Eradicating invasive plants: Hard-won lessons for islands

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Abstract The record of eradicating invasive plants, whether on islands or continents, consists of few clear victories, some stalemates, and many defeats. Instructive, if hard-won, lessons have nevertheless been learned. (1) The ideal eradication campaign would see all individuals of a potentially-invasive species destroyed immediately upon their arrival. Few immigrants meet this fate; more usually there is a failure to act until damage has been inflicted by the invader. (2) Failing the destruction of all immigrants upon their entry, maximum effort should be lodged against the immigrants' small, isolated foci. As with (1), implementing this sound advice has often proven difficult. However, the radical reduction of the range of *Striga asiatica* in North Carolina (USA) represents the clearest sustained application of this principle. (3) Eradication or even effective control of invasive species requires repeatedly surveying the same area for surviving plants. Virtual eradication of *Schinus terebinthifolius* and other invasive species on tiny islands in Bermuda has clearly succeeded through such diligence. (4) Control or even eradication of a single invasive species may ultimately produce little benefit, if its demise only sparks the rise of another non-indigenous species (e.g., the role reversals of invasive aquatic macrophytes in southern Florida, following biological control). Islands, with their intrinsic borders and geographic isolation, provide excellent locations for experimentation within which these lessons can be honed, thereby identifying both the effective and ineffective components of any eradication effort.

Keywords nascent foci; *Centaurea trichocephala*; *Salvinia molesta*; invaders; invasion; Bermuda; Hawaii.

INTRODUCTION

From the perspective of their human colonisers, oceanic islands have native floras that did not satisfy human needs or at least their desires in plants (Mack 1999, 2001). This perceived paucity in the native floras of islands has long sparked the transfer of huge numbers of non-indigenous species to islands from widely separated regions, mainly continents (e.g., Streets 1962; Whistler 1991). Islands' lack of biotic diversity compared with continents has also meant that they are poorly protected by biotic barriers to naturalisation and invasion (Mack 1996). They commonly lack native representatives for many of the taxa from which arise the natural enemies potentially capable of extirpating immigrants. For example, Hawaii has no native isopterans; Fiji has no native bruchids, both important plant foragers. The combination of increased propagule pressure and greater intrinsic vulnerability to invaders has meant that oceanic islands have become, in the past 500 years, among the most floristically-transformed land-masses. For example, Bermuda, a British possession since the early 17th century, has long been a transit point and provisioning station between the Western Hemisphere and Europe (Craven 1938). With few native species that were deemed commercially valuable, plant introduction was rampant; by 1918, more than 300 species were listed as naturalised (Britton 1965), all packed onto approximately 50 km^2. Oceanic islands commonly contain several-fold more non-indigenous plant species than an equivalent area of mainland (Lonsdale 1999). In a sense, deliberate eradication (i.e., the total elimination of a species) compensates, however feebly, for the inability of the islands' native biota to provide resistance to plant naturalisation. On the other hand, the possibility of preventing re-infestation is greater on islands, so eradication campaigns may be more fruitful.

The central questions we pose here deal with how eradication of non-indigenous plants can be implemented on islands. In assembling and evaluating the answers, we have drawn on examples from islands and continents, seeking common denominators among both successful and unsuccessful eradication efforts. Best practices in eradication (*sensu* Wittenberg and Cock 2001) are applicable to islands and continents. Furthermore, we have too few well-documented cases to exclude automatically examples based simply on the size of the land-mass in which they occurred. For much the same reason, examples from oceanic islands are applicable to ecological islands (i.e., habitats surrounded by distinctly-different environments), (e.g., lakes and ponds). From the standpoint of evolution, dispersal, biogeography (MacArthur and Wilson 1967) and the efficacy of eradication, these ecological islands are analogous to geographic islands.

Eradication of non-indigenous species anywhere is a quintessential application of science to technology. A clear distillation is needed of simply, 'what has worked' and, 'what has not.' Consequently, we strive here to assemble and discuss relevant examples in order to form basic lessons that can be placed readily into practice.

DEFEATS IN ERADICATING INVASIVE PLANTS: OPPORTUNITIES LOST

Islands, as well as continents, provide ample examples of prime opportunities lost in the eradication of introduced species that later became invaders. Among the least-justifiable episodes was the establishment and spread of the sprawling shrub, *Clidemia hirta* in Hawaii. The well-named "Koster's curse", was first collected on Oahu in 1949 but reputedly detected in Hawaii in 1941 (Anon.

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1954). As early as 1949 it was escaping from gardens. The same population had grown to cover less than 100 ha in 1952. As recently as 1961, Plucknett and Stone included *C. hirta* among "…several species which seem to be spreading in certain areas but which at present cannot be classed as dangerous or even common weeds." This lack of concern about the shrub's future proved unfortunate. By 1977, *C. hirta* had covered about 31,350 ha on Oahu (Wester and Wood 1977). In the most recent range estimation for Oahu of which we are aware, it reportedly occupies 100,000 ha (Smith 1992). Moreover, it has now spread to five other main islands in the state (http://www.hear.org/AlienSpeciesInHawaii/maps/CliHirHI.htm). Even by 1949 the reputation of *C. hirta* should have prompted its vigorous eradication. Koster's Curse had already caused so much damage in Fiji that it was declared a noxious weed by 1920 (Patel 1971). The Hawaiian governmental response in 1954 was not an eradication plan, which was then still possible, but control through release of a thrip, *Liothrips urichi*. This action simply copied a biological control effort begun on Fiji in 1930. Putatively, the thrip prevented the shrub from entering croplands and pastures; it did not, however, prevent its spread into native forest. Given the extent that *C. hirta* has now spread in Hawaii, coupled with its dispersal by birds (Wester and Wood 1977), its eradication seems exceedingly unlikely.

Other species (and the date by which they were first collected) that represent lost opportunities for eradication in Hawaii are *Hypericum perforatum* (1961), *Pistia stratiodes* (1932 or 1933), *Mollugo cerviana* (1975), and *Carduus pycnocephalus* (1986) (Wagner *et al*. 1990). *Olea europaea* subsp. *europaea* was first collected in 1982 (Wagner *et al.* 1990), although it may have entered Hawaii much earlier and remained unrecognised amongst specimens of *O. europaea* subsp. *africana*. Detection of the generalist hemiparasite *Cuscuta campestris* (Parker and Riches 1993) in 1955 and its subsequent naturalisation are particularly galling. Dodders form one of the few genera prohibited from entry into the United States (Westbrooks and Eplee 1999). Because *C. campestris* is native to the U.S. mainland, its arrival in Hawaii did not merit the eradication campaign that would have been triggered if it originated outside the U.S. (Westbrooks and Eplee 1999), a ludicrous gap in federal law. Important here is that by the time each species was first detected and was still confined to a few small populations, its propensity for invasion elsewhere had already been well documented.

EARLY DETECTION AND ERADICATION: OPPORTUNITY GRASPED

Lessons from another arena

The difficulty of detection and eradication of non-indigenous plants is forcefully illustrated by an example from an unexpected quarter – the Counter Cannabis Field Operation of the U.S. Drug Enforcement Administration in Hawaii. Conventional wisdom maintains that if a non-indigenous species is to be eradicated, it must be detected soon after its entry into a new range. Equally well understood, however, is that most non-indigenous species are exceedingly difficult to detect in low numbers. They are partly hidden under native species, or their habit, stature, leaf colour, morphology and texture, or any other visible features are indistinguishable from their native neighbours, except under close examination. Yet so strong is this link between early detection and eradication that much effort has been devoted to finding tools that aid detection. Limitations of the current range of tools to facilitate early detection have been discussed elsewhere (Mack 2000). In what is likely the most concentrated effort in the early detection of a non-indigenous plant species – the surveillance for illicit *Cannabis sativa* in Hawaii – visual detection from the air remains the tool of choice. Field agents of the U.S. Drug Enforcement Administration in Hawaii develop a search image for *C. sativa*. They detect plants primarily from the air in either fixed wing aircraft or helicopters flying at approximately 150 m altitude and usually above dense forest. Eradication either by aerial spraying or destruction of plants by ground teams follows immediately after aerial detection. Despite a level of surveillance and follow-up destruction that has been rarely, if ever, matched in the eradication of invasive species, *C. sativa* continues to be found in rural Hawaii (Anonymous source, Honolulu District Office, U.S. Drug Enforcement Administration).

This account may seem a far cry from the eradication of invasive plants, but we contend that some parallels do exist. Even though *C. sativa* is not even naturalised in Hawaii (Wagner *et al.* 1990), much less invasive, it persists through animal (i.e., human), dispersal into suitable habitat. Thus, its spread is analogous to the spread in Hawaii of other non-indigenous species, such as *Clidemia hirta*, *Miconia calvescens* or *Psidium cattleianum* (Cuddihy and Stone 1990). Despite the apparent inability of *C. sativa* to persist without cultivation in Hawaii, its illicit cultivation ensures that it occurs in small, widely scattered populations. This distribution obviously hampers its detection, also producing a distribution similar to a bird-dispersed non-indigenous species, early in its residence in a new range (e.g., *Miconia calvescens*). Furthermore, the difficulty of early detection for any non-indigenous species beneath a forest canopy is certainly illustrated with illicit *Cannabis* cultivation in Hawaii, in which the growers hide their plots in dense forest. Twenty-four years after initiation of this extensive aerial surveillance and eradication programme, the results represent broad-sense control but hardly eradication. In 1999, 3,413,083 *Cannabis* plants were destroyed in the U.S. (http://www.dea.gov/programs/marijuana.htm); more than half in Hawaii. The result of this initiative has caused a reduction in the amount of *Cannabis* (or the comparative ease of its detection) in the field compared with results in the past. The species is too widespread, has a long-lived repository for propagules for re-infestation (as long-lived as the human desire to take illicit drugs), and is too readily dispersed for eradication to be a realistic goal. It is worrying that many invasive species in Hawaii fit this template.

Plant eradication: a few slender victories

Can eradication of potentially-invasive plants ever be achieved? In citing the record for invasive animal eradications in Florida, Simberloff (1997) found no analogous list of successful pest plant eradication programmes for the United States or elsewhere. A list can indeed be assembled, although it is a slender one so far. These examples are instructive because they share common features – they consistently involve very small plant populations, usually a few hundred individuals, comprising one or only a few foci. Detection was apparently very early in the immigration; the decision to destroy all the immigrants was swift, and most important – repeated field operations at least reduced the non-indigenous species below levels of detection, even if extirpation cannot always be demonstrated.

Eupatorium serotinum is a herb native to North America. It was first detected in Australia in 1962 at Nerang, Queensland where it was growing in the cattle salesyard on a railway property. The population grew from occupying 9 m^2, to occupying about 49 m^2 just a year later. Although its identity was not confirmed until September 1963, authorities from the Queensland Department of Lands began to destroy the population four months earlier. This action was timely because some plants were already producing abundant seeds. The site was surveyed annually until 1980, and specimens were routinely collected (e.g., A. J. Tomley (s.n.), Alan Fletcher Research Station, Brisbane, Queensland). Any newly emergent plants were hand-pulled, and by 1980 no additional plants had been found for several consecutive years (corres. of K. L. Kay, Biological Branch, Queensland Department of Lands, 1980).

Similar conscientiousness in eradication was also displayed in Queensland's Mt. Tarampa District upon first detection in 1967 of several small populations of *Helenium amarum*, another composite herb native to North America. Here again, upon detection the populations were treated swiftly with herbicide, the site was searched annually for residual plants until 1992 (corres. of B. Whyte, Queensland Department of Lands), and specimens were routinely collected (T. A. Cole (s.n.), Alan Fletcher Research Station, Brisbane, Queensland). Both these Queensland examples of eradication are remarkable because the non-indigenous species were unknown in Australia beforehand, and the species have not been detected in the country since eradication campaigns were launched against them.

Action on a similar scale led to the eradication of the perennial herb, *Centaurea trichocephala*, in the western United States. At least eight *Centaurea* spp. are already naturalised, if not invasive, in the U. S. (Whitson *et al*. 1996). As a group, they are considered among the most noxious weeds in the arid western U.S. because they readily invade rangelands, provide no forage for livestock, and their control is difficult, if not intractable (Watson and Renney 1974). The reputation of the congeners led to unusually swift action when a previously-undetected *Centaurea* species was identified near Tampico (Yakima County), Washington in 1985. Once identified, the only known population (approx. 300-400 adult plants) was destroyed with herbicide in 1986. Additional herbicide treatment followed in 1987, and the site was inspected annually for any remaining *C. trichocephala* through 1990. In 1990 the Yakima County Noxious Weed Board made an official declaration that *C. trichocephala* had been eradicated from its only known site (M. Slaugh, pers. comm.). No other plants have been detected in the United States.

Salvinia molesta, the highly-aggressive tropical aquatic invader (Thomas and Room 1986), was not discovered in the United States (apart from in a few botanical gardens) until 1995 (Myers 1982 as cited in Nelson 1984). Its first field detection was in a 0.6 ha pond near Walterboro (Colleton County), South Carolina (Johnson 1995); conditions from which it could have spread readily. Eradication began several months after its detection. Hand removal of *S. molesta* in the pond was followed by herbicide application by the South Carolina Department of Natural Resources and the federal Animal Plant Health and Inspection Service (APHIS). As a result, the species was eradicated from the site and potentially from the U.S. (R. G. Westbrooks, pers. comm.). Unfortunately, this victory was short-lived: *S. molesta* appeared in Texas in 1997 and in more states in 1999. Most recently it has been found in North Carolina, plus new locales in other states, including Hawaii (Anon. 2000, http://nas.er.usgs.gov/plants/sa_molesta/docs/sa_mol.html). Its further spread in the U.S. is certain without extraordinary effort at containment and eradication, which may already be unattainable.

New Zealand has been exemplary in its successful national eradication programmes for non-indigenous terrestrial and aquatic species. So far, the terrestrial species *Acroptilon repens* and *Chondrilla juncea*, and the aquatic macrophytes, *Zizania palustris*, *Menyanthes trifoliata*, and *Pistia stratiodes*, have been eradicated from all known sites in the country (P. D. Champion, pers. comm.). Furthermore, the scourges *Sorghum halepense*, *Eichhornia crassipes* and *Salvinia molesta* have been reduced, such that the only occurrences are newly-detected sites. For *E. crassipes* and *S. molesta* the new sites are presumably the products of their illegal culture. All these eradication efforts are testimony to the zeal that New Zealand has applied in scrutinising the entry of non-indigenous species, then controlling and eliminating those that prove harmful under its biosecurity legislation, as consolidated in the Biosecurity Act of 1993 and its amendments (http://rangi.knowledge-basket.co.nz/gpacts/public/text/1993/an/095.html).

Other eradication programmes are underway; for some, the objective appears to be attainable. It is remarkable that the once deliberate spread of the salt-tolerant shrub *Bassia* (or *Kochia*) *scoparia* in Western Australia has not only been halted but even reversed. Although this Eurasian species was well known as an aggressive weed in North America (Whitson *et al.* 1996), dubious claims as to its forage value (e.g., Grimson *et al.* 1989; Mir *et al.* 1991) led to its introduction in Western Australia in 1990. Soon sufficient grower concern had arisen about its invasive character that its sale was halted, and control measures were begun in 1992. From an estimated maximum extent of 3277 ha in 1993, the new Australian range of *B. scoparia* has been reduced to about 5 ha. The plant has not reappeared in treated areas for at least three years (Wittenberg and Cock 2001). As laudatory as this effort has been, a stringent standard for eradication (total destruction of all detectable plants), is necessary. Too often, a non-indigenous species has been on the verge of eradication, the effort has slackened, and the species has rebounded in its new range (e.g., *Berberis vulgaris* in the U.S., Mack 2000). As satisfying as the reduction of an invasion to a few small remnant populations is, these remnants may become the nascent foci from which a re-invasion can emerge (Moody and Mack 1988; Higgins *et al.* 2000).

Even though these examples are drawn from widely-separated areas and involve taxonomically-unrelated species, these eradication campaigns share common traits. The non-indigenous species consisted of one, or at most a few, small populations. Eradication efforts began promptly after first detection, sometimes (as in the case of *S. molesta* in South Carolina) a few months later. And equally important, there were repeated surveys of the treated site(s) to detect and destroy any new or previously overlooked plants. Eradication, not simply control, was recognised as attainable, provided the effort was initiated quickly. These examples include both island and continental case histories: islands may offer some advantages in eradication, given their geographic and ecological boundaries, but even a non-indigenous species on a continent can be eradicated if action is swift.

STRATEGY COMPARED WITH TACTICS: ATTENTION TO NASCENT FOCI

The spatial pattern of adventitious and naturalised species falls into two functional categories: species that reproduce but do not expand their range beyond the point of introduction (e.g., *Rotala indica* at the Biggs Experiment Station, Davis, California (USA), Barrett & Seaman 1980), and those that disperse into new locales. A non-indigenous species with many widely-separated foci is much more difficult to eradicate than a single infestation. But detection and eradication of all nascent foci of a potential invasion are more important than attacking large centres of the infestation, whether control or eradication is the goal. Ignoring these small foci of an introduced species, while treating only major infestations in a new range, simply provides time for these once-inconspicuous populations to flourish. This contention, which seems at first counter-intuitive, has been amply illustrated in theory (Moody and Mack 1988; Higgins *et al.* 2000) and more importantly demonstrated in successful control campaigns that may yet be transformed into eradication.

The pending eradication in the United States of the introduced herbaceous hemiparasite, *Striga asiatica*, through emphasis on the destruction of nascent foci, is remarkable. The prospects for control, much less containment and potential eradication, would have seemed bleak when this aggressive plant parasite was first detected in North Carolina in 1956. The plant had already infested maize in a four-county area and would subsequently be detected in northern South Carolina (Westbrooks and Eplee 1999). In addition to establishing an effective quarantine to the export of *S. asiatica* seeds from the infected region, APHIS destroyed outlier populations and nascent foci before attempting to shrink the main infestation (Eplee 1981). Through this strategy, the invasion had been reduced in 1999 to about 6000 ha from its maximum extent of 177,000 ha in the early 1960s (Westbrooks and Eplee 1999). Across four decades the cost of destroying *S. asiatica* in the U.S. has totalled USD250 million (R. E. Eplee, pers. comm.): good value compared with the crop losses an unchecked *Striga* invasion would have caused. Much greater savings would have been realised over the last 50 years, however, had the same money been applied to a national early detection/eradication network that would have included the detection and eradication of all damaging non-indigenous species, including *Striga,* upon their entry.

Eradication may not be attainable in other cases, but attacking nascent foci nevertheless remains the key to the areal containment of the invader. *Casuarina equisetifolia* and *Melaleuca quinquinervia* are devastating tree invaders within the Everglades, an ecological and floristic island in southern Florida. They each occupy 100,000 ha of habitat (Schmitz *et al*. 1997). Despite the immense infestations that each species forms, emphasis in their effective control has centred on eradicating their nascent foci (Doren and Jones 1997). Slowly, these invasions may be first blunted and then potentially shrunk; attention can be then shifted to destruction of centres of the infestation.

Perhaps more challenging has been the attempt to control spread of *Mimosa pigra* (catclaw mimosa) in the Northern Territory, Australia. Its dispersal has been aided by water transport of its seeds, and until recently, by the introduced Asian water buffalo, which is itself the subject of a control/eradication programme (Lonsdale *et al.* 1989). The ability to detect nascent foci in the Australian new range of *M. pigra* is even more daunting than for introduced trees in the Everglades – the potential area of search is enormous and remote. Nevertheless, multi-year control of *M. pigra* in these satellite populations has successfully prevented the development of large stands of the invader in Kakadu National Park (Cook *et al.* 1996).

Attention to small foci has also contributed to containing catclaw mimosa in southern Florida. The potential for its invasion remains high; it currently occurs in approximately 395 ha (Schmitz and Westbrooks 1997), scattered across three counties. So far, only two populations have been eradicated, but almost 20,000 plants are destroyed annually. Eradication of one of the populations involved removing all seeds from the 20-30 trees in the stand, felling the trees, treating the stumps with herbicide, and surveying the site for post-treatment seedlings (R. Kipker, pers. comm.). Eradication is an attainable goal for *M. pigra* in southern Florida, but 15 years into the campaign, it is clear this will be a long-term venture. Other long-term eradication campaigns include *Chromolaena odorata* (eight years and continuing) and *Orobanche ramosa* (five years and recommended to continue for at least 10 more years), both in Australia.

The detection and destruction of outlier, and particularly nascent isolated populations, remain among the most important lessons learned from attempts to eradicate potentially invasive species. This lesson applies equally well to invasions anywhere – islands or continents.

POST-ERADICATION SURVEYING AND RE-ERADICATION: THE ESSENTIAL POSTSCRIPT

In a strict sense, use of the term 'eradication' in combating non-indigenous plants can be a misnomer. It can be confidently applied to those cases in which all individuals of a species are destroyed in a new range (Westbrooks 1993), but more commonly it refers to destruction of individuals below a level of detection. This distinction is important because even if all vegetative plants are destroyed, the species may persist in a seed bank. Repeated surveying of the area is essential, no matter its size, to detect remnant or newly-emergent individuals.

Post-eradication surveys for non-indigenous species have a distinguished but apparently long-forgotten past. The 1876 centenary celebration of the United States included an international exhibition in Philadelphia at which many nations exhibited their country's livestock and crops. The United States Centennial Commission had the remarkable foresight to recognise this venue as an opportunity for the inadvertent introduction of non-indigenous species. Local botanists carefully surveyed the exhibition grounds for *four years* after its close, in order to detect any immigrant species that might gain a foothold. Thirteen adventives were detected (and presumably destroyed); these included the herbs *Crepis tectorum*, *Centaurea nigra* and *Lepidium sativum* (LeConte *et al.* 1881). Had this episode sparked diligent early detection and eradication of inadvertent plant immigrants in the U.S., the scope and magnitude of invasions into the U.S. in the following 120 years might have been much different.

As noted above, repeated survey of the site of a putative eradication is not only prudent but should be mandatory. The initial eradication effort for *Eupatorium serotinum* and *Helenium amarum* in Queensland did not completely eliminate these immigrant species. Had the detection sites not been repeatedly inspected and treated, these species might still be in the Queensland flora and even naturalised. Similar attention to survivors has been a hallmark of successful eradication efforts.

While repeated surveying becomes much more difficult as the treated area becomes larger, it is still obligatory. In the ambitious programme to restore large fractions of the Cape Floristic Province in South Africa to their native floristic condition, all invasive species are progressively removed or destroyed *in situ* within much of the Cape Peninsula National Park. Since the early 1980s, each block in the extensive eradication design has been progressively cleared of invasive plants, then surveyed thoroughly every two years to destroy any plants emerging from these species' depleting seed banks. By the mid-1980s, more than 6700 ha had been treated in this manner (Clark 1985, Macdonald 1989). This re-surveying has continued. One of us (RNM) viewed restored sites in 2000; no invasive species were detected.

The distinction between eradication and control, albeit diligent control, blurs where there is a continual threat of the re-entry of invasive species. The vegetation of Bermuda has been almost totally transformed by introduced species (Wingate 1992). Particularly devastating was the 1940s entry of North American scale insects, which rapidly reduced the once-prominent Bermuda cedar *Juniperus bermudiana* to a few remnant trees (Challinor and Wingate 1971). Unfortunately, re-establishment of forest cover in Bermuda followed the same foolhardy practices adopted elsewhere (Mack 2001); native species were ignored in the name of expediency to re-establish forest cover. As a result, subtropical trees were introduced, principally *Casuarina equisetifolia* (Nolan 1980), which has in the past decade become invasive (D. B. Wingate, pers. comm.). Worse still, *Schinus terebinthifolius* (Brazilian pepper) escaped from gardens in the 1950s and became invasive (Challinor and Wingate 1971). Today, *S. terebinthifolius* is invading the upland margin of Bermuda's mangrove swamps (Thomas 1993), as well as other habitats. Introduced starlings (*Sturnus vulgaris*) repeatedly carry the bright red fruits of *S. terebinthifolius* throughout the island group. As a result, even tiny islands offshore can be readily re-infested with *S. terebinthifolius* (D. B. Wingate, pers. comm.).

Nonsuch Island is only 6 ha, yet it is important as a reserve of natural vegetation (Wingate 1992). The Bermuda Department of Agriculture has removed all *S. terebinthifolius* (along with *Asparagus densiflorus*, *Livistonia chinensis*, *Pimeta dioica*, *Eugenia uniflora* and *Citharexylum spinosum*) on Nonsuch Is. for more than 20 years. Eradication has been possible for all species, except the bird-dispersed *S. terebinthifolius* and *A. densiflorus*. Seedlings of all introduced species are removed as they appear. Any delay in removal only diminishes the likelihood of eradication; for example, two-year old Brazilian pepper will produce seeds. As many as 500,000 Brazilian pepper seedlings must be removed on this small island annually in order to hold the invasion in check (D. B. Wingate, pers. comm.).

IS SINGLE SPECIES ERADICATION WORTHWHILE? THE SISYPHUS EFFECT

In addition to the trials and tribulations of ever achieving a species' eradication, removal of a single species can have an unintended effect. In conservation biology, the implicit goal in control or eradication of invaders is the prospering of native species (Van *et al.*1998). But does that outcome always occur? Could the demise of the target species instead facilitate the emergence in the community of another long-suppressed non-indigenous species? We liken this hypothetical phenomenon to a Sisyphean experience: success (eradication) brings failure (another non-indigenous species is inadvertently fostered). So far, this general hypothesis has not been dissected into testable hypotheses, much less experimentally examined. We do have, however, observations from widely-scattered locales that could be explained by a scenario in which a non-indigenous species became more abundant coincident with the control or eradication of another non-indigenous species:

■ The coincident increase of one non-indigenous species with the decline in another was quantified during the biological control of *Hypericum perforatum* (St. John's wort) in California more than 40 years ago. As *H. perforatum* steadily declined in plots through the 1950s through attack by *Chrysolina quadrigemina*, other non-indigenous species became more abundant, including the now abundant invader *Centaurea solstitialis* (Huffaker and Kennett 1959). Tisdale (1976) maintains that a similar emergence of invasive annual grasses (primarily *Bromus tectorum* and *Taeniatherum caput-medusae*) occurred on sites in northern Idaho once dominated by *H. perforatum* as *C. quadrigemina* reduced the biomass of St. John's wort from 1110 kg/ha to less than 100 kg/ha.

■ *Acacia saligna* is being controlled in South Africa with the release of the fungus *Uromycladium tepperianum* (Morris 1999). Although *A. saligna* and *Acacia pycnantha* are not sympatric in their native ranges in Australia (Costermans 1983), they share a common ecological amplitude. And the two wattles can occur in the same habitats in South Africa (J. H. Hoffman, pers. comm.). As a result, *A. pycnantha* could occupy sites once dominated by *A. saligna*. Fortunately, *A. pycnantha* is itself controlled by a hymenopteran, *Trichilogaster* sp. (Dennill et al. 1999; J. H. Hoffman, pers. comm.) so this replacement may be fortuitously blunted.

■ Two of the most destructive invaders in the tropics and subtropics, *Chromolaena odorata* and *Lantana camara* also commonly occupy similar habitats throughout their huge new range, including sites in India (Muniappan *et al.* 1989) and southern Africa (cf. distribution maps in Baars and Neser 1999; Zachariades *et al.* 1999). Consequently, the removal of one through any form of control provides opportunity for increase in cover of the other (J. H. Hoffman, pers. comm.).

■ In South Africa the invasive leguminous tree *Sesbania punicea* has been brought under effective control with release of three phytophagous insects (Hoffman and Moran 1999). Unfortunately, in several of the sites monitored in this control effort, decline of *S. punicea* has coincided with rise in the abundance of *Lantana camara* (J. H. Hoffman, pers. comm.).

Similar events occur amongst invasive aquatic macrophytes. These species often invade communities that have few if any native macrophytes; for example, there is a conspicuous paucity of floating macrophytes in the Australian flora (Jacobs 1999). The low species richness of vascular plants in these communities lends itself to detection of changes in the abundance of a handful of non-indigenous species (Mack 2000). As is apparent with the terrestrial examples cited above, the results suggest (but do not demonstrate) that the removal of one invasive species sparked the rise in abundance of another.

■ Lake Seminole is a large impoundment created in 1957 by damming the Chattahoochee and Flint Rivers in Florida. Soon after the lake's creation, *Eichhornia crassipes* entered, and by 1960, covered approximately 2500 ha of the now-submerged Flint River arm of the lake. Extensive herbicide treatment from 1960 through 1962 devastated water hyacinth; coincident with its decline, another invasive macrophyte, *Alternanthera philoxeroides* soon expanded its coverage. Alligator weed was attacked subsequently by a flea beetle (*Agasicles* sp.) in 1968, and its coverage declined. Apparently as a result, water hyacinth rapidly re-expanded its coverage from a few hectares to 2030 ha in less than a year. By 1975 both these invasive species were still in the Flint River arm of the lake, although in reduced coverage. Simultaneously, the native grass, *Zizaniopsis miliacea*, was becoming prominent along the shoreline (Anon. 1980). Through a combination of herbicide application and insect grazing, the populations of these two invasive aquatic macrophytes have radically fluctuated for over 20 years, each taking up the slack left by the other.

■ Elsewhere in southern Florida, the widespread invader *Hydrilla verticillata* has been controlled in canals with a combination of grass carp and herbicides (R. Stocker, pers. comm.). But its reduction in coverage has coincided with an increase in the equally-unwanted invader *Hygrophila polysperma* (Sutton 1995).

■ The ability of one invasive aquatic macrophyte to increase its role as another is brought under control has long been recognised in the biological control of these species in Australia. Consequently, programmes were undertaken simultaneously for the biological control of the four most prevalent aquatic invaders (*Eichhornia crassipes*, *Pistia stratiodes*, *Salvinia molesta*, and *Hydrilla verticillata*) (Harley 1988).

Even though most of the examples involve species that were the target of a biological control programme, biological control is not in any sense a target of our query here. Similar results arise with selective herbicide treatment, especially in crop fields. Application of broad-leaf herbicides would readily cause competitive release among grasses and other weedy monocots (Aldrich and Kremer 1997). Such an outcome may also be rendered less likely by simultaneous application of multiple or broad-spectrum herbicides that remove potential replacement species among the non-indigenous flora.

CONCLUSIONS

Eradication is almost totally dependent on the early detection of a non-indigenous plant species in a new range, including an island.

In setting priorities within an eradication effort, emphasis should first be placed on destroying the immigrant's nascent foci, rather than the oldest or most-concentrated population(s) in the new range.

Even after the species has putatively been eradicated, continual surveying for it is essential. A lapse in such surveying could allow remnant foci to grow and undo the entire eradication effort. Eradication campaigns for invasive plant species typically need to last for 10 years or more.

Because the prediction of a species' invasive ability is problematic, it is better to err on the side of eradicating a non-indigenous species that later proves to be innocuous, than to withhold eradication until the species' fate is clear (Lonsdale & Smith 2000). Delay greatly reduces the prospects for eradication.

If the goal in eradication is environmental conservation, attention should focus on creating a *zone sanitaire*, rather than on undertaking single species removals, and on establishing the desired native vegetation as rapidly as possible.

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