Chapter 15
Plant Invasions in Protected Areas of Tropical Pacific Islands, with Special Reference to Hawaii

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Abstract Isolated tropical islands are notoriously vulnerable to plant invasions. Serious management for protection of native biodiversity in Hawaii began in the 1970s, arguably at Hawaii Volcanoes National Park. Concerted alien plant management began there in the 1980s and has in a sense become a model for protected areas throughout Hawaii and Pacific Island countries and territories. We review the relative successes of their strategies and touch upon how their experience has been applied elsewhere. Protected areas in Hawaii are fortunate in having relatively good resources for addressing plant invasions, but many invasions remain intractable, and invasions from outside the boundaries continue from a highly globalised society with a penchant for horticultural novelty. There are likely few efforts in most Pacific Islands to combat alien plant invasions in protected areas, but such areas may often have fewer plant invasions as a result of their relative remoteness and/or socio-economic development status. The greatest current needs for protected areas in this region may be for establishment of yet more protected areas, for better resources to combat invasions in Pacific Island countries and territories, for more effective control methods including biological control programme to contain

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intractable species, and for meaningful efforts to address prevention and early detection of potential new invaders.

Keywords Feral ungulates • Grass-fire cycle • Haleakala National Park • Hawaii Volcanoes National Park • National Park of American Samoa • South-eastern Polynesia

15.1 Introduction

The Pacific Ocean is enormous 20,000 km across from Singapore in the west to Panama in the east. It contains approximately 25,000 islands; more than the rest of the world's oceans combined (Gillespie et al. 2008); fewer than 800 of those are considered habitable by humans (Douglas 1969). Its isolated islands are known for high levels of endemism, collectively contributing a very significant portion of the earth's biodiversity (Myers et al. 2000). Its islands are notable for large numbers of endangered species and high rates of extinction. Although less famously documented than predatory and herbivorous alien animals (rats, cats, ungulates and others), plant invasions significantly contribute to undermining biodiversity on islands (e.g. Meyer and Florence 1996; Kueffer et al. 2010a).

An analysis of 25 of the world's 'biodiversity hotspots' by Myers et al. (2000) found that, collectively, the Pacific islands of Polynesia/Micronesia (excluding New Zealand), with slightly over half their flora endemic to the region, have an endemic flora comprising 1.1 % (3,334 species) of the world's flora; the remaining primary vegetation was estimated to cover 10,024 ha (21.8 % of its original extent), with 49 % in protected areas. The large continental island of New Caledonia (part of Melanesia), with 5,200 ha (28 % of the original) of remaining primary vegetation but only 10 % of it in protected areas, was noted as an exceptionally rich hotspot, with an astounding 1,865 endemic plant species.

The inherent vulnerability of Pacific islands to biological invasions was recognised as early as Darwin's observations in the Galapagos archipelago and elsewhere in the 1830s (Darwin 1859), and re-emphasised by Charles Elton in 1958. Pacific island ecosystems are recognised as typically having higher representation of alien species than mainland systems, and the severity of the impact of invasions (i.e. detrimental effects on native species) on islands generally increases with isolation of the islands, though rigorous explanation of these phenomena is elusive (D'Antonio and Dudley 1995). Isolated islands, and particularly their 'protected areas', provide extraordinary living museums of speciation and evolutionary radiation; the limitation in our ability to fully protect such areas from degradation has at least provided dynamic laboratories for better understanding invasions and their interactions with ecological processes (Vitousek et al. 1987).

Here we present and evaluate efforts by managers and researchers to address plant invasions in relatively well-studied protected areas among Pacific Islands and to highlight associated contributions to invasion biology and management.

We focus largely on several notable protected areas where substantial effort to combat invasions has been possible, both to illuminate the various approaches to the problems posed by, and the solutions applied to address the impacts of alien plant invasions on native floras of the Pacific Islands.

15.2 Evolution of the Protected Areas Network and Strategies for Addressing Plant Invasions

Hawaii Volcanoes National Park (NP), island of Hawaii (often called the "Big Island" with its one million ha of terrestrial surface), Hawaii, USA, has the longest history of management of any terrestrial protected area in the tropical Pacific Islands. It was originally established as the largest unit of Hawaii NP in 1916, primarily to protect its volcanic scenery and for geologic study. It currently covers 1,293 km^2, following the Kahuku addition in 2004 (see below), which added 60 % to the park's previous area. This represents about 12.5 % of the area of Hawaii Island or about 7.8 % of the land area of the state of Hawaii; this area is larger than most Pacific Islands. About 80 % of the total protected area is now comprised of fire derived, degraded grasslands dominated by alien species, or sparsely to unvegetated, volcanic terrain (including the upper slopes of 4,169 m a.s.l. Mauna Loa, a shield volcano and the second – to nearby 4,205 m a.s.l. Mauna Kea - highest peak of the tropical Pacific Islands). The remaining area of about 250–300 km^2 contains diverse plant communities. Hawaii Volcanoes NP has been on a trajectory of management for protection of native/endemic biodiversity since the early 1970s when Park managers set out to eliminate the entire population of about 15,000 feral goats (*Capra hircus*) with the aid of fencing. This single event was a 'game changer', initially opposed by almost everyone, including (in 1970–1971) the Director of the U.S. National Park Service (Sellars 1997), as well as by State government agencies in Hawaii.

Herbivory and associated disturbance by feral goats had been rampant in the area of Hawaii Volcanoes NP for nearly two centuries since their introduction to the islands in the 1780s, and more than 70,000 goats had been removed from the Park area since its establishment in 1916, with no long-term population control (Sellars 1997). Conventional wisdom at the time was that any serious effort toward biological conservation in Hawaii was impractical, if not impossible. The impetus for eliminating goat populations came from a strong movement (described in Sellars 1997), mostly within the U.S. National Park Service, to steer the national parks (nationwide) toward biological preservation. The goat eradication effort gradually received increasing and sustained local support, with feral goats largely eliminated in the Park by the end of the 1970s. Arguably, actions and rationale used by the National Park Service in the 1970s at Hawaii Volcanoes NP provided the critical momentum for the rise of 'active' conservation in the State of Hawaii by federal, state, and private entities, with meaningful public support. A generally environment-friendly climate in the USA in the 1970s and the federal Endangered

Species Act of 1973 added impetus to active conservation in Hawaii. By the late 1990s, there was broad buy-in to the concept of biological preservation in Hawaii and over 25 % of the state's land area had been incorporated under varying degrees of protected area management (Loope and Juvik 1998). Nevertheless, the national parks set the standard for what may be possible, and for developing reasonable and thoughtful strategies to try to achieve identified goals.

In Hawaii, alien plant management in protected areas started on a significant scale at Hawaii Volcanoes NP in the 1980s, at which time a sophisticated strategy was developed and articulated. A 1986 symposium entitled "Control of Introduced Plants in Hawaii's Native Ecosystems" was held in conjunction with Hawaii Volcanoes NP's Sixth Conference in Natural Sciences. A book edited by Stone, Smith, and Tunison, "Alien Plant Invasions in Native Ecosystems of Hawaii: Management and Research" (Stone et al. 1992), produced 6 years later, provides remarkable documentation of the development of Hawaii Volcanoes NP's strategy for addressing alien plant issues, as well as of Hawaii's state-wide situation. It was stated that the "The severity of the alien plant problem and the fact that it is so widespread in the Islands make a rigorously organised approach based on relevant information especially necessary. Moreover, development of a variety of approaches to weed control to deal with different situations… are necessary components of weed management programs." (Tunison et al. 1992a).

Tunison et al. (1992a) described the key features of the Hawaii Volcanoes NP alien plant control strategies "to protect native species assemblages." In summary: (i) controlling feral pigs and goats; (ii) excluding fire; (iii) mapping the distribution of important alien plants; (iv) controlling localised alien plants throughout the Park; (v) controlling all disruptive alien plants in Special Ecological Areas (the most diverse and intact areas in the Park); (vi) confining one widespread species, *Cenchrus setaceus* (= *Pennisetum setaceum*, fountain grass), to the area it currently infests; (vii) developing herbicidal control methods for target species; (viii) developing biological controls for some widespread species; (ix) researching the ecology, seed biology, and phenology of important alien plant pest species; (x) educating the public to the importance of alien plant control; and (xi) working with other agencies and groups in alien plant management.

Haleakala NP, island of Maui, Hawaii, was at first a disjunct part of Hawaii NP, established in 1916 and protecting the 3,055 m Haleakala volcano above about 2,000 m elevation. It was designated as Haleakala NP in 1961. The important addition of Kipahulu Valley, a highly pristine rainforest watershed stretching from sea level to above 2,000 m, and other adjacent lands, have resulted in a current Park area of 121 km^2. In the 1980s, this smaller, but more topographically diverse Park, eliminated goats with the aid of fencing and initiated alien plant management, building on the experience of Hawaii Volcanoes NP.

Points that deserve emphasis are the importance and difficulty of creating protected areas in the first place and the challenges most Pacific island countries have in funding management of such areas despite the widely acknowledged need to do so. The addition of the 48,245 ha Kahuku lands to Hawaii Volcanoes NP was enormously important for biodiversity protection in Hawaii, and the Park has been able to devote sufficient resources to take steps to initiate management on those added lands. The National Park of American Samoa (NPSA) was established in 1988 through an innovative and effective concept involving a 50-year lease to the U.S. National Park Service for the park land by the local Samoan village councils (Cox and Elmqvist 1991); the agreements were finalised on the island of Tutuila in 1993, with expansion to other islands in 2002. As such, NPS efforts in both Hawaii and American Samoa can be viewed as useful and distinct models for creating protective areas in other island nations across the Pacific.

South-eastern Polynesia is likely representative of most of the Pacific island regions regarding potential obstacles in establishing and managing protected areas. An important reason why SE Polynesia has few and small protected areas (e.g. less than 2 % of French Polynesia's land area) is that in the main inhabited islands (e.g. Tahiti, Pitcairn, Rarotonga) the land tenure situation is problematic. Most of the land is privately owned by families or clans with multiple beneficiaries that may not be capable of reaching consensus regarding establishment of protected areas. In French Polynesia for instance, the small atolls of Manuae (Scilly) and Motu One (Bellinghausen) in the Society Islands and the islets of Mohotani (Motane), Eiao and Hatutaa (Hatutu) in the Marquesas were more easily declared natural reserves in 1971 because they were uninhabited and public lands. This was also the case for the TeFaaiti Natural Park (750 ha) on Tahiti in 1989, the Vaikivi Natural Park and Reserve (240 ha) on the island of Ua Huka in 1997, and the Temehani Ute Ute plateau (69 ha) on the island of Raiatea in 2010 (Table 15.1). The lack of local capacity (long-term funding support to manage these protected areas, trained conservation managers and scientists, influential nature protection NGOs, etc.), but also the weak local political will and public support are important constraints in SE Polynesian countries and territories (pers. comm. from many people working in the South Pacific, to J.-Y. Meyer). Unfortunately, there are few or no management efforts or programmes to combat invasive alien plants in protected areas in SE Polynesia, although many of these islands provide extraordinary case-studies for illustrating both the impacts of invasive alien plants on biodiversity and cultural assets (e.g. the monumental stone statues in Easter Island or Rapa Nui), and for potential habitat restoration projects. Paradoxically, there are many fencing and weeding projects in French Polynesia recently conducted by local authorities, communities and NGOs in Tahiti, Raiatea (Society Is.) and Rapa Iti (Australs), but primarily in unprotected areas (J.-Y. Meyer, unpub. data).

15.3 How Successful Have Strategies for Alien Plant Management Been in Hawaii Volcanoes NP and Other Protected Areas?

In this section, we report progress in implementation of a generalised version of the 11 items of Tunison's (1992b) visionary strategy 20 years later.

Table 15.1 Characteristics of protected areas in South-eastern Polynesian islands, their dominant invasive plants, and other major ecological threats

Island(s)	Protected area and date of creation	Area (ha)	Main habitat and vegetation type(s)	Dominant invasive plant(s)	Other dominant threats	Source reference
Rapa Nui (Easter Island, Chile)	Rapa Nui National Park (1935), UNESCO World Heritage Site (1995)	6,650	Coastal vegetation and forest, Low and Mid-elevation grassland, savannas and shrubland (0–511 m)	*Cirsium vulgare*, *Crotalaria grahamiana*, *Melinis minutiflora*, *Psidium guajava*	Horses, cattle, human presence and frequentation	Meyer (2008, 2012) and unpub. data
Henderson (Pitcairn Is., UK)	UNESCO World Heritage Site (1988)	3,700	Coastal and raised limestone vegetation and forest (0–30 m)	None identified	Pacific rats	Florence et al. (1995) and Brooke et al. (2004)
Rarotonga (Cook Islands)	Takitumu Conservation Area (1996)	155	Low and Mid-elevation valley forest (50–250 m)	*Ardisia elliptica*, *Cestrum nocturnum*, *Psidium cattleianum*, *Syzygium jambos*, *Lantana camara*, *Spathodea campanulata*, *Merremia peltata*, *Mikania micrantha*, *Cardiospermum halicacabum*	Black and Pacific rats, human frequentation	Meyer (pers. obs. 1997, 2002); E. Saul (pers. com. 2011)
French Polynesia						
Tahiti (Society)	TeFaaiti Natural Park (1989)	750	Mid-elevation valley rainforest (70–500 m) and Mid-elevation plateau (500–700 m), High elev. cloud forest and subalpine vegeta-tion on steep slopes (up to 2110 m)	*Miconia calvescens*, *Rubus rosifolius*, *Spathodea campanulata*, *Tecoma stans*	Black and Pacific rats, human frequentation	Meyer (pers. obs. 1998–2004)

Raiatea (Society)	Temehani Ute Ute Management Area (2010)	69	High-elevation plateau (415–817 m)	*Rhodomyrtus tomentosa*, *Miconia calvescens*, *Psidium cattleianum*, *Cecropia peltata*, *Rubus rosifolius*	Feral pigs, black and Pacific rats	Meyer (1996a)
Manuae (Scilly) and Motu One (Bellinghausen) (Society)	Natural reserves (1971)	1180	Atoll coastal vegetation and forest (0–2 m)	*Stachytarpheta cayennensis*, *Cenchrus echinatus*	Coconut plantations, human presence	Sachet (1983)
Ua Huka (Marquesas)	Vaikivi Natural Park and Natural Reserve (1997)	240	Mid-to high elevation rainforest (880 m)	*Coffea arabica*, *Stachytarpheta cayennensis*, *Psidium guajava*	Feral goats, horses, Pacific rats, human frequentation	Meyer (1996a, 2005)
Eiao (Marquesas)	Natural reserve (1971), Management Area (2004)	3,920	Coastal vegetation and forest, and semi-dry and mesic forest (0–577 m)	*Acacia farnesiana*, *Leucaena leucocephala*, *Annona squamosa*	Feral sheep and pig, black and Pacific rats	Meyer (unpubl. data 2010)
Mohotani (Marquesas, French Polynesia)	Natural Reserve (1971), Management Area (2004)	1,280	Coastal forest, and semi-dry and mesic forest (0–531 m)	*Senna occidentalis*, *Pityrogramma calomelanos*	Feral sheep, Pacific rats	Meyer (1996a, 2000)
Hatutaa (Marquesas)	Natural Reserve (1971), Management Area (2004)	660	Coastal vegetation and forest, and semi-dry forest (0–428 m)	*Passiflora foetida*, *Senna occidentalis*	Pacific rats	Meyer (unpubl. data 2010)
7 atolls: Fakarava, Aratika, Niau, Raraka, Taiaro, Kauehi, Tou (Tuamotu)	UNESCO Biosphere Reserve (Taiaro in 1972 and 6 other atolls in 2006)		Atoll and raised limestone coastal forests (0–6 m)	*Stachytarpheta cayennensis*	Black and Pacific rats, human presence, coconut plantations (fires)	Meyer (unpubl. data 2007)

15.3.1 Feral Ungulates and Implications for Managing Plant Invasions

Hawaii and other isolated islands lack an evolutionary history of ungulate presence, though large flightless birds may have filled similar ecological niches. Ungulates are still absent in wildland areas of many Pacific islands (Merlin and Juvik 1992), but many others have them. Feral goats – as well as deer (*Axis axis*), sheep (*Ovis aries*), mouflon (*Ovis musimon*), and other ungulates – continue to deplete biodiversity outside fenced areas in Hawaii but protected areas have become increasingly fenced (though at great cost) over the past three decades. Feral pigs (*Sus scrofa*) are currently considered primary modifiers of remaining Hawaiian rainforest and have substantial effects on other ecosystems. Although pigs were brought to the Hawaiian Islands by Polynesians roughly a millennium ago, the current severe environmental damage inflicted by pigs apparently began much more recently and seems to have resulted entirely from release of domestic, non-Polynesian genotypes (Diong 1982). Polynesian pigs were much smaller, more docile, and less prone to taking up a feral existence than those introduced in historical times (Tomich 1986). Much of the damage to plants by pigs is direct, involving physical rooting and feeding. Much damage also occurs from invasion of opportunistic plant species, often alien, that contribute to further displacement of native species. Seeds of alien plants are carried on pigs' coats or in their digestive tracts, and they thrive upon germination on the forest floor where pigs have exposed mineral soil (Diong 1982; Medeiros 2004).

Feral pigs have proven much harder to eliminate than goats. Hawaii Volcanoes NP has established 13 pig-free fenced units (often corresponding with SEAs – see below) with a combined area of approximately 16,000 ha plus. At Haleakala NP, feral pigs were eliminated in remote Kipahulu Valley in the late 1980s with fencing and snaring (Anderson and Stone 1993), and the entire Park has since been largely pig-free.

Once aggressive plant invaders have obtained a new foothold in the forest – often as a result of feral ungulate disturbance and dispersal – they spread opportunistically, aided by pigs and alien birds. Removal of pigs stops the mechanical damage and some of the seed dispersal, and is essential for halting direct degradation of biodiversity, but experience has showed that it does not stop plant invasions (e.g. Huenneke and Vitousek 1990; Medeiros 2004). Frequently, after pigs are removed from an area, native species may undergo various degrees of recovery, but plant invasions occupy the sites that had been kept bare by pig-digging. This trend was recently documented by Cole et al. (2012) who measured a substantial increase in cover (51 %) of common native species within a large (1024 ha) fenced exclosure in Hawaii Volcanoes NP from which pigs had been excluded for 16 years; within the same exclosure, the invasive tree *Psidium cattleianum* (strawberry guava) underwent a fivefold increase. Similar effects have been noted with response of alien vegetation to feral goat removal (e.g. Kellner et al. 2011).

Haleakala NP currently faces serious invasive plant problems in the remote Kipahulu Valley, especially with *Clidemia hirta* (Koster's curse), *Hedychium* *gardnerianum* (Kahili Ginger), and *P. cattleianum*. The original expectation was that removal of feral pigs from Kipahulu Valley would not only reduce direct pig impact on native vegetation but would also reduce expansion of these invasions to some degree, ideally facilitating effective management by mechanical/chemical means (Loope et al. 1992); in retrospect, removal of pigs has allowed substantial recovery of native vegetation, but *H. gardnerianum* and *C. hirta* have also expanded substantially even with concerted control effort and are currently posing severe problems, with biological control urgently needed (Medeiros 2004; Arthur Medeiros, U.S. Geological Survey, pers. comm.).

Feral pigs have recently become a serious problem in the National Park of American Samoa (NPSA), creating disturbance to native vegetation; substantial control effort has been made, but the problem persists (Tavita Togia, NPSA, pers. comm.). Ungulates are also present and create serious disturbance, facilitating alien plant invasions in some protected areas of SE Polynesia (Meyer, pers. obs.). Feral sheep infest Eiao and Mohotani, as do feral goats and horses (*Equus caballus*) in the Vaikivi Natural Park of Ua Huka in the Marquesas; feral pigs thrive on the Temehani Ute Ute plateau on Raiatea in the Society Islands, threatening rare endemic plants (Jacq and Meyer 2012); and cattle (*Bos taurus*) and horses are causing forest destruction and facilitating invasion by light-demanding weeds on Rapa Nui (Meyer, pers. obs.). The reserves on the dry, uninhabited islands of Eiao (with long-standing serious ungulate problems, Fig. 15.1) and Hatutaa (without ungulates) in the Marquesas provide a dramatic comparison of the persistence of native vegetation where ungulates are absent and the degradation of native vegetation with replacement by alien plant species in the presence of ungulates (Merlin and Juvik 1992; Meyer, pers. obs.).

15.3.2 Fire Management and the Intractable Grass-Fire Cycle at Hawaii Volcanoes NP

From the inception of Hawaii Volcanoes NP in 1916 until the 1960s, fire had a small and infrequent footprint across park lands. Although ignition sources were present (e.g. lava flows, lightning, human activity), vegetation was either too discontinuous or of too high a moisture content to provide adequate fuels to burn. In the subsequent decades following 1960, however, fire frequency increased tenfold and the extent of fires increased an astounding 60-fold (Tunison et al. 1995) in the extensive mid-elevation seasonal fire management unit (also known as the seasonal submontane zone). What caused the fire regime to change so dramatically? It was the local establishment and proliferation during the 1960s of alien C_4 grass species such as *Andropogon virginicus* (broomsedge) from the south-eastern US, *Schizachyrium condensatum* (beardgrass) from South America, and the spread of the *Melinis minutiflora* (African molasses grass) during the 1980s. Each of these species exhibit attributes that make them very prone to burning and very adept at re-establishing following fire; all create a continuous fuel bed,

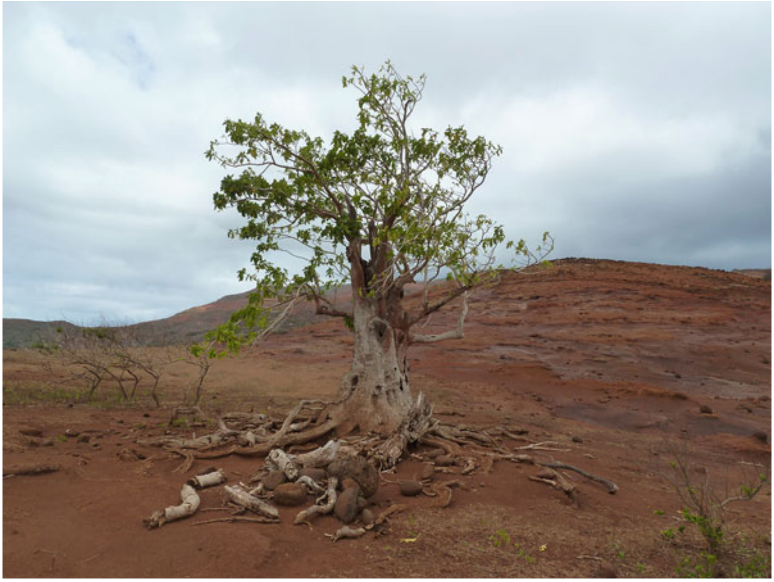

Fig. 15.1 Isolated native tree *Pisonia grandis*, in severely eroded landscape overgrazed by feral sheep, island of Eiao, Marquesas Islands, French Polynesia (Photo J-Y Meyer, November 2010)

maintain high dead-to-live biomass ratios throughout the year, and exhibit high extinction moisture content, allowing them to burn at high relative humidity (Hughes et al. 1991). These alien grasses readily invaded the interstices of what had been woodlands and shrublands dominated by native woody plants such as *Metrosideros polymorpha* (‘Ohi’a lehua) and *Leptecophylla tameiameiae* (Puhatikiei) among others. Once invaded by alien grass species, these systems became exceedingly prone to fire, and when they did almost inevitably burn, the grasses rapidly recolonised the burned areas (Hughes et al. 1991); in general alien grass cover increased 33 %, and grass biomass increased 2- to 3-fold following fire (Tunison et al. 1995; D’Antonio et al. 2000). *Melinis minutiflora* in particular increased dramatically from pre-fire cover and biomass values (Hughes et al. 1991).

In stark contrast, an average of 55 % of *M*. *polymorpha* individuals suffered mortality following fire, and this is likely an inflated survivorship given that many of the surviving trees were located on rocky outcrops and thus experienced little in the way of fire effects (D’Antonio et al. 2000). Post-fire *M*. *polymorpha* seedling recruitment and establishment is non-existent (Tunison et al. 1995), and most of the common native shrub species were sharply reduced with respect to both cover and stem density (Fig. 15.2; Hughes et al. 1991) immediately following fire as well as after two decades of post-fire succession (D’Antonio et al. 2011). Successive fires lead to increased dominance of grasses and further diminution of native woody and herbaceous species populations. Collectively, alien grasses now dominate extensive areas of dry and seasonally dry habitats in Hawaii. They have been demonstrated to

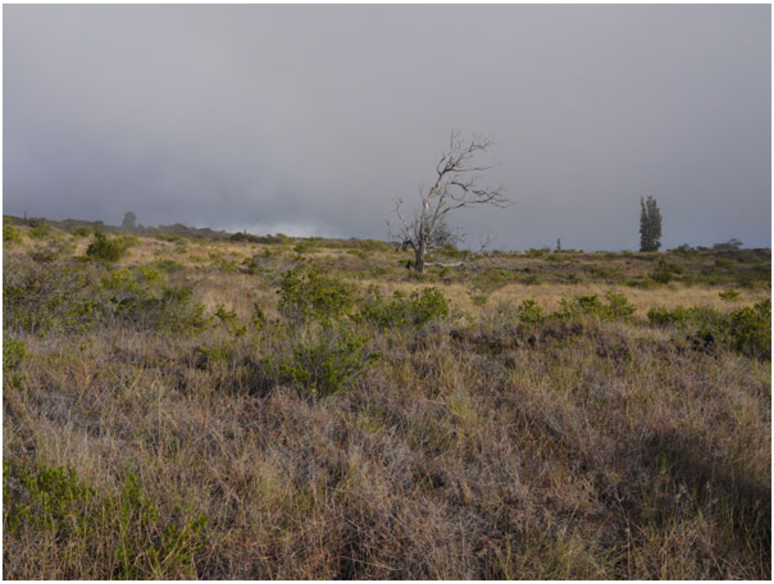

Fig. 15.2 Fire-degraded grass-shrubland, elevation 900 m, Hawaii Volcanoes National Park, Hawaii. Shrubs in foreground are native *Dodonaea viscosa*, surrounded by a matrix of alien C4 grass species. Note large, dead *Metrosideros polymorpha* tree in middle ground (Photo RF Hughes, January 2013)

effectively compete with native species (D'Antonio et al. 1998) and alter both light regimes and soil nutrient dynamics (Hughes and Vitousek 1993; D'Antonio and Mack 2006). A more recent study (D'Antonio et al. 2011) documenting long-term patterns of post-fire succession in the absence of subsequent fire events demonstrated that replacement of native woody species by alien grasses, even in the absence of fire, persists over the long-term; results indicated that in spite of multiple 'fire-free' decades of post-fire succession, grasses maintained their dominance and native species failed to recover. As such, fire suppression by itself is inadequate to restore these systems to any sort of a native-dominated state.

As a consequence, the Hawaii Volcanoes management rule regarding wildfire – particularly in the mid-elevation seasonal fire management unit – has been one of active and concerted fire suppression (Hawaii Volcanoes National Park 2005). This is primarily in order to limit disturbance to, and mortality of, non-fire adapted native species and limit further proliferation of pyrophytic alien grasses. An exception to blanket suppression is in the dry coastal lowlands where fire effects studies and prescribed burns have demonstrated the positive effect of fire on the native grass *Heteropogon contortus* (Spear Grass; Tunison et al. 1994; D'Antonio et al. 2000). In these areas fire may be allowed to occur with minimal interference as a way to enhance cover of this native grass. Recently, large-scale efforts have been undertaken to plant a suite of native species that exhibit fire tolerant characteristics into burned areas. These planting efforts have met with success in terms of the establishment and survival of meaningful population sizes, and it is hoped that these relatively fire-tolerant native plant populations will persist and sustain themselves in the event that such areas experience successive fires in the future (Rhonda Loh, Hawaii Volcanoes NP, pers. comm.).

15.3.3 Mapping of Important Alien Plant Species

Adequate knowledge regarding the abundance, extent (i.e. hectares invaded) and distribution of invasive species is critical for developing effective control strategies and establishing workload requirements. Hawaii Volcanoes NP undertook a systematic programme to map the distribution of 38 widespread alien plant species in 1983–1985 (Tunison et al. 1992b). Results were successful in determining locations of many untreated populations, helping to assess feasibility of possible local eradication, shifted priorities, and showing that eight species were too widespread for control with the resources available – so that efforts for these species were shifted from a parkwide emphasis to control in selected areas with high biological value. Mapping and monitoring of an expanding suite of alien plant species continues at Hawaii Volcanoes NP. An important recent report (Benitez et al. 2012) reviews the expanding survey/mapping work and control history since 2000, reporting on results for 134 species surveyed by foot, vehicle and helicopter; 33 of those are widespread species and beyond park-wide control.

15.3.4 Park-Wide Control and Eradication of Localised but Potentially Problematic Alien Plant Species

In the 1980s, Hawaii Volcanoes NP adopted a strategy of controlling certain localised alien plant species on a park-wide basis while controlling widespread alien species in Special Ecological Areas (SEAs, see below). The purpose of the former effort has been to prevent the spread of potentially disruptive alien species while they are still manageable. Of the 41 species that were initially targeted, mapped, treated with appropriate herbicides and monitored, control of 21 was considered highly effective, with partial control for 17 additional species; three species were recalcitrant to control (Tunison and Zimmer 1992). By 2004, at least 15 of the initial species were considered eradicated and workloads reduced for most of the others (Timothy Tunison, Hawaii Volcanoes NP (retired), pers. comm.). The local eradication strategy has been extended opportunistically to prevent encroachment of new invaders, especially high impact species such as *Falcataria moluccana* (batai wood). A current analysis (Benitez et al. 2012) details progress/setbacks for 134 alien species (101 of them localised), including 16 species newly established in the past decade; the most problematic new species for attempted containment may be the shrub *C. hirta*, first detected in 2003, and *Cyathea* (= *Sphaeropteris*) *cooperi* (Australian tree fern), first detected in 2000. The Park's alien plant programme (including eradication/control of localised species and control in SEAs) has expanded significantly in scope and complexity over the past 3 decades. Since the early 1980s, the annual number of worker days spent in the field searching for and removing weeds has increased from $<$200 to $>$500 by the early 1990s and exceeds 1,200 worker days currently (Benitez et al. 2012).

15.3.5 Controlling All Disruptive Alien Plant Species in Selected High-Value Areas

In Hawaii Volcanoes NP, management units called Special Ecological Areas (SEAs) (Tunison and Stone 1992) were first established in 1985 to control 20+ highly disruptive invasive plant species recognised as too widespread for park-wide eradication to be feasible. SEAs are prioritised for intensive weed management based on their (i) ecological representativeness or rarity, (ii) manageability (accessible and with high recovery potential for native species), (iii) species diversity and rare species, and (iv) value for research and interpretation. Control methods varied from manual uprooting to chemical treatments depending on species. Typically, initial search and knockdown of weeds (knockdown phase) by control crews is followed by subsequent revisits (normally at 1–5 year intervals) as needed to keep infestations at low or manageable levels (maintenance phase) in SEAs.

The SEA concept has proved remarkably effective and flexible to date – whereas initial weed control may focus on only a few prime areas, the number and size of units can be expanded with time as opportunities become available. It has provided Hawaii Volcanoes NP with the ability to protect key biodiversity sites, even when alien plant species are uncontrollable on a large scale. The SEA programme started in 1985 with six SEAs and a total of 5,000 ha; by 2007, it had been expanded to 27 SEAs and 27,500 ha. Meanwhile, costs per ha declined from roughly \$28/ha to \$8.15/ha (Tunison and Stone 1992; Loh and Tunison 2009).

While we are not aware that any other protected area in the Pacific islands has formally adopted an SEA approach, many areas focus invasive plant control effort in areas where rare/endangered species are being threatened by plant invasions.

15.3.6 Local Eradication or Containment of Certain High-Impact Species

Fountain grass has for decades been considered one of the most disruptive alien plant species in Hawaii. It is believed capable of invading all Hawaii Volcanoes NP's plant communities, except closed rainforest, from sea level to 2,500 m elevation. It increases fuel loadings, thus increasing fire potential and most notably invades and reaches high densities on largely barren lava flows, which normally are pristine sites with few aliens, potentially making extensive new areas vulnerable to fire (Tunison 1992a). Hawaii Volcanoes NP's strategy in general has targeted problematic widespread species for biological control (believed to be unlikely for grasses) and/or for conventional control in Special Ecological Areas. However, a single exception was made, beginning in 1976 and intensified in 1983, for an 8,000 ha infestation of fountain grass, largely localised in the south-western corner of the Park, with outliers, especially along roads. The strategy for fountain grass has involved controlling all outlying populations, scouting the areas between these populations by helicopter, and controlling the periphery of the main infestation; the grass's pattern of spread suggested that such strategy could succeed (Tunison 1992a). Workloads were initially large and expensive but have gradually declined substantially in effort and cost over more than three decades of sustained effort (Benitez et al. 2012). At this point in time, the considerable investment to contain fountain grass seems justifiable and sustainable.

The only other case of such an ambitious strategy being applied in a Pacific island protected area may be the example of *F. moluccana* in the National Park of American Samoa (Case study 1).

15.3.6.1 Case Study 1: *Falcataria moluccana* at the National Park of American Samoa

Falcataria moluccana (= *Paraserianthes falcataria*, *Albizia falcataria*) is a very large, nitrogen-fixing tree of the legume family (Wagner et al. 1999). As an invasive species, it is daunting as the fastest growing tree species in the world, capable of 2.5 cm of growth per day (Walters 1971; Footman 2001). Individuals reach reproductive maturity by the age of four and subsequently produce copious amounts of wind dispersed seed (Parrota 1990). Canopies of single mature trees extend over 0.5 ha, and canopies of multiple trees commonly coalesce across multiple hectares up to square kilometres (Hughes and Denslow 2005).

Although valued by some in the Pacific, *F. moluccana* has become invasive in forests and developed landscapes across many Pacific islands. An archetypical early successional (i.e. pioneer) species, *F. moluccana* is generally found in mesic to wet forest environments and favours open, high light environments such as disturbed areas; its capacity to readily acquire nitrogen via its symbiotic association with *Rhizobium* bacteria makes it able to colonise even very young, highly N-limited lava flows such as those found on Hawaii Island (Hughes and Denslow 2005).

Previous research on the impacts of *F. moluccana* invasion on native Hawaiian forests demonstrated that wherever it invades, it utterly transforms the entire ecosystem by substantially increasing inputs of nitrogen, facilitating invasion by other weeds, while simultaneously suppressing native species. Hughes and

Denslow (2005) described the impacts of *F. moluccana* invasion on some of the last intact remnants of native wet lowland forest ecosystems undergoing primary succession in Hawaii. Nitrogen inputs via litterfall were 55 times greater in *F. moluccana* stands compared to native-dominated forests (Hughes and Denslow 2005), and at 240 kg N ha^{-1} year^{-1}, were commensurate to typical fertilizer N inputs of industrialised corn cropping systems of the US Midwest (Jaynes et al. 2001). Changes in nutrient status coincided with dramatic compositional and structural changes as well; *Falcataria moluccana* facilitated an explosive increase in densities of understory alien plant species, particularly *Psidium cattleianum*. In contrast, native species, especially the keystone over-story tree *Metrosideros polymorpha* suffered widespread mortality to the point of effective elimination from forests that they formerly dominated. Even where *F. moluccana* populations are killed and/or removed, native species are typically challenged to grow quickly enough to outpace the rapid recruitment of abundant *F. moluccana* seedlings that promptly germinate in response to increased understory light availability. Based on these findings, Hughes and Denslow (2005) concluded that the continued existence of native-dominated lowland wet forests in Hawaii largely will be determined by the future distribution of *F. moluccana*. As such, detection and control of individuals or small numbers of *F. moluccana* has become the default approach in protected areas such as Hawaii Volcanoes NP.

In American Samoa *F. moluccana* has invaded large areas of the forests of the National Park (NPSA) and neighbouring areas on Tutuila Island. The species likely was first introduced to Samoa on the island of Upolu perhaps as early as the 1830s and is thought to have spread to Tutuila Island in the early 1900s. By the 1980s *F. moluccana* was noted as locally common within NPSA boundaries (Whistler 1980, 1994) and by 2000, approximately 35 % of Tutuila Island (6,725 ha) including much of NPSA, was infested with *F. moluccana*. This prompted NPSA efforts to begin aggressive measures to control this species (Fig. 15.3; Hughes et al. 2012). Research results indicate that *F. moluccana* displaces native Samoan trees; although aboveground biomass of intact native forests did not differ from those invaded by *F. moluccana*, greater than 60 % of the biomass of invaded forest plots was accounted for by *F. moluccana*, and biomass of native species was significantly greater in intact native forests. Following removal of *F. moluccana* (i.e. killing of mature individuals), a number of native Samoan trees grew rapidly, filling the resulting light gaps, achieving secondary succession without a reinvasion of *F. moluccana*. The presence of successional native tree species appeared to be the most important reason why *F. moluccana* removal is likely a successful management strategy. Once *F. moluccana* is removed, native tree species grow rapidly, exploiting the legacy of increased available soil N – left from *F. moluccana* litter inputs – and available sunlight. In addition, recruitment by shade intolerant *F. moluccana* seedlings was severely constrained to the point of being non-existent, likely a result of the shade cast by re-establishing native trees in management areas (Hughes et al. 2012). Like many Pacific islands, American Samoa commonly experiences large-scale, cataclysmic disturbances from cyclones (Mueller-Dombois and Fosberg 1998) as

Fig. 15.3 The large, non-native, N-fixing tree *Falcataria moluccana* has spread over extensive areas in and around the National Park of American Samoa. The Park has responded with a remarkably successful campaign of killing trees in place and relying on reproduction and growth of pioneer-type native tree species to quickly fill resulting light gaps. Photo from June 2006, 8 months after girdling of the *F. moluccana* trees (Photo Tavita Togia, NPSA)

well as more frequent but less cataclysmic disturbances in the form of cyclone 'near misses', tropical storms, and tropical depressions. Collectively these create what has been termed a 'chronic disturbance' regime (Webb et al. 2011), which has played a potent evolutionary role in shaping the composition, structure, and function of Samoa's native forests (Webb et al. 2006). Indeed, nearly 40 % of the common native trees in forests of American Samoa could be classified as successional (i.e. regenerating readily in disturbed forest) (Hughes et al. 2012). This is a critical evolutionary feature for the forest species of American Samoa, and one that makes large-scale control of *F. moluccana* feasible. Further, this scenario stands in stark contrast to results from experimental removal of the alien, N2-fixing tree, *Morella faya* (faya tree) from forests of Hawaii Volcanoes NP, where successful reestablishment and recovery of native forest species following control of the *M. faya* is much less certain given the presence of highly competitive alien species and the relatively slow-growing character of the native species – species that have not evolved in such a frequent storm disturbance environment (Loh and Daehler 2008).

Yet, if native Samoan tree species are so well adapted to small, forest gap forming disturbances, as well as large-scale disturbances such as cyclones, how did *F. moluccana* attain recent dominance in forest stands in the first place? Moreover, and what is to keep it from returning to dominance following disturbance in the future? Answers can be found in the growth characteristics of

F. moluccana. Because this species becomes very tall, very quickly (Walters 1971; Parrota 1990), and its seedlings accompany recruitment of native species, *F. moluccana* will likely outpace other species in the race to canopy dominance. In addition, since mature *F. moluccana* individuals attain heights well above those exhibited by most of the native Samoan tree species (Whistler 2004), *F. moluccana* will maintain overstory canopy dominance. As long as cyclones occur at sufficient frequencies, *F. moluccana* populations will likely persist and expand in the absence of on-going management practices. However, removal of mature *F. moluccana* individuals, re-establishment of native Samoan tree species, and exhaustion of the *F. moluccana* seed bank prior to subsequent large-scale disturbances may suffice to break the cycle of *F. moluccana* establishment and proliferation.

15.3.7 Develop Optimally Effective Herbicidal Control Methods

Any protected area in the Pacific that seriously addresses plant invasions must use herbicides effectively as part of the overall strategy. Hawaii Volcanoes NP conducted formal experiments in the 1980s to develop a safe, effective and efficient arsenal of treatments appropriate for a large number of target species for which there were no standard treatments (e.g. Santos et al. 1992). In some cases standard methods may not be sanctioned by National Park Service or other protected area guidelines. Over the past three decades, standard treatments have become increasingly available (e.g. Langeland and Stocker 1997; Motooka et al. 2002). Common active ingredients used in natural area weed management have included 2,4-D, triclopyr, glyphosate, metsulfuron methyl, imazapic and imazapyr; picloram and hexazinone were proven to be effective in Hawaii as well, but have less utility now due to restrictions in their use (James Leary, University of Hawaii, pers. comm.). The new herbicide active ingredients aminopyralid and aminocyclopyrachlor are proving to be highly effective on many target weed species in Hawaii, particularly those in the Fabaceae or legume family (J. Leary, pers. comm.). Individual plant treatment techniques include directed foliar applications, basal bark applications and basal injections. Additionally, on-going research in Hawaii has identified effective injection techniques for 16 invasive woody species so far (e.g. *F. moluccana*), where effective doses are less than 0.5 g active ingredient for large mature specimens (J. Leary, unpubl. data). Furthermore, an herbicide injection is a clean, safe, and efficient technique for delivering very small aliquots of concentrated formulations directly to the target. Besides proven target efficacy, this provides a practical use pattern in remote natural areas where a total payload that weighs a fraction of one kilogram is enough to treat hundreds of individuals in an all-day effort.

Benitez et al. (2012) summarise current herbicide treatments used for targeted alien plant species at Hawaii Volcanoes NP.

15.3.8 Facilitate Biological Control for Otherwise Intractable Species

Biological control was recognised in the early 1980s as a potentially major component of effort to address the most severe plant invasions in national parks and other protected areas in Hawaii. Hawaii already had a long history (dating back to about 1900) of biological control to assist against insect pests to protect agriculture, and some plants had been successful targets. The National Park Service (NPS) entered an agreement in the early 1980s with the U.S. Forest Service, the Hawaii Departments of Land and Natural Resources and of Agriculture, and University of Hawaii to "intensify biological control efforts on forest pests in the State" (Tunison 1992b). A quarantine (containment) facility was constructed at 1,200 m elevation at Hawaii Volcanoes NP for plant biocontrol using insects; the facility was completed and became operational in 1984 (Markin et al. 1992). Hawaii Department of Agriculture (HDOA) had the only other biocontrol containment facility in the state in Honolulu, which was capable of accommodating insects and fungal agents for testing (Markin et al. 1992).

Some details of progress (and lack thereof) with biocontrol are given for *Morella* (= *Myrica*) *faya* and *Hedychium gardnerianum* in Case study 2 and 3, respectively. The challenges of conducting a biocontrol programme in the public arena are raised below. Experience with *Clidemia hirta*, another high-priority target as one of Hawaii's most aggressive invaders, has been particularly discouraging; of 17 agents tested for host specificity in Hawaii, six arthropods and one fungal agent were field released: a thrips (1953), a fungus (1986), a beetle (1988), and four moths (one in 1970, three in 1995) (Conant 2002). Five established, but none were truly successful. Biotic interference (by alien ants and/or parasitoids) has been demonstrated for the thrips and strongly suspected for the moths (Conant 2002). DeWalt et al. (2004) reported observing promising potential biocontrol agents in *C. hirta*'s native range in Costa Rica. A nematode (*Ditylenchus gallaeformans*), under consideration for biocontrol of closely-related *M. calvescens* (miconia), may be the most promising agent for *C. hirta* in the long run (Johnson 2010, Tracy Johnson, U.S. Forest Service, pers. comm.); Hawaii has no native melastomes.

Experience with the recent release of an extremely important biocontrol agent for strawberry guava provided meaningful insights into the importance (and limitations) of public outreach. After 15 years of exploration and testing in Brazil (Wikler and Smith 2002), *Tectococcus ovatus* (Homoptera: Eriococcidae), a leaf-galling scale insect, was brought to Hawaii for intensive experimental testing to ensure its safety as a biocontrol agent; Tracy Johnson of the U.S. Forest Service conducted Hawaii-specific laboratory testing in the Hawaii Volcanoes NP biocontrol facility, beginning in 1999. In 2005, a release permit application was issued. The Hawaii Board of Agriculture held public meetings in 2005–2007, and federal and state permits for release were obtained in early 2008. However, Johnson then applied for additional permission for release of *T. ovatus* on state land (in order to facilitate intensive post-release monitoring), which triggered the need for a state

Environmental Assessment (Warner and Kinslow 2011). After an unexpected additional 3 ½ years of contentious, ‘high-profile’ interaction, primarily with a local (Hawaii-island) critic and his supporters (described in some detail by Warner and Kinslow 2011), *T. ovatus* was finally released on state land near Hawaii Volcanoes NP in December 2011. Whereas there was organised opposition on Hawaii island, public opinion on Maui was generally positive toward the release.

Although biological control of plant invaders has proved very effective in some countries of the world (see Van Driesche and Center 2014, this volume for a synthesis), it has had few successes for helping protected areas in Hawaii during nearly three decades of good intention and very significant effort. In general, after initial enthusiasm for biocontrol was tarnished by some early failures and dead ends, there has been an apparent tendency among key agencies to fund on-the-ground mechanical/chemical plant control more generously than biological control. The two existing containment facilities are far less than adequate to accommodate state-wide needs, especially given that HDOA biocontrol efforts are primarily targeted at agricultural pests. The regulatory process for biological control has become more rigorous, and some would say unnecessarily slow, perhaps especially for Hawaii (Messing and Wright 2006). Most recognise the importance of much increased efforts to monitor the fate of biocontrol releases (e.g. Denslow and D’Antonio 2005). Biological control continues to have much promise for addressing Hawaii’s most serious invasive plant issues in protected areas and elsewhere, but an injection of major resources (not easily obtained, especially in harsh economic times) will be required for sustained success.

15.3.8.1 Case Study 2: *Morella faya*

Morella (= *Myrica*) *faya* is a small evergreen tree (5–10 m tall), an actinorrhizal nitrogen-fixer, native to the Canary Islands, Azores, and Madeira in the North Atlantic. It was brought to Hawaii by Portuguese immigrants in the 1880s, probably as an ornamental, and later was planted on multiple islands by the Territorial Department of Forestry for watershed reclamation in the 1920s and 1930s. Its aggressive invasiveness was recognised by the 1930s, by which time it was present on five of the six major Hawaii islands. The worst invasion is in Hawaii Volcanoes NP, where it colonises early successional open-canopied forests, on young volcanic substrates, achieving drastic alteration of N-levels and eventually forming nearly monospecific, closed canopy stands (Vitousek and Walker 1989). Feral pigs may have assisted its spread (before localised pig eradication), but ample bird-dispersal is highly efficient (Woodward et al. 1990). *Morella faya* increased from one tree found in 1961 to an estimated 12,200 ha by 1985, to 15,800 ha by 1992, and 30,495 ha at present, with about half of that total comprised of dense infestations (Benitez et al. 2012). Analysis of airborne imaging spectroscopy, focusing on a 1,360 ha forested area (originally endemic *Metrosideros polymorpha*) in the heart of Hawaii Volcanoes NP at 1200 m elevation, suggests that about 28 % of the landscape is dominated by *M. faya*, with an additional 23 % undergoing transformation as *M. faya* grows into the canopy (Asner and Vitousek 2005). In these sites, the rate of diameter growth is 15-fold more rapid than that of *M. polymorpha*; the rate of nitrogen-fixation in dense *M. faya* stands was measured at 18 kg N ha ^{1} year ^{1}, resulting in soil N-levels of about five times that of uninvaded *M. polymorpha* stands (Vitousek and Walker 1989).

Morella faya is apparently seriously invasive only in Hawaii, though it has been introduced to Australia, New Zealand and elsewhere. It has been a target of biological control, one of the first under Hawaii's interagency biocontrol agreement. An expedition to the *M. faya* home range in 1984 was relatively unsuccessful in finding promising agents. However, two moth species were brought back to Hawaii and tested, and one was released but found ineffective (Smith 2002).

An invasive leafhopper (*Sophonia rufofascia*) from Asia that was first recorded in Hawaii in 1987 has a very broad host range and attacks both *M. faya* and (to a lesser extent) *M. polymorpha*. Negative effects of the leafhopper on *M. polymorpha* are damaging and have been shown to be more severe where there is adjacent *M. faya* (Lenz and Taylor 2001); *M. faya* has shown significant mortality. Hawaii Volcanoes NP has explored options for optimal control of *M. faya* and concluded that girdling *M. faya* trees and leaving the dead trees in place is the most promising methodology for restoring native species (e.g. Loh and Daehler 2007).

15.3.8.2 Case Study 3: *Hedychium gardnerianum*

Hedychium gardnerianum (Fig. 15.4) is a serious invader of rainforests at both Hawaii Volcanoes and Haleakala NPs as well as elsewhere in the Hawaiian Islands. Its case is unusual in that the first known collection in the state was made in the Hawaii Volcanoes NP employees' housing area in 1943; it apparently became well-established during the 1960s–1980s (Linda Pratt, U.S. Geological Survey, pers. comm.). It is a large herb (1–3 m tall) that occupies about 3,000 ha (Benitez et al. 2012) at 750–1,300 m elevation in Hawaii Volcanoes NP and tends to establish a monospecific understory, gradually smothering native understory species and generally preventing native tree seedling recruitment (Minden et al. 2010a, b). The invasive tree *Psidium cattleianum* is the only species that appears able to reproduce successfully through a dense thicket of *H. gardnerianum* (Minden et al. 2010b). Experience at Haleakala NP (where it was discovered in the mid-1980s and is rapidly spreading in spite of control efforts) shows that *H. gardnerianum* is fully capable of smothering rare Lobeliaceae (Stephen Anderson, Haleakala NP, pers. comm.). Analysis of airborne imaging spectroscopy, combined with ground-based analyses, indicated that *M. polymorpha* forest over-stories had significantly lower leaf N concentrations in areas with *H. gardnerianum* understory – likely a competitive effect (Asner and Vitousek 2005).

Fig. 15.4 *Hedychium gardnerianum*, a large herbaceous ginger from the Himalayan region, is becoming an increasingly serious invader of Hawaiian middle and high-elevation rain forests, capable of smothering and preventing reproduction of most other species in invaded stands (Photo RF Hughes, January 2013)

Hedychium gardnerianum, native to high elevations of Nepal and India, is a serious invader in many rainforest areas worldwide, especially on islands (Kueffer et al. 2010a). Exploration and preliminary testing for biocontrol agents is underway through a multi-country collaborative project with CAB International (Djeddour and Shaw 2011).

15.3.9 Support/Encourage Research on Biology and Control Strategies of Alien Plant Species

Scientific research regarding the ecology, impact and control of alien plant species is critical for intelligent, efficient, and effective management of protected areas. It can be instrumental in helping to understand and document the nature and relative threat posed by respective alien species; in its absence, managers risk basing management decisions on inaccurate conventional wisdom, assumptions, and hearsay. A relatively rich collection of scientific literature has helped inform and prioritise management efforts in Hawaii Volcanoes NP. Vitousek and Walker (1989) and Asner and Vitousek (2005) documented ecosystem-scale influences of *Morella faya* invasions, and Loh and Daehler (2007, 2008) addressed successional trajectories following *M. faya* removal. Huenneke and Vitousek (1990) documented mechanisms of *Psidium cattleianum* invasion and their impact for native forest management. Asner and Vitousek (2005) and Minden et al. (2010a, b) elucidated the community and ecosystem impacts of *Hedychium gardnerianum* invasion into Hawaii's native forest based on remote sensing and plot-level investigations. La Rosa (1992) described the characteristics of the alien vine, *Passiflora tarminiana* (= *P. mollissima*, banana passion flower/banana poker) as well as the mechanisms of its invasion into mesic to wet native forests. The grass fire cycle has been the topic of numerous publications that have helped managers determine the most appropriate approaches to protect native woodland/shrubland ecosystems, initially by Hughes et al (1991) and most recently D'Antonio et al. (2011). In Haleakala NP, Medeiros (2004) addressed patterns and mechanisms of *P. cattleianum*, *H. gardnerianum*, and *Clidemia hirta* invasion. Diong (1982) addressed the synergy of feral pigs with increased dominance of *P. cattleianum* in Haleakala NP. Elsewhere, research in Tahiti (Meyer 1996a, 2010 and many other papers) has been very important in alerting Hawaii and for developing strategies against *M. calvescens*. Research by Hughes and Denslow (2005) concerning Hawaii and Hughes et al. (2012) concerning American Samoa – discussed in Case study 1 – illustrates particularly well how the importance of understanding certain plant invasions in depth can illuminate strategies for managing those invasions when circumstances are right.

Despite a rich literature documenting the threats posed by invasive alien species, recent research has addressed the supposition that alien species may prove unavoidable components of island ecosystems and should be "embraced" where appropriate (Hobbs et al. 2006, 2009). Lugo (2004) and Kueffer et al. (2010b) demonstrated in forests of Puerto Rico and the Seychelles, respectively, the potential usefulness of alien trees for providing suitable conditions for native plant recruitment following major anthropogenic disturbances such as deforestation associated with mining or conversion to agriculture. Clearly, any potential benefits incurred from alien plant species will be determined by the particular characteristics of the species as well as the characteristics of the ecosystems they happen to inhabit, and costs and/or benefits should be evaluated on a case by case basis with an underpinning of ecological understanding of that alien plant/ecosystem interaction. It appears very unlikely that such aggressive invaders in the Pacific as *F. moluccana*, *H. gardnerianum*, and *M. calvescens* will ever prove beneficial for native biodiversity, but keeping open minds to conservation possibilities for 'novel ecosystems' may be warranted.

15.3.10 Education Within Agencies and Outreach to the Public

Resource managers at Hawaii Volcanoes and Haleakala NPs have tried to thoroughly educate co-workers in the National Park Service so that they will continue to support alien plant control programmes. Alien plant problems are a minor theme of some park interpretive programmes that emphasise geological, cultural, and other biological messages to the public. The Resources Management Division at Hawaii Volcanoes NP has an active volunteer programme but has been more successful at generating interest in assignments with wildlife (especially sea turtles) than with alien plant control activities (Tunison 1992b, R. Loh, pers. comm.). On the whole, conservation groups and scientists are well informed about alien plant problems, but it has been difficult to interest the local public about damage caused by alien plants and the need to control them. One reason may be that Hawaii Island (although it has native ecosystems more accessible to the public than any other island) has had a largely rural and small-town culture with little interest in conservation (Tunison 1992b). Fortunately, that situation has been improving as schools at every level have begun including Hawaiian natural history topics in their syllabi. There is much greater awareness of invasive species issues for communities in and around the Park than in more distant parts of Hawaii Island (R. Loh, pers. comm.). Establishment and progress of Watershed Partnerships and the Big Island Invasive Species Committee over the past two decades have also stimulated considerable interest. On the island of Maui, public support for combating alien plants may be more developed, at least partly because of perceived need for watershed protection. (Hawaii Island has relatively few watersheds in relation to its size, because of the relatively gentle topography and porous substrates.) Maui County funding has been very important for the campaign to combat *M. calvescens* on East Maui, for example, where the species threatens not only biodiversity but ecosystem services in watersheds. At a state-wide level, the population is largely urban (mostly on Oahu island); many citizens of the state may have little contact with or knowledge of Hawaii's endemic biota.

The control of *F. moluccana* in the NPSA of Tutuila Island (Case study 1) provides an instructive example of how to effectively involve the public in a meaningful manner that both builds substantive support and accomplishes stated objectives to protect native biodiversity. From the outset, managers addressed the need for action, for example, control of *F. moluccana* populations in national park boundaries - with the surrounding village chiefs who are the relevant bodies of authority (T. Togia, NPSA, pers. comm.). Second, widespread public knowledge of, and support for, the control effort was cultivated through the use of local media outlets on a consistent basis. Lastly, and perhaps most importantly, the majority of funds acquired for *F. moluccana* control were dedicated to the employment of large numbers of young people from the respective surrounding villages to actually carry out the control efforts. This approach created a truly meaningful connection between the villages at large and execution of control efforts, galvanizing strong and tangible support in a manner that would be difficult to engender by other means. To date, NPSA field crews have killed over 6,000 mature trees and restored approximately 1,500 ha of native Samoan forest in the process. This is a model that bears consideration when contemplating invasive species control efforts elsewhere in the Pacific.

15.3.11 *Collaborative Work with Other Agencies to Try to Address the Alien Plant Problem at Its Roots*

15.3.11.1 Early Detection and Rapid Response Outside Protected Area Boundaries

Individuals in Hawaii became aware in the late 1970s and early 1980s that an aggressively invasive tree, *M. calvescens*, with likely serious implications for extinguishing biodiversity, was undergoing a rampant invasion in Tahiti and that the species was already present and spreading on the island of Hawaii. They were unfortunately unable to stimulate sustained agency or collective action until about 2 years after the species was discovered by a Haleakala NP employee on East Maui, about 8 km from the Park's remote Kipahulu Valley rainforest, in 1988. By the time a sustained alarm was raised, increasing information was becoming available from Tahiti's *M. calvescens* invasion (culminating in the accounts of Meyer 1996b and Meyer and Florence 1996). Based on the behaviour of the species in Tahiti, it was concluded that remote Kipahulu Valley's rainforest would not be defendable from a wave of *M. calvescens* invasion, so that a more proactive strategy was urgently needed. Haleakala NP employees first conducted surveys and removal efforts in 1991, other agencies helped, and a small interagency organization, the "Melastome Action Committee" (later the Maui Invasive Species Committee, MISC) coalesced, to publicise the problem and try to marshal resources to address the local and state-wide invasion of *M. calvescens* and other serious weeds. It became evident that partnerships are the only opportunity to have a chance of dealing effectively with such enormous shared threats. From the beginning, containment of *M. calvescens* was regarded as a holding action until biological control, first investigated in 1993, could become available (Medeiros et al 1997). Two decades later, *M. calvescens* is under containment by MISC on windward East Maui (only a few small plants have been found and removed in Haleakala's lower Kipahulu Valley over the years), but at considerable sustained cost (Meyer et al. 2011). Haleakala NP has been a major funding source and provider of expertise and manpower for *M. calvescens* containment. Unfortunately, though *M. calvescens* biocontrol efforts are progressing in recent years and can soon put forward multiple agents (Johnson 2010), biocontrol has not received nearly the level of resources that on-the-ground control has received. The best hope is for continued *M. calvescens* containment as biocontrol agents are released and monitored on Maui to assure that Maui's *M. calvescens* invasion is not allowed to 'explode' before the biocontrol agents can take over the job.

The concept of "stopping the next miconia" (i.e. locating and targeting potentially serious plant invasions early, while eradication is still a possibility, wherever on an island they arise) has gained traction in Hawaii in the past decade, especially on Maui, with development of pragmatic early detection methodology and actual eradications by MISC (Loope et al. 2004; Kueffer and Loope 2009; Penniman et al. 2011). Kraus and Duffy (2010) have described Hawaii's current relatively effective “existing functional management model for the eradication of incipient populations of invasive species that avoids reliance on official governmental response”. The individual island-based model “involves formation of informal multi-partner committees that utilise outside funding to achieve pest-management goals”. The model has evolved from the efforts to address the *M. calvescens* problem, first on Maui and later on other islands.

French Polynesia has devoted substantial education and regulatory effort to prevent *M. calvescens* from spreading (seeds can be easily spread via contaminated construction equipment, hiking boots, etc.) from the hub of Tahiti to other high islands. As a result, no other island has been invaded since 1997 (Meyer 2010). The National Park of American Samoa has conducted public outreach with the aim of reducing the spread of well-known plant invaders to its islands as well as eliminating small populations of barely established species such as *P. cattleianum* (T. Togia, NPSA, pers. comm.) The Pacific Invasives Learning Network (PILN, www.sprep.org) and collaborating organizations have encouraged similar efforts throughout the Pacific.

15.3.11.2 State-Wide and Countrywide Efforts at Reducing New Invasions

Given Hawaii’s high vulnerability to invasions, there is an obvious need to continually reassess possibilities to improve Hawaii’s network for prevention of new invasions of all taxa. Hawaii’s Coordinating Group on Alien Pest Species (CGAPS, www.hawaiiinvasivespecies.org/cgaps/) was launched in 1995 (Holt 1996) with hopes of “an alliance of biodiversity, health, agriculture, and business interests for improved alien species management in Hawaii”. The CGAPS has evolved into an important voluntary forum of 14 State, Federal, and private organizations directly involved in or with a major stake in invasive species prevention and/or management in Hawaii. It is well integrated with related efforts such as the relatively new and governmental Hawaii Invasive Species Council (HISC, www.hawaiiinvasivespecies.org/hisc/). Interagency communication has been greatly facilitated CGAPS, and progress is continually taking place. However, it must be said that its task is Herculean, given the forces of globalization, Hawaii’s vulnerability to invasions, and perhaps inevitably fragmented government response (CGAPS 2009; Kraus and Duffy 2010). Notably, there has been exceptionally little progress in the past two decades in implementation of regulations to restrict high-risk plant imports into Hawaii, in spite of such efforts as the Hawaii-Pacific weed risk assessment (Daehler et al. 2004; Denslow et al. 2009), good public information on the subject, support from an influential segment of the plant industry (Kueffer and Loope 2009), and even seemingly good support in the state legislature (Mark Fox, The Nature Conservancy of Hawaii, pers. comm.).

The Pacific Invasives Initiative (PII, www.pacificinvasivesinitiative.org) and associated programmes are working to facilitate improvement of biosecurity capacity in Pacific island countries.

15.4 Overview of Plant Invasions of Protected Areas in South-Eastern Polynesia

We use SE Polynesia as a region that may be somewhat representative of the vast complexity of Pacific islands and their protected areas, though obviously this is an oversimplification. Meyer (2004) has reviewed the status of the invasive plant threat to native flora and vegetation of the region. The islands of SE Polynesia include French Polynesia (a French overseas territory formed by five archipelagos, namely the Australs, Marquesas, Society, Tuamotu and Gambier, and comprising 120 islands), Cook Islands (an independent country in free association with New Zealand, 16 islands), Pitcairn Islands (a UK overseas territory with four islands, namely Pitcairn, Henderson, and Oeno and Ducie atolls), and Easter Island or Rapa Nui (a Chilean territory). These archipelagoes are comprised of relatively small tropical islands, the largest being Tahiti (1,045 km^2) in the Society Islands. Except for the National Park of Rapa Nui (6,660 ha) created in 1930 and a World Heritage Cultural Site since 1995, and the uninhabited raised atoll of Henderson (3,700 ha) declared a World Heritage Natural Site in 1988, there are few protected areas in SE Polynesia (Table 15.1). Only one is found in the main island of Rarotonga (Cook Islands), the "Takitumu Conservation Area," which is a community-based management area of about 155 ha. There are eight in French Polynesia with different protection status (natural parks, reserves, 'management areas', and one UNESCO Biosphere Reserve comprising seven atolls in the Tuamotu), but their size is relatively small (total area of about 10,000 ha, less than 2 % of French Polynesia's terrestrial surface).

Unfortunately, the number of invasive alien plants is high in many islands; some of them are dominant such as the small tree *M. calvescens* in low to mid-elevation rainforest in Te Faaiti Natural Park; the thorny shrub *Acacia farnesiana* (klu bush, kolu) in dry coastal areas in Eiao Management Area, a remote islet in the northern Marquesas; the small tree *Rhodomyrtus tomentosa* (downy rose myrtle) on the high elevation wet plateau (400–800 m) of Temehani Ute Ute Management Area (Fig. 15.5); the shrubs *Ardisia elliptica* (shoebutton) and *Cestrum nocturnum* (night-blooming jasmine) in the lowland rainforests of Takitumu Conservation Area. Most of Rapa Nui National Park is covered by grassland, which has become invaded by *M. minutiflora*, (Fig. 15.6), *Cirsium vulgare* (bull thistle), *Crotalaria grahamiana* (rattle-pod) and *Psidium guajava* (common guava). In addition to the presence of grazing and browsing alien ungulates in some reserves, natural disturbances such as cyclones may also facilitate plant invasions such as the vines *Merremia peltata* (merremia), *Mikania scandens* (climbing hempweed) and *Cardiospermum grandiflorum* (balloon vine) in the Takitumu Conservation Area (E. Saul, pers. comm., Fig. 15.7), or the spiny shrub *Rubus rosifolius* (thimbleberry) and the African tulip tree *Spathodea campanulata* (African tulip tree) in Te Faaiti Natural Park.

There are very few programmes to manage invasive alien plants in protected areas in SE Polynesia, perhaps because the proportion of protected areas occurring across the areas is commensurately small. Manual and mechanical control of the

Fig. 15.5 Temehani Plateau, Raiatea, French Polynesia. Invasion of native shrubland by the alien shrubs *Chrysobalanus icaco* and *Rhodomyrtus tomentosa* (Photo J-Y Meyer, April 2009)

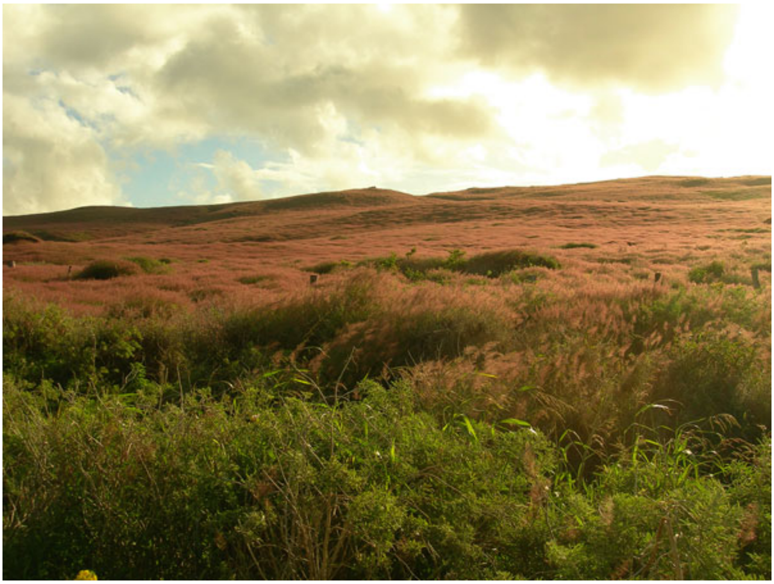

Fig. 15.6 Landscape dominated by the alien grass *Melinis minutiflora* near Anakena, Rapa Nui National Park, Rapa Nui (Easter Island) (Photo J-Y Meyer, February 2012)

Fig. 15.7 Invasion of native rain forest near Te Manga, island of Rarotonga (Cook Islands), South-eastern Polynesia, by the alien vines *Merremia peltata*, *Mikania micrantha*, and *Cardiospermum halicacabum* (Photo J-Y Meyer, October 2009)

weeds *Asclepias curassavica* (Mexican butterfly weed), *Cenchrus clandestinus* (= *Pennisetum clandestinum*, kikuyu grass), *Crotalaria grahamiana* (bushy rattlepod), and *Melinis minutiflora* has started in Rano Raraku in 2011 by Rapa Nui National Park (CONAF) in collaboration with the international branch of the French National Forestry Office (ONF-International). A pilot project to eradicate the thorny tree *Robinia pseudoacacia* (black locust) in the Rano Kau crater (Fig. 15.8) has been launched in 2012 (Meyer, unpub. data), and habitat restoration by reintroducing native and endemic plants (including the famous *Sophora toromiro* (toromiro) which was extinct in the wild) is planned. A biological control programme to contain *M. calvescens* with a fungal pathogen *Colletotrichum gloeosporioides* forma specialis *miconiae*, successfully released (Fig. 15.9) in Tahiti in 2000, has resulted in the recruitment of native and endemic plants in the understory of heavily invaded upland rainforests, such as in Te Faaiti Natural Park (Meyer et al. 2011).

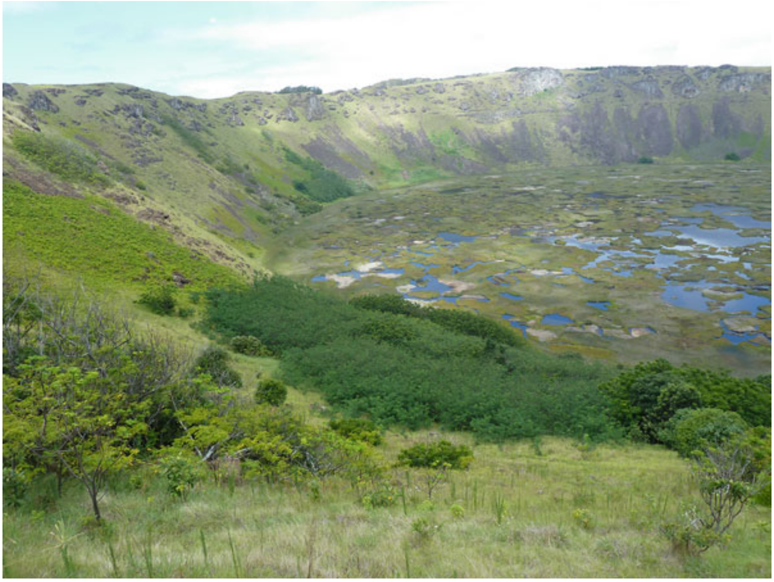

Fig. 15.8 Dense forest of invasive alien *Robinia pseudoacacia*, RanoKau crater, Rapa Nui National Park, Rapa Nui (Easter Island) (Photo J-Y Meyer, February 2012)

Fig. 15.9 Sapling of the highly invasive alien tree *Miconia calvescens* showing effects of the biocontrol fungal pathogen *Colletotrichum gloeosporioides* forma specialis *miconiae*, Fare Mato, Tahiti, French Polynesia (Photo J-Y Meyer, November 2009)

15.5 Conclusions

From an invasive plants standpoint, protected areas in the Hawaiian Islands, compared with those of other Pacific island governments, have had the advantage of a relatively favourable economic situation. They also have the disadvantages of a highly globalised economy, a history of purposeful alien plant introductions with an astounding 10,000 species introduced to the islands between 1906 and 1960 (Woodcock 2003), and a society that values horticultural novelty (Meyer and Lavergne 2004; Denslow et al. 2009; Kueffer et al. 2010a). Most protected areas in Pacific islands outside Hawaii have little invasive plant management, but may often have fewer plant invasions as a result of being ‘off the beaten path’, though that may not remain the case in the future. Increased regulation pertaining to the importation of alien plant and animal material is called for in addition to increased enforcement of existing importation regulations.

Hawaii has provided, and continues to provide, a useful laboratory for experiencing, understanding, and combating invasions. Hawaii Volcanoes NP has led conceptually and by example, and the strategies adopted there 2 or 3 decades ago serve as credible approaches for slowing invasions though not stopping them. However, there are largely intractable situations of the grass-fire cycle and certain invasive plant species. These problems are enormous, and the inability of the conservation community to work successfully through the political process to find a practical strategy for substantially reducing continued plant invasions has been a major disappointment, though this shortcoming is by no means unique to Hawaii and Pacific islands. Fortunately, improved herbicide technology offers promising tools for eradications and control as well for resurrection of native biodiversity after catastrophic invasions (e.g. Baider and Florens 2011).

Accelerated efforts involving biocontrol are warranted, along with more restrictive plant material importation rules and enforcement to help stem the tide of the most serious invasions and reduce the probability of new ones. If Hawaii’s biocontrol efforts can thrive in coming decades, the prospect of enhanced international collaboration with Pacific island biocontrol projects is appealing.

Historically, the common conservation strategy has been to set up protected areas in sites with high ecological values, often in pristine habitats or remnants of natural ecosystems where native and endemic species are dominant. In some cases in the Pacific Islands, however, protected areas have been set in historically human-disturbed areas (e.g. in South-eastern Polynesia, lowland rainforests have been modified by Polynesians during the past 1,000 years) or are currently found in alien-dominated forests as a result of plant succession after anthropogenic or even natural disturbances (such as fires, floods or cyclones). Innovative conservation strategies in these ‘novel’ ecosystems (sensu Hobbs et al. 2006, 2009) should be explored where possible and appropriate.

A major question for the future, however, is whether plant invasions will become even less manageable and more problematic in protected areas of Pacific Islands with increasing manifestations of climate change. Oceanic islands may benefit to some degree from moderation of the rate of change by maritime influences, in comparison with continental areas. Nevertheless, some, and perhaps many, invasive plant species on islands will likely have an extra edge in competitive ability due to increased CO_2 availability, disturbance from extreme climate events, and ability to invade higher elevation habitats as climates warm (Bradley et al 2010).

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References

Anderson SJ, Stone CP (1993) Snaring to control feral pigs *Sus scrofa* in a remote Hawaiian rainforest. Biol Conserv 63:195–201

Asner GP, Vitousek PM (2005) Remote analysis of biological invasion and biogeochemical change. Proc Natl Acad Sci U S A 102:4383–4386

Baider C, Florens FBV (2011) Control of invasive alien weeds averts imminent plant extinction. Biol Invasions 13:2641–2646

Benitez DM, Loh R, Tunison T et al (2012) The distribution of invasive plant species of concern in the Kilauea and Mauna Loa strip areas of Hawaii Volcanoes National Park, 2000–2010. Technical report 179. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu

Bradley BA, Blumenthal DM, Wilcove DS et al (2010) Predicting plant invasions in an era of global change. Trends Ecol Evol 25:310–318

Brooke de M L, Hepburn L, Trevelyan RJ (2004) Henderson Island world heritage site management plan 2004–2009. Foreign and Commonwealth Office, London, 40 pp

CGAPS (Coordinating Group on Alien Pest Species) (2009) CGAPS vision and action plan. http://www.hawaiiinvasivespecies.org/cgaps/pdfs/20100204cgapsactionsbyagency.pdf

Cole RJ, Litton CM, Koontz MJ et al (2012) Vegetation recovery 16 years after feral pig removal from a wet Hawaiian forest. Biotropica 44:463–471

Conant P (2002) Classical biological control of *Clidemia hirta* (Melastomataceae) in Hawaii using multiple strategies. In: Smith CW, Denslow J, Hight S (eds) Proceedings of workshop on biological control of native ecosystems in Hawaii. Technical report 129. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu. pp 13–20

Cox PA, Elmqvist T (1991) Indigenous control of tropical rain-forest reserves: an alternative strategy for conservation. Ambio 20:317–321

Daehler CC, Denslow JS, Ansari S et al (2004) A risk assessment system for screening out invasive pest plants from Hawaii and other Pacific Islands. Conserv Biol 18:360–368

D'Antonio CM, Dudley TL (1995) Biological invasions as agents of change on islands versus mainlands. In: Vitousek P, Loope L, Adsersen H (eds) Biological diversity and ecosystem function on islands. Ecological studies 115. Springer, New York, pp 103–119

D'Antonio CM, Mack MC (2006) Nutrient limitation in a fire-derived, nitrogen-rich Hawaiian grassland. Biotropica 38:458–467

D'Antonio CM, Hughes FR, Mack M et al (1998) The response of native species to removal of invasive exotic grasses in a seasonally dry Hawaiian woodland. J Veg Sci 9:699–712

D'Antonio CM, Tunison JT, Loh RK (2000) Variations in impact of exotic grasses and fire on native plant communities in Hawaii. Austral Ecol 25:507–522

D'Antonio CM, Hughes RF, Tunison JT (2011) Long-term impacts of invasive grasses and subsequent fire in seasonally dry Hawaiian woodlands. Ecol Appl 21:1617–1628

Darwin CR (1859) On the origin of species by means of natural selection, or the preservation of favoured races in the struggle for life. John Murray, London

Denslow JS, D'Antonio CM (2005) After biocontrol: assessing indirect effects of insect releases. Biol Control 35:307–318

Denslow JS, Space JC, Thomas PA (2009) Invasive exotic plants in the tropical Pacific islands: patterns of diversity. Biotropica 41:162–170

DeWalt SJ, Denslow JS, Ickes K (2004) Natural enemy release facilitates habitat expansion of the invasive tropical shrub *Clidemia hirta*. Ecology 85:471–483

Diong CH (1982) Population biology and management of the feral pig (*Sus scrofa* L.) in Kipahulu Valley, Maui. Ph.D. dissertation, Department of Zoology, University of Hawaii, Honolulu. http://www.hear.org/articles/diong1982/

Djeddour D, Shaw R (2011) Biological control of kahili ginger, *Hedychium gardnerianum*. CABI annual report, Europe UK 2010. CABI, Egham. http://www.cabi.org/uploads/file/Centrereports/UK_Centre_Report_2010.pdf

Douglas G (1969) Checklist of Pacific oceanic islands. Micronesica 5:327–464

Florence J, Waldren S, Chepstow-Lusty AJ (1995) The flora of the Pitcairn Islands: a review. Biol J Linn Soc 56:79–119

Footman T (2001) Guinness world records 2001. Bantam Books, New York

Gillespie RG, Claridge EM, Roderick GK (2008) Biodiversity dynamics in isolated island communities: interaction between natural and human-mediated processes. Mol Ecol 17:45–57

Hawaii Volcanoes National Park (2005) Fire management plan. Division of Resources Management, National Park Service, Volcano

Hobbs RJ, Arico S, Aronson J et al (2006) Novel ecosystems: theoretical and management aspects of the new ecological world order. Glob Ecol Biogeogr 15:1–7

Hobbs RJ, Higgs E, Harris JA (2009) Novel ecosystems: implications for conservation and restoration. Trends Ecol Evol 24:599–605

Holt A (1996) An alliance of biodiversity, health, agriculture, and business interests for improved alien species management in Hawaii. In: Sandlund OT, Schei PJ, Viken A (eds) Proceedings of the Norway/UN conference on alien species. Directorate for Nature Management and Norwegian Institute for Nature Research, Trondheim, pp 155–160. http://www.hear.org/AlienSpeciesInHawaii/articles/norway.htm

Huenneke LF, Vitousek PM (1990) Seedling and clonal recruitment of the invasive tree *Psidium cattleianum*: implications for management of native Hawaiian forests. Biol Conserv 53:199–211

Hughes RF, Denslow JS (2005) Invasion by a N_2-fixing tree alters function and structure in wet lowland forests of Hawaii. Ecol Appl 15:1615–1628

Hughes RF, Vitousek PM (1993) Barriers to shrub reestablishment following fire in the seasonal submontane zone of Hawaii. Oecologia 93:557–563

Hughes F, Vitousek PM, Tunison T (1991) Alien grass invasion and fire in the seasonal submontane zone of Hawaii. Ecology 72:743–746

Hughes RF, Uowolo AL, Togia TP (2012) Effective control of *Falcataria moluccana* in forests of American Samoa: managing invasive species in concert with ecological processes. Biol Invasions 14:1393–1413

Jacq F, Meyer J-Y (2012) Taux de mortalité et causes de disparition de *Apetahia raiateensis* (Campanulaceae), une plante endémique de l'île de Raiatea (Polynésie française). Revue d'Ecologie (Terre et Vie) 67:57–72

Jaynes DB, Colvin TS, Karlen DL et al (2001) Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. J Environ Qual 30:1305–1314

Johnson MT (2010) Miconia biocontrol: where are we going and when will we get there? In: Loope LL, Meyer J-Y, Hardesty BD et al (eds) Proceedings of the international Miconia conference, Keanae, Maui, Hawaii, May 4–7, 2009. Maui Invasive Species Committee and Pacific Cooperative Studies Unit, University of Hawaii, Honolulu, 11pp. http://www.hear.org/miconia2009/pdfs/johnson.pdf

Kellner JR, Asner GP, Kinney KM et al (2011) Remote analysis of biological invasion and the impact of enemy release. Ecol Appl 21:2094–2104

Kraus F, Duffy DC (2010) A successful model from Hawaii for rapid response to invasive species. J Nat Conserv 18:135–141

Kueffer C, Loope L (eds) (2009) Prevention, early detection and containment of invasive, non-native plants in the Hawaiian Islands: current efforts and needs. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu. Technical report no. 166. http://www.botany.hawaii.edu/faculty/duffy/techr/166

Kueffer C, Daehler CC, Torres-Santana W et al (2010a) A global comparison of plant invasions on oceanic islands. Perspect Plant Ecol Evol Syst 12:145–161

Kueffer C, Schumacher E, Dietz H et al (2010b) Managing successional trajectories in alien-dominated, novel ecosystems by facilitating seedling regeneration: a case study. Biol Conserv 143:1792–1802

Langeland KA, Stocker RK (1997) Control of non-native plants in natural areas of Florida. Institute of Food and Agricultural Sciences publication SP242. University of Florida, Gainesville

LaRosa AM (1992) The status of banana poka in Hawaii. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 271–299

Lenz L, Taylor JA (2001) The influence of an invasive tree species (*Myrica faya*) on the abundance of an alien insect (*Sophonia rufofascia*) in Hawaii Volcanoes National Park. Biol Conserv 102:301–307

Loh R, Daehler CC (2007) Influence of invasive tree kill rates on native and invasive plant establishment in a Hawaiian wet forest. Restor Ecol 15:199–211

Loh RK, Daehler CC (2008) Influence of woody invader control methods and seed availability on native and invasive species establishment in a Hawaiian forest. Biol Invasions 10:805–819

Loh RK, Tunison JT (2009) Long term management of invasive plant species in Special Ecological Areas at Hawaii Volcanoes National Park: a review of the last 20 years, or where do we go from here? In: Kueffer C, Loope L (eds) Prevention, early detection and containment of invasive, non-native plants in the Hawaiian Islands: current efforts and needs. Technical report no. 166. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu, pp 33–35

Loope LL, Juvik SP (1998) Protected areas. In: Juvik SP, Juvik JO (eds) Atlas of Hawaii, 3rd edn. University of Hawaii Press, Honolulu, pp 154–157

Loope LL, Nagata RJ, Medeiros AC (1992) Introduced plants in Haleakala National Park. The status of banana poka in Hawaii. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 551–576

Loope LL, Starr F, Starr KM (2004) Protecting endangered plant species from displacement by invasive plants on Maui, Hawaii. Weed Technol 18:1472–1474

Lugo AE (2004) The outcome of alien tree invasions in Puerto Rico. Front Ecol Environ 2:265–273

Markin GP, Lai P-Y, Funasaki GY (1992) Status of biological control of weeds in Hawaii and implications for managing native ecosystems. The status of banana poka in Hawaii. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 466–482

Medeiros AC (2004) Phenology, reproductive potential, seed dispersal and predation, and seedling establishment of three invasive plant species in a Hawaiian rain forest. Ph.D. Dissertation, Department of Botany, University of Hawaii, Honolulu. http://www.hear.org/articles/medeiros2004dissertation

Medeiros AC, Loope LL, Conant P et al (1997) Status, ecology, and management of the invasive tree *Miconia calvescens* DC (Melastomataceae) in the Hawaiian Islands. In: Evenhuis NL, Miller SE (eds) Records of the Hawaii biological survey for 1996. Bishop Mus occasional paper 48:23–35

Merlin MD, Juvik JO (1992) Relationships among native and alien plants on Pacific islands with and without significant human disturbance and feral ungulates. The status of banana poka in Hawaii. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 597–624

Messing RH, Wright MG (2006) Biological control of invasive species: solution or pollution? Front Ecol Environ 4:132–140

Meyer J-Y (1996a) Espèces et espaces menacés de la Société et des Marquises. Contribution à la biodiversité de la Polynésie française N 1-5 [unpublished report]. Délégation à l'Environnement/Délégation à la Recherche, Papeete, 245 pp

Meyer J-Y (1996b) Status of *Miconia calvescens* (Melastomataceae), a dominant invasive tree in the Society Islands (French Polynesia). Pacific Sci 50:66–76

Meyer J-Y (2000) Rapport de mission aux Marquises Sud (Hiva Oa, Fatu Hiva, Mohotani) du 6 au 20 février 2000 [unpublished report]. Délégation à la Recherche, Papeete, 19 pp

Meyer J-Y (2004) Threat of invasive alien plants to native flora and forest vegetation of Eastern. Polynesia. Pacific Sci 58:357–375

Meyer J-Y (2005) Rapport de mission botanique à Ua Huka (Marquises, Groupe Nord) du 24 au 31 juillet 2005 et description de la végétation et de la flore de la réserve naturelle de Vaikivi [unpublished report]. Délégation à la Recherche, Papeete, 29 pp

Meyer J-Y (2008) Rapport de mission d'expertise à Rapa Nui du 02 au 11 juin 2008: plan d'action stratégique pour lutter contre les plantes introduites envahissantes sur Rapa Nui (Île de Pâques) [Strategic action plan to control invasive alien plants on Rapa Nui (Easter Island) [unpublished report]. Délégation à la Recherche, Papeete, 62 pp

Meyer J-Y (2010) The miconia saga: 20 years of study and control in French Polynesia (1988–2008). In: Loope LL, Meyer J-Y, Hardesty BD et al (eds) Proceedings of the international Miconia conference, Keanae, Maui, Hawaii, May 4–7, 2009. Maui Invasive Species Committee and Pacific Cooperative Studies Unit, University of Hawaii at Manoa. http://www.hear.org/miconia2009/pdfs/meyer.pdf

Meyer J-Y (2012) Rapport de mission d'expertise à Rapa Nui du 14 au 20 février 2012: projet de restauration écologique dans des parcelles de démonstration sur Rapa Nui (île de Pâques) [unpublished report]. Délégation à la Recherche, Papeete, 50 pp

Meyer J-Y, Florence J (1996) Tahiti's native flora endangered by the invasion of *Miconia calvescens* DC (Melastomataceae). J Biogeogr 23:775–781

Meyer J-Y, Lavergne C (2004) Beautés fatales: Acanthaceae species as invasive alien plants on tropical Indo-Pacific Islands. Divers Distrib 10:333–347

Meyer J-Y, 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. IUCN, Gland, pp 91–96. http://www.issg.org/pdf/publications/Island_Invasives/pdfwebview/1Meyer.pdf

Minden V, Hennenberg KJ, Porembski S et al (2010a) Invasion and management of alien *Hedychium gardnerianum* (kahili ginger, Zingiberaceae) alter plant species composition of a montane rainforest on the island of Hawaii. Plant Ecol 206:321–333

Minden V, Jacobi JD, Porembski S et al (2010b) Effects of invasive alien *Hedychium gardnerianum* on native plant species regeneration in a Hawaiian rainforest. Appl Veg Sci 13:5–14

Motooka PL, Ching G, Nagai G (2002) Herbicidal weed control methods for pastures and natural areas of Hawaii. CTAHR, Cooperative Extension Service, University of Hawaii. Technical bulletin WC-8. http://www2.ctahr.hawaii.edu/oc/freepubs/pdf/wc-8.pdf

Mueller-Dombois D, Fosberg FR (1998) Vegetation of the tropical Pacific islands. Springer, New York

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Parrotta JA (1990) *Paraserianthes falcataria* (L.) Nielsen. Batai, Moluccan sau. Leguminosae (Mimosoideae) Legume family. Report SO-ITF-SM-31. US Department of Agriculture, Forest Service, Southern Forest Experiment Station, Institute of Tropical Forestry, New Orleans, 5 pp

Penniman TM, Buchanan L, Loope LL (2011) Recent plant eradications on the islands of Maui County, Hawaii. In: Veitch CR, Clout MN, Towns DR (eds) Island invasives: eradication and management. IUCN, Gland, pp 325–331. http://www.issg.org/pdf/publications/Island_Invasives/pdfHQprint/3Penniman.pdf

Sachet M-H (1983) Végétation et flore terrestre de l'atoll de Scilly (Fenua Ura). J Soc Océanistes 77:29–34

Santos GL, Kageler D, Gardner DE et al (1992) Herbicidal control of selected alien plants in Hawaii Volcanoes National Park. The status of banana poka in Hawaii. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Hawaii, pp 341–375

Sellars RW (1997) Preserving nature in the National Parks: a history. Yale University Press, New Haven

Smith CW (2002) Forest pest biological control program in Hawaii. In: Smith CW, Denslow J, Hight S (eds) Proceedings of workshop on biological control of native ecosystems in Hawaii. Technical report 129. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu, pp 91–102. http://www.botany.hawaii.edu/faculty/duffy/techr/129.pdf

Stone CP, Smith CW, Tunison JT (eds) (1992) Alien plant invasions in native ecosystems of Hawaii: Management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu. http://www.hawaii.edu/hpicesu/book/1992_chap/default.htm

Tomich QC (1986) Mammals in Hawaii: a synopsis and notational bibliography, 2nd edn. B.P. Bishop Museum special publication 76, Honolulu

Tunison JT (1992a) Fountain grass control in Hawaii Volcanoes National Park: management considerations and strategies. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 376–393

Tunison JT (1992b) Alien plant control strategies in Hawaii Volcanoes National Park. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 485–505

Tunison JT, Stone CP (1992) Special Ecological Areas: an approach to alien plant control in Hawaii Volcanoes National Park. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 781–798

Tunison JT, Zimmer NG (1992) Success in controlling localized alien plants in Hawaii Volcanoes National Park. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 506–524

Tunison JT, Smith CW, Stone CP (1992a) Alien plant management in Hawaii: conclusions. In: Stone CP, Smith CW, Tunison JT (eds) Alien plant invasions in native ecosystems of Hawaii: management and research. University of Hawaii Press for Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu, pp 821–833

Tunison JT, Whiteaker LD, Cuddihy LW et al (1992b) The distribution of selected localized alien plant species in Hawaii Volcanoes National Park. Technical report 84. Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu

Tunison JT, Leialoha JAK, Loh RK et al (1994) Fire effects in the coastal lowlands, Hawaii Volcanoes National Park. Technical report 88. Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu

Tunison JT, Loh RK, Leialoha JAK (1995) Fire effects in the submontane seasonal zone, Hawaii Volcanoes National Park. Technical report 97. Cooperative National Park Resources Studies Unit, University of Hawaii, Honolulu

Van Driesche R, Center T (2014) Chapter 26: Biological control of invasive plants in protected areas. In: Foxcroft LC, Pyšek P, Richardson DM, Genovesi P (eds) Plant invasions in protected areas: patterns, problems and challenges. Springer, Dordrecht, pp 561–597

Vitousek PM, Walker LR (1989) Biological invasion by *Myrica faya* in Hawaii – plant demography, nitrogen-fixation, ecosystem effects. Ecol Monogr 59:247–265

Vitousek PM, Loope LL, Stone CP (1987) Introduced species in Hawaii: biological effects and opportunities for ecological research. Trends Ecol Evol 2:224–227

Wagner WL, Herbst DR, Sohmer SH (1999) Manual of the flowering plants of Hawai'i. University of Hawaii Press, Honolulu

Walters GA (1971) A species that grew too fast – *Albizia falcataria*. J For 69:168

Warner KD, Kinslow F (2011) Manipulating risk communication: value predispositions shape public understandings of invasive species science in Hawaii. Public Underst Sci. doi:10.1177/0963662511403983

Webb EL, van de Bult M, Chutipong W et al (2006) Composition and structure of lowland rain-forest tree communities on Ta'u, American Samoa. Pacific Sci 60:333–354

Webb EL, Seamon JO, Fa'aumu S (2011) Frequent, low-amplitude disturbances drive high tree turnover rates on a remote, cyclone-prone Polynesian island. J Biogeogr 38:1240–1252

Whistler WA (1980) The vegetation of Eastern Samoa. Allertonia 2:45–190

Whistler WA (1994) Botanical inventory of the proposed Tutuila and Ofu units of the NPSA. CPSU technical report 87. University of Hawaii, Honolulu

Whistler WA (2004) Rainforest trees of Samoa. Isle Botanica, Honolulu

Wikler C, Smith CW (2002) Strawberry guava (*Psidium cattleianum*) prospects for biological control. In: Smith CW, Denslow J, Hight S (eds) Proceedings of workshop on biological control of native ecosystems in Hawaii. Technical report 129. Pacific Cooperative Studies Unit, University of Hawaii, Honolulu, pp 108–116

Woodcock D (2003) To restore the watersheds: early twentieth-century tree planting in Hawai'i. Ann Assoc Am Geogr 93:624–635

Woodward SA, Vitousek PM, Matson K et al (1990) Use of the exotic tree *Myrica faya* by native and exotic birds in Hawai'i Volcanoes National Park. Pacific Sci 44:88–93