Chapter 25
Eradication: Pipe Dream or Real Option?

Daniel Simberloff

Abstract Invasive alien plant populations have often been eradicated from very small areas, but pessimism about eradication of widely distributed plants pervades the management community. Contributing to this view are several legendary and expensive failed eradication campaigns, the inconspicuous nature of many plants, the existence of soil seed banks, and the perceived expense of eradication over large areas. However, if several years' worth of the cost of maintenance management campaigns could instead be devoted to a one-shot, well-funded eradication effort, projects that currently seem impossible might be brought within the range of feasibility. Factors in addition to cost that must be considered are whether adequate lines of authority can compel cooperation and prevent sabotage, whether there is sufficient knowledge of the target species to have identified a feasible approach to eradication that advances the goal of restoration, and the need for intensive monitoring and possible follow-up operations. Especially for PAs, the likelihood of reinvasion from nearby sites is a concern. If an eradication campaign would employ the same general methods as those that would have been used if the goal was maintenance control, there is likely little cost and much potential benefit to attempting eradication. Gradual improvement has occurred in plant eradication programmes through accumulated experience and incremental improvement of longstanding methods. However, the field of invasive plant management (including eradication) has not seen the advent of remarkably innovative new approaches and greatly improved records of eradication success that currently foster optimism and enthusiasm among managers dealing with invasive animals.

Keywords Extirpation • Monitoring • Small populations • Target species

D. Simberloff (✉)
Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, TN, USA
e-mail: dsimberloff@utk.edu

L.C. Foxcroft et al. (eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges*, Invading Nature - Springer Series in Invasion Ecology 7,
DOI 10.1007/978-94-007-7750-7_25,

25.1 Introduction

By ‘eradication’, invasion scientists mean removing every individual of a discrete, more or less isolated population. This is distinct from ‘extirpation’, which means eliminating a segment of a population, but with conspecific individuals still present in contiguous or nearby populations. Unfortunately, the term ‘eradication’ is used quite colloquially, particularly in media reports and political statements advocating or announcing ‘eradication’ of some weed or pest species, when what is meant is really extirpation of the species in some defined area. Sometimes ‘eradication’ is used to mean simply killing a lot of individuals, not even all of them at the same site. Part of a general scepticism about the feasibility of eradication stems from the fact that campaigns colloquially announced as ‘eradication’ campaigns were never meant to be that, so of course they failed to eradicate the target population (Simberloff 2003a).

Another factor generating scepticism about the possibility of eradication is the history of several high-profile, expensive true eradication campaigns that not only failed to eradicate their target species but had enormous damaging non-target impacts. An example is the failed campaign in the United States to eradicate white pine blister rust, introduced in the early twentieth century on white pine seedlings from Germany (Maloy 1997). The campaign aimed to eliminate the fungus by eradicating both native and introduced species of *Ribes*, the alternate host. Labour costs alone were over $150 million and were particularly heavy during World War I. During World War II, prison inmates as well as German and Italian prisoners of war dug up *Ribes* and spread chemicals, including in wetlands and stream-sides. Non-target impacts were massive, and the campaign failed utterly. Another continent-wide campaign, this time to eliminate *Berberis vulgaris* (European barberry), was similarly motivated. In the United States, *B. vulgaris* is an alternate host of stem rust of cereals, which inflicted enormous losses on wheat growers. The campaign, detailed by Campbell and Long (2001) and Mack and Foster (2009), began in 1918, lasted 60 years, employed thousands, and rendered *B. vulgaris* a rare plant in much of the United States, even today. As the methods included use of rock salt, kerosene, and dynamite (Mack and Foster 2009), one can speculate about non-target impacts.

In a widely cited paper, Rejmánek and Pitcairn (2002) found a decade ago that eradication of agricultural weed populations smaller than a hectare is usually feasible, and that for infestations between one hectare and 1,000 ha, between a fourth and a third of attempts they surveyed had succeeded. However, in their survey the cost of eradications rose so rapidly with area that they felt it was unlikely that eradications of plant populations occupying more than 1,000 ha would be feasible. Panetta and Timmins (2004) agreed with this threshold and suggested that the prospects for eradication of plants from natural areas would tend to be much dimmer than those for agricultural weeds. Gardener et al. (2010) recently cast further doubt on the feasibility of most plant eradication projects other than very small ones, an assertion subsequently publicised in a high-visibility news report

(Vince 2011). However, the past decade has seen dramatic progress in eradication of invasive animal populations (see for example Genovesi 2011a, b) on ever-larger islands, plus further experience with invasive plant management. It is thus timely to reconsider Rejmánek and Pitcairn's pessimism regarding large-scale plant eradications and also ask under what circumstances should attempted eradication be the preferred response to a plant invasion of a PA as opposed to some sort of maintenance management, such as biological, chemical, mechanical, or physical control.

It is a commonplace that plants are generally harder to eradicate than animals, especially vertebrates. Seed banks may persist in the soil for many years (Panetta 2004), numbers of plant individuals may be enormous, individuals – even seedlings of trees – may be small and cryptic, and the attractive baits and traps that have so aided animal eradication are not applicable to plants. Thus, the degree of optimism that has begun to infuse the community of managers and policymakers dealing with invasive animals (e.g. Genovesi 2011a, b) has largely failed to engage those managing invasive plants. It is telling that, in a recent international conference on eradication of invasive species on islands (Veitch et al. 2011), of 94 papers, 89 were about eradicating animals and only 6 about eradicating plants; of 45 abstracts, 44 were on eradicating animals and none were on eradicating plants.

25.2 Successes and Failures

Many small plant invasions have been eradicated from sites other than PAs. Mack and Lonsdale (2002) describe eradication in Australia of small populations of North American *Eupatorium serotinum* (late boneset) in a cattle sales yard, as well as eradication of Old World *Centaurea trichocephala* (feather-head knapweed) in a degraded pasture in Washington state, USA. In the marine realm, the "killer alga" (*Caulerpa taxifolia*) was eradicated from two sites in California (Anderson 2005; Woodfield and Merkel 2006). At one site, about 0.13 ha of the alga was distributed widely among 42.3 ha of a 100.6 ha lagoon, and the other consisted of a group of shallow ponds totalling 1.1 ha connected to a harbour. In South Australia, *C. taxifolia* was eradicated from an artificial marine water body 7 km long by a few hundred meters wide (Walters 2009).

Some small populations of invasive plants have been eradicated in PAs. For instance, Rejmánek and Pitcairn (2002) cite two eradications of small populations in the Channel Islands National Park, California, while *Oryza rufipogon* (Asian common rice) was eradicated from an area of 0.1 ha in Everglades National Park (Westbrooks 1993). Macdonald (1988) reports ten invasive plant species as having been eliminated from Kruger National Park, South Africa. He identified only four of these species: *Opuntia aurantiaca* (jointed cactus), *Acacia dealbata* (silver wattle), *Bidens formosa* (cosmos), and *Nicotiana glauca* (tree tobacco). However, L. Foxcroft (personal communication, 2013) reports that the latter species is present cyclically. The Bermuda Department of Agriculture has eradicated *Livistona chinensis* (Chinese fan palm), *Pimenta dioica* (allspice), *Eugenia uniflora*

(Barbados cherry), and *Citharexylum spinosum* (fiddlewood) from Nonsuch Island (Bermuda), a wildlife sanctuary of 5.7 ha (Mack and Lonsdale 2002). In 1972, the New Zealand government targeted 29 non-native plant species for removal from 2,943 ha Raoul Island, a designated nature reserve. For seven species that occupied relatively small areas, including highly invasive ones such as *Cortaderia selloana* (pampas grass), success is believed to have been achieved, although continued monitoring is undertaken to ensure that resurgence does not occur from a soil seed bank (West 2002). However, for the seven main target species, all originally quite widespread on the island, progress toward eradication has been more gradual, with occasional setbacks as new infestations are detected (Holloran 2006). In the Galapagos, four non-native plant species have been eradicated from Santa Cruz Island (two of these are not found elsewhere in the archipelago), each from an area less than 0.1 ha (Gardener et al. 2010). Although the great majority of Santa Cruz is part of the Galapagos National Park, at least one of eradications took place on private land. *Cenchrus echinatus* (sandbur) was eradicated from 64 ha on Laysan Island (Hawaiian Islands; 411 ha), managed as a PA by the US Fish and Wildlife Service, in a 10-year campaign beginning in 1991 (Flint and Rehkemper 2002; E. Flint, personal communication, 2007).

A much larger success, although not in a PA, was the eradication of the pasture pest *Bassia scoparia* (burning bush) from several thousand ha distributed over a linear distance of 900 km in western Australia (Randall 2001; Dodd 2004), no doubt helped by the fact that locations of all plantings had been recorded. Perhaps the most ambitious current plant eradication programme rivals the *Ribes* and *Berberis* eradication campaigns of the early twentieth century. This is the attempt to eradicate a parasitic agricultural weed, *Striga asiatica* (witchweed), which is ongoing after over 50 years (Eplee 2001; Mack and Foster 2009) and has reduced the infested area from 162,000 ha to less than 1,000 ha in North and South Carolina. Success is likely within a decade (Mack and Foster 2009).

Many more attempted plant eradications have failed than have succeeded. Gardener et al. (2010), for example, cite failure to eradicate (so far) 26 targeted plant species in the Galapagos, comparing this record to the four successes cited above. For at least two of these failures, no campaign was actually implemented. For *Caulerpa taxifolia*, several eradication efforts have failed, as against the three successes noted above (Walters 2009).

25.3 Criteria for Success

Myers et al. (2000) and Simberloff (2002a, b, 2003a, b) have suggested several criteria that characterise successful eradications and that should be met before eradication is attempted. Of course the idiosyncrasies of each case will weigh heavily, but the following factors should always be borne in mind:

1. Economic resources. Are resources sufficient to complete the eradication as planned, and are those resources encumbered in such a way that they will be available for the duration of the project, even as the target population and its perceived impact are greatly reduced? Costs of removing the last few individuals may exceed those of removing all the rest, and funding agencies may be inclined to reduce support once the problem is lessened (Mack and Lonsdale 2002). Are resources needed to manage the species in areas near the target protected area to prevent reinvasion, and, if so, are they available?
2. Adequate lines of authority. Eradication is, by its nature, an all-or-none phenomenon. By contrast, in maintenance management by chemical or mechanical control, for instance, the refusal of a few landowners to permit the project to be carried out on their property would not necessarily prevent substantial reduction of the target species. However, an inviolable sanctuary for the target would prevent eradication by definition. Do such sanctuaries exist adjacent to or near the target protected area?
3. Enough must be known about the biology of the target species that a route to eradication can be identified that is feasible with available resources.
4. The eradication project, even if successful, must not produce an undesirable condition. For PAs, the ultimate goal would almost certainly be restoration of a semblance of the natural ecological community and the dynamic trajectory it was following before the invasion. Thus, for instance, high likelihood that an eradicated plant species would simply reinvade quickly or be replaced by another introduced species would weigh heavily against attempting eradication (although this would not necessarily be decisive; see an example below). It is also possible the method used in an attempted eradication would have a high risk of non-target impacts that would prevent the restoration goal from being achieved. Massive use of some persistent herbicide, for instance, or tremendous damage from machinery used in a scorched-earth operation, might so damage the prospects for restoration as to be untenable.

25.4 Monitoring: Determining Success and Detecting Reinvasion

The main issue concerning eradication of any invasive plant population is whether it is feasible and at what cost. If so, particularly if the campaign is costly, it then becomes important to consider whether reinvasion is likely, whether it would be detected quickly, and what could be done about it if it occurs. Protected areas adjacent to unprotected lands pose particular problems in this regards, whereas island refuges are obviously at an advantage. Of course to a great extent likelihood of reinvasion depends on the location of the site relative to extant populations and the means by which propagules of the eliminated species might arrive. Constant vigilance is needed first of all to ensure that an eradication effort really was successful, and secondly, to note and deal with any newly arrived individuals.

To know that every last individual of a plant species is gone is fraught with many difficulties, enumerated by Panetta and Timmins (2004). With plants, the existence of a soil seed bank poses particular problems (Panetta 2004) and, depending on seed longevity, can mean that many years must pass before one can ascertain that eradication had occurred. For animals, depending on the species, it is common practice to declare success (or concede failure) quite quickly, for example, often 1 year for rats and 4 years for the Asian longhorn beetle (*Anoplophora glabripennis*) in Chicago (Haack et al. 2010). For plants, sometimes 4 years of absence has been chosen as the criterion for success, for example, *C. taxifolia* in California (Woodfield and Merkel 2006), while for *Bassia scoparia* in Western Australia the criterion was 3 years (Randall 2001). However, several announcements of eradication have been premature. For instance, in Queensland, a 40-plus-year campaign to eradicate several small populations of the North American herb *Helenium amarum* (yellow sneezeweed), first detected in 1953, was declared successful in 2002 after annual searching for survivors failed to detect any (Mack and Lonsdale 2002; Csurhes and Zhou 2008). But in 2007 several individuals were found; these were removed and the area continues to be monitored (Csurhes and Zhou 2008).

25.5 When Should Eradication Be Attempted in Protected Areas?

Not all potential eradication projects that meet the above criteria for high likelihood of success can be undertaken, if only because resources would likely not suffice. However, in assessment of alternative management possibilities – in essence, (i) do nothing, at least for the present, (ii) attempt some sort of maintenance management, or (iii) attempt eradication – several factors suggest that eradication deserves more consideration than it often gets.

First, if an invasion is recent and the invaded area still small, it is likely that eradication is feasible, as suggested by the data in Rejmánek and Pitcairn (2002; see Pluess et al. 2012). Furthermore, the cost would be far less than if the effort were made after the invasion had spread. The likelihood of damaging non-target impacts would be less, both because the invader is unlikely to have established important interactions with native resident species and because whatever eradication method is attempted will not be employed over a large area. Finally, it is risky to wait to see if the species begins to spread or cause problems and, if it does, only then undertake an eradication campaign. Many introduced species, including plants such as *Schinus terebinthifolius* (Brazilian pepper) and *Arundo donax* (giant reed), have remained restricted for long periods, even decades, before rather suddenly spreading widely (see Crooks 2005). Because eradication campaigns typically take time to plan and implement, one could easily miss a window of opportunity by delaying an eradication attempt. Also, some invasive plants have major ecosystemic impacts that are nevertheless sufficiently subtle that they are not detected quickly; plants that fix nitrogen or concentrate phosphorus can fall in this category (Simberloff 2011). Waiting until such impacts become evident may allow a species to spread to a point at which eradication is vastly more expensive and perhaps not feasible. *Crupina vulgaris* (common crupina) in the American West and *Clidemia hirta* (Koster's curse) in the Hawaiian Islands, two non-native plants that were discovered soon after arrival and almost certainly could have been eradicated, without likely reinvasion, were allowed to spread while authorities questioned whether they would be very damaging. Both proved highly invasive and were well beyond the stage when they could have been eradicated by the time it was agreed that they should be controlled (Simberloff 2003b). The alga *Caulerpa taxifolia*, which has now spread throughout much of the near-shore western Mediterranean, could also almost certainly have been controlled had a campaign been undertaken soon after discovery (Meinesz 1999).

For PAs, the status of the target plant in neighbouring areas is a particular concern, as the funding for management, including attempted eradication, in the PA is unlikely to allow efforts beyond that area. Thus, for instance, the State of Florida and US federal agencies have mounted a promising programme using chemical and mechanical means to reduce or eliminate *Melaleuca quinquenervia* (broad-leaved paperbark tree) from state and federal lands (including PAs) in south Florida (F. Laroche, personal communication). However, by statute public funds cannot support such efforts on private lands adjacent to government properties. Three biological control insects have been released and, of course, do not respect property boundaries. These may contribute to an effective maintenance management programme in this case, but biological control in otherwise untreated areas would be unlikely to lead to eradication of this or other invasive plant species if this were the goal.

For widespread invasions, including longstanding ones, the expense of an eradication campaign can be forbidding even if the technology exists to suggest that success is possible. However, a comparison to ongoing costs of maintenance management in some cases leads to speculation about whether attempting eradication might be the truly most cost-effective approach (Simberloff 2003a). For instance, the United States spends \$45 million annually on management of *Lythrum salicaria* (purple loosestrife) and \$3 million to \$6 million annually on control of *M. quinquenervia*). Thus, over a 10-year period, ongoing maintenance management costs tens or hundreds of millions of dollars. One can imagine that having such resources available over a much shorter period for an eradication attempt might make an eradication attempt feasible that would have been impossible with just a few million dollars. Another possible resource that has not been devoted to plant eradication is volunteer or prisoner labour. Such sources are now routinely used in a number of effective maintenance management programmes, especially in PAs (see Simberloff 2003a), and allow managers to marshal many more workers than could possibly have been paid. Use of vast amounts of manpower might make it possible to eradicate much more widespread invasions than would have been deemed feasible based on personnel costs alone.

If an eradication campaign uses the same method that would have been used had maintenance management been the goal, it may well be more cost-effective to invest added resources and attempt to eradicate the invader. This is because even failure to eradicate would be no great loss and would probably entail more complete maintenance management. One would have to tally the costs and potential benefits of the added effort. An excellent example is the project, begun in 1992, to eradicate *Ammophila arenaria* (European beachgrass) and hybrids of two African ice-plant species (*Carpobrotus edulis* × *C. chilensis*) from an 11 ha area of Lanphere Dunes in Humboldt Bay National Wildlife Refuge in California (Pickart 2013). The stated goal in terms of these invasive plants was eradication, and the goal for the system was restoration of the ecosystem to its trajectory before European modification, by restoring abiotic processes that maintain a dynamic dune ecosystem. Herbicides were precluded by local community objections, and the impact of heavy machinery on native vegetation, including two federally listed species, would have been too great, so the method chosen was digging and pulling by hand. *Ammophila arenaria* was almost wholly eliminated after 2 years, and ice-plant after 5. However, rare resprouts of both species are seen, and there is occasional reinvasion from nearby areas. Annual spot treatments control these at very low densities. Thus, complete eradication has not yet been achieved for either species (or else reinvasion quickly occurs), but the ultimate restoration goal has been met, and, if maintenance management rather than total eradication had been the stated goal, the method that was implemented would have been exactly the same. Further, the fact that quick reinvasion is likely does not invalidate the approach in this instance, as annual monitoring and spot treatments are feasible and inexpensive.

25.6 Further Advances?

Just within the last decade, animal eradication has advanced greatly, with projects that would have seemed impossible a decade or two ago now well within the realm of possibility (Genovesi 2011a, b), with better methods of avoiding non-target impacts (e.g. Caut et al. 2009) and with important conservation benefits (McGeoch et al. 2010). Some of these advances result from new technologies and others from incremental improvement of existing techniques, combined with ambition (Simberloff et al. 2013). Even if one grants the difficulties that are peculiar to invasive plant eradication, it seems as if greater successes are possible by the same routes that are forging progress in animal eradication. For instance, greater efforts using the same techniques that had previously failed to control invasive plant species on Motuopao Island (New Zealand) are leading towards successful eradication of several species (Beauchamp and Ward 2011). Assiduous application of longstanding techniques has led to eradication of small infestations of 12 non-native plant species from single islands in the Hawaiian archipelago and the imminent elimination of eight others (Penniman et al. 2011).

What do not seem to have arisen in invasive plant control generally, and eradication attempts in particular, are highly innovative new technologies. Meyer et al. (2011) suggest a new strategy, focused on preventing fruit production, which might permit eradication of small infestations of previously intractable *Miconia calvescens* (miconia). But absent are plant analogues to completely novel approaches such as the development of attractive pheromones that have greatly advanced sea lamprey management (Fine and Sorensen 2008), toxic micro-beads that have cleared some water facilities of zebra mussels (Aldridge et al. 2006), and the battery of genetic manipulations currently under way in attempts to eradicate populations of fishes (e.g. Thresher 2008) and insects (e.g. Pollack 2011). It seems unlikely that the biology of plants differs in characteristic ways from that of animals so as to inhibit the development of radically new control technologies.

References

Aldridge DC, Elliott P, Moggridge GD (2006) Microencapsulated BioBullets for the control of biofouling zebra mussels. Environ Sci Technol 40:975–979

Anderson LWJ (2005) California's reaction to *Caulerpa taxifolia*: a model for invasive species rapid response? Biol Invasion 7:1003–1016

Beauchamp AJ, Ward E (2011) A targeted approach to multi-species control and eradication of escaped garden and ecosystem-modifying weeds on Motuopao Island, Northland, New Zealand. In: Veitch CR, Clout MN, Towns DR (eds). Island invasives: eradication and management. Proceedings of the international conference on island invasives. IUCN, Gland, Switzerland and CBB, Auckland, New Zealand, pp 264–268

Campbell CL, Long DL (2001) The campaign to eradicate the common barberry in the United States. In: Peterson PD (ed) Stem rust of wheat: from ancient enemy to modern foe. APS Press, St. Paul, pp 16–43

Caut S, Angulo E, Courchamp F (2009) Avoiding surprise effects on Surprise Island: alien species control in a multi-trophic level perspective. Biol Invasion 11:1689–1703

Crooks JA (2005) Lag times and exotic species: the ecology and management of biological invasions in slow-motion. Écoscience 12:316–329

Csurhes S, Zhou Y (2008) Pest plant risk assessment. Bitter weed, *Helenium amarum*. Queensland Government, Department of Primary Industry and Fisheries, Brisbane

Dodd J (2004) *Kochia* (*Bassia scoparia* (L.) A.J. Scott) eradication in Western Australia: a review. In: Sindel BM, Johnson SB (eds) Proceedings of the fourteenth Australian weeds conference. Weed Science Society of New South Wales, Sydney, pp 496–499

Eplee RE (2001) Coordination of witchweed eradication in the USA. In: Wittenberg R, Cock MJW (eds) Invasive alien species: a toolkit of best prevention and management practices. CAB International, Wallingford, p 36

Fine JM, Sorensen PW (2008) Isolation and biological activity of the multi-component sea lamprey migratory pheromone and new information in its potency. J Chem Ecol 34:1259–1267

Flint EN, Rehkemper C (2002) Control and eradication of the introduced grass, *Cenchrus echinatus*, at Laysan Island, central Pacific Ocean. In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species. IUCN Species Survival Commission, Gland, pp 110–115

Gardener MR, Atkinson R, Rentería JL (2010) Eradications and people: lessons from the plant eradication program in Galapagos. Restor Ecol 18:20–29

Genovesi P (2011a) Eradication. In: Simberloff D, Rejmánek M (eds) Encyclopedia of biological invasions. University of California Press, Berkeley, pp 198–203
Genovesi P (2011b) Are we turning the tide? Eradications in times of crisis: how the global community is responding to biological invasions. In: Veitch CR, Clout MN, Towns DR (eds) Island invasives: eradication and management. Proceedings of the International conference on island invasives. IUCN, Gland, Switzerland and CBB, Auckland, New Zealand, pp 5–8
Haack RA, Hérard F, Sun J et al (2010) Managing invasive populations of Asian longhorned beetle and citrus longhorned beetle: a worldwide perspective. Annu Rev Entomol 55:521–546
Holloran P (2006) Measuring performance of invasive plant eradication efforts in New Zealand. NZ Plant Protect 59:1–7
Macdonald IAW (1988) The history, impacts and control of introduced species in the Kruger National Park, South Africa. Trans R Soc S Afr 46:251–276
Mack RN, Foster SK (2009) Eradicating plant invaders: combining ecologically-based tactics and broad-sense strategy. In: Inderjit (ed) Management of invasive weeds. Springer, Dordrecht, pp 35–60
Mack RN, Lonsdale WM (2002) Eradicating invasive plants: hard-won lessons for islands. In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, pp 164–172
Maloy OC (1997) White pine blister rust control in North America: a case history. Annu Rev Phytopathol 35:87–109
McGeoch MA, Butchart SHM, Spear D et al (2010) Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. Divers Distrib 16:95–108
Meinesz A (1999) Killer algae. University of Chicago Press, Chicago
Meyer J-L, Loope L, Goarant A-C (2011) Strategy to control the invasive alien tree *Miconia calvescens* in Pacific islands: eradication, containment or something else? In: Veitch CR, Clout MN, Towns DR (eds) Island invasives: eradication and management. Proceedings of the international conference on island invasives. IUCN, Gland, Switzerland and CBB, Auckland, New Zealand, pp 91–96
Myers JH, Simberloff D, Kuris AM et al (2000) Eradication revisited: dealing with exotic species. Trends Ecol Evol 15:316–320
Panetta FD (2004) Seed banks: the bane of the weed eradicator. In: Sindel BM, Johnson SB (eds) Proceedings of the 14th Australian weeds conference. Weed Society of New South Wales, Wagga Wagga, NSW, Australia, pp 523–526
Panetta FD, Timmins SM (2004) Evaluating the feasibility of eradication for terrestrial weed incursions. Plant Protect Q 19:5–11
Penniman TM, Buchanan L, Loope L (2011) Recent plant eradications on the islands of Maui County, Hawai'i. In: Veitch CR, Clout MN, Towns DR (eds). Island invasives: eradication and management. Proceedings of the international conference on island invasives. IUCN, Gland, Switzerland and CBB, Auckland, New Zealand, pp 325–331
Pickart A (2013) Dune restoration over two decades at the Lanphere and Ma-le'l Dunes in northern California. In: Martinez ML, Gallego-Fernandez JB, Hesp PA (eds) Restoration of coastal dunes. Springer, New York, pp 159–171
Pluess T, Jarošík V, Pyšek P et al (2012) Which factors affect the success or failure of eradication campaigns against alien species? PLoS ONE 7:e48157
Pollack A (2011) Mosquito bred to fight dengue fever shows promise in a field test. New York Times, October 31, p B1
Randall R (2001) Eradication of a deliberately introduced plant found to be invasive. In: Wittenberg R, Cock MJW (eds) Invasive alien species: a toolkit of best prevention and management practices. CAB International, Wallingford, p 174
Rejmánek M, Pitcairn MJ (2002) When is eradication of exotic pest plants a realistic goal? In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, pp 249–253

Simberloff D (2002a) Today Tiritiri Matangi, tomorrow the world! Are we aiming too low in invasives control? In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, pp 4–12

Simberloff D (2002b) Why not eradication? In: Rapport DJ, Lasley WL, Ralston DE et al (eds) Managing for healthy ecosystems. CRC/Lewis Press, Boca Raton, pp 541–548

Simberloff D (2003a) Eradication – preventing invasions at the outset. Weed Sci 51:247–253

Simberloff D (2003b) How much information on population biology is needed to manage introduced species? Conserv Biol 17:83–92

Simberloff D (2011) How common are invasion-induced ecosystem impacts? Biol Invasion 13:1255–1268

Simberloff D, Martin J-L, Genovesi P et al (2013) Impacts of biological invasions – what's what and the way forward. Trends Ecol Evol 28:58–66

Thresher RE (2008) Autocidal technology for the control of invasive fish. Fisheries 33:114–121

Veitch CR, Clout MN, Towns DR (eds) (2011) Island invasives: eradication and management. Proceedings of the international conference on island invasives. IUCN, Gland, Switzerland and CBB, Auckland, New Zealand

Vince G (2011) Embracing invasives. Science 331:1383–1384

Walters L (2009) Ecology and management of the invasive marine macroalga *Caulerpa taxifolia*. In: Inderjit (ed) Management of invasive weeds. Springer, Dordrecht, pp 287–318

West CJ (2002) Eradication of alien plants on Raoul Island, Kermadec Islands, New Zealand. In: Veitch CR, Clout MN (eds) Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, pp 365–373

Westbrooks RG (1993) Exclusion and eradication of foreign weeds from the United States by USDA APHIS. In: McKnight BN (ed) Biological pollution. Indiana Academy Science, Indianapolis, pp 225–241

Woodfield R, Merkel K (2006) Final report on the eradication of the invasive seaweed *Caulerpa taxifolia* from Agua Hedionda Lagoon and Huntington Harbour, California. Southern California Caulerpa Action Team, Long Beach, California