KEYNOTE ADDRESS

Today Tiritiri Matangi, tomorrow the world! Are we aiming too low in invasives control?

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Abstract Eradication of invasive non-indigenous species is often viewed as an impossible goal and an approach historically typified by high-profile failures. However, there have been a surprising number of successful eradications of animals, plants, and even microorganisms. Although the majority of successes have concerned geographically-circumscribed invasions (e.g., on small islands), others have rid substantial continental areas of invaders (e.g., *Anopheles gambiae* from north-eastern Brazil, or smallpox from the entire Earth). Successful eradications share three features: (1) sufficient economic resources must exist for the project to be completed, (2) clear lines of authority must exist; someone must be in charge and must be able to compel cooperation, and (3) the biology of the target organism must be adequately researched and appropriate. For many but not all eradication attempts, probability of rapid re-invasion must be low for success to ensue. Further, even when the above criteria are met, an eradication attempt, even if successful, can lead to unforeseen problems, such as mesopredator release or a proliferation of non-indigenous weeds at the expense of native plants. Finally, not only can attempted eradication of widely distributed invaders be costly, but it can generate non-target impacts (e.g., on human health or species of conservation concern), the importance of which will be weighed differently by different stakeholders. Thus, successful eradication may be as much a function of political skill and public education as of technology. When eradication is feasible, a benefit-cost analysis may help indicate when it is the best management strategy. To date, eradication has been a rather idiosyncratic matter, often resting on the drive and ingenuity of one person or a few people. This has partly resulted from lack of public interest in invasions. Other developments in management of invasions should increase the appeal of eradication attempts. The evolution of more comprehensive monitoring and reporting systems, as well as more rapid response procedures, should lead to the more frequent eradication of invasions before they become metastatic. However, even invasions that escape initial elimination and spread widely may be susceptible to eradication. Many invasions that would, *a priori*, appear suitable by the above criteria for eradication have not been attacked because no one has mustered the enthusiasm to try it or generated the political support to provide the necessary resources and framework. Moreover, we do not know the geographic limits of current technologies. For example, just how great an investment would be required to rid a large island or substantial continental region of a pestiferous mammal? As with many other aspects of the invasion problem, eradication may largely be a victim of an unwarranted fatalism that could generate the very outcome that is most feared – in fact, we are not doomed to the biotic homogenisation of the Earth, but we will surely lose this war if we do not aim high.

Keywords defeatism; invasion economics; re-invasion; restoration; side effects.

INTRODUCTION

As biologists and the public worldwide increasingly recognise the damage caused by invasive non-indigenous species (Mooney 1999), they usually assume that maintenance management is the appropriate response. "Maintenance management" means controlling an invader at a density low enough that we can tolerate the damage it causes. Maintenance options typically include mechanical, chemical, and biological control, plus ecosystem management (Simberloff 2002). Although politicians occasionally call for eradication of a new invader, the total removal of every single individual remains a controversial goal (e.g., Myers *et al.* 1998), and much of the scientific community views it as a bad idea (e.g., Dahlsten 1986) for three reasons: it is seen as unlikely to succeed, it may be costly, and it may impose substantial collateral damage. Some famous failed eradications exemplify these problems. Probably the most notorious was the 14-year eradication project for the imported fire ant (*Solenopsis invicta*) in the southeastern United States (Davidson and Stone 1989), a legendary fiasco in terms of collateral damage (including to humans) and expense (over USD200 million) termed "the Vietnam of entomology" by E. O. Wilson (Brody 1975). The biology of the ant rendered successful elimination over very large areas impractical. This campaign probably worsened the fire ant invasion by causing greater mortality for its natural enemies than for the fire ant itself.

However, many invaders have been successfully eradicated (Myers *et al.* 2000; Simberloff 2001). To my knowledge, the earliest insect eradication was the elimination of the tse-tse fly (*Glossina* spp.) from the 126 km^2 island of Principe in the Gulf of Guinea (Lapeyssonie 1988). The flies were introduced in cargo from Africa in 1825, and sleeping sickness was noted beginning in 1859, ultimately reducing the human population ten fold. A four-person team completely eradicated the fly (and the disease) between 1911 and 1914. In 1956, a tse-tse fly was again noticed on Principe, and a large scientific team was immediately dispatched to the island, where they captured 66,894 flies in two months. With the aid of traps, insecticides,

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extensive brush-clearing, and massive hunting to reduce populations of pigs and wild dogs, the fly was again eradicated at a cost of £7500 and has not been seen since. Principe is an island (though not a tiny one), and many successful eradications have occurred on islands. These range from small ones, such as the elimination of the screw-worm fly (*Cochliomyia hominivorax*) from Curaçao (Baumhover *et al*. 1955), Asian citrus blackfly (*Aleurocanthus woglumi*) from Key West (Hoelmer and Grace 1989), Oriental fruit fly (*Dacus dorsalis*) from Rota and Guam (Steiner *et al*. 1955, 1965, 1970), and Pacific rats (*Rattus exulans*) from Tiritiri Matangi (Veitch 2002), to very large ones, such as nutria (*Myocaster coypus*) from Great Britain (Gosling 1989), yellow fever from Cuba (Fenner *et al*. 1988), and the melon fly (*Bactrocera cucurbitae*) from the entire Ryukyu Archipelago, including Okinawa (Iwahashi 1996; Kuba *et al*. 1996).

Though many of the most striking recent eradications have removed various mammals from islands (e.g., Veitch and Bell 1990; Chapuis and Barnaud 1995; Day and Daltry 1996a, 1996b; Pascal 1996; Day *et al*. 1998; Pascal *et al*. 1998; Varnham *et al.* 1998; Bell 1999; Donlan *et al*. 1999), successful eradication is not just an island phenomenon. The most widespread eradication eliminated smallpox from the face of the Earth (Fenner *et al*. 1988). One of the most impressive continental eradications was that of the African mosquito (*Anopheles gambiae*), a vector of malaria, from 31,000 km^2 of north-eastern Brazil (Soper and Wilson 1943; Davis and Garcia 1989). Other eradications from large parts of continents include the screw-worm (first from Florida, then from the southeastern United States, then from Mexico, and most recently from several Central American nations (Reichard *et al.* 1992; Galvin and Wyss 1996)), the cattle tick (*Boophilus annulatus*) from over a million km^2 of the United States (Klassen 1989), and bovine contagious pleuropneumonia from the United States (Fenner *et al*. 1988). For the cattle tick example, there is occasional re-invasion (see below). Eradication from smaller continental areas is fairly common, such as that of the giant African snail (*Achatina fulica*) from a region of south Florida (Mead 1979) and part of Queensland, Australia (Colman 1978), the medfly (*Ceratitis capitata*) from 20 Florida counties (references in Simberloff 1997a), yellow fever from Panama (Fenner *et al*. 1988), karoo thorn (*Acacia karoo*) from Western Australia and Victoria, and Taurian thistle (*Onopordum tauricum*) from Victoria (Weiss 1999; R. Groves, pers. comm. 2000).

Of course, besides famous failures such as the fire ant campaign, there are many other attempted eradications that have not resulted in the complete elimination of an invader; surely there are more such cases than total successes. I have not attempted a tally, because the literature is too scattered and grey, and because colloquial use of the term "eradication" makes it difficult to assess exactly what is a failure (Simberloff 1997a, 2001). Often public figures (e.g., Chiles 1996) and even scientists (e.g., Langland and Sutton 1992) use "eradication" to mean partial removal and substantial control. In these instances total eradication was never even attempted. Should such a campaign be viewed as a failure? This assessment seems unduly harsh if the same method used in the eradication campaign would have been used for maintenance management, and if substantial control results even though elimination is not complete, as in the attempt to eradicate *Spartina* spp. from New Zealand (Nicholls 1998).

In the remainder of this paper I attempt to parse the successes and failures to seek guidance as to when eradication is feasible. Do common features characterise successful campaigns? Do similar problems plague many failures? At the outset, I emphasise that I am not addressing whether society as a whole wants a particular invader removed or even controlled. Often one faction wants to eliminate a species that others see as a boon – note the battle in Australia over *Echium plantagineum*, termed Paterson's curse by ranchers and Salvation Jane by apiarists (Cullen and Delfosse 1985). Rather, assuming that society does want to control a particular species, I will ask what is the best means.

ECONOMIC RESOURCES

Eradication on a small scale may not require enormous resources; the enthusiasm and hard work of a single person or a small, non-governmental organisation may even suffice. For example, a dedicated group of scientists (the Island Conservation & Ecology Group) has succeeded in removing various combinations of feral cats (*Felis catus*), Norway and black rats (*Rattus norvegicus* and *R. rattus*), house mice (*Mus musculus*), rabbits (*Oryctolagus cuniculus*), goats (*Capra hircus*), sheep (*Ovis aries*), and burros (*Equus asinus*) from nine islands in north-west Mexico (Donlan *et al*. 1999). However, for large areas, costs are often huge. For 50 infestations of 16 plant pests of California, Rejmánek *et al.* (2000) found that log (cost) increased linearly and rapidly with log (infested area). Successful large regional eradications have been supported by significant government resources and/or private investment. The Brazilian eradication of *Anopheles gambiae* was funded by the Rockefeller Foundation and the Brazilian government (Davis and Garcia 1989), the screw-worm eradication in the United States and Mexico cost United States taxpayers USD750 million (Reichard *et al*. 1992), while the reduction of the African root parasite witchweed (*Striga asiatica*) in the Carolinas from 162,000 ha in the 1950s to *c*. 2800 ha now entailed the massive support and cooperation of the United States government and the state governments of North and South Carolina (Westbrooks 1993). Of course, huge budgets do not ensure success – witness the fire ant eradication disaster. However, for eradication over substantial areas, big budgets are generally a prerequisite (Myers *et al*. 2000; Simberloff 2001b).

The fact that expense increases rapidly as area of an invasion increases leads to the dictum that it is best to eradicate early (e.g., Simberloff 1997a; Weiss 1999; Myers *et al.* 2000). Although some longstanding, widespread invasions have been eradicated, likelihood of success is obviously improved and cost minimised if an invasion is nipped in the bud. This fact argues for effective early warning and rapid response machinery (Simberloff 1997b; Weiss 1999), a subject beyond the scope of this paper. Two cases exemplify the benefits of acting very quickly when eradication is the goal. The Caribbean black-striped mussel (*Mytilopsis sallei*), was discovered in 1999 in Cullen Bay (600 megalitres, 12.5 ha), Darwin Harbour, within six months of its arrival and before it had spread further in Australia. Within nine days the bay had been quarantined and treated with 160,000 *l* of liquid bleach and 6000 metric tonnes of $CuSO_4$. All living organisms were believed killed, and the mussel population was eradicated (Myers *et al*. 2000; Bax *et al*. 2002). The tropical alga *Caulerpa taxifolia* could almost certainly have been eliminated in the Mediterranean soon after its discovery, when it was restricted to a few square metres in front of the Oceanographic Museum of Monaco, but the effort was delayed for years and the alga now infests several thousand hectares of the coasts of Spain, France, Monaco, Italy, and Croatia (Meinesz 2001). By contrast, an effort to eradicate a small infestation of the same alga near San Diego within a year of its discovery seems promising (Meinesz 2001). An attempt to combat a much larger infestation near Los Angeles using similar methods is more problematic.

Some expenses of eradication campaigns can be substantial and not obvious at the outset (Myers *et al.* 1998). Killing the first 99% of a target population can cost less than eliminating the last 1%. This fact can become a problem with governmental funding authorities, who may be inclined to lessen support for a programme once the problem subsides, rather than see it through to completion (Schardt 1997; cf. Mack and Lonsdale 2002). Costs of monitoring may increase when pest densities are very low, yet intensive monitoring is the only effective way to determine when to end an eradication campaign. Depending on the target species and the means employed to remove it, an expensive public relations campaign may be needed to ensure public support, and lawsuits may have to be contested (Myers *et al.* 1998). For instance, for just part of a California medfly eradication project, 14,000 claims were filed for damage to car paint, and the state of California paid USD3.7 million (Getz 1989).

LINES OF AUTHORITY

It is always difficult to induce large groups of people with diverse interests to support a programme when the benefits seem unequally distributed, and eradication frequently falls in this category. Because eradication can, by its nature, be subverted by one or a few individuals, some government agency or interagency entity must have the ability to compel cooperation (Myers *et al.* 2000; Simberloff 2001b). In nations or regions with strong distrust of government, such authority will automatically generate opposition (cf. Perkins 1989). Specific concerns about the eradication techniques may be so vehement that only a strong governmental authority can enact the programme. Aerial spraying of malathion to eradicate medflies fostered widespread complaints about discomfort or threats to human health in California (Penrose 1996) and Florida (Anon. 1997). Killing large vertebrates by trapping, hunting, or poisoning often generates vocal opposition – witness the outcry over snaring feral pigs (*Sus scrofa*) in the Hawaiian islands (Van Driesche and Van Driesche 2000), trapping nutria in Great Britain (Gosling 1989), and shooting monk parakeets (*Myiopsitta monachus*) in the United States (Simberloff 1997a).

When human health is at stake, as in the tse-tse eradication on Principe or in Nigeria (Oladunmade *et al*. 1986) or the malaria mosquito eradication in Brazil, even heavy-handed government control is less likely to generate opposition. When an eradication campaign directly benefits agriculture, and the costs and possible side-effects are borne by the entire public as in spraying malathion to kill medflies, perceived inequities are more likely to generate conflict (Simberloff 2001b). Most eradications attempted for conservation purposes have occurred on small islands, often with little or no human population, and opposition has usually been minimal. Until conservation achieves a higher value in the eyes of the entire public, I predict that attempts to eradicate ecological pests over wide areas will engender hostility because of economic or emotional costs or side-effects. On a small scale, the local attempts to eradicate Asian long-horned beetles (*Anoplophora glabripennis*) by felling urban trees in Chicago and New York, and to eradicate citrus canker in Florida by destroying citrus trees, gave a foretaste of complaints that will arise if this campaign must be greatly extended (e.g., Stout 1996; Toy 1999; Sharp 2000); of course, the ultimate purpose in these instances is silvicultural or agricultural more than ecological. I know of no large-scale eradication projects conducted solely for conservation purposes, though some carried out primarily for agricultural or silvicultural reasons are perceived as having conservation benefits (e.g., that of the gypsy moth (*Lymantria dispar*)(Myers *et al*. 2000)).

BIOLOGY OF THE TARGET SPECIES

A sufficiently-determined effort can probably eradicate almost any species in a small enough area, but certain biological features can make a target less tractable. When eradication must be conducted over a large region, the biology of the target species may be particularly crucial and the scientific knowledge must be profound (Fenner *et al.* 1988; Myers *et al*. 2000; Simberloff 2001b). Some traits conducive to successful eradication are obvious – for example, large mammals are far easier to find than small insects, while plants with a soil seed bank are more difficult to eliminate than those without this feature (Simberloff 2001b). However, key biological traits often require substantial research, usually in the vein of natural history. Biological features figure large in successful eradications: smallpox has no non-human reservoir or long-term carriers (Fenner *et al.* 1988); the giant African snail does not self-fertilise (Mead 1979); *Anopheles gambiae* in Brazil was found almost exclusively near buildings (Hoelmer and

Grace 1989); while citrus canker (caused by *Xanthomonas axonopodis* pathovar *citri*), eradicated in the south-eastern United States in the early 20th century, had a very restricted host range and required movement of infected hosts by humans to spread (Merrill 1989). A recent successful eradication resting on carefully-determined biology of a pest and host was that of an introduced sabellid polychaete (*Terebrasabella heterouncinata*), parasitising abalone (*Haliotis* spp.) and other molluscs in Cayucos, California (Culver and Kuris 2000). The worms are specific to gastropod shells, especially large individuals of two common species, while the gastropod hosts have pelagic larvae, ensuring their rapid re-colonisation. The removal of 1.6 million highly susceptible hosts reduced the threshold host density below a point at which the worm could persist.

PROBABILITY OF RE-INVASION

Is the effort to eradicate an invader worth it if rapid re-invasion is likely? One reason so many eradication attempts have been on islands is that their isolation suggests immunity from rapid re-invasion. In many circumstances, even a successful eradication campaign can be a wasted effort because of re-invasion. In Washington state, an intensive campaign rid Long Lake (130 ha) of Eurasian water milfoil (*Myriophyllum spicatum*) (Thurston County Department of Water and Waste Management 1995). However, a public boat ramp permitted quick re-invasion, and the county switched to a programme of maintenance management by hand-pulling (M. Swartout, pers. comm. 1999). Other times, the probability of deliberate subversion of an eradication (Perkins 1989) is so high that the attempt may be futile. The reappearance of northern pike (*Esox lucius*) in Lake Davis, California, after its apparently successful eradication (Anon. 1999) probably resulted from sabotage (P. Moyle, pers. comm. 1999). The ease with which a single individual can subvert an eradication of some species (e.g., Davis 1990) may be an argument against the attempt when the goal is controversial.

In general, whether the probability of re-invasion should forestall an eradication campaign rests on a full assessment of the likely costs and benefits. There may be reasons to attempt eradication even if re-invasion is probable. For instance, sometimes the benefit of an eradication campaign may be a biologically artificial one, in that trade regulations may prohibit importation of some good unless its region of origin is certified as free of a pest. In such instances, the economic benefits may be so great that certain re-invasion would not argue against eradication attempts. This is the reason government officials repeatedly mount expensive eradication campaigns against the medfly in California and gypsy moth in parts of the United States and Canada in spite of a high probability of rapid re-infestation (Myers *et al*. 2000). This is not to say that the ecological and/or economic benefits of either of these campaigns might not suffice to justify them even in the absence of trade regulations. The point I am making is that low-level maintenance management, as opposed to eradication, is not an option because of trade regulations, even if maintenance management would achieve greater real control and/or cost less.

Independent of trade regulations, an eradication campaign can have sufficient economic, ecological, health, or even symbolic benefits to warrant the cost even if quick re-invasion is certain. In the successful eradication of the cattle tick from the United States, described above, re-infestation into the lower Rio Grande River region of Texas continually occurs through movement of infected animals from Mexico; leading to frequent small control operations (Klassen 1989). No one doubts the value of this programme. The Alberta rat control programme (Bourne 2000; Holubitsky 2000) is an inspirational eradication example despite frequent re-invasion. Norway rats (*Rattus norvegicus*) were first discovered on the eastern border of Alberta in 1950. Because rats destroy crops, every landowner and municipality in Alberta is mandated to destroy them, but the provincial government now pays all costs. The bulk of the activity is conducted by pest control inspectors hired and supervised by municipalities along the Alberta-Saskatchewan border. Every premise within a 29 x 600 km border zone is inspected at least annually, and control is effected primarily by eliminating food sources, extensive use of anticoagulant baits, and hunting by a team of seven provincial rat patrol officers. The cost is about CA$350,000 annually. Of course, re-invasion is continual, and every year between 36 and 216 infestations are discovered and destroyed. However, Alberta is so rat-free that discovery of a single rat in Edmonton or Calgary receives full media coverage. Aside from the benefit to agriculture of eliminating crop loss to rats, the programme has engaged the population of the entire province and sensitised them to the potential dangers of failing to deal promptly and comprehensively with invading species.

POSSIBILITY OF RESTORATION

Simply removing an invader does not constitute restoration (Towns *et al*. 1997). An ecological restoration scheme founded on eradication may be defeated by re-invasion or other problems (Simberloff 2001). Key species may be extinct and no acceptable functional equivalents available. Restoration efforts are sometimes mysteriously unsuccessful. For instance, after eradication of predators, re-introduction of stitchbirds (*Notiomystis cincta*) to New Zealand islands has failed to produce self-sustaining populations, and reasons are not apparent (Towns *et al*. 1997). Our knowledge of community structure and function is inadequate to predict with assurance the impacts of removing a prominent member of an ecological community. Thus, unforeseen impacts of eradication abound (references in Towns *et al*. 1997). Mouse densities increased greatly following eradication of Norway rats from Mokoia Island. Even control of top predator densities at levels far above eradication can lead to increases in densities of intermediate predators (“mesopredator release”; Terborgh *et al*. 1999) with various further effects throughout the community. Elimination of a predator can also lead to increased herbivore populations and damage; eradication of Pacific rats (*Rattus exulans*) from Motuopao Island to protect a native snail resulted in detrimental increases in a non-indigenous snail instead. Removal of an introduced herbivore can lead to proliferation of non-indigenous weeds rather than restoration of the native plant community. Eradication of rabbits from Motunau Island led to increases of introduced boxthorn (*Lycium ferocissimum*), while removal of grazing livestock from Santa Cruz Island (California) caused dramatic increases in fennel (*Foeniculum vulgare*) and other introduced plants (Dash and Gliessman 1994). Such changes in vegetation structure following elimination of an herbivore can, in turn, affect animal populations. For example, removal of cattle in both Nebraskan prairie (Ballinger and Watts 1995) and Mana Island, New Zealand (Newman 1994) has decreased native lizard populations by modifying vegetation.

Some impacts of eradication described in the previous paragraph might have been predicted, but others are so idiosyncratic that even a substantial scientific research project might not have suggested them. Thus, eradication is often a large, uncontrolled experiment, and we should expect unforeseen outcomes (Simberloff 2001b).

ECONOMICS OF ERADICATION

So far, I have addressed primarily the feasibility of eradicating a pest, with some attention to benefits that might accrue even if an eradication attempt is unsuccessful, as well as to unforeseen problems. I have thus avoided the key question of whether eradication is an appropriate approach even if it is feasible. Of course the prospect of permanent removal of an invader from a region, and thus the elimination of annual management costs as well as the danger of some delayed impact, must be very seductive. However, given the great costs that may be associated with successful eradication, especially over a substantial area, society cannot undertake to eradicate every pestiferous invader for which there is a high probability of eradication success. Prioritisation of invaders for management action is a general problem, and eradication decisions are just a part of that problem. Which invaders cause, or are likely to cause, the most damage, and under what circumstances is eradication the best of available management options? Typically such decisions are based on benefit-cost analyses (Arrow *et al*. 1996), but benefit-cost analyses of many natural resource issues, particularly those related to conservation, are problematic because there is often no market, as there is for an agricultural commodity (LeVeen 1989; Simberloff 1992). In the new field of invasion economics, benefit-cost analyses are especially problematic and have rarely if ever been adequately performed (Perrings *et al*. 2000). One problem is the great difficulty in predicting the trajectory of invasions, while another is the difficulty of predicting the impacts of various kinds of control measures. Surely benefit-cost analyses will have extremely wide confidence limits for many years to come.

Nevertheless, in certain circumstances, it seems that an eradication attempt would surely be justified by a comprehensive benefit-cost analysis. For smallpox (Fenner *et al*. 1988), the entire annual national and international cost of the eradication from the inception of a full-fledged campaign in 1967 to its success in 1979 was only USD23 million, while the annual cost of the disease (not counting control efforts) during this period to underdeveloped nations alone was at least USD1.07 billion, and worldwide was estimated as USD1.35 billion. The annual cost of control efforts before the eradication campaign just in the United States was USD150.2 million. Even if the campaign had not succeeded, so long as it had even a moderate probability of success, it would seem to have been an appropriate investment.

Just a rapid glance at the annual current management costs (not including losses and damages) estimated for some invaders in the United States (Pimentel *et al*. 2000) suggests that even an expensive eradication campaign might be appropriate, so long as the prospects of success were even moderate and the attempt would not substantially interfere with, or foreclose, other effective management techniques. Every year, the United States spends USD45 million on purple loosestrife (*Lythrum salicaria*) control, USD3-6 million on management of *Melaleuca quinquenervia*, USD4.6 million to manage the brown treesnake (*Boiga irregularis*) on Guam, and USD100 million to deal with Dutch elm disease (*Ophiostoma ulmi*). However, the real prospects of successful eradication of any of these species would have to be assessed based on detailed knowledge of its biology, and alternative methods (e.g., the recently released biological control agents for the first two species) may end up producing adequate control at far lower than current expenditures. My point in listing these examples is that each one entails an enormous annual expenditure, and I wonder if the possibility has been considered that total, long-lasting eradication could be achieved for, say, 10 or 20 times the current annual control cost, plus future costs of prevention. Do resource managers typically think this big?

CONCLUSIONS

There are some spectacular large-scale eradication successes. And there is a growing string of smaller successes. Further, a wide array of techniques has been successfully deployed – sterile insect release, male annihilation, traps, pathogens, vaccination, chemical sprays and baits, hunting, dogs, Judas goats, host removal, fire, and many other gory procedures. Nevertheless, eradication is almost a stepchild of management of invasives, often not considered as a possible solution even when the specifics of a situation might augur well for success. I see two main reasons for this disconnect:

- First, the literature on eradication is scattered and often very grey. Eradication of mammals is published in different outlets from insect eradication, and plant eradication histories, when published at all, are found in yet other sources. This conference is the first international conference spanning the entire field of eradication, and the number and high quality of presentations shows that the organisers have struck a very responsive chord. I predict that the published proceedings will go a long way towards both unifying the field and attracting the attention of policy makers, managers, and invasion biologists. In addition, leaders of eradication projects must recognise high-quality, international publication as a normal part of the job. If we want eradication to become a real option in managing invasive species, we have to publicise the methods and results better.

- Second, the entire problem of introduced species seems so overwhelming that it has induced a sort of fatalism – the forces arrayed against us, particularly the growing movement of cargo and people in the free-trade era, seem so overwhelming that some authors see us doomed to an eventual global homogenisation (e.g., Quammen 1998). Eradication, both because of publicised failures and because it is, in a sense, the management approach that aims the highest, falls victim to this fatalism even more acutely than other methods. But surely this sense of unavoidable doom is unwarranted. We know that eradication can work because it has. It has worked despite the relatively poor lines of communication I have outlined above and despite what would often seem to be the awesome biological powers of the target invader. New Zealanders have even developed an export industry of advice on, and application of, island mammal eradication techniques. What we do not know are the limits of most of these technologies. Just how large an island could be cleared of rodents by the techniques developed in New Zealand and northwestern Mexico? If the political will and economic support could be mustered, could nutria be completely eradicated in North America? Rabbits in Australia? What about invasive plants – under what circumstances could the witchweed approach be replicated? If smallpox and citrus canker can be eradicated, are insects on continents really out of the question?

I do not know the answers to these questions, but the inspirational stories from the literature and this conference suggest that we should not sell ourselves short. It is worthwhile to reflect on the defeatism expressed by the distinguished scientist René Dubos (1965) as he reflected on human disease eradication on the eve of the successful campaign to eliminate smallpox: "…it is easy to write laws for compulsory vaccination against smallpox, but in most parts of the world people would rather buy the vaccination certificate than take the vaccine; and they shall always find physicians willing to satisfy their request for a small fee. For this reason, and many others, eradication programs will eventually become a curiosity item on library shelves, just as have all social utopias." One thing is certain – we will surely lose the war against invasive nonindigenous species if we consider eradication an impossible fantasy and not an attainable reality.

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