid names with another dozen retained as *species incertae sedis*; of these 80 taxa, about 40 are presumed natives, 33 are known aliens, and the remaining species are of uncertain origin (Blakemore 2003, 2004a). The Japan/Ryukus diversity is very similar to that of the Korean peninsula including volcanic Cheju (= Quelpart) Island that has 93 known species (70 native); and these totals compare (Tab.1 and Appendix 1) with 71 species from Taiwan (44 native); 152 from China (112 native composed of about 90 from the mainland and 22 from Hainan). The totals are modified slightly from Shih *et al.* (1999) and from Tsai *et al.* (2000a) to include recently described *Amynthas* species from Korea (some being synonyms) e.g. by Hong and James (2001) and several new, mainly pheretimoid species from subtropical Taiwan as listed by Blakemore (2005) and Blakemore *et al.* (in press).

The diversity of earthworms differs considerably in non-Asian countries. The British Isles, for example, have 48 taxa which are mostly composed of common Lumbricidae (re-)introduced from continental Europe since the last Ice Age (Sims and Gerard 1999) with ca. 30 of these same species now in Australasia and the Americas (Blakemore 2002). Currently about 180 taxa (ca. 120 natives) are to be found in North America, the

Table 2 Relative proportions of Lumbricidae (originally from temperate Eurasia or North America) and Megascolecidae (mainly from subtropical Asia/Australasia).

	Aust. (excl. Tas.)	**Tas.**	**N.Z.**	**Japan (excl. Ryuku)**	**Ryu-kyu Isls.**	**Korea (inc. Cheju)**	**Taiwan**	**China (incl. Hainan)**	**SE Asia**	**U.S.A. and Canada**
Approx. No. Lumbricidae as % total aliens	35%	60%	67%	39%	0	57%	22%	23%	6%	50%
Approx. No. Megascolecidae as % total aliens	37%	18%	22%	46%	77%	39%	66%	55%	48%	30%

northern parts of which were similarly glaciated, and the volcanic Hawaiian Islands have 50 taxa listed, probably only 33 being reasonably valid names and all presumed to be post-Columbian introduced species (Hendrix 1995, Anon. 2003, Blakemore 2005). New Zealand's North and South islands have 199 species (172 natives), while ca. 715 (ca. 650 natives) are known from Australia, including ca. 203 natives from the cool temperate island state of Tasmania that is roughly the same size as Ireland, Hispaniola, or Hokkaido (Lee 1959, Blakemore 1999, 2000, 2002, 2005). If neoendemics (as defined by Blakemore 1999) and taxa that are native to the region but believed also introduced outside of their natural range within the region were included with the ca. 65 aliens (<10% of total with just 3% lumbricids), then Australian and Tasmania would have a combined total of nearly 80 non-wholly native taxa. These figures are remarkably high compared to diversities in Europe, Asia or the Americas, especially considering the relatively brief exposure to international trade and communication with Australia since 1788 and somewhat delayed start to eco-taxonomic surveys.

ECOLOGICAL/ECONOMIC RISK OF ALIEN SPECIES

The beneficial and deleterious effects of invasive alien earthworms in North America are presented in a summary by Hendrix and Bohlen (2002). Alien species for which there are reports of some adverse effects include *Pontoscolex corethrurus* (Müller. 1856) that often dominates newly colonised tropical lowlands (e.g. Tsai *et al.* 2000b). It has yet to be confirmed in mainland Japan although it is newly reported here in Yona, northern Okinawa [collected by R.J.B. on 20.xi.2005 from soil by storm-drain in *Castanopsis sieboldii* (Makino) forested hills above Ryukyu University Forestry Research Centre]. Gates (1972: 183) noted that *Pontoscolex corethrurus* along with a lesser population of *Polypheretima elongata* (Perrier, 1872) were implicated in rendering a South Indian soil cloddy and unproductive. Similarly, seepage from taro patches in Kauai, Hawaii, from rice paddies in Taiwan, and from 2,000-year-old mountain rice terraces in Ifugao, Philippines were all attributed to excessive burrowing by *Polypheretima elongata* morphs by Gates (1972).

Dichogaster annae (Horst, 1893), reported as its probable junior synonym *D. curgensis* Michaelsen, 1921, has been indicted as a serious pest of rice terraces in the Philippine Cordilleras (Barrion and Listinger, 1997). Although known from Africa, India, Southeast Asia, South America and recently recorded from Australia (Blakemore 1999, 2002), it is not yet reported from Japan.

The semi-aquatic South American *Eukerria saltensis* is a new species record in Japan (Appendix 2). It is considered a pest in aerially-sown rice in Australian paddies with crop failures from up-rootings usually occurring between tillering and harvest, caused by increased water turbidity and reduced soil compaction attributed to the worms' activities (Stevens and Warren 2000). These authors also found an indirect affect on the rice due to the worms attracting ibis (*Threskiornis* spp.) and other waterbirds that trample the young plants as they hunt for prey. Rotations with dryland crops such as winter cereals appeared effective in controlling both these worm problems (Stevens 2003; see also *http://www.ogtr.gov.au/rtf/ir/biologyrice1.rtf* February, 2005). To what extent this species presents a threat to Japanese rice production, if at all, is currently unknown.

Furthermore, the alien lumbricid *Eiseniella tetraedra* (Savigny, 1826) is newly recorded from a riverbank in Toyama-ken central Honshu, Japan (identified by R.J.B. from a specimen delivered to M.T. Ito, December, 2005) and from running water of Chichawan Stream on Wuling Farm, Shei-Pa National Park, northeastern Taiwan (several specimens from Dr. J.-H. Chen, February, 2004). This limicolous species is probably native to the western Palaearctic but is now widespread in mainly temperate regions in both hemispheres of the world. It is not known to present any environmental risks (Blakemore 2002, Csuzdi & Zicsi 2003).

ACKNOWLEDGEMENTS

This study was supported in part by 21st Century Program COE Environmental Risk Management for Bio/Ecosystems at YNU where an International Conference on "Assessment and Control of Biological Invasion Risks" was held 26 – 29 August, 2004 at Yokohama, Japan [*http://bio-eco.eis.ynu.ac.jp/jpn/top/topic_sheet/symposium20040827/eng.html*]. The organisers, anonymous referees and the journal editors are thanked for constructive criticism. Dr Nikita Eriksen-Hamel kindly commented on the draft manuscript.

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APPENDIX 1

Records of alien earthworms from Australasian and Oriental regions compared to continental North America. (Family classification after Blakemore 2000; full synonymies listed in Blakemore 2002).

Families (origins) and Species from Regions	**Aust. (excl. Tas.)**	**Tas.**	**N.Z.**	**Japan (excl. Ryuku)**	**Ryu-kyu Isls.**	**Korea (inc. Cheju)**	**Taiwan**	**China (incl. Hainan)**	**SE Asia**	**U.S.A. and Canada**
MONILIGASTRIDAE (Oriental & Indian)										
Desmogaster sinensis Gates, 1930								+		
Drawida barwelli (Beddard, 1886)	*							*(H)	+	
Drawida japonica (Michaelsen, 1892)				+		+	+		+	
Drawida longatria longatria Gates, 1925									+	
Drawida nepalensis Michaelsen, 1907									+	
GLOSSOSCOLECIDAE (Neotropical)										
Pontoscolex corethrurus (Müller, 1856)	+		+	+(BI)	*		+	+	+	+
ALMIDAE (Circum-tropical)										
Glyphidrilus papillatus (Rosa, 1890)								+(H)		
CRIODRILIDAE (Southwestern palaearctic)										
Criodrilus lactuum Hoffmeister, 1845										+
HORMOGASTRIDAE (Mediterranean)										
Hormogaster redii Rosa, 1887										+
LUMBRICIDAE (Holarctic)										
Allolobophora chlorotica (Savigny, 1826)		+	+							+
Allolobophoridella eiseni (Levinsen, 1884)		*	+							+
Aporrectodea caliginosa (Savigny, 1826)	+	+	+	+		+	+?	+		+
Aporrectodea icterica (Savigny, 1826)										+
Aporrectodea limicola (Michaelsen, 1890)	B+									+
Aporrectodea longa (Ude, 1885)	+	+	+							+
Aporrectodea rosea (Savigny, 1826)	+	+	+	+		+		+		+
Aporrectodea trapezoides (Dugès, 1828)	+	+	+	+		+	+	+	+	+
Aporrectodea tuberculata (Eisen, 1874)	+		+	+?		+?	+?	+?		+
Bimastos parvus (Eisen, 1874)	+		+?	+		+	+	+	+	+?
Dendrobaena attemsi (Michaelsen, 1902)	?	?								+
Dendrobaena hortensis (Michaelsen, 1890)	B+	*								+
Dendrobaena octaedra (Savigny, 1826)				+						+
Dendrobaena pygmaea (Savigny, 1826)				*						+
Dendrobaena veneta (Rosa, 1886)	B+		*							+
Dendrodrilus rubidus rubidus (Savigny, 1826)	+	*	+	+		+		+		+
D. rubidus subrubicundus (Eisen, 1874)	+									+?
D. rubidus tenuis (Eisen, 1874)	+(HI)	+(MI)		+		+				+?
Eisenia andrei Bouché, 1972	+?		+?	+?		+?				+?
Eisenia fetida (Savigny, 1826)	+	*	+	+		+	*	+	+	+
Eisenia japonica (Michaelsen, 1892)				+?		+?				
Eisenia nordenskioeldi (Eisen, 1879) sub-spp.						+?		+		
Eiseniella tetraedra (Savigny, 1826)	+	+	+	*			*			+
Lumbricus castaneus (Savigny, 1826)	*	*	+							+
Lumbricus festivus (Savigny, 1826)	+?									+
Lumbricus friendi Cognetti, 1904										+
Lumbricus rubellus Hoffmeister, 1843	+	+	+			+?				+
Lumbricus terrestris Linnaeus, 1758		*	+			+?				+
Murchieona minuscula (Rosa, 1906)										+
Octolasion cyaneum (Savigny, 1826)	+	+	+							+
Octolasion tyrtaeum lacteum (Örley, 1881)	*?		+					+		
O. tyrtaeum tyrtaeum (Savigny, 1826)	+									+
Satchellius mammalis (Savigny, 1826)										+

(APPENDIX 1 continued)

FAMILIES (ORIGINS) and Species from Regions S	Aust. (excl. Tas.)	Tas.	N.Z.	Japan (excl. Ryuku)	Ryu-kyu Isls.	Korea (inc. Cheju)	Taiwan	China (incl. Hainan)	SE Asia	U.S.A. and Canada
OCNERODRILIDAE										
(Tropical America & Africa)										
Gordiodrilus elegans Beddard, 1892	*								+	+
Nematogenia panamaensis (Eisen, 1900)									+	
Ocnerodrilus occidentalis Eisen, 1878	*	*		+	+			+	+	+
Eukerria kuekenthali (Michaelsen, 1908)	+(CI)								+	
Eukerria saltensis (Beddard, 1895)	+	*?		*					+	+
Malabaria levis (Chen, 1938)								+(H)	+	
Thatonia exilis Gates, 1945									+	
Thatonia gracilis Gates, 1942									+	
ACANTHODRILIDAE (Pangean?)										
Microscolex dubius (Fletcher, 1887)	+	*	+							+
Microscolex kerguelarum (Grube, 1877)	+(HI)									
Microscolex macquariensis (Beddard, 1896)		+(MI)								
Microscolex phosphoreus (Dugès, 1837)	+	*	+	+						+
Rhododrilus kermadecensis Benham, 1905	B+?	*								
Rhododrilus queenslandicus Michaelsen, 1916	+									
OCTOCHAETIDAE										
(Circumtropical, Australasian)										
Dichogaster affinis (Michaelsen, 1890)	*							+(H)	+	+
Dichogaster annae (Horst, 1893)	*								+	
Dichogaster bolaui (Michaelsen, 1891)	+				+		+	+(H)	+	+
Dichogaster corticis (Michaelsen, 1899)									+?	
Dichogaster modiglianii (Rosa, 1896)									+	+
Dichogaster saliens (Beddard, 1893)	*				+				+	+
Dichogaster sp. nov?	*									
Lennogaster pusillus (Stephenson, 1920)									+	
Octochaetona beatrix (Beddard, 1902)	*								+	
Octochaetona surensis Michaelsen, 1910									+	
Ramiella bishambari (Stephenson, 1914)	+(CI)							+	+	
MEGASCOLECIDAE (mostly Indo-Australasia)										
Argilophilus marmoratus Eisen, 1893	?									
Pontodrilus litoralis (Grube, 1855)	+		+	+				+(H)	+	+
Perionyx excavatus Perrier, 1872	*	*	*	+		+	+		+	+
Amynthas aspergillum (Perrier, 1872)							+	+		
Amynthas carnosus (Goto & Hatai, 1899)						+?	+?	+?	+?	
Amynthas corticis (Kinberg, 1867)	+	*	+	+	+	+	+	+	+	+
Amynthas glabrus (Gates, 1932)				+	+				+	
Amynthas gracilis (Kinberg, 1867)	+		+	+	+		+	+?	+	+
Amynthas hupeiensis (Michaelsen, 1895)	+(T)?		?+	+	+	+	+	+	+	+
Amynthas incongruus (Chen, 1933)							+	+		
Amynthas lautus (Horst, 1883)					+?		+?	+?		
Amynthas loveridgei (Gates, 1968)										+
Amynthas minimus (Horst, 1893)	+			+	+		+	+	+	+
Amynthas morrisi (Beddard, 1892)	+			+	+		+	+	+	+
Am. morrisi group sp. nov.?	*									
Amynthas papulosus (Rosa, 1896)				+	+		+	+	+	
Amynthas robustus (Perrier, 1872)				+	+	+	+	+	+	
Amynthas rodericensis (Grube, 1879)	+							+?	+	+
Amynthas taipeiensis (Tsai, 1964)							+?	+?		
Amynthas tokioensis (Beddard, 1892)						+?				+
Anisochaeta dorsalis (Fletcher, 1887)		+								
Anisochaeta gracilis (Fletcher, 1886)		+?								
Anisochaeta sebastiana (Blakemore, 1997)		+								
Begemius queenslandicus (Fletcher, 1886)	+									
Didymogaster sylvatica Fletcher, 1886			+							
Lampito mauritii Kinberg, 1866	+(CI)							+	+	
Metaphire agrestis (Goto & Hatai, 1899)						+?				+
Metaphire bahli (Gates, 1945)	+								+?	
Metaphire californica (Kinberg, 1867)	+			+	+		+	+	+	+
Metaphire hilgendorfi (Michaelsen, 1892)						+?				+
Metaphire houlleti (Perrier, 1872)	+							+	+	+
Metaphire javanica (Kinberg, 1867)	+?							+?	+?	
Metaphire peguana (Rosa, 1890)					+?				+	
Metaphire posthuma (Vaillant, 1868)	+(CI)						+	+	+	+
Metaphire schmardae macrochaeta (Mich., 1899)				+				+?		
Metaphire schmardae schmardae (Horst, 1883)				+?	+		+	+	+	

(APPENDIX 1 continued)

FAMILIES (ORIGINS) AND SPECIES FROM REGIONS	Aust. (excl. Tas.)	Tas.	N.Z.	Japan (excl. Ryuku)	Ryu-kyu Isls.	Korea (inc. Cheju)	Taiwan	China (incl. Hainan)	SE Asia	U.S.A. and Canada
Metaphire soulensis (Kobayashi, 1938)				+?		+?		+?		
Pheretima darnleiensis (Fletcher, 1886)	+(T,CI)								+	
Pheretima montana Kinberg, 1867									+	
Pithemera bicincta (Perrier, 1875)	+			+	+		+		+	+
Polypheretima brevis (Rosa, 1898)	+(CI)									
Polypheretima elongata (Perrier, 1872)	+				+		+		+	+?
Polypheretima taprobanae (Beddard, 1892)	+									
EUDRILIDAE (West African)										
Eudrilus eugeniae (Kinberg, 1867)	*									+

Aust – Mainland Australia; Tas. – Tasmania; N.Z. – New Zealand. **+ -** present as an alien species; ***** - first records from principal author's (R.J. Blakemore) studies; ? - indicates some ambiguity of taxonomic description, endemicity, or veracity of report; B – J. C. Buckerfield of Adelaide pers. comm.; (BI) Bonin Island; (CI) Christmas Island; (H) Hainan; (HI) Heard Island; (MI) Macquarie Island; (T) Torres Straits Islands.

Notes: The above table is adapted from Blakemore (1999, 2000, 2002, 2005) and various other sources as noted there within. Rosa's *constricta*, once part of *Bimastos parvus* (syn. *beddardi*) is now included in *D. rubidus. Rhododrilus kermadecensis* is probably endemic to Kermadec. *Anisochaeta spp.* and *Didymogaster* are endemic to Australia. *Amynthas indicus* (Horst, 1883) listed from Christmas Island (and Torres Straits?) was said by Sims and Easton (1972: 263) to be *Pheretima darnleiensis.* It was though by Easton (1981) that *Amynthas lautus* was a synonym of *A. robustus*, but Tsai *et al*. (2000a: 286) disagree. *Metaphire javanica* is possibly synonymous with *M. californica* that has page priority (pers. obs.) whereas *Pithemera bicincta* may comprise more than one taxon (pers. obs.) as its synonym *Pithemera violacea* (Beddard, 1895) perhaps merits specific status.

APPENDIX 2

Eco-taxonomic description of *Eukerria saltensis* (Oligochaeta : Ocnerodrilidae).
Eukerria saltensis (Beddard, 1895)

Kerria saltensis Beddard, 1895: 225. [Type locality Valparaiso, Salto, Chile. Types in US National Museum (21025) and British Museum (1904:10:5:928)]; Michaelsen, 1900: 371; Michaelsen, 1907: 23 (syn. *sydneyensis*).
Acanthodrilus sydneyensis Sweet, 1900: 124. [From Sydney, Australia. Types?]
Kerria gunningi Michaelsen, 1913: 1. [From tropical or subtropical Africa? Types in Hamburg: 7490].
Kerria nichollsi Jackson, 1931: 121, Pl. XVI, figs. 5,8,9,11. [From WA. Types?].
Eukerria saltensis: Michaelsen, 1935: 40 (syns. *nichollsi*, *sydneyensis*); Gates, 1942: 67 (syn. *gunningi*); Gates, 1972: 270; Blakemore, 2002 (syn. *sydneyensis*, *gunningi*, *nichollsi*).

Taxonomic note: Often misdated as "Beddard, 1892" when other *Eukerria* species were described by that author, e.g. the morphologically similar *Eu. halophila* (Beddard, 1892: 357).
Distribution: South America and spread worldwide by human and other agencies: Spain (e.g. Valencia), USA [e.g., Oregon, Texas, Georgia, Florida, and North Carolina by Gates (1972: 270)], Chile (Salto, Valparaiso, Coquimbo, Quillota, also mid-Pacific Easter Island and the Iles San Fernandez: Robinson Crusoe Island, Santa Clara Island, and Alexander Selkirk Island), Argentina (Bella Vista, Cordoba), Brazil (e.g. Minas Gerais and from Sao Paulo), Myanmar (Pyinmana, Mandalay, Lower Chindwin), Vietnam (Thai, 2000), South Africa (Cape Province, Natal, Transvaal), New Caledonia. In Australia previously reported from NSW (Sydney including Sydney Harbour, Paramatta and Blue Mtns.) and Victoria, WA, now confirmed from Qld, and possibly also present in Tasmania (museum specimens). In Japan: from Kamakura and Machida, Tokyo. These are the first records from the Far East.

Locality of Examined Material: Samford (e.g. ANIC: RB.95.4.6), Closeburn, CSIRO Narayen (collector R.J.B. as detailed in Blakemore, 1994) all in Qld; Whitton near Griffiths (collector J. Blackwell); Woodburn Island/Maclean (e.g. ANIC: RB.95.13.1), Lismore, Whitton and Deniliquin (ANIC: RB.95.2.1-2 collected 7.xii.1994 by M. Stephens), NSW; in QVM collection, Tasmania - new Australian records. From drain at Kuzuharagaoka Shrine, (founded 1333) at Kamakura, collected 13.vi.2004 by R.J.B., Amanda Reid and Yuko Hiramoto; also besides Sakaigawa creek east of Hashimoto station, Machida-ku, Tokyo, the boundary between Tokyo and Kanagawa-ken, collected 18.viii.2004 by R.J.B. - new Asian records. (For details of ANIC Canberra, ACT, Australia collection, see Blakemore, 2005).
Habitat: limicolous, generally in irrigated or sodden soil, besides water courses or in drains, under rice, sugarcane, and pasture soil. Can survive in clay soils (Blakemore, 1994).

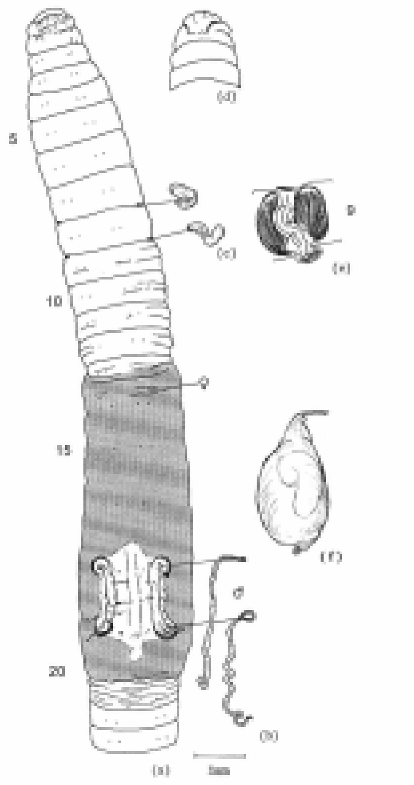

Figure 1 *Eukerria saltensis* Qld specimen, (a) anterior view with (b) prostates and (c) spermatheca *in situ*, (d) prostomium, (e) laterally paired ocnerodrilid diverticula in 9, (f) cocoon with embryo visible. (After Blakemore 2002, fig. 1.7).

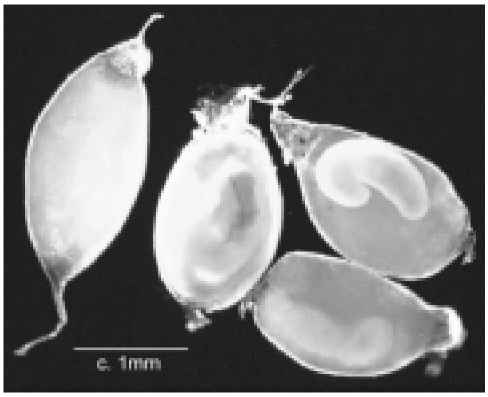

Figure 2 *E. saltensis* cocoons, lengths 2mm (photo courtesy Dr Mark Stevens, NSW Agriculture from adult specimens identified by R.J.B.).

Registration No.: Japanese specimens to be deposited in National Science Museum, Tokyo.
Length: 30-95mm.
Width: generally ca. 1mm.
Segments: 97-131.
Colour: unpigmented but red from blood and dark from soil in gut; anterior faint, with brilliant, blue iridescence, some worms may appear white from coelomocytes in body cavity. Clitellum yellowy or pale.
Behaviour: fairly docile although white prostomium probes inquisitively; body readily extends and is easily broken; when in water specimens aggregate in coiled masses; specimens coil on preservation and produce much mucus.
Prostomium: epilobous, closed or open.
First dorsal pore: none (consistent with aquatic habitat).
Setae (ratio of aa:ab:bc:cd:dd:U): 8 per segment, **ab** absent on 17, **b** absent on 18-19. (3:1:3:1:10:0.44).
Nephropores: not visible (in **ab** lines?).
Clitellum: 13½, 14-20; mostly annular but thin or absent near male field.
Male pores: acanthodriline, on 18 in slightly inwardly bowed seminal grooves between pairs of prostates equatorial in **ab** in 17 and 19.
Female pores: 14, variously: paired longitudinal slits anterior to setae **a** almost at 13/14; longitudinal slits just anterior to **b**; only a single pore found in two Qld specimens just anterior to **b** line on right hand side.
Spermathecal pores: inconspicuous in 7/8 and 8/9 lateral between **b** and **c** lines, often closer to **c**.
Genital markings: none.
Septa: 5/6-11/12 present and thick.
Dorsal blood vessel: single, continuous onto pharynx.
Hearts: 9, 10 and 11; supraoesophageal vessel with commissurals in 10 and 11.
Gizzard: weakly muscular barrel or pear-shaped gizzard in 7.
Calciferous glands or diverticula: paired in 9, ventro-laterally discharging into oesophagus at 9/10, (glands are supplied by small capillaries and have thick walls and central lumen in cross section).
Intestine origin: commences between 11-13 (caeca, typhlosole absent).
Nephridia: holoic, commencing from around 7, avesiculate (consistent with aquatic habitat).
Testis/sperm funnels: free and iridescent in 10; paired seminal vesicles in 9 or 9 and 11 [or funnels in 11, seminal vesicles in ?11 and 12].
Ovaries: large pair palmate in 13.
Prostates: two pairs of thin elongate tubular prostates with short muscular ducts in 17 and 19, intercoiled and extending back several segments.
Spermathecae: two pairs in 8 and 9; moderately small; ampullae may be bent at right angles to longer duct; adiverticulate; non iridescent.
Gut contents: fine soil and colloidal organic matter (consistent with habitat).
Reproduction: Gates (1972) provides data that shows this species to have both biparental and parthenogenetic reproduction; there is some evidence to suggest (internal?) self-fertilisation of some isolated specimens in laboratory experiments.

Figure 3 *E. saltensis* known World distribution (original).

Genetic profiles: not yet known to have been sequenced.

Notes: collected in relatively large numbers at CSIRO Narayen, Qld from sodden soil beside a water tank, where they were difficult to extract since they produced a viscous, "gummy" exudate that adhered to their bodies which snapped in two when stretched. From Samford, they were found in very moist clay soil, coiled at 5-10cm depth or active in the root zone in two locations and in association with several other earthworm species, including *Aporrectodea trapezoides.* Maclean specimens were collected form the delta of Clarance River in irrigated alluvial soil under sugarcane, found in association with *Zacharius zacharyi* Blakemore, 1997. Specimens of *Eu. saltensis* from Whitton and Deniliquin were collected from rice paddies where they were abundant and thought to be rather problematic as they attracted wading birds such as ibis (*Threskiornis* spp.) which muddied the water (M. Stevens *pers. comm*. 1994, and see Stevens & Warren, 2000; Stevens, 2003). In Japan, several specimens were collected from a drainage channel beside a shrine at Kamakura and a riverbank at Machida (similar specimens collected from Kochi, Shikoku Island by R.J.B. in 2004 were too damaged to reliably identify to species). It is not known if their spread into Japanese rice fields is likely to be problematic or not, as Japanese rice is generally transplanted, unlike in Australia where it is sown.

In moist habitats, this species may be easily confused with chironomid larvae (Diptera: Chironomidae) or aquatic microdriles such as tubificids that are called "bloodworms"; although of similar size, these larvae and microdriles have body appendages unlike true 'earthworms' and can be more serious pests of rice paddies (e.g. *http://www.rirdc.gov.au/reports/RIC/99_141.doc*).

This species was deliberately introduced along with lumbricids into cultivated soils in NSW, but failed to clearly demonstrate beneficial effects although air permeability of the red-brown earth soil was reportedly increased (Blackwell & Blackwell, 1989). In a series of laboratory and glasshouse trials on studies of about 30 native and alien species by Blakemore (1994; 1997), they were found to have slight to negligible effect on mesocosm plant yield and soil structure, and their small size and susceptibility to injury made them difficult to handle. They also seem to require high moisture for survival. Although found to be of negligible benefit in these plant growth studies, like many other peregrine species, the ubiquity and range of distribution of *Eukerria saltensis* in Australia, and now Asia, from a supposed South American origin is quite remarkable.

Key to alien/peregrine *Eukerria* species originating from South America

Spermathecal pores in setal ab lines
.... *E. kuekenthali* (Michaelsen, 1908)
Spermathecal pores slightly median to setal c lines
.... *E. saltensis* (Beddard, 1895)

Accidental introduction of symbionts with imported freshwater shrimps

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Abstract Many specimens of the symbiotic branchiobdellidan, *Holtodrilus truncatus* have been recorded on the freshwater atyid shrimp, *Neocaridina denticulata*, in central Japan since 2003. *H. truncatus* had been restricted to China and the new localities in Japan are located within a small area where recreational fishing sites are concentrated and large amounts of the potential host shrimps have been introduced from China for use as live baits. It is therefore likely that the branchiobdellidans were unintentionally introduced into Japan associated with the introduction of the host shrimps. Japanese and continental populations of the *N. denticulata* are closely related to each other and hybridisation between them is predicted.

Keywords: Accidental introduction; alien species; branchiobdellidan; *Holtodrilus truncatus*; freshwater shrimp; *Neocaridina denticulata*; *Scutariella*

INTRODUCTION

Symbionts accompanying an intentionally introduced alien species are often ignored unless they result in outbreaks of diseases. Nevertheless, they may alter the biodiversity of the area where the hosts are introduced. In Japanese freshwater environments, multiple phyla of non-pathogenic alien invertebrate symbionts have been recorded in association with the introduction of alien crayfish (Yamaguchi 1933, Tamura *et al.* 1985, Smith and Kamiya 2001, Ohtaka *et al.* 2005).

In the course of an ecological study in 2003 of the freshwater atyid shrimp, *Neocaridina denticulata* in the Sugo River, Yumesaki River system, Hyogo Prefecture, Japan, many symbiotic worms were found on the shrimps' body surfaces. The symbionts consisted of two taxonomic groups: branchiobdellidans (Annelida, Clitellata) and temnocephalidans (Platyhelminthes, "Turbellaria"). The branchiobdellidans were identified as *Holtodrilus truncatus* hitherto known only from China (Niwa *et al.* 2005). Niwa *et al.* (2005) suggested an unintentional introduction of the branchiobdellidans, basedon the frequent introduction of Chinese shrimps into the area where the branchiobdellidans occurred.

In this paper, the distribution of the branchiobdellidans in central Japan is shown and the possibility of accidental introduction is discussed in relation to the present introduction of the potential host shrimps from the Eurasian Continent to the Japanese archipelago.

MATERIAL AND METHODS

The freshwater shrimp, *N. denticulata* (Fig. 1), is distributed in east Asia covering China, Korea, Taiwan and Japan. A series of subspecies have been described for *N. denticulata*, based on morphological and geographical differences (cf. Cai 1996). The Japanese native population, distributed in the western parts of the Japanese Islands, has traditionally been ascribed as a subspecies *N. denticulata denticulata* (e.g. Kubo 1938). However, subspecific recognition of *N. denticulata* is often difficult and the infraspecific taxonomy is still controversial (Cai 1996, Liang 2004, Nishino and Niwa 2004). In the present paper, therefore, we ascribe every material of this species to *N. denticulata* without denoting subspecific or other infraspecific names.

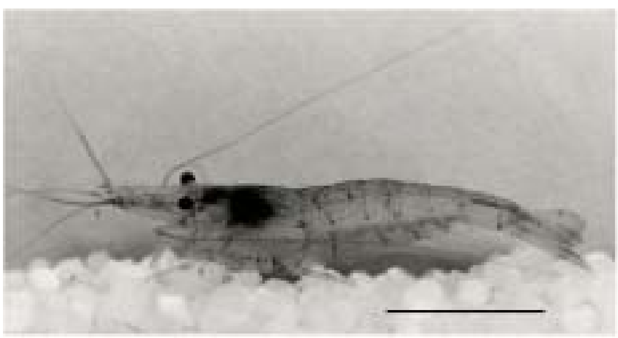

Figure 1 A live female specimen of *Neocardina denticulata* (Decapoda; Atyidae) from the Sugo River, on 10 September, 1993. Scale bar, 10mm.

Pages 182-186. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

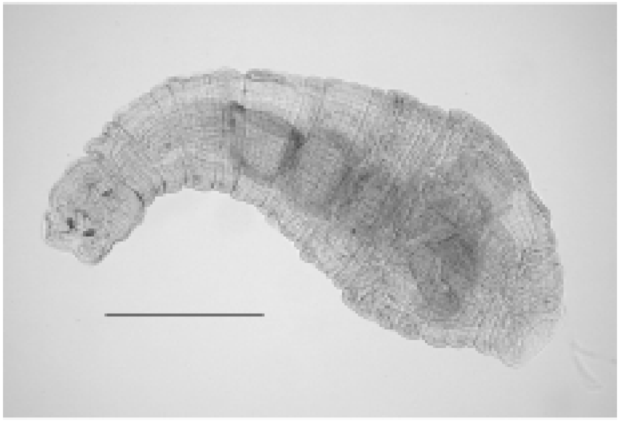

Figure 2 A fixed slide specimen of *Holtodrilus truncatus* (Branchiobdellida) from Sugo River 16 September, 2003. Scale bar, 0.5mm.

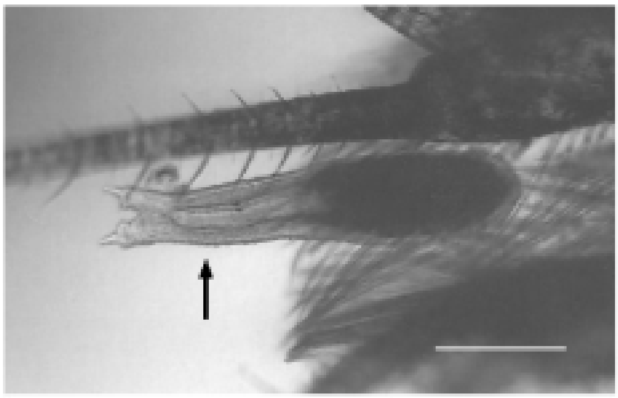

Figure 3 A live specimen of *Scutariella* sp. (Temnocephalida) (arrow) on the body surface of *N. denticulata* from Sugo River on 9 May, 2004. Scale bar, 0.5mm.

To inspect the ectosymbionts, specimens of *N. denticulata* (Fig. 1) were randomly collected from September 2003 to May 2004, from 24 sites in 17 rivers in Kinki and Chugoku districts in central Japan. The collecting surveys were carried out twice in Yumesaki R., Sugo R., Ibo R.and Asahi R., and once in the other rivers. All of the shrimp specimens collected at a locality at one time of collection were placed together in a single container, and fixed in 10 % formalin solution. The shrimp's external surfaces, including the branchial cavity, together with any debris at the bottom of the storage containers were examined for branchiobdellidans and temnocephalidans, using a dissection microscope. All branchiobdellidans were *H. truncatus* (Fig. 2), and the temnocephalidans were *Scutariella* sp. (Scutariellidae) (Fig. 3). Mean abundance of symbionts was calculated for each sample, as the total number of symbionts per total number of host shrimps examined. In addition, ectosymbionts of imported shrimps arriving at Kansai Airport were examined in October 2003.

Recent history and statistical information on the importation of shrimps into Japan were obtained (personal communication, anonymous importer in Akashi, Hyogo Prefecture, central Japan).

RESULTS AND DISCUSSION

Distribution of branchiobdellidans and temnocephalidans in central Japan

Symbiotic *Holtodrilus truncatus* were found at 10 sites in 7 rivers (Tab. 1, Fig. 4). The localities were all within a 60km range bordered by Akashi R. (No. 5) in the east and Chikusa R. (No.12) in the west. All localities were in Hyogo Prefecture, in rivers flowing into the Setonaikai Sea. The mean abundance of branchiobdellidans per host shrimp was lower than 0.6 in all localities, except for Sugo R. (No. 8) where mean abundance was as high as 4.5.

Temnocephalidan *Scutariella* sp. were found at 21 sites in 17 rivers, more widely distributed than branchiobdellidans (Table 1, Fig. 4). Mean abundance of the temnocephalidans per host shrimp ranged from 0 to 8.1. The highest value was found in the Sugo R., in which branchiobdellidans were also abundant. Localities of *Scutariella* sp. included all 10 of the sites at which branchiobdellidans were also found. Many shrimp specimens carried both branchiobdellidans and temnocephalidans at every site where both ectosymbionts occurred sympatrically. Shrimps from two sites, one each in Maruyama R. (No. 13) and Asahi R. (No. 17), did not carry any ectosymbionts.

History of unintentional introduction of alien symbionts associated with the introduction of decapod crustaceans in Japanese freshwater habitats

Branchiobdellidans are obligate, ectosymbionts living on freshwater astacoidean crayfish and occasionally on atyid shrimps in the Holarctic (Gelder, 1999, Brinkhurst and Gelder 2001). Thirteen native and endemic branchiobdellidan species have been described from Japan on the country's only endemic crayfish, *Cambaroides japonicus,* which is distributed across Hokkaido and the northern part of Honshu (Yamaguchi 1934, Gelder and Ohtaka 2000a). North American branchiobdellidans, *Cambarincola okadai,* were the first alien branchiobdellidan species to be reported in Japan (Yamaguchi 1933, Gelder and Ohtaka 2000b) following introduction of North American crayfish in the 1920s. Two other North American branchiobdellidan species, *Sathodrilus attenuatus* and *Xironogiton victoriensis,* have been recorded from introduced North American signal crayfish, *Pacifastacus leniusculus* (Kawai *et al.* 2004,

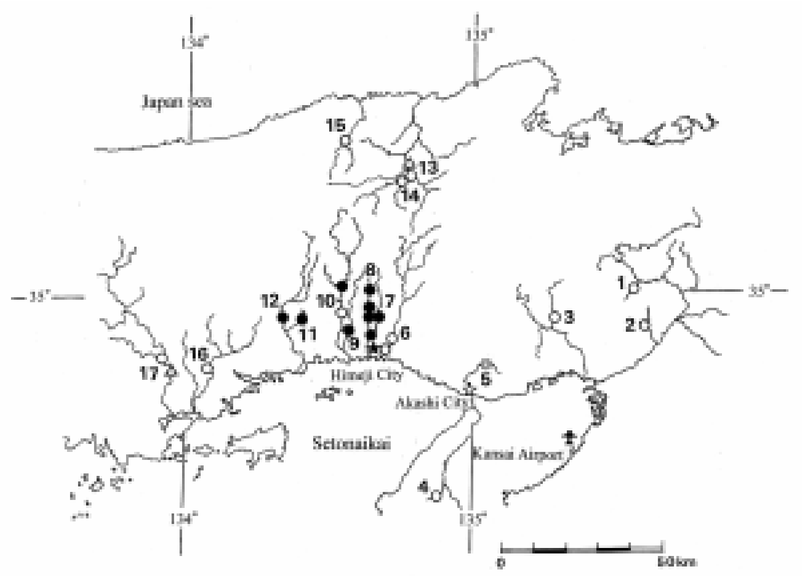

Figure 4 Occurrence of symbiotic branchiobdellidans and temnocephalidans on *Neocardina denticulata* in Kinki and Chugoku districts in Central Japan. Solid circle, localities in which both branchiobdellidans (*Holtodrilus truncatus*) and temnocephalidans (*Scutariella* sp.) occurred; double circle, those in which only branchiobdellidans occurred; open circle, those in which only temnocepalidans occurred; triangle, any ectosymbionts did not occur. Centres of Akashi City (open star) and Himeji City (solid star) are also denoted. River No. 1: Inukai R., Kyoto Prefecture; 2: Akuta R., Osaka Prefecture; 3: Hatsuka R., 4: Sumoto R., 5: Akashi R., 6: Ichikawa R., 7: Yumesaki R., 8: Sugo R., 9: Hayasida R., 10: Ibo R., 11: Yano R., 12: Chikusa R., 13: Maruyama R., 14: Ohya R., and 15: Yada R., Hyogo Prefecture; 16: Yoshii R. and 17: Asahi R., Okayama Prefecture.

Table 1 Mean abundance of symbionts on *Neocardina denticulata* shrimp from 17 rivers in central Japan. Results of surveys in Sep.-Nov., 2003 were combined, except for Sugo R. in which results for April-May, 2004 are shown.

Location in Fig.4	Name of River	Date of sampling	No. of shrimps examined	Mean abundance	
				Holtodrilus truncatus	***Scutariella* sp.**
1	Inukai R.	2003/10/22	65	0	1.2
2	Akuta R.	2003/10/30	81	0	0.7
3	Hatsuka R.	2003/11/3	295	0	0.6
4	Sumoto R.	2003/10/23	42	0	0.02
5	Akashi R.	2003/9/14	200	0.01	0
6	Ichikawa R.	2003/10/11	315	0	2.9
7	Yumesaki R.	2003/11/3	121-285	0.4-0.5	1.5-1.7
8	Sugo R.	2004/4/25,5/15	81-199	0.5-4.5	3.1-8.1
9	Hayashida R.	2003/10/11	500	0.04	0.6
10	Ibo R.	2003/9/13,11/13	51-75	0-0.08	0.6-1.2
11	Yano R.	2003/10/11	50	0.1	1.1
12	Chikusa R.	2003/10/11	50	0.1	4.1
13	Maruyama R.	2003/10/26	252	0	0.3
14	Ohya R.	2003/10/26	122	0	0.4
15	Yada R.	2003/10/26	4	0	1.0
16	Yoshii R.	2003/10/18	125	0	1.0
17	Asahi R.	2003/10/18,11/2	10-208	0	0-0.6

Ohtaka *et al*. 2005). The former branchiobdellidan species was confirmed from voucher crayfish samples deposited since 1930s. Another branchiobdellidan, *Holtodrilus truncatus*, was found and described for the first time from the Chinese shrimp, *Neocaridina denticulata sinensis* from a spring near Sichwan, Hunan Province, China (Liang 1963). The second record of this species was made from the Chanjiang River near Shaoguan, Guangdong Province, China, on the same species of shrimp (Liu 1984). The recent finding of *H. truncatus* on *N. denticulata* in Japan (Niwa *et al.* 2005) was the first record of Eurasian continental branchiobdellidans in Japan, and also the first record of branchiobdellidans found on a non crayfish host in Japan.

Closely similar to branchiobdellidans, temnocephalidans are symbiotic on freshwater crayfish and other decapod crustaceans, and occasionally on vertebrates. They are restricted mainly to the southern continents, but the distributions of some groups extend to the northern hemisphere, including Japan (Gelder 1999). According to the recent bibliographic study by Kawakatsu *et al*. (personal communication), three species of temnocephalidans have been known from Japanese fresh waters. A native species, *Scutariella japonica*, has been recorded on freshwater atyid shrimps in Honshu and more southern islands in Japan. This species has also been recorded in Taiwan and Korea (Kawakatsu *et al.* personal communication). Another native species, *Temnosewellia* sp., which closely resembles *T. semperi*, has been recorded on freshwater crabs in Kyushu and the Southwest Islands, Japan. An Australian temnocephalidan, *Temnosewellia minor,* was recorded in 1985 on an alien crayfish, *Cherax tenuimanus,* introduced from Western Australia into Ibusuki City, Kagoshima Prefecture (Tamura *et al.* 1985).

It is also probable that alien temnocephalidans have already been introduced into Japan in accordance with the introduction of the host shrimps. However, in the case of *Scutariella japonica,* the alien populations may not be easily distinguished from native ones because this species is distributed naturally in Japan as well as in neighbouring areas.

Ectocytherid ostracods are other symbionts living on crustacean hosts, including crayfish, isopods, amphipods and crabs. Smith and Kamiya (2001) recorded the North American entocytherid ostracod, *Uncinocythere occidentalis*, on *Pacifastacus leniusculus* from Hokkaido, northern Japan. This is the first record of this species outside western North America.

Import of the potential host shrimps and associated problems

Live *N*. *denticulata* have been imported into Japan from continental East Asia, to be used as baits (called "Butsuebi") in recreational fishing for marine and brackish fish, such as black sea bream, *Acanthopagrus schlegeli* (Sparidae), and the black rockfish, *Sebastes inermis* (Scorpaenidae). Although *N*. *denticulata* does not survive in seawater, it (and its symbionts) will survive and may spread when released into freshwater habitats. In addition, live specimens of the alien *N. denticulata* can also be bought as "cleaners" for ornamental aquaria via internet sales as well as from many aquarium shops in Japan. However, authoritative statistical data on the quantity and distribution of imported live shrimps are currently scarce, because the majority of the shrimps are traded privately and no distinction is made in the records between those for food use and those for other use.

At present there are about ten trading companies dealing with live *N*. *denticulata* in Japan; three in Tokyo, five or six in Kansai district and one in Kyushu. According to personal communications from an anonymous importer in Akashi, Kinki district, live *N*. *denticulata* have been imported from Korea as bait for fishing since the 1970s, with a change to China as a source since the 1990s. The localities of origin of the imported shrimps change with the seasons and with demand. In the 1998-2004 period, the annual amount of imported shrimps from one of the importers ranged from 2,780 to 7,867kg fresh weight (an anonymous importer, personal communication).

In the present study, in October 2003, 365 individuals of *Scullariella* sp. were obtained from 629 live *N*. *denticulata* from China, at Kansai Airport. In Kansai district, the majority of the imported shrimps arrived at Kansai Airport and were then distributed to neighbouring areas. Akashi and Himeji, in Hyogo Prefecture, are famous fishing areas in the Kinki District and a large amount of imported *N*. *denticulata* has been routinely transported there. The fishing area overlaps with the localities in which the branchiobdellidan, *H*. *truncatus,* has been found. It is, therefore, likely that the recent occurrence of *H*. *truncatus* in central Japan is an unintentional introduction from continental China associated with the introduction of live shrimps from China.

In addition, since the Japanese and Chinese populations of *N*. *denticulata* are closely related to each other, hybridisation between them can also be expected. In recent years, *N*. *denticulata* specimens, whose morphologies closely resemble Chinese *N*. *d. sinensis*, have been recorded from several localities in Japan. These localities extended beyond the natural distribution range of the Japanese native *N*. *denticulata* (Nishino and Niwa 2004). This also suggests that alien or hybrid shrimps have successively established in various water systems. Considering the nationwide increase in the popularity of recreational fishing and aquaculture, the import of alien shrimps is expected to increase, and the opportunity for unintentional introduction of alien symbionts is therefore also expected to be increasing.

ACKNOWLEDGEMENTS

We thank Dr. Machiko Nishino of Lake Biwa Environmental Research Institute, Otsu, Japan, for her invaluable suggestions for study of *Neocaridina*. An anonymous importer in Akashi, Japan, provided information about imported shrimps from China and Korea. Our thanks are also due to Dr. Masaharu Kawakatsu of Sapporo, Japan, for his suggestion on taxonomy and distribution of Temnocephalida. We also thank Messrs. Mr. Tatsuya Yokoyama of Osaka Water Museum, Osaka Pref., Toshio Doi of Kobe Municipal Suma Aqualife Park, Hyogo Pref. and Hiroyuki Yoshida of Wesco Okayama Pref., all in

Japan, for their assistance in the field. We also extend our appreciation to Dr. Katsuki Nakai of Lake Biwa Museum, Shiga Pref., Japan, for providing us with the opportunity for presentation of this study for the International Conference on Assessment and Control of Biological Invasion Risks.

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Distribution and status of the introduced red-eared slider (*Trachemys scripta elegans*) in Taiwan

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Abstract From 2001 to 2005, I conducted an island wide survey to assess the current status and distribution pattern of the introduced aquatic turtle, *Trachemys scripta elegans*, in Taiwan. A total of 265 individuals were captured from 31 sites. Among them, only one was obtained in southern Taiwan, and none from the eastern region. This introduced turtle is more common in the northern and central regions of Taiwan, especially in urban and suburban areas. The population sizes in most surveyed sites were still small, and few small-sized juveniles have been found in the wild, indicating that the establishment of feral populations is still rare. Meanwhile, it has been confirmed that this turtle has established reproductive populations in northern and central regions of Taiwan. Compared to the results of a previous study conducted in 1995-1996, the relative abundance of *T. s. elegans* did not increase significantly at two sites, but it increased greatly in the Keelung River, a main study site with high environmental disturbance in northern Taiwan. In the analyses of stomach contents, I did not find clear evidence to claim that *T. s. elegans* may compete for food resources with sympatric native turtles, mainly *Ocadia sinensis* and *Mauremys mutica*. It is more carnivorous, ingesting more animal food items than other coexisting turtle species.

Keywords: *Trachemys scripta elegans*; distribution; current status; population trend; dietary composition; Taiwan

INTRODUCTION

The red-eared slider (*Trachemys scripta elegans*) [Fig. 1 (a)], a subspecies indigenous to the southern United States (Iverson 1992, Ernst *et al*. 1994), has been reported to be introduced to the wild outside its natural range, including various countries and regions in Asia, Europe, Australia, and Africa as a result of release of unwanted pets (Newberry 1984, Uchida 1989, Da Silva and Blasco 1995, Ota 1995, Luiselli *et al*. 1997, Chen and Lue 1998*a*, Cox *et al*. 1998, Liat and Das 1999, Cadi *et al*. 2004). This turtle is well known as a successful invasive species and regarded as one of the world's worst invasive alien species (Lowe *et al*. 2000). Although it is generally argued that introductions of *T. s. elegans* may cause detrimental impacts on native turtles or fauna (HSUS 1994, Da Silva and Blasco 1995, Moll and Moll 2000), little evidence is available to support this claim. In captivity, the introduced *T. s. elegans* is reported to compete for basking sites and detrimentally impact on other sympatric turtles (Cadi and Jolly 2003, 2004). Detailed studies of potential impacts of the introduced alien turtles on the environment are still lacking.

In Taiwan, the introduction of foreign commercial and pet animals has been a common

(a) (b) (c)

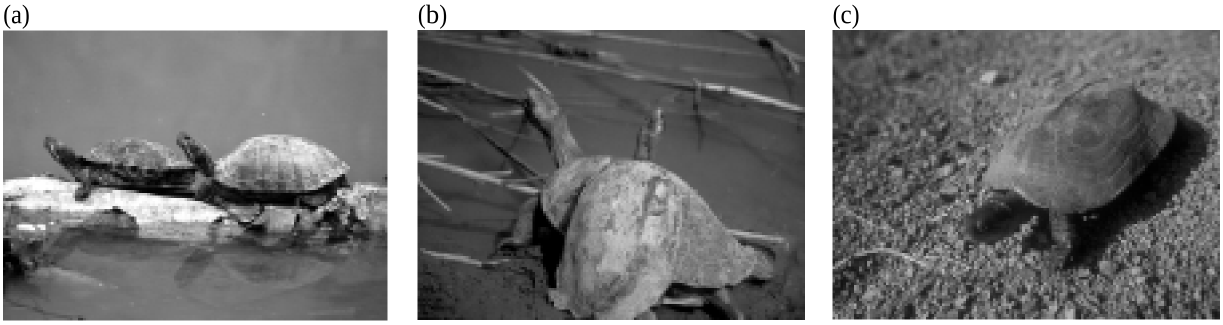

Figure 1 (a) Two basking red-eared sliders (*Trachemys scripta elegans*) in the Keelung River, northern Taiwan. (b) The most common turtle, the Chinese stripe-necked turtle (*Ocadia sinensis*), usually coexisting with *T. s. elegans* in Taiwan. (c) The Asian yellow pond turtle (*Mauremys mutica*) coexisting with *T. s. elegans* in montane areas.

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practice (Shao and Tzeng 1993). In addition to the release of unwanted pets, red-eared sliders are usually set free in Buddhist mercy ceremonies, resulting in its wide spread in various aquatic habitats. Although the history of introduction of this turtle is poorly documented in Taiwan, it is believed that the practice has lasted for decades (Shao and Tzeng 1993). It is reported that this alien turtle has established feral populations in the Keelung River, in northern Taiwan (Chen and Lue 1998*a*).

The purpose of this study is to investigate the status and distribution patterns of the introduced *T. s. elegans* in Taiwan. I also compared the community structure of aquatic turtles in the several sites in northern Taiwan during 1995-1996 and 2001-2003, and concentrated the population study in the Keelung River. Stomach contents of *T. s. elegans* and two sympatric native turtles, *Ocadia sinensis* and *Mauremys mutica* [Fig. 1 (b), (c)] were obtained and analysed to investigate potential food competition between the species.

METHODS

I conducted the fieldwork from February 2001 to August 2005 in various aquatic habitats of Taiwan. Based on 1:25,000 maps published by the Ministry of Interior, Republic of China, I identified the potentially suitable habitats for aquatic freshwater turtles, such as ponds, rivers, agricultural ditches, or wetlands. Because of high frequency of lost traps associated with them, I avoided the man-made ponds located in urban areas, although the occurrence of alien turtles was higher than in remote areas. Turtles were collected with funnel traps baited with canned cat food. Three to five traps were set at each site along the riverbanks or margins of aquatic regimes and were checked once to twice each week. When no turtles were captured after two weeks of trapping, I regarded the site as one where no turtles were present, or where they were at low population density. Since this study was carried out in conjunction with a general survey of freshwater turtles and analysis of population genetics of native turtle species, I moved from one site to another after one month of trapping, except at four main study sites: Keelung River, Shenaokeng (1), Gongliao (2), and Ermei (3) (See Appendix 1 for list of sites). As the trapping efficiency was highly related to season and weather condition, I did not estimate the trapping effort in this study.

The maximum straight carapace length (CL) of each individual was measured to the nearest 0.1mm with vernier callipers. Animals were sexed based on secondary sexual characteristics by examining the position of the cloacal opening and elongation of the foreclaws. The captured turtles were marked with individual notching on the marginal scutes, using a hand saw, for further identification on subsequent recapture.

As I conducted a field study of *T. s. elegans* in the middle section of the Keelung River in 1995-1996, I concentrated my work on the community structure of freshwater turtles in the same area. Originally, the river in this study area was narrow, meandering, and sluggish, with a width of about 25m and depth of 1.5m. The water level of the study area was highly variable, with occasional flood inundating the adjacent urban areas. A large-scale flood control project was carried out from 1998 to 2000. Riverine and riparian habitats have been much modified and disturbed as a result, mainly through the construction of levees, flood walls, cement trapezoidal channels and channel dredging. Hence, I had the opportunity to investigate the trend in community structure in this highly disturbed habitat. During 1995 and 1996, I also conducted a similar survey in parts of northern Taiwan (Chen and Lue 1998*a*), which gave me the opportunity to monitor the community structure of aquatic turtles at two other study sites in addition to the Keelung River.

The stomach contents of turtles captured were obtained by using stomach flushing (Legler 1977, Parmenter 1980), and were preserved in 70% ethanol for further identification. Each food item was identified to the lowest taxon, as possible. Fragments of litter were assumed to have been accidentally ingested and were excluded from the analysis. As traps were checked at intervals of three to seven days at most sites, there was a high frequency of empty stomachs. Stomach samples were taken for *T. s. elegans* and coexisting native species, pooled from various sites in northern Taiwan.

In the analyses of correlations of capture frequency between introduced *T. s. elegans* and native species, turtle capture data were log-transformed [$\log_{10}(x+0.5)$] to improve normality. Statistical tests follow Sokal and Rohlf (1996).

RESULTS

Distribution patterns and population sizes

In the trapping programme conducted from February 2001 to August 2005, a total of 1,928 aquatic turtles were captured from 105 sites in various regions around Taiwan, including an island near mainland China (Tab. 1). Among those turtles captured, 265 *Trachemys scripta elegans* were captured from 31 sites (Fig. 2). This introduced turtle was more abundant in

Table 1 The results of trapping programme of *Trachemys scripta elegans* and native turtles from various regions of Taiwan in 2001-2005.

Regions	No. of sampling sites	*Ocadia sinensis*	*Mauremys mutica*	*Chinemys reevesii*	*Pelodiscus sinensis*	*Trachemys s. elegans*	Other species
Northern Taiwan	54	1157	235	-	17	241	4
Central Taiwan	17	91	17	-	7	22	-
Southern Taiwan	20	100	-	-	-	1	-
Eastern Taiwan	8	2	11	-	7	-	-
Kinmen Island	6	-	-	14	1	1	-
Subtotal	105	1350	263	14	32	265	4

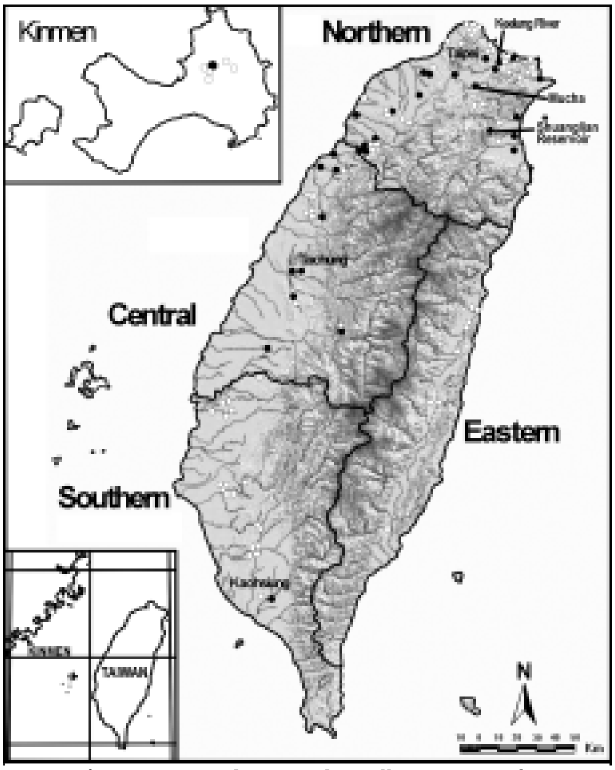

Figure 2 Map showing the collection sites for *Trachemys scripta elegans* in Taiwan in 2001-2005. The black dots indicate collection sites for *T. s. elegans*; the white dots indicate sites without *T. s. elegans*.

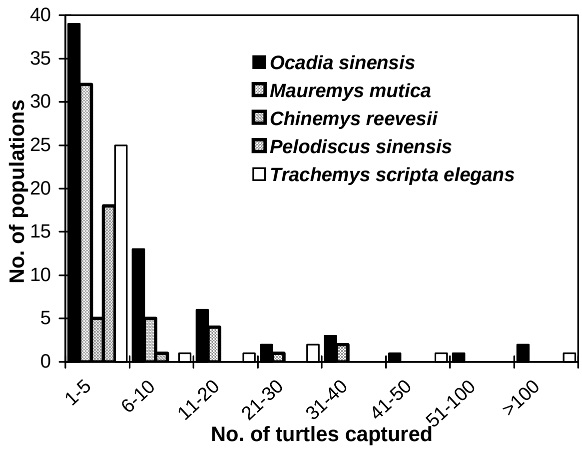

Figure 3 Histogram showing numbers of *Trachemys scripta elegans* and other sympatric native species collected at different sampling sites in 2001-2005

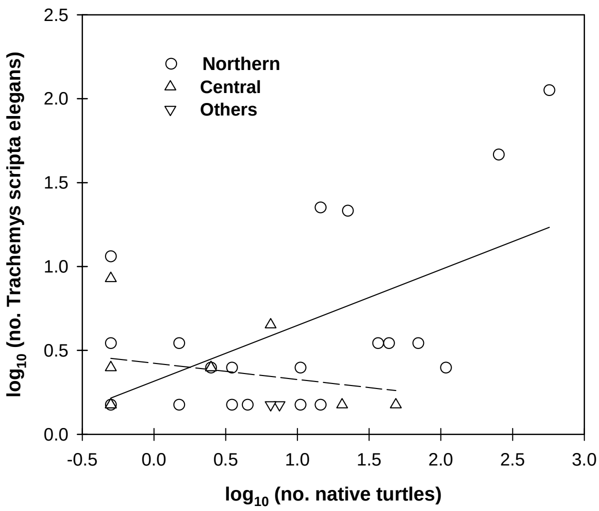

Figure 4 Scatter diagram of numbers of *Trachemys scripta elegans* and other sympatric native species collected in 2001-2005. Northern Taiwan (solid line): $y = 0.32 + 0.33x$, $R^2 = 0.29$; Central Taiwan (dashed line): $y = 0.42 - 0.10x$, $R^2 = 0.08$.

northern and central regions than in other regions in Taiwan; it has been captured at 20 sites in northern Taiwan and at nine sites in central Taiwan (Fig. 2). Only one individual was captured in southern Taiwan, and none have been captured in the eastern region. On Kinmen Island, the introduction of *T. s. elegans* was also evidenced; one individual has been captured in 2004. At 12 study sites, the Yilan River, Banciao, Sijhih (1), Lujhu (1), Lujhu (2), Bade, Jhubei, Jhunan, Sanyi, Wurih, Dali, and Douliou, the capture rate for introduced *T. s. elegans* was equal to, or higher than that for sympatric native turtles (Appendix 1).

Based on the results of the trapping programme, most aquatic turtle population sizes were small (Fig. 3); only a few populations with a number of turtles

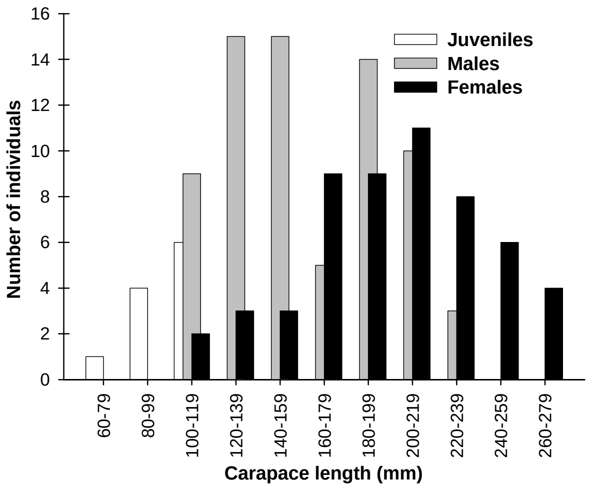

Figure 5 Histogram showing carapace length distribution of *Trachemys scripta elegans* from Taiwan except for the main study site in the Keelung River, collected in 2001-2005.

captured >20 have been found: *O. sinensis* from nine sites, *M. mutica* from three sites, and *T. s. elegans* from four sites. There was a significant positive relationship between the numbers of *T. s. elegans* captured and other turtles captured from northern Taiwan (Fig. 4; $F_{1, 18} = 7.31$, $P < 0.05$, $R^2 = 0.29$), suggesting that this study did not provide confirmation for a general prediction of exclusive competition between this introduced turtle and native turtles. In central Taiwan, the relationship was not statistically significant either, though there was a weakly negative slope ($F_{1,7} = 0.60$, $P > 0.05$, $R^2 = 0.08$).

Four other species of introduced turtles: *Graptemys kohnii*, *Pseudemys nelsoni*, *P. concinna* and *Cuora trifasciata* were also captured in this study. The capture rate was very low, with only one single case found for each species.

Table 3 The number of introduced *Trachemys scripta elegans* and native *Ocadia sinensis* for each sex and unsexed juveniles captured from the Keelung River during the two different study periods (1995-1996 and 2001-2002).

Species/period	Sex			Total captured
	Juvenile	Male	Female	
Trachemys scripta elegans				
1995-1996	9	28	31	68
2001-2002	2	78	32	112
Ocadia sinensis				
1995-1996	133	322	243	698
2001-2002	8	413	146	567

Population trends of *Trachemys scripta elegans* in northern Taiwan

The carapace length distribution of *T. s. elegans* captured, excluding data from the Keelung River is shown in Fig 5. The size composition was dominated by large individuals. Though small-sized juveniles (< 100mm CL) only occupied 4.7% of *T. s. elegans* captured, they have been found at five sites, including Shuanglian Reservoir, Banciao, Lujhu (1), Yangmei (1), and Wurih, suggesting that this turtle has established reproductive populations.

Compared to the results of the previous study at three sites in northern Taiwan (Tab. 2), the capture rate of *T. s. elegans* did not increase at two sites (Mucha: $G = 0.40$, $P > 0.05$; Shuanglian Reservoir: $G = 0.12$, $P > 0.05$), but increased significantly in the Keelung River ($G = 19.35$, $P < 0.01$). The recapture rate of *T. s. elegans* at these three sites was low: no recaptured individuals have been found between the two study periods.

In the major study area, the Keelung River, the structure of the turtle community had changed greatly (Tab. 2). The population compositions have changed significantly between the two study periods for both *O. sinensis* ($G = 156.17$, $P < 0.01$) and *T. s. elegans* ($G = 18.52$, $P < 0.01$) the two most important species (Tab. 3). The proportions of small-sized juveniles and females have decreased in both species,

Table 2 The trapping results of *Trachemys scripta elegans* and native turtles in three sites during the two different study periods (1995-1996 and 2001-2002).

Sampling site	Periods	*Ocadia sinensis*	*Mauremys mutica*	*Pelodiscus sinensis*	*Trachemys s. elegans*
Keelung River	1995-1996	698	1	3	68
	2001-2002	567	1	1	112
Mucha	1995-1996	2	0	0	3
	2001-2002	3	0	0	2
Shuanglian Reservoir	1995-1996	12	21	0	2
	2001-2002	32	37	0	3

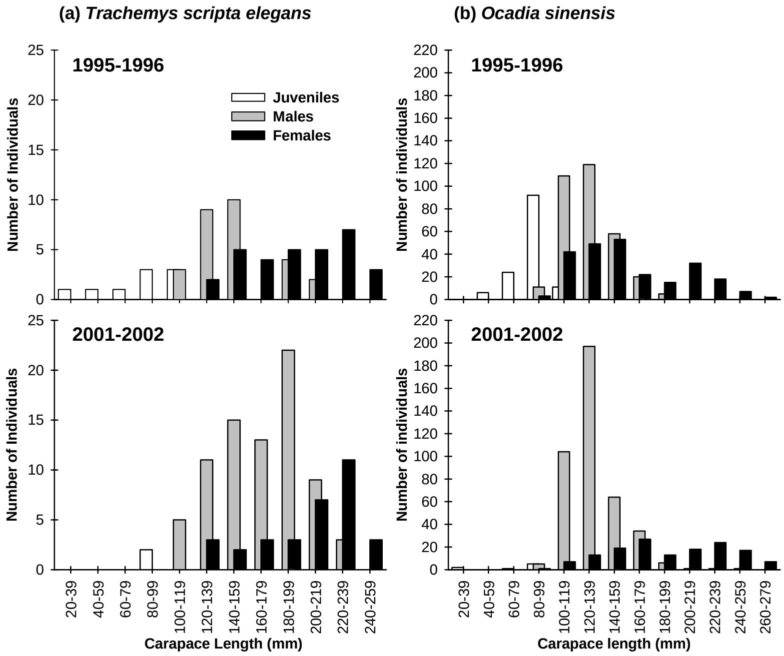

Figure 6 Histogram showing carapace length distribution of *Trachemys scripta elegans* (a) and coexisting *Ocadia sinensis* (b) collected from the Keelung River in 1995-1996 and 2001-2002, respectively.

and that of males has increased greatly (Fig. 6). In May 2002, three female *T. s. elegans* were found attempting to lay eggs on the riverbank of the Keelung River, indicating the viability of this population.

Dietary composition of *Trachemys scripta elegans* in the wild

A comparison of stomach contents of *T. s. elegans* with those from coexisting native turtle species, mainly *O. sinensis* and *M. mutica*, shows that *T. s. elegans* is more carnivorous, ingesting animal food more frequently (Tab. 4). I did not find clear evidence of dietary overlap between *T. s. elegans* and the other two turtles. For *T. s. elegans*, animal materials occurred in 79.5% of the stomach samples, and plant materials appeared in 71.8% of them. For all three species, plant material was the most frequently item ingested: 93.3% for *O. sinensis*, 85.5% for *M. mutica*, and 71.8% for *T. s. elegans*. Meanwhile, the occurrence of fish (64.1%) was high in the stomach samples of *T. s. elegans*. It has to be pointed out that *T. s. elegans* is a more aggressive predator than the native turtle species - small fish, shrimps and frog eggs have been found in stomach samples.

DISCUSSION

Although this study has not found evidence of direct impact of the introduced *T. s. elegans* on the native turtles, it shows that this turtle is widely distributed in Taiwan. This introduced turtle has become a common species in Taiwan and it has been captured in various aquatic habitats, especially in the northern and central regions. Furthermore, the establishment of reproductive populations has been recorded in the Keelung River (Chen and Lue 1998*a*), and possibly at some other sites, such as Shuanglian Reservoir, Banciao, Lujhu (1), Yangmei, and Wurih, where small unsexed juveniles have been found during trapping.

In the past decades, hatchlings of *T. s. elegans* have been imported into Taiwan in huge numbers as pets. It is reported that at least 182,200 individuals have been imported from the United States to Taiwan during 1994-1997 (Salzberg 1998), and HSUS (2001) reported that the number was as high as 153,203 individuals in 1997 alone. The practice of releasing alien and native animals is common in Taiwan (Shao and Tzeng 1993), and turtle was one of the most common animals released. As well as native turtles, red-eared sliders and other alien turtles may be released to various aquatic habitats as unwanted pets by their owners or may be set free in some religious practices. In Taiwan, Buddhists usually release turtles to the wild in mercy practices, especially in remote mountain areas (Ling 1972). This introduced turtle has been observed for sale near a temple and released in religious ceremonies (Lu *et al.* 1996). This may be part of the reason for the distribution of *T. s. elegans* in some remote sites, such as Su-ao, Shuanglian Reservoir, Sijhih (2), Ermei (1), Ermei (2), and Sun

Table 4 The percentage of occurrence of food items in the stomach samples of *Trachemys scripta elegans* and other two sympatric turtles collected in northern Taiwan in 2001 and 2002.

Food items	***Ocadia sinensis* (N = 104)**	***Mauremys mutica* (N = 14)**	***Trachemys s. elegans* (N = 39)**
Plant Materials	93.3	85.7	71.8
Filamentous algae	1.9	--	--
Gramineae	52.9	78.6	56.4
Murdannia keisak (leaves)	12.5	--	15.4
Polygonum sp.	3.8	--	--
Eclipta prostrata	7.7	--	--
Wedelia trilobata (leaves)	12.5	--	--
Ageratum conyzoides	7.7	--	--
Solanum nigrum	9.6	--	--
Ludwigia sp.	8.7	7.1	7.7
Flowers and fruits	13.5	14.3	23.1
Plant roots and shoots	11.5	14.3	2.6
Animal Materials	33.7	57.1	79.5
Gastropod	1.0	--	2.6
Insecta	9.6	50.0	20.5
Coleoptera	1.0	--	5.1
Diptera larvae and pupae	5.8	7.1	5.1
Lepidoptera larvae	5.8	7.1	--
Odonata larvae	1.0	7.1	--
Unidentifiable insects	7.7	28.6	12.8
Oligochaeta	1.9	--	--
Hirudinea	1.9	--	--
Decapoda (Shrimp and crab)	--	--	5.1
Fish	11.5	7.1	64.1
Frog eggs	--	--	2.6

Moon Lake. For most freshwater turtle species, dispersal is usually limited to specific river drainages or wetland systems. It is reported in a mark-recaptured study that the distance movement for *T. s. elegans* might extend from 0.2 to 9km (Gibbons *et al*. 1990); it ranged from 0.7 to 2.2km by radio-tracking in the Keelung River (Chen unpublished data). The island-wide distribution of *T. s. elegans* in Taiwan might therefore be the result of the practice of releasing turtles by pet owners or Buddhists.

At most of the sites surveyed in this study, the capture rate of *T. s. elegans* was relatively low (Appendix 1), indicating that the establishment of reproductive populations is still rare or that it is difficult to establish breeding colonies in low population density. The capture rate of small-sized individuals was low in the present study by funnel traps. The presence of hatchlings might be underestimated due to sampling bias. As a result, it is possible that there might be more established populations of this introduced turtle than shown in my study. It is generally argued that there might be a negative correlation between the richness of native and invasive alien species, since more competitive alien species can gain dominance and replace native species. In this study however, I did not find a negative correlation between the richness of *T. s. elegans* and coexisting native turtles (Fig. 4). At some sites with the larger population size, such as the Keelung River, the relative abundance of *T. s. elegans* might have increased greatly (Tab. 2). It has been pointed out that some native species may be disadvantaged by alien species in highly disturbed habitats (Byers 2002). Feral *T. s. elegans* populations are likely to establish in areas which have been intensively disturbed by human activity, such as urban and suburban areas, where native turtle populations have been seriously impacted. Vacant niches in disturbed habitats might be occupied by introduced turtles. In the case of the Keelung River, the riverine and riparian habitats have been modified dramatically as part of flood control projects. This may have contributed to the increase of relative abundance of *T. s. elegans* in the turtle community. From comparisons of population structures of *T. s. elegans* and *O. sinensis* between two different periods (Tab. 3 and Fig. 6), it seems that native turtles are more sensitive to environmental disturbance than introduced ones. This is especially true for juveniles and females, which might be more severely impacted by the flood control projects. Due to its aggressive and generalised habits, red-eared sliders may compete for food and space with native turtle species (Moll and Moll 2000). In addition, because red-eared sliders can reproduce more offspring than the coexisting *O.*

sinensis and M. mutica (Chen and Lue 1998*a*, 1998*b*, Chen *et al*. 2000), this population growth might be faster than that of native species, once it has established breeding colonies.

Red-eared sliders are known as opportunistic omnivores, consuming a wide variety of animal and plant food items (Parmenter and Avery 1990). Chen and Lue (1998*a*) noted that *T. s. elegans* ingested animal materials more frequently than plants in northern Taiwan. In this study, there was no clear evidence to conclude that this introduced turtle competes for food resources with the sympatric *O. sinensis* and *M. mutica* (Tab. 4). In the stomach samples, some small-size aquatic animals, such as guppies, freshwater shrimps and frog eggs, have been identified, which have never been found in the stomach samples of native turtles. The impacts of introduced turtles on non-chelonian aquatic fauna might be greater than that on native turtles.

In Europe, it is reported that red-eared sliders may compete for basking places of the native turtles (Cadi and Jolly 2003). However, suitable basking sites are not likely to be a limiting factor in most aquatic habitats of Taiwan, and it is unlikely that there is competition for available basking space with native turtles. The introduction of *T. s. elegans* may also account for mortality and weight loss of native turtles in Europe (Cadi and Jolly 2004). Generally, most invasive alien species are not noticed until they have caused serious damage to the environment. In the case of the brown tree snake in Guam, its detrimental impacts were ignored until about 25 years after the snake's initial colonisation (Savidge 1987). As the possible impacts of introduced *T. s. elegans* on the native fauna and other environment components in Taiwan is still unclear, more detailed studies and further monitoring of this species are strongly recommended.

ACKNOWLEDGEMENTS

I would like to thank many people for kind provision field information and Liu S.L. and Lue Y..D. for their assistance with field work. This research was supported by the Council of Agriculture [91AS-2.1.4-FC-R1 (11), 92AS-4.1.4-FC-R1, 93AS-4.1.1-FB-e2, 94AS-9.1.7-FB-e1] and the National Council of Science (NSC89-2621-Z-396-003, NSC90-2313-B-396-004), Republic of China.

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APPENDIX 1

The results of trapping programme of aquatic turtles from various regions of Taiwan and adjacent island from 2001 to 2005.

Sampling site	Habitat	Species					
		Ocadia sinensis	***Mauremys mutica***	***Chinemys reevesii***	***Pelodiscus sinensis***	***Trachemys s. elegans***	**Other species**
Northern Taiwan							
Keelung River	River	567	1	--	2	112	1
Lioudu	Pond	2	1	--	--	--	--
Badouzih	Pond	1	--	--	--	--	--
Shenaokeng (1)	Pond	37	16	--	5	--	--
Shenaokeng (2)	Pond	--	--	--	--	3	--
Yuemeishan	Pond	3	--	--	--	--	--
Shuanglian Reservoir	Pond	32	37	--	--	3	--
Dajiaosi	Pond	1	2	--	--	--	--
Neicheng	Pond	--	1	--	--	--	--
Toucheng	Pond	7	7	--	--	1	--
Yilan River	River	9	5	--	--	22	--
Dongshan River	River	2	1	--	--	1	--
Su-ao	Pond	6	4	--	--	1	--
Mucha	Stream	3	--	--	--	2	--
Banciao	Ditch	21	--	--	1	21	--
Feitsui Reservoir	Ditch	--	12	--	--	--	--
Jingualiao Stream	Stream	--	--	--	1	--	--
Rueifang	Pond	--	9	--	--	--	--
Gongliao (1)	Pond	--	8	--	--	--	--
Gongliao (2)	Pond	1	--	--	--	--	--
Gongliao (3)	Wet land	62	12	--	--	2	--
Shuangsi (1)	Pond	2	1	--	--	--	--
Shuangsi (2)	Pond	--	7	--	--	--	--
Pingsi	Pond	--	1	--	--	--	--
Sijhih (1)	Pond	-	--	--	--	1	1
Sijhih (2)	Pond	10	--	--	--	2	--
Wanli (1)	Pond	16	5	--	--	--	--
Wanli (2)	Pond	--	2	--	--	--	--
Wanli (3)	Pond	8	4	--	--	--	--
Shihmen (1)	Pond	9	1	--	--	--	--
Shihmen (2)	Pond	--	--	--	--	--	1
Shihmen (3)	Pond	3	--	--	--	--	--
Shihmen (4)	Pond	4	2	--	--	--	--
Shihmen (5)	Pond	2	3	--	--	--	--
Shihmen (6)	Pond	1	3	--	--	--	--
Sanjhih	Pond	--	--	--	1	--	--
Danshuei	Pond	3	--	--	--	--	--
Linkou	Pond	--	--	--	1	--	--
Lujhu (1)	Pond	1	--	--	--	3	--
Lujhu (2)	Pond	--	--	--	1	1	--
Bade	Pond	--	--	--	--	11	--
Longtan	Ditch	20	--	--	--	--	--
Shihmen Reservoir	Pond	11	3	--	--	--	--

(APPENDIX 1 continued)

Sampling site	Habitat	Species					
		Ocadia sinensis	***Mauremys mutica***	***Chinemys reevesii***	***Pelodiscus sinensis***	***Trachemys s. elegans***	**Other species**
Sinwu	Stream	2	--	--	--	--	--
Yangmei (1)	Pond	2	34	--	--	3	--
Yangmei (2)	Pond	--	12	--	1	--	--
Jhubei	Stream	1	--	--	1	2	--
Beipu (1)	Pond	1	3	--	--	1	--
Beipu (2)	Pond	--	1	--	--	--	--
Ermei (1)	Pond	2	3	--	--	--	--
Ermei (2)	Pond	1	2	--	--	--	--
Ermei (3)	Wet land	226	27	--	--	46	--
Ermei (4)	Pond	39	3	--	1	3	--
Ermei Reservoir	Reservoir	6	1	--	1	--	--
Central Taiwan							
Touwu	Pond	--	--	--	--	1	--
Jhunan	River	2	--	--	--	2	--
Houlong	River	15	--	--	5	1	--
Shihtan	Reservoir	7	--	--	--	--	--
Sanyi	Stream	--	--	--	2	2	--
Tongsiao (1)	Pond	--	1	--	--	--	--
Tongsiao (2)	Pond	--	3	--	--	--	--
Yuanli (1)	Pond	6	2	--	--	--	--
Yuanli (2)	Pond	5	--	--	--	--	--
Yuanli (3)	Pond	--	9	--	--	--	--
Wurih	Stream	--	--	--	--	8	--
Dali	Stream	--	--	--	--	2	--
Yuchih	Pond	--	2	--	--	--	--
Sun Moon Lake	Reservoir	6	--	--	--	4	--
Nantou	Stream	2	--	--	--	--	--
Fenyuan	Stream	48	--	--	--	1	--
Douliou	Ditch	--	--	--	--	1	--
Southern Taiwan							
Dalin	Ditch	2	--	--	--	--	--
Shueishang	Pond	1	--	--	--	--	--
Taibao	Pond	29	--	--	--	--	--
Yijhu	Pond	16	--	--	--	--	--
Baihe	Pond	10	--	--	--	--	--
Dongshan	Ditch	1	--	--	--	--	--
Sinhua (1)	Pond	11	--	--	--	--	--
Sinhua (2)	Pond	2	--	--	--	--	--
Sinhua (3)	Pond	1	--	--	--	--	--
Yujing (1)	Pond	3	--	--	--	--	--
Yujing (2)	Pond	1	--	--	--	--	--
Guanmiao	Pond	3	--	--	--	--	--
Longci	Pond	1	--	--	--	--	--
Yanchao (1)	Pond	1	--	--	--	--	--
Yanchao (2)	Pond	1	--	--	--	--	--
Cishan (1)	Pond	1	--	--	--	--	--
Cishan (2)	Ditch	7	--	--	--	--	--
Gaoping River	River	1	--	--	--	--	--
Wandan	Ditch	1	--	--	--	--	--
Neipu	Pond	7	--	--	--	1	--
Eastern Taiwan							
Hualien (1)	Ditch	1	2	--	5	--	--
Hualien (2)	Ditch	1	--	--	--	--	--
Shoufong	Pond	--	2	--	--	--	--
Guangfu (1)	Pond	--	1	--	--	--	--
Guangfu (2)	Pond	--	2	--	--	--	--
Rueisuei (1)	Pond	--	--	--	1	--	--
Rueisuei (2)	Pond	--	4	--	--	--	--
Luye	Stream	--	--	--	1	--	--
Kinmen Island							
Doumen (1)	Pond	--	--	1	--	--	--
Doumen (2)	Pond	--	--	6	--	1	--
Doumen (3)	Pond	--	--	1	--	--	--
Doumen (4)	Pond	--	--	1	--	--	--
Yangjhai	Pond	--	--	2	1	--	--
Longling Lake	Pond	--	--	1	--	1	--

Rapid range expansion of the feral raccoon (*Procyon lotor*) in Kanagawa Prefecture, Japan, and its impact on native organisms

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Abstract The distribution of feral raccoons (*Procyon lotor*) was surveyed in Kanagawa Prefecture, central Japan. Information was collected mainly through use of a questionnaire to municipal offices, environment NGOs, and hunting specialists. The raccoon occupied 26.5% of the area of the prefecture, and its distribution range doubled over three years (2001 to 2003). The most remarkable change was the range expansion of the major population in the south-eastern part of the prefecture, and several small populations that were found throughout the prefecture. Predation by feral raccoons on various native species probably included endangered Tokyo salamanders (*Hynobius tokyoensis*), a freshwater Asian clam (*Corbicula leana*), and two large crabs (*Helice tridens* and *Holometopus haematocheir*). The impact on native species is likely to be more than negligible.

Keywords: Feral raccoon; *Procyon lotor*; distribution; questionnaire; invasive alien species; native species; Kanagawa Prefecture

INTRODUCTION

The first record of reproduction of the feral raccoon in Kanagawa Prefecture was from July 1990, and it was assumed that the raccoon became naturalised in this prefecture around 1988 (Nakamura 1991). Damage by feral raccoons is increasing and the number of raccoons, captured as part of the wildlife pest control programme, is also rapidly increasing.

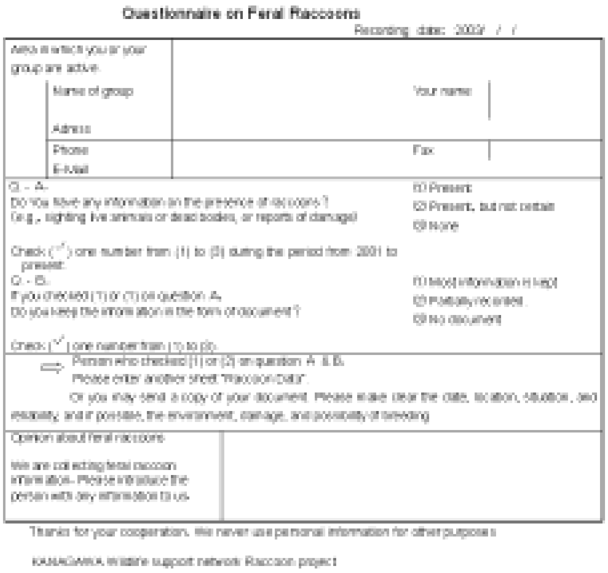
The distribution range of feral raccoons in Kanagawa Prefecture was reported, based on information from 1998 to 2000 (Kanagawa Prefecture 2001). However, the more recent distribution range since this survey had not been determined. Distribution and its rate of expansion are essential information required in addressing invasive alien species. We, therefore, decided to investigate the present distribution range through questionnaires and interviews.

There is much information available on the agricultural damage and nuisance to humans caused by feral raccoons in Kanagawa Prefecture, and other prefectures; but, evidence of impact on native species is unclear in Kanagawa Prefecture. Predation on native species was, therefore, determined through questionnaires and interviews.

METHODS

Questionnaires

We collect information through the use of questionnaires. In the main sheet of the questionnaire (Fig. 1), questions were concerned mainly with the presence of feral raccoons between 2001 and 2003 and the reliability of the information. One of the issues relating to reliability is possible confusion with the native raccoon dog (*Nyctereutes procyonoides*; Canidae), which has a similar facial pattern with a black band around the eyes, and a similar body size to the raccoon. An important external feature characteristic to the raccoon is the stripe on the tail, which is absent in the raccoon dog. The masked palm civet (*Paguma larvata*; Viverridae), another invasive alien mammal, is also sometimes confused with the raccoon. Therefore, we attached a sheet with


Questionnaire on Feral Raccoons

Recording date: 200 / /

Area in which you or your group are active

Name of group | Your name

Adress

Phone | Fax

E-Mail

Q. - A.
Do you have any information on the presence of raccoons ?
(e.g., sighting live animals or dead bodies, or reports of damage)
(1) Present
(2) Present, but not certain
(3) None

Check (✓) one number from (1) to (3) during the period from 2001 to present.

Q. - B.
If you checked (1) or (2) on question A,
Do you keep the information in the form of document ?
(1) Most information is kept
(2) Partially recorded
(3) No document

Check (✓) one number from (1) to (3).

⇨ Person who checked (1) or (2) on question A & B,
Please enter another sheet "Raccoon Data".
Or you may send a copy of your document. Please make clear the date, location, situation, and reliability, and if possible, the environment, damage, and possibility of breeding.

Opinion about feral raccoons

We are collecting feral raccoon information. Please introduce the person with any information to us.

Thanks for your cooperation. We never use personal information for other purposes.

KANAGAWA Wildlife support network Raccoon project

Figure 1 Questionnaire used. Original questions were in Japanese.

Pages 196-199. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

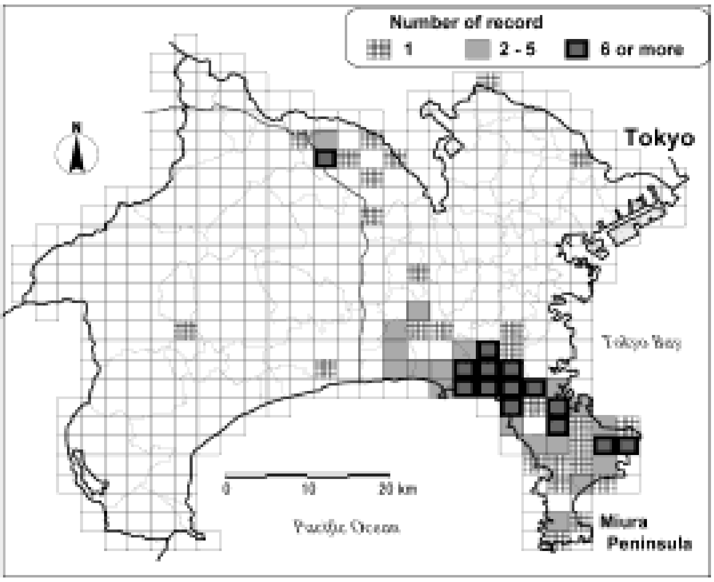


Figure 2 Distribution of feral raccoons by 2001 (based on information in 1998-2000). (From: Kanagawa Prefecture (2001)).

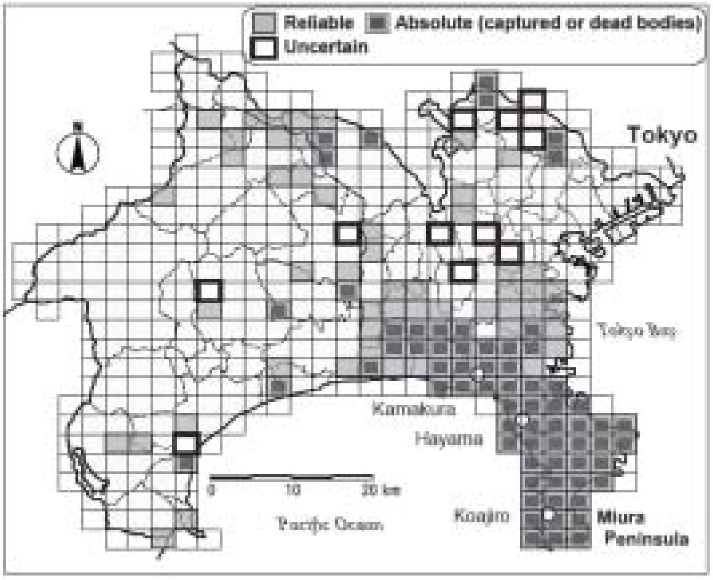


Figure 3 Distribution of feral raccoons by 2004 (based on information in 2001 - 2003). Three circles (Kamakura, Hayama, Koajiro) represent the places where predation on native species was reported.

illustrations of the external features and footprints, and with behavioural characteristics of the four carnivores (raccoons, raccoon dogs, palm civets, and badgers (*Meles meles*; Mustelidae)).

On another sheet, respondents were asked to choose one or several answers, on methods of identification (dead body, captured, sighting, footprints), confidence of identification, habitat, damage and reproduction (sightings of young). When the identification of the raccoon was not certain, we interviewed the respondents. If we could not confirm the identification even after an interview, the information was dealt with as "uncertain". Finally, we classified identification reliability into three categories: capture or dead bodies as "absolute", "reliable" information from sightings and/or footprints, and "uncertain".

The questionnaire (Fig. 1) was sent to the division of all municipal offices for wildlife conservation and management. It was also sent to staff of wildlife conservation and park management of the municipal government, and the environment NGOs. The questionnaire for municipal offices was sent through the Department of Environment and Agriculture, Kanagawa Prefectural Government.

Distribution range

In the previous distribution survey (Kanagawa Prefecture 2001, Fig. 2), a questionnaire was used to obtain the location of raccoons. These locations were then indicated on a plot map. We followed the same method to show changes in the distribution range of feral raccoons.

All information was plotted on a map drawn to a 1:25,000 topographic map. Each plot was about 2.85 x 2.3km in size, which was 1/16 of the area size of the 1: 25,000 topographic map.

Information on predation of native species

We interviewed environmental NGOs in the Miura Peninsula at the south-eastern part of the prefecture (Hayama Town, and Cities of Kamakura, Zushi, Yokosuka and Miura), and also obtained information from local newspapers and local reports of raccoon feeding on native species.

RESULTS AND DISCUSSION

Distribution pattern

A total of 882 records containing information on feral raccoons were obtained. Reliability of information was variable, but captures under the wildlife pest control programme certified raccoon presence. Figure 3 shows the distribution between 2001 and 2003, with the reliability of information classified into the three categories (“absolute”, “reliable”, and “uncertain”). A concentrated distribution range existed in the south-eastern part of the prefecture (Fig. 3). Based on the number of plots (2.85 x 2.3km), this major population occupied 524.4 km^2, and represented 67.2% (80 of 119 “absolute” and “reliable” plots) of all plots with raccoons. About 2/3 of its population boundary was formed by the coastline. This main population not only extended over a large area, but was estimated to have a high density, based on capture under the wildlife pest control programme, which had been in most of those plots (83.6%; 56 of 67 capture plots; Fig. 3). Small populations were found in other parts of the prefecture, mostly situated inland. Among them, the second largest population was at the north-western part of the prefecture.

Table 1 Number of plots showing the presence of raccoons. Each plot was about 2.85 x 2.3km in size. Presence represents the sum of “absolute” and “reliable”. Number in parenthesis represents number of “absolute” plots.

Survey years	Presence	Uncertain	Total	Whole plots
2001–2003	119 (67)	11	130	448
1998–2000[a]	58	0	58	448

[a] Kanagawa Pref. (2001)

Range expansion

Raccoons were found in 12.9% of the area of the prefecture by 2001, and in 26.5% of the area by 2004 (Tab. 1). In other words, their distribution range increased 2.05 times over these three years. Because the effort put into obtaining the distributions was different between these two surveys, this value of 2.05 times is not absolute, but it is sufficient to infer raccoons are expanding rapidly. In comparison with

Table 2 Presumed predation by the feral raccoon on native species on the Miura Peninsula, Kanagawa Prefecture, Japan. (Information from 2001 to 2003, based on interviews and questionnaires).

Location	Observation
Hayama Town, and Yokosuka City	Eleven partially eaten bodies of endangered salamanders (*Hinobius tokyoensis*) were left at their spawning sites. Many footprints of raccoons were left there simultaneously (observations by M. Kaneda and M. Ohno).
Hayama Town	More than ten egg sacs of endangered salamanders (*H. tokyoensis*) were eaten or broken, and many footprints of raccoons were left (observations by M. Kaneda).
Chuo Park, Kamakura City	Recent sightings of medium-sized mammals during the night were mostly of feral raccoons, instead of native raccoon dogs (*Nyctereutes procyonoides*).
Kamakura City	Freshwater Asian clams (*Corbicula leana*) were eaten. Such predation was not observed before the raccoon invasion.
Koajiro, Miura City	Numbers of the sesarma crab (*Holometopus haematocheir*) decreased after footprints of raccoons increased. Footprints of raccoons were found widely. A claw was found in the excrement of a medium-sized mammal on a fallen tree (observation by M. Kaneda).
Koajiro, Miura City	Numbers of the grapsid crab (*Helice tridens*) decreased after footprints of raccoons increased. Footprints of raccoons were found widely at the habitats of the crab (an article in a local newspaper, Kanagawa Shinbun on 7 July, 2003).

the previous survey (Fig. 2), the biggest increase in distribution range was from the major population, in the north and northwest direction. The range of the major population seems to have joined up with that of the second largest population in the northwest.

It should be noted that several small patchy isolated populations in the north and north-eastern parts of the prefecture in 2001 did not disappear, and seemed to have spread in the present study. New isolated populations were also found at the south-western part of the prefecture. These new small populations suggest that raccoons emigrated from the major population under high population density and/or were released by people. After they settled, they increased in numbers presumably through natural reproduction, resulting in an expanded distribution range.

Impact on native species

Records collected suggest predation by feral raccoons on native salamanders, freshwater clams, and two large crabs (Tab. 2). Almost all of these species live in and around water. The grapsid crab *Helice tridens* and the sesarma crab *Holometopus haematocheir* are semi-terrestrial, but spawn in seawater. Except for the spawning period, adult *H. haematocheir* live in forests, whereas adult *H. tridens* live in mud flats and salt marshes.

In the Miura Peninsula situated on the south-eastern part of the prefecture, it is possible that the increase in feral raccoons was involved in a decrease of native raccoon dogs which may have a similar niche to raccoons. This indicates the possibility of competition, directly through physical contact and/or indirectly, through food sources. Different use of microhabitat between these two sympatric species was reported in northern Japan (Abe *et al*. 2006). Mange epizootics occurred recently among the raccoon dog population in Kanagawa Prefecture (Shibata and Kawamichi 1999), where the number of raccoon dogs infected with mange epizootic has been high since 1992, and the number of raccoon dogs that were hunted or trapped abruptly reduced by 1995, indicating the decline of population density.. Because mange epizootics apparently contributed to a decrease of raccoon dogs, the ecological relationship between raccoons and raccoon dogs may be complicated, and this makes any conclusions about competition more difficult.

Although we did not obtain direct observations of predation on native species by raccoons, many footprints of raccoons were left at predation sites (Tab. 2). Most information indicated a decrease in numbers of native species and a simultaneous increase in the feral raccoon (Tab. 2), based on careful observations over many years by local naturalists. These observations should be given serious consideration, although further research is required to clarify the impact of feral raccoons on native species.

ACKNOWLEDGEMENTS

We greatly appreciate other members of Kanagawa Wildlife Support Network, Raccoon Project, particularly Jynichi Asami, Yoshiyuki Isiwatari, Teruo Kitabayasi, Hisasi Kuwahara, Yasuko Neagari, Akira Fujii, Ken-iti Lee, Miwa Yamamoto, and Yukika Yosinomoto, for participating in our research. The Greenery Policy Division, the Wildlife Department of Environment and Agriculture, Kanagawa Prefectural Government, kindly help us with distribution and collection of questionnaires. We would like to thank the Kanagawa Prefecture Natural Environment Conservation Centre, Kamakura bird-watching group of the Wild Bird Society of Japan, The Group for Study of Miura Peninsula Natural History, persons in charge of cities, towns and villages of Kanagawa Prefecture, environment NGOs, Kanagawa Natural History Museum, staff working on wildlife conservation in the municipal government and natural observation leaders, for their help in this research. Drs. Fumito Koike and Takeo Kawamichi greatly helped us to improve our draft. This research was financially supported by the Pronatura fund.

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Invasion of an alien palm (*Trachycarpus fortunei)* into a large forest

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Abstract Range expansion of an alien palm (*Trachycarpus fortunei*) was studied in a continuous forest in Kanagawa Prefecture, Japan. Life history was studied in a mature population close to a residential area. A 1km x 1km plot in continuous forest was divided into 100m x 100m subplots and the presence or absence of adults and juveniles was recorded in each subplot.

The palm formed an erect trunk after reaching 120cm in leaf length and started to produce flowers after the trunk length reached 2m. Maximum height was 6m. Adult plants were found in subplots close to residential and agricultural areas. Juveniles were found around the subplot with adults, and their distribution was spreading into the forest. In the forest the rate of range expansion from the estimated colonisation kernel was slower than in the fragmented suburban landscape. A smaller probability of flowering under the forest canopy may be a reason for shorter colonisation distance. Despite strong shade tolerance in juveniles, the demand for light in the flowering stage, and the small maximum height suggest that the palm will not dominate inside the natural forests. However they may reproduce in abandoned coppice forest, disturbed forest (canopy dieback), forest edges and cliff side forests, and the juveniles can persist for many years under a closed canopy. The species composition of forests is likely to be changed by this alien palm.

Keywords: Miura peninsula; Kanagawa; Japan; *Trachycarpus fortunei*; colonisation kernel; seed dispersal; woody alien plant

INTRODUCTION

Woody alien plants are gradually increasing (Martin 1999, Fine 2002, Healey *et al.* 2002, Maesako *et al.* 2003) and they are likely to cause serious impacts because tall and shade tolerant plants are likely to gain dominance in communities and suppress other species (Keddy 1990, Koike 2001).On the subtropical Ogasawara Islands in Japan, an intentionally introduced tree (*Bischofia javanica*) invaded the climax forests, and became dominant, changing the climax forest ecosystems on this oceanic island.

In suburban, fragmented, forests of Kanagawa Prefecture, two woody alien species are spreading. They are *Trachycarpus fortunei* (Hook.) H. Wendl. (Palmae) and *Ligustrum lucidum* Ait. (Oleaceae). Both species were introduced for ornamental use. The palm (*T. fortunei*) is native in China (Murata 1994), and was often planted in gardens because of its strong cold tolerance (Fig. 1). This palm is now very common in fragmented forests (Komuro and Koike 2005). But naturalisation in large, continuous, forest has not hitherto been reported.

In the research reported here, the range expansion of the palm was studied in a large forest in Kanagawa Prefecture, and the rate of range expansion was compared with that in the urban landscape.

METHODS

Study site

Research was conducted in a forest in upper Morito River in Miura Peninsula (latitude 35°17′N, longitude 139°36′E, Fig. 2). Annual mean temperature is 16.1°C, and annual precipitation is 1634 mm. The area is hilly with an altitude between 20m and 200m. Slopes are very steep with many small cliffs. The area is covered with plantations of an evergreen conifer (*Cryptomeria japonica* (L.fil.) D.Don), and abandoned coppice forests of the deciduous tree *Quercus serrata* Thumb. ex. Murray. The forest floor of the abandoned coppice forests were often covered by a dwarf

Figure 1 The alien palm *Trachycarpus fortunei*. Adult plant in an abandoned coppice forest (left) and trunk-less juveniles on the forest floor (right).

Pages 200-203. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

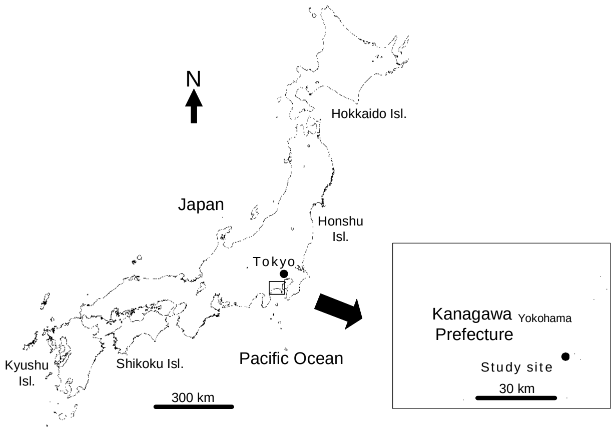

Figure 2 Study area.

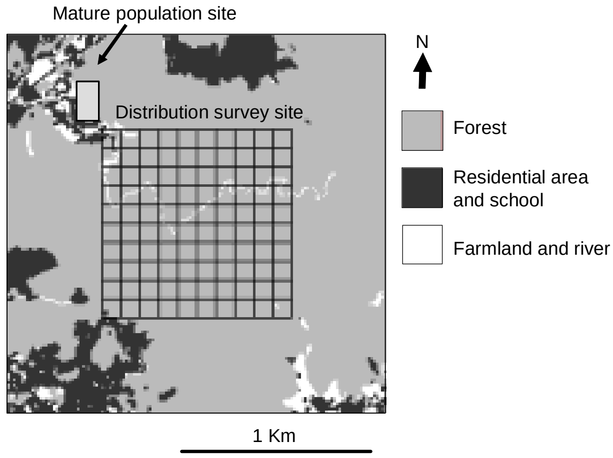

Figure 3 Land use map and research plots. The large school and sports grounds in the north of the distribution research site, and the residential area in the south-west were developed recently. Villages in north-west and south-east have a long history. Land use data was mapped using Minna de GIS (Koike 2004) based on Geographical Survey Institute (1997).

bamboo (*Pleioblastus chino* (Franch. *et* Savat.) Makino). The conifer plantations are dying back for unknown reasons. Fallen logs were frequently found and the canopy was not closed in many parts of the conifer forest.

Life history

All field surveys were made in the summer of 2003. The longest leaf length, trunk length, and presence of inflorescence were recorded for *T. fortunei* palms in a site close to the residential area with an existing mature population (Fig. 3). This study site included forests, forest edges and agricultural lands. The palm does not produce a trunk when it is small. It does not produce sprouts from the base of the trunk and never regenerates vegetatively. It regenerates only by seeds, which are dispersed by birds (Wild Bird Society of Japan, Kanagawa Branch. 1992). Usually the palm is dioecious, but it sometimes bears both male and female flowers on one plant (Nakanishi 1997); gender of plants was therefore not recorded in this research. Leaf size in relation to trunk length and minimum size at first reproduction were analysed. Change in the probability of flowering with stem length was evaluated by logistic regression with an additional parameter of maximum probability of flowering. Maximum likelihood estimation of parameters was made using Microsoft Excel, *Solver*, and the threshold plant size required for flowering was determined.

Colonisation kernel

A 1km x 1km plot was situated in the forest (Fig. 3), and was divided into a 100m x 100m subplot. An observer walked at least 100m in each subplot, and the presence or absence of adult and juvenile palms was recorded for each subplot. Adults and juveniles were distinguished, based on the threshold stem length for flowering.

Using these data, distance dependent colonisation to empty subplots was analysed. A subplot with adult palms was considered as a seed source (source subplot). Centre-to-centre distance from the nearest source subplot was obtained for all subplots. When a subplot had both adult and juvenile plants, the distance from the nearest source was assumed to be zero. The colonisation kernel was determined using the method described by Komuro and Koike (2005). Presence (value=1) or absence (value=0) was assigned as a dependent variable, and the distance from the nearest seed source subplot was assumed as an independent variable. If the distance to the outer edge of the whole research plot was smaller than the distance to a plot with adults, those data were not used, because there might be source plants outside the research area. Logistic regression was used to determine the colonisation kernel.

$$z(r) = \frac{1}{1 + e^{ar+b}} \qquad (1)$$

Where $z(r)$ represents the probability that a subplot located r m distant from the nearest seed source has at least one juvenile in 1 ha, and a and b are regression coefficients. The slope a is usually positive, and a large value of a represents a steep decrease of colonisation probability along with the distance from the source subplot. The significance of coefficients was evaluated by boot strap re-sampling, and the semi inter-quartile range (SIQR, the range between the 75 and 25 percentile in the frequency distribution, corresponding to "standard deviation" in a parametric description) was obtained.

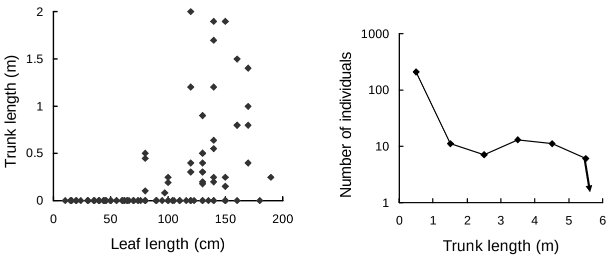

Figure 4 Trunk production by juveniles (trunk length <2m) against leaf size growth (left); and frequency distribution of trunk length in the whole population (right). No plant having a trunk longer than 6m was found. Data was from a survey in the mature site (Fig. 3).

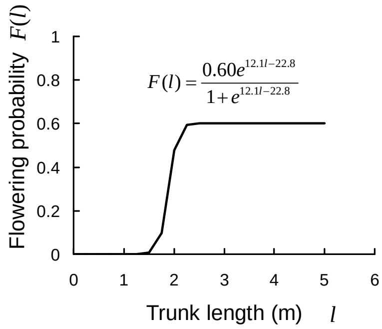

Figure 5 Flowering probability against trunk length. Logistic regression was used to fit the curve. Data was based on the survey in the mature population site (Fig. 3).

RESULTS

Life history of the palm

The palm usually forms a trunk after the leaf length attained 120cm (Fig. 4). Juveniles without trunks made up 72% of the population. No plants taller than 6m were found in the studied area, but many dead trunks of less than 6m height were observed. We therefore estimated the maximum height of the species to be 6m. Palm plants with a stem length larger than 2m bore flowers (Fig. 5). In the year of study about 60% of adult plants flowered. Palms larger than 2m in stem length were considered adult; smaller plants were considered juveniles.

Distribution and range expansion

Adult plants were found in two distinct places in the studied plot (Fig. 6). Both were close to residential or agricultural areas (Fig. 3). Adult palms were found in valleys as well as on ridges, and no habitat preference was found. Juveniles were often found in a subplot close to adults. The colonisation kernel obtained was as follows (Fig. 7);

$$z(r)=\frac{1}{1+e^{0.017r-2.426}} \qquad (2)$$

The slope of the colonisation kernel was 0.017 $\pm$0.003 (median $\pm$SIQR) in boot strap sampling.

DISCUSSION

Distribution of adults and juveniles suggests that the palm is spreading into the forest from adjacent residential and agricultural areas (Fig. 6). Shade tolerance and maximum height are important traits for plant survival in natural forests (Koike 2001). Juveniles without trunks are likely to have strong shade tolerance, given that growing juveniles are often observed on the forest floor. However, the flowering probability for adult plants in the forest (60%, Fig. 5) is likely to be lower than that for plants in gardens and at the forest edges, where almost all tall palms have flowers (Koike personal observation). This suggests that the palm in its flowering stage requires

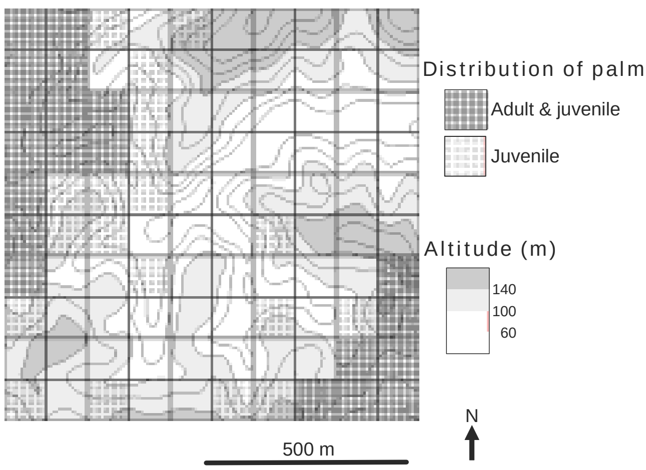

Figure 6 Distribution of juvenile and adult palm (*Trachycarpus fortunei*) in 1 km x 1 km area of continuous forest in Kanagawa Prefecture, Japan.

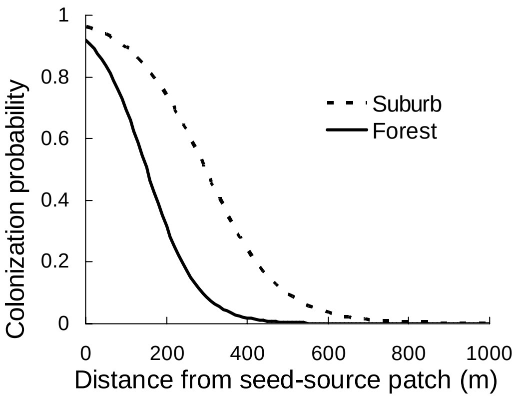

Figure 7 Colonisation kernel of the alien palm (*Trachycarpus fortunei*) in continuous forest and in the suburban landscape. Data for the suburb from Komuro and Koike (2005) for a study in Yokohama, Japan.

light. The estimated maximum height of 6m is lower than that for the most pronounced dwarf sub-canopy species of *Eurya japonica* Thumb. (Koike and Hotta 1996). This suggests that the palm will not become the dominant species in natural forests. They can reproduce on sites that are at least slightly open, such as in disturbed forests, under deciduous canopies, along riversides, forests close to cliffs, and at the forest edge. Juveniles dispersed from such habitats will persist for many years due to their strong shade tolerance. In this study area range expansion of the palm was enhanced due to the steep topography and die back of planted conifers.

The slope of the colonisation kernel for the same species in the suburban landscape of Yokohama was 0.011 $\pm$0.003 (median +SIQR, Komuro and Koike 2005), and smaller than in the continuous forest studied in this research. Thus the colonisation distance in continuous forest was smaller than that in the fragmented suburban landscape, and the palm spread slower in continuous forest than suburb. The reason for such phenomena is probably the smaller seed production in forests, due to the demand for light at the flowering stage. As shown in Komuro and Koike (2005), the number of seeds greatly affects the colonisation distance even if the dispersal distance for each single seed is the same.

Although the palm will not dominate inside the natural forests, the species composition of the forests will be changed by this alien species. The dominance of native species could be reduced to some extent. In the long term, accumulation of such shade tolerant sub-canopy alien species could possibly cause a serious reduction in native plants, through neutral replacement (Hubble 2001). Removing adult palms can easily be done by cutting the trunk, because adult plants never sprout. However, when juvenile plants are cut they produce new leaves, and removing juveniles is hence labour intensive. It is recommended to stop selling the palm, and to develop suitable methods for control of the palm population.

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Line census and gnawing damage of introduced Formosan squirrels (*Callosciurus erythraeus taiwanensis*) in urban forests of Kamakura, Kanagawa, Japan

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Abstract The Formosan squirrel (*Callosciurus erythraeus taiwanensis*) is a tree squirrel introduced to Japan from Taiwan. Its distribution in Japan is currently expanding and its impacts on Japanese ecosystems are considerable. To obtain basic ecological information on this species, we conducted line censuses and observations of gnawing damage in the urban forests of Kamakura City, Japan. We set a 2.5km transect line in a preserved green area, the Hiromachi Green Reserve, where sighting, hearing and nesting points of squirrels were recorded in the summer and autumn of 2002－2004. Gnawing damage on trees was recorded along nine transect lines in various forests in Kamakura City in the summer of 2004. Formosan squirrels were sighted at almost all surveys in the Hiromachi Green Reserve for two years, but the population trend has not yet been determined. Although squirrels were observed throughout the census lines, regardless of vegetation or topography, they significantly preferred evergreen broad-leaved forest for their nesting sites. Gnawing damage on trees was widely observed. Two modes of gnawing were observed on the tree bark: scratch and spread gnawing. Scratch gnawing was found in evergreen trees of *Ilex integra*, *Camellia japonica*, *Elaeocarpus sylvestris*, and *Quercus myrsinifolia*, while spread gnawing in *Cinnamomum japonicum*, *Neolitsea sericea*, and *Rhus succedanea*. Both types of gnawing were found in *Cornus controversa* and *Machilus thunbergii*.

Keywords: Formosan squirrel; *Callosciurus erythraeus taiwanensis*; line census; gnawing damage; introduced species; Kamakura

INTRODUCTION

The Formosan squirrel (*Callosciurus erythraeus taiwanensis*) is a tree squirrel originated in Taiwan (Fig. 1). It was introduced to Japan around 1930s as a zoo animal, and is now established in several areas of the forests through south and central Japan. In Kanagawa Prefecture, the Formosan squirrel was introduced in the 1950s and since then its distribution has rapidly expanded (Shiozawa *et al*. 1985, Kobayashi 1987, Furuuchi *et al*. 1990, Kamiya and Noguchi 1995) and its population has showed exponential growth (Tamura 2004). Its distribution continues to expand and rapid growth of local populations is also reported (Fujita *et al*. 1990).

Some of the economic impacts of Formosan squirrels are gnawing damage on garden trees, house exteriors and electric cables. They also interact with Japanese ecosystems in various ways, and ecological impacts are also considerable. Their intake of seeds and fruits contributes to the mortality of native plants, and may disrupt the regeneration process. Formosan squirrels are also known to gnaw and damage the tree bark, which may cause physiological disruption to trees although the reason for bark gnawing is not known. Gnawing damage increases during winter when the availability of food decreases (Yamada, personal observation), so they may feed on the inner bark or phloem. The predators of Formosan squirrels, such as snakes and birds of prey may also be affected although their predatory pressure is lower than that in Taiwan (Tamura 1989). Competition may occur with native Japanese squirrels (*Sciurus lis*) in future. Currently the habitat of the Japanese squirrel is

Figure 1: An adult Formosan squirrel in Kamakura, photographed 16 February, 2002.

Pages 204-209. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

restricted in suburban forests (Yatake and Takahashi 1987, Tamura 2000), and the distribution areas of these two species do not overlap. It is important to understand the risk of competition with Japanese squirrels by gathering sufficient ecological information.

The ecological habits of Formosan squirrels in Japan are different from those in their native range, due to cold weather (Tamura 1989). Nesting sites are an important and limiting resource for rodents. Although a large-scale habitat preference model is available (Tamura *et al.* 2004, Okubo *et al.* 2005), detailed information on nesting-site preference has not yet been recorded in Japan. Our research aimed at obtaining basic ecological information on Formosan squirrels in their central distribution area in Japan. To assess tree gnawing impacts and habitat use in Japanese forests, line censuses were carried out.

METHODS

Study area

The study area is located in and around Kamakura City, which is an ancient capital of Japan. Its historic landscape, including suburban forest areas, has been well preserved (Fig. 2). These green areas offer favourable habitats for Formosan squirrels. The mean annual temperature is 15.9 °C and the mean annual precipitation is 1448mm (during 1992 - 2000, at the closest meteorological station in Tsujido, Japan Meteorological Agency).

Line censuses were conducted in the Hiromachi Green Reserve (139°30'E, 35°18'N), which is one of the reserved forests in Kamakura. This reserved area is located in an isolated urban forest surrounded by residential areas, and consists of about 100ha of abandoned rice fields and forests. The altitude ranges from 15-75m above sea level. The management of both rice fields and forests was abandoned about 10 years ago. The current forest vegetation is roughly classified as natural forest of evergreen broad-leaved trees, secondary forest of deciduous broad-leaved trees, and artificial plantation of conifer trees.

Bark gnawing was investigated in nine isolated transect lines in various forests in and around Kamakura City. These transect lines passed through evergreen broad-leaved forest, deciduous secondary forest, and artificially planted evergreen coniferous forest.

Squirrel census

A 2.5km transect line was established through the abandoned rice fields and forests in the Hiromachi Green Reserve. The squirrel survey started in March 2002, and has been conducted every month or twice a month with an interval of more than two weeks, from the beginning of summer to the end of autumn each year. Observers walked at a rate of 1km/hour from 6:30am. During the survey, the points of sighting, hearing and locations of nests of Formosan squirrels were recorded. Their alarm call is characteristic, and clearly distinguished from sounds of other animals. Their spherical nests built with twigs and leaves on tree branches are easily detected. No distinction was made between nests that were currently utilised and those that were not.

Nest site preference was analysed with a GIS (Koike 2005). Geographical coordinates of the observed nest points were read from the map. Additional points were taken at 50m intervals on the transect line (transect line points), and these points were used to represent vegetation along the line. The percentage cover of vegetation (such as evergreen broad-leaved forest, deciduous broad-leaved forest, evergreen conifer plantation, other forests such as bamboo, abandoned paddy field, and residential area - from Biodiversity Centre of Japan 2004, Fig. 5a) was obtained within a 25m radius around the nest points and the transect line points. Existence or absence of a nest at the points was used as the dependent variable, assuming that nest was absent at the transect line points. Percentage cover of each vegetation type and four topographic variables (altitude, slope steepness, laplacian representing convexity, and catchment area representing wetness of the site) were used as independent variables. A stepwise variable selection

Figure 2 Location of the study area. Kamakura city is indicated by a black arrow on the upper map. On the large scale map, shaded areas are forests, and white areas include commercial, residential or agricultural areas. The solid lines with numbers in forests are the transect lines for gnawing research.

Figure 3 Gnawing damage by Formosan squirrels in Kamakura. Scratch gnawing on *Ilex integra* (left) and spread gnawing on *Cornus controversa* (right). Photographed on July 24, 2004.

procedure of logistic regression (SPSS 13.0J) was used to evaluate nest site preference.

Census of gnawing damage on forest trees

Gnawing damage on tree stems and branches was surveyed in nine forests in and around Kamakura City (Fig. 2). The length of the transect lines varied depending on the area of forest. The survey was conducted in the summer of 2004. For a damaged tree, species name, height, diameter at breast height, gnawing mode and the proportion of the damaged area to the total surface of the tree trunk were recorded.

Two kinds of gnawing modes were designated; scratch and spread (Fig. 3). Scratch gnawing leaves trace of horizontal lines about 1cm width on the bark of the tree, as if carved by a wood chisel. Usually many are found on a tree and the length is variable. Spread gnawing shows as a patch with the bark removed. There are no agents which can cause such damage on the trees, other than Formosan squirrels.

We recorded the dominant tree species along the transect lines to avoid the bias of observation to specific vegetation, but we did not count the total frequency of each tree species to analyse the exact preference (the proportion of damaged trees of the total frequency).

RESULTS

Census of Formosan squirrel

Formosan squirrels were sighted during almost all surveys in the Hiromachi Green Reserve, though the observed number of squirrels varied among the surveys (Fig. 4). The average numbers of observed squirrels along the census line was 4.1 individuals in 2003, and decreased to 1.6 individuals in 2004, while the frequency of squirrels heard did not differ between these two years (3.2 times in 2003 and 3.3 times in 2004 on average). The frequent canopy trees along the census line were evergreen *Castanopsis sieboldii* and deciduous *Quercus serrata*, *Cornus controversa*, *Prunus jamasakura*, and *Celtis sinensis* var. *japonica*. The observation points of Formosan squirrels were scattered through the transect line (Fig. 5b). The distribution of the points did not show a clear relationship with vegetation or topography. Contrary to this, the nesting points were significantly concentrated on evergreen broad-leaved forest ($P<0.001$) (Fig. 5c). *Castanopsis sieboldii*, a broad-leaved evergreen tree was dominant there and often utilised for nesting. Other vegetation and topographic variables were not significantly different for points with and without squirrels ($P>0.05$).

The gnawing damage on forest trees

Gnawing damage was found along all nine transect lines (Table 1). Formosan squirrels were also sighted on every survey. Dominant canopy species of evergreen trees were *Ilex integra*, *Machilus thunbergii* and *Castanopsis sieboldii*, and those of deciduous trees were *Quercus serrata*, *Cornus controversa*, *Prunus jamasakura*, and *Celtis sinensis* var. *japonica*. *Chamaecyparis obtusa* and *Cryptomeria japonica* were used in plantations. Although both evergreen and deciduous trees were observed along all transect lines, *C. controversa* was scarce along

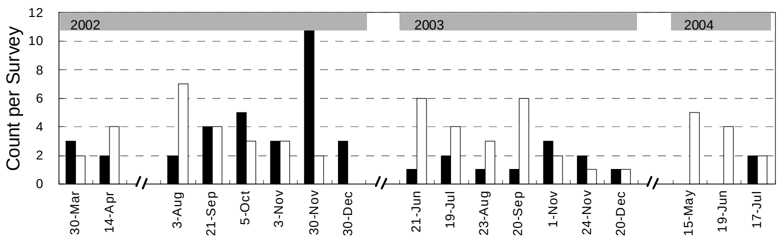

Figure 4 Observed number of sightings and hearings of Formosan squirrels during the monitoring in Hiromachi Green Reserve. Black bars are the sightings while open bars are the hearings. The horizontal axis is broken if the interval between the surveys was more than two months.

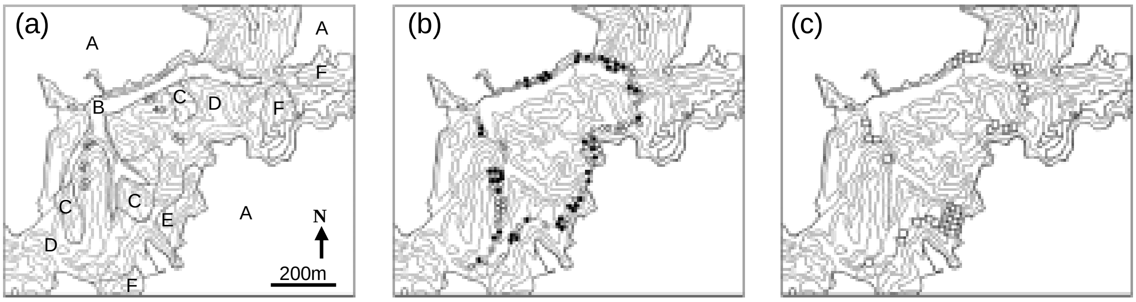

Figure 5 (a) Vegetation map of Hiromachi Green Reserve (Biodiversity Centre of Japan 2004). A: residential area, B: abandoned paddy field, C: evergreen coniferous forest, D: deciduous broad-leaved forest, E: evergreen broad-leaved forest, F: other forests; (b) points of sightings (black circles) and hearings (open circles) of Formosan squirrels on the line census; (c) nesting points of Formosan squirrels. All points throughout the surveys are indicated in (b) and (c). The transect line is indicated as a broken line. The contour interval within the Hiromachi Green Reserve is 10m.

transect 3, 4 and 5. The gnawing damage was found most frequently on *I. integra* (Table 1, Fig. 6). Squirrels also gnawed *C. controversa* and *M. thunbergii*. Among the dominant trees, *Q. serrata*, *P. jamasakura*, *C. sinensis* and coniferous trees were not damaged.

Usually only one type of gnawing, either scratch or spread, was found if a tree was damaged and the frequency of gnawing modes differed between species (Fig. 6). For example, damage of *I. integra* were mostly by scratch gnawing, while for *Cinnamomum japonicum* and *Neolitsea sericea* there was more spread gnawing. On *C. controversa* and *M. thunbergii*, both scratch and spread gnawing were found. The degree of damage varied from a few centimetres in diameter of spread gnawing to the removal of more than half of the trunk surface. However, there were no cases of dead trees with presumed cause of death through gnawing damage.

DISCUSSION

Nest site and habitat preference

The habitat preference of Formosan squirrels was not clear in Hiromachi. In Taiwan, Formosan squirrels inhabit the subtropical evergreen forests (Tamura *et al*. 1989), but in Kamakura, they have adapted to other types of vegetation such as grassland or deciduous woods (Sonoda and Tamura 2003).

Okubo *et al.* (2005) reported that, although

Table 1. Frequencies of trees damaged by Formosan squirrel gnawing along each transect line. Locations of transect lines are indicated in Fig. 2, with corresponding numbers. Tree species are ordered by the total count of damaged trees.

Tree Species	Transect line ID (length, km)									Total
	1 (1.0)	2 (0.9)	3 (1.2)	4 (0.9)	5 (1.0)	6 (1.2)	7 (4.2)	8 (2.1)	9 (1.2)	
Ilex integra	7	1	1				38	10	5	62
Cornus controversa	12	6	1			11	1	6	5	42
Machilus thunbergii	1	1		1	1	7	9	1	5	26
Cinnamomum japonicum	4							7	1	12
Neolitsea sericea							1		6	7
Rhus succedanea			6					1		7
Camellia japonica							6			6
Elaeocarpus sylvestris	5									5
Quercus myrsinaefolia								3		3
Quercus glauca								2		2
Acer palmatum				1				1		2
Aphananthe aspera								2		2
Morus australis		1						1		2
Mallotus japonicus	1									1
Ficus erecta		1								1
Zanthoxylum ailanthoides	1									1
Stachyurus praecox						1				1
Castanopsis sieboldii							1			1
Total	31	10	8	2	1	19	56	34	22	183
Density (count/km)	30.2	11.2	6.6	2.1	1.0	16.5	13.2	16.1	18.0	

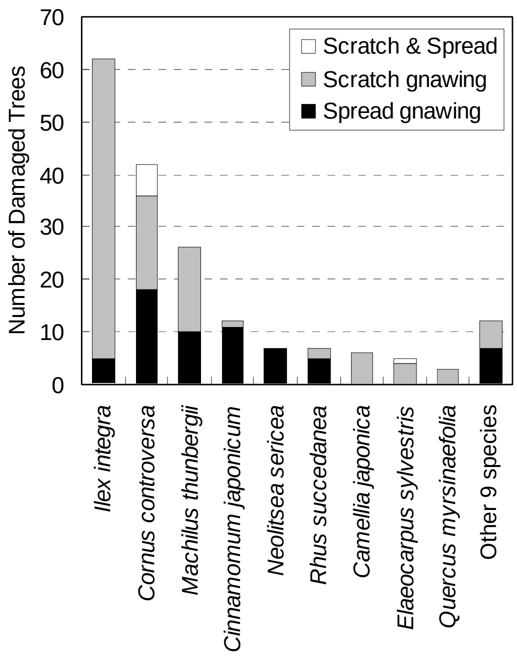

Figure 6 Number and gnawing modes of damaged trees by Formosan squirrels, along the transect lines around Kamakura City. Nine additional species, of which only one or two individual trees were damaged, are pooled into "Other 9 species". All species names are given in Tab 1.

Formosan squirrels use various types of forests, they prefer evergreen broad-leaved forest if it is available in their home range. In the present study, they preferred evergreen trees for nesting. Evergreen broad-leaved forest may be important core habitat for reproduction. Preference for evergreen trees for nesting behaviour was also found for other tree squirrels, for example, Japanese squirrels (*Sciurus lis*) preferred evergreen trees (Tamura 1998), while Abert squirrels (*S. aberti*) preferred closed stand of conifer trees (Halloran and Bekoff 1994). This nesting behaviour may occur because tree crowns can hide the nest from birds of prey. Formosan squirrels mate throughout the year in Japan (Tamura *et al*. 1988), and so evergreen trees may offer more secure nesting places than deciduous trees.

Gnawing damage

The Formosan squirrel seems not to gnaw *Quercus serrata*, *Prunus jamasakura*, *Celtis sinensis* var. *japonica*, *Chamaecyparis obtusa* and *Cryptomeria japonica*, although these species are common in secondary and plantation forests in and around the study area. The factors influencing gnawing preference and gnawing mode require further research. Nutrient content or the presence of defensive chemicals, such as polyphenols and flavanols may need to be considered. (Tamura and Ohara 2002).

In total, eighteen tree species were damaged by gnawing. Although there was no evidence of "whole tree mortality", it is likely that tree growth suffered to some degree from die back of branches or damage to the trunk's cambium. Furthermore, species specific gnawing might alter the tree species composition of forests. To estimate the impact, it is necessary to investigate the growth and survival of the damaged trees for a longer period.

A comparative study of native and introduced populations of the Formosan squirrel revealed that it feeds mainly on fruits and seeds both in Japan and in Taiwan; though food availability is lower in Japan because of lower temperature (Tamura *et al*. 1989). Furthermore, under the temperate climate in Japan, food availability decreases during winter, which may cause shift in food preference to nutrient poor sources like tree bark. The relationship between squirrels and their food source and factors influencing feeding behaviour should be studied further.

ACKNOWLEDGEMENTS

We thank N. Tamura for her useful advice on our research. K. Fukuoka provided considerable support and F. Koike provided helpful comments on the original manuscript. J. Higuci helped us during the field survey. The members of Hiromachi Forest Monitoring Group gave us many comments on the research and they deserve our special thanks.

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Short paper

Balanus glandula: a new alien barnacle from the west coast of North America, established on the northeast coast of Japan

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INTRODUCTION

A common littoral barnacle, *Balanus glandula* Darwin, 1854, (Crustacea, Cirripedia) from the west coast of North America was first reported in Mar del Plata port, Argentina in 1974 (Bastida *et al*., 1980). Further studies showed that this species established along the shore between 37˚S and 47˚S (Vallarino and Elías, 1997, Elías and Vallarino, 2001, Rico *et al*., 2001, Orensanz *et al*., 2002). The species was also found along the opposite side of the Pacific; in the littoral zone between 38˚30'N and 42˚40'N on the northeast coast of Honshu, Japan (Kado, 2003). It seems interesting that *B. glandula* was introduced in the late 1960s or afterwards in the two regions. In the case of Japan, the vectors of *B. glandula* for the introduction were inferred to be cargo-ships carrying lumber from the NW of the US, (Kado, 2003). From the first introduction to the present (probably less than 30 years), *B. glandula* successfully established in the littoral zone, especially in big commercial ports. In both regions *B. glandula* seems to have some ecological superiority over native communities. In Japan, however, the distribution is not continuous along the coast between harbours, and population densities of *B. glandula* do not always correlate with distances from invaded harbours. At some locations *B. glandula* was common, even far from these ports. These new facts suggest that local spread of this species does not depend only on its larval dispersal. This study reports on the ecological properties of *B. glandula* on the northeast coast of Honshu, Japan and investigates the role of barges, tugboats and similar vessels within Japan as likely vectors of *B. glandula* for further spread.

METHODS

Field surveys and experiments were carried out at Kamihira fishing port in Ofunato Bay, Iwate Prefecture during autumn, in the period from 2000 to 2003. Reproductive conditions were investigated monthly by checking brooding condition of a sample of barnacles. Three reproductive categories were assigned: no embryos, early embryos with yellowish colour, and late embryos with brownish colour. Parameters of water quality such as temperature, salinity and Chlorophyll-a concentration were measured at the same time by means of a mercury thermometer, a conductivity meter (LF340, WTW), and a fluorometer (Field fluorometer 10-005, Turner), respectively. Seasonality of settlement was examined using monthly immersed triplicate test plates with frosted surface. Test plates were fixed at three different sea levels: mean sea level (MSL) and 40cm above and below the MSL. Test plates were changed monthly and settled spat and juveniles on their surface were counted and recorded separately.

Settlement depth of *B. glandula* was examined by means of paired poly-vinyl chloride pipes hung vertically from the quay wall.

To estimate growth and age of the *B. glandula* population in this Bay, a growth analysis was performed for a population settled around 40cm below the MSL on the concrete mooring slope at Kamihira fishing port. Shell size was measured monthly with digital callipers, at low tide. Growth and age composition were analysed with the Solver method (packaged in MS-Excel (Microsoft)).

The hull and fenders of barges and tugboats were checked for fouling by *B. glandula*. We also collected information on locations they visited.

RESULTS

On the sheltered shores of Ofunato Bay, *Balanus glandula* had established a dense population over the whole range of the littoral zone. *Balanus albicostatus,* which is endemic to temperate Japanese and adjacent waters, was out-competed there, except for the upper fringe of the zone. On exposed rocky shores, to which *B. glandula* had just started to expand its distribution, the species was starting to compete for space with two temperate-subtropical endemics, *Tetraclita japonica* and *Chthamalus challengeri* in the mid and upper littoral fringe, and with a sub-arctic endemic, *Semibalanus cariosus* in the mid and lower littoral zone.

In Ofunato, *B. glandula* had a long breeding season throughout most of the year with monophasic

Pages 210-211. In Koike, F., Clout, M.N., Kawamichi, M., De Poorter, M. and Iwatsuki, K. (eds), *Assessment and Control of Biological Invasion Risks*. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland, 2006.

peak between February and June. It also had a longer settling season from April to July, compared to other native balanid species which settled only in July. *B. glandula* could settle on 74% of the littoral range (160cm). In the Kamihira population in Ofunato four cohorts were observed with shell size from juvenile to 20mm basal diameter.

Barges and tugboats which have Ofunato as their home port, have often been sent to other harbours and other coastal areas to construct facilities such as quays, Tsunami guards, wave-dissipating blocks, or to carry out dredging. They even visited as far as Hachinohe (ca. 190km north) and Kesen-muma (ca. 20km south). Hulls and fenders of the barges and tugboats were settled by *B. glandula*. We confirmed that *B. glandula* scraped off from fenders of the tug boats visiting Okkirai Bay on 24 April, 2003 held egg masses inside the shell, and when egg masses were put into seawater larvae hatched out.

DISCUSSION

B. glandula has many potential advantages over native barnacles and sessile animals, such as longer seasons in breeding and settling, wider range of settling, smaller size when breeding, and greater adaptability to both sheltered and exposed environments. It was also fortunate for *B. glandula* that the coastal areas where it was introduced were located on the fringes of the distribution area for both temperate, subtropical, and sub-arctic endemics, resulting in less competition. It seems, therefore, unlikely that there would be interference with *B. glandula* expanding its range to the exposed rocky shores, where indigenous biota still remains. Frequent visits of barges and tugboats from ports that are settled heavily by *B. glandula* to these new locations should also be expected to play a role given that such vessels are vectors for further spread within Japan. In addition, the greater tolerances of *B. glandula* to water temperature and salinity (Kado, unpublished) is also likely to facilitate the spread further south up to 36˚N, which is a southern fringe of mid-temperate zone in the Pacific coast of Japan.

ACKNOWLEDGEMENTS

We gratefully thank the students of the Laboratory of Marine Primary Production, School of Fisheries Sciences, Kitasato University during field and laboratory work for their cooperation throughout the years. We also thank the editors for improving the manuscript**.**

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Subject index

Taxonomical index

ISBN978-4-87974-604-7

Published by :
SHOUKADOH Book Sellers, Japan
The World Conservation Union (IUCN), Switzerland

Printed on 100% recycled paper with soybased inks.