

Critical issues and new challenges for research and management of invasive plants in the Pacific Islands

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Invasive alien species are recognized as a major threat to island biodiversity and ecosystem functions worldwide, with well-documented, detrimental impacts on the native biota of Oceania. Despite their high number and rapid increase in the Pacific Islands, invasive alien plants (IAP) have received less attention by researchers, managers, and the general public compared to invasive animals (e.g., predatory mammals). Indeed, although lists of IAP in natural and agro-ecosystems have been set up in most island countries and territories, their ecological and socio-economical impacts are still not well documented and/or popularized. Very few IAP eradication successes have been reported, and post-control monitoring and cost-benefit analysis are often missing. Moreover, most of the published studies have been conducted in the Hawaiian and the Galápagos islands. This essay is a call for more research and management efforts on IAP in Oceania, especially in the small tropical Pacific Islands. Focal areas should not only include species biology, control strategies and methods and prioritization systems (including risk assessments), but also better understanding of island ecosystems functioning (e.g. forest dynamics and resilience), with the integration of past and present anthropogenic and natural disturbances. The importance of “novel” ecosystems, where natural habitats have been partially or totally modified by humans, and the potential effects of climate change on terrestrial ecosystems should be addressed, and new conservation and management strategies defined in the Pacific Islands, in order to try to halt biodiversity erosion in highly vulnerable island ecosystems.

Key words: globalization, invasive alien plants, novel ecosystems, Pacific Islands, prioritization, resilience, restoration

INTRODUCTION

DUE to their geographic isolation which restricts gene flow among populations, islands are often celebrated as natural laboratories for evolution (e.g., Carlquist 1974; Grant 1998), favoured sites for biogeographical and ecological studies (e.g., Vitousek *et al.* 1995; Whittaker and Fernandez-Palacios 2007), and recognized as “biodiversity hotspots” (Myers *et al.* 2000). Islands are also preferential areas to document the global biotic homogenization phenomenon through introduction of non-native species (McKinney and Lockwood 1999; Olden 2006; Castro and Jaksic 2008). Indeed, the number of introduced species in the Pacific Islands is now equalling or exceeding the number of native and endemic species in many islands or archipelagos (Table 1), and is dramatically increasing with time and with recent species inventories. High

vulnerability of island ecosystems to biological invasions is attributed to the impoverishment of the native biota (taxonomic disharmony, lack of certain functional groups), evolution in long isolation from outside influences (Loope and Mueller-Dombois 1989; Keppel *et al.* 2014), competitive advantages of introduced species under particular environmental conditions (Daehler 2003), and by the fact that most introduced species have been introduced without their natural enemies (DeWalt *et al.* 2004).

Invasion by alien (or non-native, exotic) plant and animal species is indisputably recognized as one of the major drivers of change in island ecosystems worldwide (Vitousek *et al.* 1996; MEA 2005; Ricciardi 2006) and a serious threat to island native biodiversity (IUCN/SSC/ISSG 2000; Reaser *et al.* 2007). In the Pacific Islands, introduced predatory animal invaders (e.g., cats,

Table 1. Comparison between native and alien flora (flowering plants and ferns) in selected Pacific tropical islands (by size of terrestrial area) and number of naturalized and invasive alien plants (including dominant or major IAP).

Island or island group	Area (km ²)	Native flora (number of indigenous species)	Alien flora (number of introduced species)	Naturalized alien plant species	Invasive alien plant species	Dominant IAP
New Caledonia	19 060	3 261 ^a	2 008 ^b	597 ^b	97 ^c	67 ^b
Fiji	18 270	1 622 ^d	977 ^d	461 ^d	107 ^e	30 ^f
Hawai'i	16 880	1 138 ^g	8 134 ^h	1 104 ⁱ	469 ⁱ	86 ^j
Galápagos	7 900	550 ^k	870 ^l	229 ^l	109 ^l	22 ^l
French Polynesia	3 519	885 ^m	> 1 700 ⁿ	593 ⁿ	-	57 ⁿ
Cook Is.	238	296 ^o	997 ^o	333 ^o	76 ^p	12 ^q
Rapa Nui (Easter Island)	166	48 ^r	370 ^s	180 ^s	-	36 ^s
Wallis et Futuna	142	351 ^t	338 ^u	151 ^u	-	18 ^u

^aJaffré *et al.* 2004, ^bMeyer *et al.* 2010, ^cHequet *et al.* 2009, ^dBrownlie 1977 and Smith 1996, ^eGISD, ^fMeyer 2000, ^gWagner *et al.* 1999, ^hStaples and Herbst 2005, ⁱStaples and Cowie 2001, ^jSmith 1985, ^kMauchamp 1997, Trueman *et al.* 2010, ^lFlorence *et al.* 2007, ^mFourdrigniez and Meyer 2008, ⁿMcCormack 2007, ^oSpace and Flynn 2002, ^pMeyer 2004, ^qDubois *et al.* 2013, ^rMeyer 2008, ^sMorat *et al.*, ^tMeyer *et al.* 2010

rats, brown tree snake, snails, tilapia, mosquitofish) have caused extinction or extirpation (local extinction of species populations in specific areas) of land and seabirds (Savidge 1987; Wiles *et al.* 2003), land molluscs and tree snails (Murray *et al.* 1988; Cowie 2001), and aquatic insects (Englund 1999). The impacts of introduced animals (feral goats, sheep, deer, pigs, horses and cattle) on native vegetation are also well documented. For example, losses of *Pitosporum tianianum* in New Caledonia (Bouchet *et al.* 1995) and *Canavalia*

kauensis in Hawai'i (Loope and Scowcroft 1985) are attributed to feral ungulates, while seed predation by rats has resulted in decline of rare endemic plants (Loope and Medeiros 1990; Meyer and Butaud 2009). A great amount of scientific, technical and popular papers have also been published on the detrimental effects of the cane toad *Rhinella marina* (syn. *Bufo marinus*), Indian mongoose *Herpetes auropunctatus*, and alien ants (e.g., yellow crazy ant *Anoplolepis gracilipes*, big-headed ants *Pheidole spp.*, Argentine ant *Linepithema humile* [syn. *Iridomyrmex humilis*],

Table 2. Most commonly cited tropical Pacific Islands IAP, by number of citations and alphabetical order of scientific name, in 10 textbooks on invasive alien species at a global scale. 1: Elton (1958), 2: Cronk and Fuller (1995), 3: Williamson (1996), 4: Wittenberg and Cox (2001), 5: Myers and Bazely (2003), 6: Davis (2009), 7: Lockwood *et al.* (2009a), 8: Booth *et al.* (2010), 9: Kiehn (2011), 10: Simberloff and Rejmanek (2011);*cited in the "100 of the World's Worst Invasive Alien Species" (Lowe *et al.* 2000); §considered a "major pest" and with a Weed Risk Assessment score ≥ 20 according to the "Pacific Islands Ecosystems at Risk" database (<http://www.hear.org/pier/wralist.htm>).

Plant name (Family)	Habit	Island(s)	Context	Textbooks	Documented major ecological impacts (References)
<i>Morella</i> (syn. <i>Myrica faya</i>) (Myricaceae)	Tree	Hawai'i	Ecology, control, "transformer"	2, 3, 5, 6, 8, 9, 10	Increases soil nitrogen [N] level; alters native plant communities (Vitousek <i>et al.</i> 1987; Vitousek and Walker 1989)
<i>Miconia calvescens</i> * (Melastomataceae)	Tree	Tahiti, Hawai'i, New Caledonia	Ecology, control, biological control using a fungal pathogen, "transformer", genetics	2, 4, 8, 9, 10	Decreases light availability and understory species diversity; favours soil erosion on steep slopes (Meyer and Florence 1996; Meyer 2010)
<i>Psidium cattleianum</i> * (Myrtaceae)	Tree	Hawai'i, Polynesia	Ecology, control	2, 3, 7, 9	Constrains native plant (Myrtaceae) recruitment; alters the future successional trajectories of rain forests (Zimmerman <i>et al.</i> 2008)
<i>Clidemia hirta</i> *§ (Melastomataceae)	Shrub	Hawai'i, Fiji, Samoa, Tonga, Palau	Ecology, control, biological control using a fungal pathogen, "transformer"	2, 5, 9, 10	
<i>Tibouchina herbacea</i> § (Melastomataceae)	Shrub	Hawai'i	Ecology, control	2, 7, 8, 10	
<i>Lantana camara</i> *§ (Verbenaceae)	Shrub	Hawai'i	Ecology, control, biological control, dispersal by alien birds	1, 2, 10	
<i>Cinchona pubescens</i> (syn. <i>C. succirubra</i>)* (Rubiaceae)	Tree	Galápagos	Ecology, control, "transformer"	2, 8, 9	Affects the diversity and cover of native vegetation (Jäger <i>et al.</i> 2009); increases nutrient concentrations (mainly nitrogen [N] and phosphorus [P] levels); favours invasion by other introduced plants (Jäger <i>et al.</i> 2013).
<i>Passiflora tarminiana</i> (syn. <i>P. tripartita</i> , <i>P. mollissima</i>)§ (Passifloraceae)	Vine	Hawai'i	Ecology, control	2, 3	
<i>Andropogon virginicus</i> § (Poaceae)	Grass	Hawai'i	Ecology, control, "fire enhancer"	2, 5	Increases fire frequency and intensity (Smith and Tunison 1992; D'Antonio <i>et al.</i> 2011)
<i>Mikania micrantha</i> (syn. <i>M. scandens</i>)§ (Asteraceae)	Vine	Pacific Islands	Ecology, control	2, 9	
<i>Mimosa diplotricha</i> (syn. <i>M. invisa</i>)§ (Mimosaceae)	Shrub	Cook Is., Micronesia, PNG, Marianna Is.	Biological control	2, 4	
<i>Leucaena leucocephala</i> * (Mimosaceae)	Tree	Hawai'i, Pacific Islands	Ecology, control	1, 2	

little fire ant *Wasmannia auropunctata*) in the Pacific region (e.g., Elton 1958; Williamson 1996; Wittenberg and Cox 2001; Simberloff and Rejmánek 2011).

Ironically, although invasive alien plants (IAP) are numerous (ca. 1,132 species according to the “Pacific Islands Ecosystems at Risk” (PIER) database <http://www.hear.org/pier/>; Denslow *et al.* 2009), and several lists of IAP and agricultural weeds in most of the 25 Pacific Islands countries and territories have been published (Waterhouse and Norris 1987; Swarbrick 1997; Waterhouse 1997; Meyer 2000; 2004; Muniappan *et al.* 2002; Motooka *et al.* 2003; Denslow *et al.* 2009), relatively few species are thoroughly studied and/or broadly recognized. A review of major textbooks dealing with biological invasions at a global scale reveals that only 12 species have been extensively studied in the Pacific Islands (Table 2). The most cited IAP include the small trees *Cinchona pubescens* (syn. *C. succirubra*), *Miconia calvescens*, *Morella* (syn. *Myrica*) *faya*, *Psidium cattleianum*, and the shrubs *Clidemia hirta*, *Lantana camara*, and *Tibouchina herbacea*, which were often studied in Hawai'i and the Galápagos. With the notable exception of *Miconia calvescens* (sometimes called the “green cancer” in Tahiti, French Polynesia, or the “purple plague” in the Hawaiian islands), plant species may be less studied than animals because: (1) their impacts are not immediate and difficult to see or to demonstrate; (2) very few endemic species extinctions caused by plant invasions have been demonstrated (Sax and Gaines 2008); and (3) there are still few documented eradication successes (e.g., Veitch and Clout 2002; Veitch *et al.* 2011). Moreover, IAP are often intentionally introduced as useful species, and thus the subject of many conflicts of interests between different stakeholders (e.g., gardeners, foresters and ranchers versus conservationists and environmentalists).

The main goal of this paper is to try to assess the progress made on research and management to address plant invasions in remnant terrestrial natural ecosystems of the Pacific Islands. By natural, I refer to habitats dominated by native and endemic species where conservation is critical. The assessment is based on literature review and 20 years of research on IAP and personal field experience in the small tropical islands of Polynesia (mainly Cook Islands, Hawaiian Islands, Easter Island or Rapa Nui, French Polynesia and Wallis and Futuna) and Melanesia (Fiji, New Caledonia), conducted in strong collaboration with local scientists, managers, policy-makers, and nature protection groups.

Although this paper is restricted to the small tropical and subtropical islands of the Pacific (excluding large continental islands over

20 000 km² such as New Zealand or Papua New Guinea) and focused on natural ecosystems, many issues raised are also relevant to other tropical and subtropical islands or archipelagos worldwide (e.g., the Caribbean and the Macaronesian islands in the Atlantic Ocean, the Mascarenes and the Seychelles in the Indian Ocean, the Bonin and the Juan Fernández Islands in the Pacific Ocean). Agricultural weeds and adventives exclusively found in urbanized and ruderal areas, cultivated lands and agricultural fields, forest plantations, mines or other highly anthropogenically modified sites are not considered here, although I recognize that the frontiers between agro- and natural ecosystems in the small Pacific Islands are not so clear-cut because of ancient (pre-European period) development of agroforestry and the creation of socio-ecological ecosystems resulting from past and present human occupation and habitat transformation (Vitousek 2006; Thaman 2014).

ESTABLISHMENT AND SPREAD OF IAP IN THE PACIFIC ISLANDS

The rapid and recent increase of introduced plants in the Pacific Islands is directly related to globalization, with the increase movement of people and goods due to the development of aerial and maritime traffic between islands and the nearest continents, and between island countries and territories, leading to an irreversible homogenization of floras and faunas. It is noteworthy that the most economically developed islands (with the highest Gross Domestic Product) are those with the highest number of introduced plants (Küeffler *et al.* 2010). Major cities, such as Honolulu in Hawai'i, Papeete in French Polynesia and Nouméa in New Caledonia, have international airports and harbours that act as transport hubs for people, and plant and animal species, accidentally or intentionally introduced from Asia, Australia, and the Americas. Moreover, these French and U. S. territories support many public and private botanical gardens that were established in the last century to acclimatize “useful” plants from other tropical and temperate countries, including many forestry and ornamental species. Garden ornamentals presently represent the major source of plant invaders and potentially new invasive species worldwide (Dehnen-Schmutz *et al.* 2007; Hulme 2011), and more particularly in the tropical Pacific islands (Staples *et al.* 2000; Meyer and Lavergne 2004; Meyer *et al.* 2008). For instance, among the 250 introduced species cultivated in the Papeari Botanical Garden in Tahiti (French Polynesia) between 1921 and 1944, nine have become dominant invaders, legally declared a threat to biodiversity in French Polynesia by a decree voted by the Assembly in 2006, including *Ardisia*

elliptica, *Chrysobalanus icaco*, *Cecropia peltata*, *Cestrum nocturnum*, *Miconia calvescens*, *Rhodomirtus tomentosa*, *Schinus terebinthifolius*, *Spathodea campanulata*, and *Waterhousea floribunda* (Meyer 2003). Others are now spreading, such as the tree *Schefflera actinophylla* or the climbing liana *Anodendron paniculatum* (Meyer 2007).

In the Hawai'ian islands, there are currently about 10 000 introduced plant species according to the "Hawai'i Biological Survey", of which "10% of them are known to be established or naturalized" (Staples and Cowie 2001). Naturalized (or established) species are defined as species being able to reproduced sexually or vegetatively without human assistance, forming self-sustainable populations, and able to disperse at considerable distance from the parent plant or population (e.g., Richardson *et al.* 2000; 2011). In less than 20 years, the number of introduced plant species has increased two-fold in the Galápagos islands, from 438 alien plants, including 185 naturalized plants (Mauchamp 1997), to 870 introduced species including 229 established species (Trueman *et al.* 2010). A recent study in New Caledonia has estimated the number of alien plants to exceed 2 000, including approximately 600 naturalized species (Hequet *et al.* 2009). By contrast, the last census published 15 years ago mentioned only 1 400 species, including approximately 360 naturalized species (MacKey 1994 cited in Meyer *et al.* 2006).

Similar increasing rates of establishment have been observed in other French territories. On the small 140 km² island of Moorea (Society archipelago, French Polynesia), a total of 767 non-native plants were recorded on Moorea during a five year intensive species inventory conducted for the "Moorea Biocode Project" (2007–2012), with 433 new island records compared to the past published data ("Base de données botaniques Nadeaud", Florence *et al.* 2007). With a total of at least 327 naturalized species, the alien flora of Moorea is now exceeding the native flora, comprising about 300 indigenous and endemic species (J.-Y. Meyer, unpublished data). In the small archipelago of Wallis and Futuna located in Western Polynesia, botanical surveys conducted in 2007 and 2008 (Meyer *et al.* 2010) added 139 new introduced species since the last survey in 1980 (Morat and Veillon 1985), of which 44 were naturalized, equating to a naturalization rate of 1.5 species per year for the past several decades.

A dominant plant invader is defined as a widely naturalized alien species forming dense cover, stands or forests in natural and semi-natural habitats, and likely to have strong ecological impacts on biotic communities or ecosystem functions or processes (Cronk and

Fuller 1995; Meyer 2000; 2004; Kueffer *et al.* 2010). The number of dominant plant invaders in each Pacific Island country and territory has increased during the past decades, following more or less the "tens rule" stating that about 10% of the introduced species may become naturalized and 10% of them may turn into invasive species (Williamson 1996). For instance, plant invaders increased in the Hawaiian Islands from 86 species reported in the 1980s to 469 nowadays (Smith 1985; Staples and Cowie 2001), 67 to 97 in New Caledonia (Meyer *et al.* 2006; Hequet *et al.* 2009), and 11 to 109 in the Galápagos (Mauchamp 1997; Trueman *et al.* 2010). However, these changes in numbers may be biased by the definitions of "invasive plants" used by different authors, and whether agricultural weeds and other ruderals are included or not. For instance, the last census of IAP in New Caledonia was enlarged to include roadside weeds (Hequet *et al.* 2009). Many so-called "invasive species" cited in global databases are in fact widely or even locally naturalized species with relatively low ecological impacts. For example, the shrub *Abrus precatorius*, an ancient Polynesian or early European introduction (Whistler 1991), is considered to be an "introduced invasive in French Polynesia" by the PIER database although it is only locally naturalized (Florence *et al.* 2007 and pers. obs.). In the same way, the IUCN "Global Invasive Species Database" (www.issg.org/database/welcome/) includes the fern *Angiopteris evecta*, a common native fern in Fijian rainforests and other South Pacific Islands (Brownlie 1977), as an "invasive species". This error may be related by the fact that the fern is recognized as naturalized in Hawai'i and other tropical islands where it has been introduced as a garden ornamental outside its native distribution range (Christenhusz and Toivonen 2008). There is thus a need to clarify which naturalized or invasive species should be considered of high priority for management, and to update databases with field observations and data.

Prioritization of the worst invaders is a necessity, especially in small Pacific Islands where funding resources are often limited (Denslow *et al.* 2009), data are limited and local capacity and political will to address invasive plants are relatively low. For instance in French Polynesia, only five of the 35 major IAP legally declared a threat to biodiversity are targeted for active management: the tree *Miconia calvescens* in the islands of Tahiti, Raiatea, Nuku Hiva and Fatu Hiva, because of its documented impacts on the native flora (Meyer and Florence 1996); the shrubs *Chrysobalanus icaco* and *Rhodomirtus tomentosa* on the plateau of Temehani of high ecological value in Raiatea; and the trees *Psidium cattleianum*, *Schefflera actinophylla* and *Spathodea*

campanulata in areas where small restoration projects are currently being conducted in Tahiti for the recovery of endangered and protected endemic plants and birds.

The PIER project has developed a Weed Risk Assessment (WRA) system for Hawai'i and the Pacific Islands (Daehler *et al.* 2004), a modified version of the Australia and New Zealand WRA to identify pest plants in managed lands and native forests and predict their invasiveness and potential impacts. It may be used to rank invasive species as a prioritization tool (Tye 2009). A total of 172 species were screened for analysis, and species are considered to be of high risk if the obtained WRA score is above six. The tropical Pacific island major pests, interpreted as species which have "substantial effects on native ecosystems or on native populations, OR substantial economic effects in managed ecosystems", and with the highest scores, include: *Lantana camara* (WRA score of 34), *Clidemia hirta* (27), *Mikania micrantha* (syn. *M. scandens*) (25), *Mimosa diplotricha* (syn. *M. invisa*) (24), *Tibouchina herbacea* (24), *Passiflora tarminiana* (syn. *P. mollissima*) (24), *Andropogon virginicus* (20). This ranking is consistent with biological invasion expert opinions (Table 2). However, other well-studied dominant plant invaders, such as *Melinis minutiflora* (18), *Psidium cattleianum* (18), *Morella* (syn. *Myrica*) *faya* (17), *Leucaena leucocephala* (15), *Miconia calvescens* (14) and *Cinchona pubescens* (9), have relatively low WRA scores, far less than other weeds having lower ecological impacts such as the small vine *Passiflora foetida* (26), showing the limits of WRA system.

MAJOR IMPACTS OF IAP

Ecological impacts

Among the dominant invasive plants, species recognized as habitat transformers because they directly or indirectly change the availability of resources for other species by causing biotic or abiotic changes (Richardson *et al.* 2000; 2011) should be given the highest priority for management. Notorious examples of species altering "the basic rules of existence for all organisms" (Vitousek 1990) in the Pacific Islands are: the nitrogen fixing tree *Morella* (*Myrica*) *faya*; the fire-tolerant and fire-enhancing grass *Andropogon virginicus* in Hawai'i; and trees forming dense monotypic forests such as the large-leaved *Miconia calvescens* in Tahiti, the quinine tree *Cinchona pubescens* in the Galápagos Islands, and the strawberry guava *Psidium cattleianum* in rainforests of many islands (Table 2). Yet more research is need on biological invasions is to determine whether IAP are drivers of changes (i.e., directly leading to native

species and habitat degradation) or just passengers of other disturbances (MacDougall and Turkington 2005).

The most frequently cited dominant IAP in the Pacific region includes *Lantana camara* (e.g., Thaman 1974; Meyer 2000). Ironically, this species is still not under control despite one century of biological control efforts in Hawai'i since 1902 and the introduction of more than 20 insects (Davis *et al.* 1992). The small tree *Leucaena leucocephala* (syn. *L. glauca*), introduced as a fodder in most Pacific islands, often invades lowland dry and mesic habitats after anthropogenic disturbances (e.g. fire, ungulates), where it forms dense monotypic stands (e.g., Wagner *et al.* 1999). *Clidemia hirta* is a serious weed both in agricultural lands and in the rainforests of Hawai'i and Fiji (Wester and Wood 1977), as well as in other Melanesian (PNG, Solomon, Vanuatu), Western Polynesian islands (Samoa, Tonga, Wallis and Futuna), Guam and Palau (Meyer and Medeiros 2011), and under partial biological control for several decades with the release of the parasitic thrips *Liothrips wichii* introduced in 1930 in Fiji and 1953 in Hawai'i (Smith 1992). The mat-forming grass *Melinis minutiflora*, also introduced as a fodder plant, is a major invader in French Polynesia, Hawai'i, New Caledonia and Rapa Nui (Hughes *et al.* 1991; Meyer 2000; D'Antonio *et al.* 2011).

There are documented examples in the Pacific Islands where IAP have directly caused drastic decline of native and endemic plant species by competition (or physical displacement) or modification of ecosystem processes (such as light availability, water regime, soil nutrients cycles) in the absence of other natural or anthropogenic disturbances. IAP that form dense stands (grass covers, shrub thickets or tree forests) have devastating effects on the survival of native and endemic plants, especially those confined to very small areas ("micro-endemics"). For example the rare endemic lobeliad *Cyanea superba* has a very restricted range on O'ahu (Hawai'i), an area now occupied by strawberry guava *Psidium cattleianum* (Smith 1985). The number of species and populations of endemic understorey plants such as *Cyrtandra* spp. have also declined in the Hawaiian islands, probably due to altered light and soil conditions caused by strawberry guava (Kiehn 2011). Other critically endangered endemic species threatened by *P. cattleianum* include *Charpentiera australis* and *Meryta brachypoda*, restricted to the two main summits of Tubuai (Austral Island, French Polynesia) that are now completely invaded (Meyer 2004).

Between 40 and 50 species strictly endemic to Tahiti, mainly understorey herbs, shrubs and small trees belonging to *Cyrtandra*, *Ophiorrhiza*,

Myrsine and *Psychotria*, among the most speciose genera in the Pacific Islands, were endangered by the massive invasion of *Miconia calvescens* on that island (Meyer and Florence 1996). Moreover, the observed faster litter decomposition of *Miconia calvescens* compared to native plants in Hawai'i causes rapid nutrient losses and impacts soil development (Allison and Vitousek 2004).

Hedychium gardnerianum a rhizomatous herb found between 0–1 700 m in Hawai'i growing in semi-shade and full shade beneath the forest canopy (Smith 1985), inhibits regeneration of native forests by affecting the germination of native species, and leading to a change in forest structure and biogeochemical processes (Asner and Vitousek 2005). It could also increase the likelihood of other invasive species, including *Psidium cattleianum*, which is the only tree species observed able to maintain itself in dense *H. gardnerianum* stands (Minden *et al.* 2010), a phenomenon known as “invasional meltdown” (Simberloff and Von Holle 1999), whereby an alien species facilitates one another's establishment and spread. The “Australian tree fern” *Sphaeropteris cooperi* (syn. *Cyathea cooperi*) is displacing native tree fern *Cibotium glaucum* in Hawaiian rainforests (Medeiros *et al.* 1992). It supports fewer epiphytic plant species than *C. glaucum* (Medeiros *et al.* 1993), produces more leaves, contains more nitrogen (N) and phosphorus (P), and decomposed faster than *C. glaucum* leaves, thus altering nutrient cycles for native plants (Chau *et al.* 2013).

Another example of ecological impacts of IAP on ecosystem processes is the effects of the large tree *Albizia* (syn. *Falcataria*) *moluccana* on remnants of native lowland rainforests in Hawai'i. The primary productivity in the form of litterfall was found to be more than eight times greater in *Albizia* dominated forest stands, N and P inputs via litterfall were greater, as well as rates of litter decomposition. *Albizia* invasion facilitates the increase of understory alien plant species, particularly strawberry guava *Psidium cattleianum*, and affects native species such as *Metrosideros polymorpha* (Hughes and Denslow 2005). Seed germination of native species in Hawai'i is also inhibited by the stems and leaves of the mat-forming grass *Melinis minutiflora* (Hughes and Vitousek 1993).

Socioeconomic impacts

Invasive species are defined by international organizations as “an agent of change that threatens native biological diversity” (IUCN 2000) or “alien species that threaten ecosystems, habitat or species. In addition, they can pose a threat to food security, human health and economic development” (UNEP/CBD 2000).

However, these socioeconomic and health impacts are sometimes difficult to assess and/or often not studied, leading some experts to exclude any connotation of impacts in their definition (Richardson *et al.* 2011; Simberloff 2013).

Economic cost is one way to bring greater attention to the problem of invasive species to resource managers and decision-makers (Lee 2007). For example, in 2009, the Hawai'i Department of Transportation spent one million dollars (USD) to remove about 1 500 *Albizia moluccana* trees along a single mile of roadway because of falling trees and branches (Hughes *et al.* 2013). *A. moluccana* is a roadside, urban forest and residential pest of major significance. A cost-benefit analysis was recently conducted on *Miconia calvescens* in Hawai'i (Burnett *et al.* 2007). If left untreated, the ecological damages from *Miconia*'s invasion will reach a total of 627 million US dollars over the next 40 years. Estimates of potential expected losses from *Miconia*'s invasion on the island of O'ahu in terms of groundwater recharge suggest that a loss of 41 million gallons per day would generate economic losses of 137 millions US dollars per year (Kaiser and Roumasset 2002).

TREATMENT OF NATIVE INVASIVES

Invasive species are often defined as alien (or no-native) species that are introduced intentionally or accidentally through human activity (Cronk and Fuller 1995; IUCN 2000; Richardson *et al.* 2011). However, the biogeographical status of some IAP in the Pacific Islands is sometimes controversial. It is indeed sometimes difficult to know if they are native (or indigenous) or ancient human introductions, as Pacific islands were colonized by humans between 60 000 and 40 000 years ago in Melanesia and Micronesia to less than 1 000 years ago in Eastern Polynesia (Kirch 2010; Keppel *et al.* 2014). The first human colonizers transported many useful animals and plants during their migrations, creating “transported landscapes” (Kirch 1984). Between 70 and 80 plant species were introduced into Polynesia, intentionally or accidentally, as adventives (Whistler 1991; Florence *et al.* 2007). The Guidelines for Invasive Species Management in the Pacific produced by the Secretariat of the Pacific Regional Environment Programme included the native species “that proliferate following environmental changes caused by human activities” (Tye 2009).

For instance, the gaiac tree *Acacia spirorbis* subsp. *spirorbis*, as well as the grasses *Imperata cylindrica* and *Heteropogon contortus*, are native in New Caledonia (Jaffré *et al.* 2004), where they are considered as weeds in pastures and

grasslands (Blanfort *et al.* 2008). In the same way, the pantropical bracken fern *Pteridium aquilinum*, indigenous in New Caledonia is considered as a fire-enhancing weed by local managers where it forms dense mats (Jaffré *et al.* 1998; Meyer *et al.* 2010). Another classic example is the climbing woody liana *Merremia peltata*, native in Fiji (Smith 1991) and other Melanesian islands, but not in Polynesia (Samoa, French Polynesia) although sometimes considered as indigenous in Cook Islands (Space and Flynn 2002; McCormack 2007), and considered as a high risk plant pest because of its invasiveness (Daehler *et al.* 2004). Even *Hibiscus tiliaceus*, a very common native tree in lowland rainforests of many Pacific Islands, might be a Polynesian introduction in Cook Islands and thus considered naturalized (McCormack 2007).

Palaeo-ecological and palynological studies may solve the status of these pseudo-indigenous or cryptogenic species (*sensu* Carlton 1996) by analysing past vegetation before and after human occupation. For instance, the ironwood *Casuarina equisetifolia*, which is considered a Polynesian introduction in eastern Polynesia (Whistler 1991), may be native on Rapa Nui where fossil pollen grains have been found (Prebble 2008). This species, known to be invasive in other tropical islands and countries (Potgieter *et al.* 2014), has spread in several atolls in the Tuamotu (French Polynesia), where it is a recent (1960s) modern introduction (J.-Y. Meyer, unpublished data). Moreover, the aboriginal introductions *Aleurites moluccana* (candle nut tree), *Schizostachyum glaucifolium* (Polynesian bamboo) and *Inocarpus fagifer* (Tahitian chestnut), widely naturalized in lowland rainforests, are now part of the natural and cultural heritage of Polynesian islands (Larrue *et al.* 2010). Restoration projects using the control or removal of IAP in the Pacific Islands should incorporate these socio-cultural and historical aspects, and clearly identify the goals to be reached (e.g., the original pre-human or the pre-European state of the ecosystem).

Native species that exhibit an invasive or aggressive behavior (Space and Flynn 2002) often belong to fast-growing pioneer species, and their demographic explosions is often related to natural disturbances (such as cyclones, floods, landslides) and/or anthropogenic pressures, such as fires or forest clearing. For instance, the common native fern *Dicranopteris linearis* forms thick stoloniferous mats and dense cover on entire slopes in many Pacific islands, often associated with anthropogenic fires. This fernland may persist for decades or centuries and seems to block the forest dynamics (Mueller-Dombois and Fosberg 1998). In the same way, *Merremia peltata* is a gap filler or a transitional

species in secondary plant succession. For an ecological (and not socio-economical) point of view, this species can be viewed as a natural element of forest dynamics in Fiji (Kirkham 2005), rather than being considered as an invasive species or aggressive weed. Removing or controlling these native species may thus alter the pathways of plant succession and be detrimental for ecosystem functioning.

MANAGEMENT INTERVENTIONS AND EFFECTIVENESS

Plant eradication and alternatives

Eradication supposes the elimination of all individuals of a species from an area to which reintroduction will not occur (Myers and Bazely 2003), including all reproductive plants, juveniles, seedlings and seeds in the ground. Eradication effectiveness depends on the area over which the plant is distributed, as follow up surveys are constrained by site accessibility, plant detectability, the species' characteristics, and funding support (Panetta and Timmins 2004). Mack and Lonsdale (2002) note that "the record of eradicating invasive plants, whether on islands or continents, consists of few clear victories, some stalemates, and many defeats." Very few IAP eradication successes on islands have been reported compared with alien animals (Veitch and Clout 2002; Veitch *et al.* 2011). Although many different control methods have been developed for dominant invaders, such as *Miconia calvescens* in French Polynesia and Hawai'i (Medeiros *et al.* 1997; Meyer *et al.* 2011), *Cinchona pubescens* in the Galápagos (Jäger and Kowarik 2010) or *Albizia* (syn. *Falcataria*) *moluccana* in Hawai'i and American Samoa (Hughes *et al.* 2013), eradication was not achieved at an island level, mainly due to the large size of infestation (hundreds to thousands of hectares) and logistic constraints.

The very few documented successful eradications of IAP in the Pacific Islands include *Cenchrus echinatus* on Laysan Island (Flint and Rehkemper 2002), a small atoll (411 ha) located in the north-central Pacific. This small annual herb was threatening the breeding habitats of two endemic birds, the Laysan finch and the Laysan duck. Control using herbicide (glyphosate) and hand-pulling reduced populations in five years to almost undetectable levels. The relative high cost (US\$150 000 per year for staff, supplies and vessel charter) was due to the difficulty to access to this remote island.

Another eradication programme conducted on the Hawaiian islands of Maui, Lana'i and Moloka'i between 2001 and 2009 focussed on 12 alien species with a small infestation extent

(< 11 ha) and a small number of individuals (between 1 and 165; Penniman *et al.* 2011). Success was achieved for species with short seed longevity (e.g., the woody *Cryptostegia grandiflora*) between 1 to 5 years, but long-term monitoring is still needed for other species, such as *Ulex europaeus*, with seed viability of over 30 years.

A pre-control feasibility study to prioritize species for eradication is often needed. In the Galápagos, three species (the vine *Pueraria phaseoloides*, the shrub *Rubus glaucus* and the tree *Citharexylum gentryi*) were selected for eradication based on their distribution (size of infestation), their life history traits (dispersal, growth rate, reproductive capacities), their potential invasiveness in the archipelago and elsewhere, and the availability of treatment methods and ease of control via site accessibility (Soria *et al.* 2002). A major obstacle for plant eradication is the existence of a soil seed bank, which can persist for less than one year to several decades. For instance, seeds of *Miconia calvescens* are viable in soil samples for at least 16 years (Meyer 2010), thus decreasing the feasibility of eradication if mature reproductive trees are found in an area. By contrast, eradication efforts may be more feasible with plants that lack persistent seed bank, such as strawberry guava *Psidium cattleianum*, which has rapid and high germination rates, post-dispersal seed predation and seed mortality (Uowolo and Denslow 2008). An alternative strategy to eradication is the juvenilization used for *Miconia calvescens* in the Pacific Islands, where all reproductive trees are targeted first to stop the massive seed rain (one mature tree may produce millions of viable seeds) and the high risk of dispersal over long distances by frugivorous birds (Meyer *et al.* 2011).

Biocontrol

Some naturalized plants and weeds have become so widespread that manual, mechanical and chemical control methods are often not practical nor economically feasible. In that case, biological control (i.e., the use of natural enemies to decrease the populations or the impacts of target species) appears to be the only sustainable solution (Myers and Bazely 2003). Several biocontrol programmes have been conducted in the Pacific Islands in the past decades (Cochereau 1972; Waterhouse and Norris 1987; Paynter 2010), with varied effectiveness. The success rate of biocontrol programmes to combat 17 invasive plants and weeds in Hawai'i between 1890 and 1985 is only 60% (Funasaki *et al.* 1988). The release of 17 natural enemies to combat *Clidemia hirta* in Hawai'i (Conant 2002) has been deceiving, as only five agents have established and none were truly successful (Loope *et al.* 2013). *Lantana*

camara is still invasive in Hawai'i and Micronesia where active biological control programmes have been conducted for almost one century, and thus recognized as “a particularly hardy and resilient plant species that demonstrates a remarkable capacity to recover from the severest of insects attacks” (Denton *et al.* 1991).

Nevertheless, there are some successful and well-documented biocontrol programmes in the Pacific region, such as the collaborative project conducted on *Miconia calvescens* in Hawai'i and French Polynesia with the release of a fungal pathogen *Colletotrichum gloeosporioides* f. sp. *miconiae* found in Brazil in 1997 (Killgore *et al.* 1999). Ten years of post-release monitoring show that biocontrol can partially restore rainforests densely invaded by *Miconia* with important conservation benefits such as endemic species recovery (Meyer *et al.* 2008b; Meyer and Fourdrigniez 2011; Meyer *et al.* 2012). Expectations have been raised from the recent release in Hawai'i in 2011 of a gall-forming scale insect, *Tectococcus ovalus*, against strawberry guava *Psidium cattleianum*, after nearly 15 years of research in Hawai'i and Brazil (U.S. Forest Service 2013), as well as the rust *Puccinia spegazzinii*, released in 2009 in PNG and Fiji to control the vine *Mikania micrantha* (Day *et al.* 2013).

A regional approach for biocontrol in the Pacific has been recommended (Waterhouse and Norris 1987; Dovey *et al.* 2004), mainly to offset the limited resources and to benefit from foreign partnerships and expertise. A recent analysis using a ranking system that includes the feasibility of success has identified the best IAP targets within the Pacific region (Paynter 2010). The release of the same biocontrol agents is often proposed for different Pacific Island countries and territories. However, a crucial issue is the presence in these islands of endemic species, and the necessity to conduct additional host-specificity tests to avoid non-target effects. For instance, complementary tests need be done on endemic species belonging to the genus *Bidens* and the endemic genus *Oparanthus* (Asteraceae) if a *Mikania micrantha* biocontrol project is decided in French Polynesia, as it was done for *Miconia calvescens*, with tests on native and endemic melastomes that are absent in Hawai'i (Killgore *et al.* 1999). The proposal to introduce the psyllid *Heteropsylla spinulosa* to French Polynesia, New Caledonia or Vanuatu to control *Mimosa diplotricha* (syn *M. invisa*) (Paynter 2010) is also complicated because of the presence of native legume trees (such as *Serianthes myriadenia* and *Schleinitzia insularum* in French Polynesia). The release of natural enemies to control *Spathodea campanulata* and *Tecoma stans* is easier as there are no native and endemic species belonging to this plant family

(Bignoniaceae) in the Pacific Islands (Smith 1991; Wagner *et al.* 1999; Florence *et al.* 2007). *Hedychium gardnerianum* and other ornamental invasive *Hedychium* species (*H. flavescens*, *H. coronarium*) are difficult targets (Anderson and Gardner 1999) because of the presence of Zingiberaceae species of economic importance such as the edible ginger *Zingiber officinale*, the shampoo ginger *Zingiber zerumbet* and *Cucurma longa*, two Polynesian introductions of cultural importance (Whistler 1991).

Managing conflicts of interest

Human perceptions of the status of alien species is variable according to geographic areas and with time (e.g., McNeely 2001; Barbault and Atramentowicz 2010;). Conflicts of interest between stakeholders can delay or prevent IAP control programmes. Many invasive trees were intentionally introduced as forestry or agricultural species (Richardson 1998), notably the tree *Albizia moluccana* (syn. *Falcataria moluccana*, *Paraserianthes falcataria*) and the Caribbean pine *Pinus caribaea* in the Pacific Islands (Meyer and Malet 2000). In New Caledonia, the Caribbean pine is the only non-native plant able to colonize ultramafic soils (Meyer *et al.* 2006) and is rapidly colonizing open areas such as fernlands and grasslands as well as the lower Temehani plateau, a site of high conservation value in French Polynesia (J.-Y. Meyer, personal observation). Both species are recognized to be invaders by New Caledonian authorities (legally declared invasive by the “Code de l’Environnement de la Province Sud” in 2009), whereas in French Polynesia, only *Albizia moluccana* is considered a threat to biodiversity (by a decree voted in 2006), mainly because of the reluctance of local foresters to recognize the situation. Australian *Acacia* spp. are often promoted by aid and development agencies as multipurpose useful trees (Low 2012), including *Acacia farnesiana*, one of the major invader in dry lowland and coastal vegetation types in the Marquesas (French Polynesia) and in Hawai’i (Smith 1985). IAP are often seen as beneficial invaders because they are valuable to people. Meanwhile in Tahiti, the dry fruits of the Brazilian pepper *Schinus terebinthifolius*, a melliferous tree (i.e., producing honey), are commonly used as a condiment in sashimi in that island and thus very appreciated by locals.

There are some cases where IAP now perform important ecological functions and are considered as ecological substitutes or surrogates (Caro and O’Doherty 1999), such as fleshy-fruited invasive plants that provide food that supports indigenous frugivore populations. This was demonstrated in Australia where several native birds appear to rely on the fruit of

camphor laurel *Cinnamomum camphora* as their principal food over part of the year (Date *et al.* 1996). *Lantana camara* has in similar ways become a keystone species for many animals, providing a greater range of resources than almost any native plants (Gosper and Vivian-Smith 2006). The small tree *Leucaena leucocephala* provides ground cover and can reduce soil erosion in badly degraded landscape (Iye 2009), and disappears when overgrown by taller trees, thus having a pioneer role as a second growth species (Mueller-Dombois 2008). Another notorious example is gorse *Ulex europaeus*, an aggressive thorny shrub in pastures at high elevation in Hawai’i (Wagner *et al.* 1999), but used by farmers as hedge, forage plant, and valuable pollen source in New Zealand. It is also known to facilitate the recruitment of some woody native trees in New Zealand by providing structural complexity or shade and shelter for seedlings, such as increased moisture availability (Hayes 2007; Wotton and McAlpine 2013). However, the benefits of these IAP should be compared to their ecological costs on a long-term base, as many species provide only temporary benefits.

Another critical issue is the cultural dimension of some introduced species in the Pacific region. For instance, the vine *Merremia peltata*, perceived as a “troublesome smothering weed” in Fijian forests (Smith 1991), is valued by local inhabitants of ‘Uvea, Wallis and Futuna, as a good ground cover in cultivated areas as it suppresses other undesirable weeds (Meyer *et al.* 2010). Strawberry guava *Psidium cattleianum* biocontrol in Hawai’i was faced with strong local opposition and manipulation by activists (Warner and Kinslow 2013), which delayed the release of the agent for several years. These different perception and recognition of IAP with reference to human value system and societies may be partially solved by consultation and dialogue with the different stakeholders, as well as information and education at all levels, from the general public to politicians (Meyer 2013).

FUTURE CHALLENGES

The next invaders

Preventing the establishment of new alien species and the early detection of potential new invaders is often given the highest priority in IAP control strategies. The concept of “stopping the next *Miconia*” (i.e., locating and targeting potential serious plant invasions early while eradication is still possible; Loope *et al.* 2013) is complicated by the fact that some species don’t show any sign of invasion for long periods of time after their introduction (thus called sometimes “sleeper weeds”; Grice and Ainworth

2003). This latency period, defined as the time lag between introduction and initiation of invasion (Kowarik 1995), has different causes including genetic variability, disturbance regime, suboptimal habitats, or lack of pollinators and dispersal agents (Williamson 1996). The lag phases may take 50 to 100 years (e.g., for *Schinus terebinthifolius* in Florida; Ewel 1986), but typically occur over a few decades when the species are introduced in disturbed areas that are in the vicinity of appropriate natural vegetation (Daehler 2009). For instance, it took more than 50 years for the liana *Anodendron paniculatum* to escape the Papeari botanical garden in Tahiti and spread in lowland rainforests (Meyer 2007), and between 25 and 35 years for *Miconia calvescens* to become invasive in Tahiti and Hawai'i after its introduction as a garden ornamental (Meyer 1998).

Future invaders include many ornamental species such as palms (Meyer *et al.* 2008a) or species belonging to popular garden plant families such as Acanthaceae (Meyer and Lavergne 2004), including the popular ornamental *Sanchezia speciosa*, which is naturalized in Tahiti, Rarotonga, Hawai'i and Fiji. The octopus tree or Queensland umbrella tree *Schefflera actinophylla* is another good example, commonly planted in the main cities of many Pacific islands (J.-Y. Meyer, personal observation), as well as in villages in American Samoa, Palau, and in urban areas and along roadsides on Viti Levu, Fiji (Keppel and Watling 2011). In Tahiti, the fleshy fruits of *S. actinophylla* are dispersed by both native and alien frugivorous birds and is rapidly spreading in mesic and rain forests up to 1 000 m elevation, threatening remnant populations of the rare endangered endemic tree *Erythrina tahitensis* (J.-Y. Meyer, unpublished data).

Shade-tolerant trees that can regenerate under closed canopy (Fine 2002; Martin *et al.* 2008), or species that spread vegetatively have the potential to invade intact forest ecosystems (Denslow 2003), should be considered as highest priority for prevention and early eradication. For instance, the shade-tolerant succulent herb *Kalanchoe pinnata* (syn. *Bryophyllum pinnatum*), that reproduces from plantlets and bulbils formed along the leaf margin, is now spreading in coastal and dry lowland forests in Hawai'i (Wagner *et al.* 1999) and the Austral islands in French Polynesia (J.-Y. Meyer, personal observation). Seedlings of the ornamental tree *Filicium decipiens*, a urban street tree known to have naturalized in Hawai'i (but with a very low WRA score of 2 according to the PIER database), are now found in the understorey of mid-elevation secondary rainforest of Tahiti up to 600 m elevation (J.-Y. Meyer, personal

observation). Forestry species such as the mahogany *Swietenia macrophylla*, introduced as a timber tree in Fiji since 1911 and widely planted (40 000 ha of plantations), are now established in lowland rainforests (Kepper and Watling 2011). *Sphagneticola* (syn. *Wedelia trilobata*), a small creeping herb introduced in the 1970s as an ornamental and/or promoted as a ground cover in many Pacific islands, has escaped cultivation and is now naturalized in many habitats, from coastal vegetation to mangroves margins (Thaman 2009). It is now found in Tahiti along roadsides and colonizing the edges of montane rainforests up to 1 100 m elevation (J.-Y. Meyer, personal observation).

Climate change and IAP

Changes in climate may significantly alter the areas at risk of invasion and the spatial pattern of distribution and abundance of plant invaders. In Oceania, natural disturbances, such as cyclones or typhoons, floods, landslides and droughts are expected to increase in frequency and intensity with global climate change (Kingsford and Watson 2011). It will result in drastic changes in forest canopy structure (treefall gaps and canopy openings) and more plant invasions in forest understorey (Loope and Giambelluca 1998), especially by light-demanding fast-growing pioneer species with good dispersal abilities or early successional species (Dukes and Mooney 1999), such as the African tulip tree *Spathodea campanulata*, *Tecoma stans* or *Lantana camara*, which are rapidly colonizing landslides and treefall gaps in many Pacific islands, including Tahiti (J.-Y. Meyer, personal observation). *Albizia* (*Falcataria*) *moluuccana* is known to benefit from cyclones in American Samoa (Hughes *et al.* 2012; Loope *et al.* 2013), and the spread of *Miconia calvescens* in rainforests in Australia was triggered by canopy openings after cyclone (Murphy *et al.* 2008). Other "winners" (*sensu* McKinney and Lockwood 1999) of forest disturbances are climbing vines and lianas, such as *Mikania micrantha*, *Merremia peltata* and *Passiflora* species.

Increased frequency of drought would involve higher incidence of fire, promoting the spread of fire-tolerant plants such as *Andropogon virginicus*, *Lantana camara* (Duggin and Gentle 1998) or *Melinis minutiflora*. The last two species have been observed to aggressively colonize burnt areas in native forests of Tahiti and Moorea up to 700 m elevation (J.-Y. Meyer, personal observation).

Changes in atmospheric carbon dioxide concentrations are predicted to expand the distribution and abundance of vines, as documented in Australia with the shrubby vine *Cryptostegia grandiflora* (Kriticos *et al.* 2003).

Range shifts will also likely occur as plant species may invade higher elevation habitats with elevated air temperatures (Bradley *et al.* 2010). Species temperature tolerance zones for both native and alien species may be shifted upward by about 300–400 m in high volcanic tropical islands, assuming a mean rate of 0.6°C every 100 m rise in elevation (Loope and Giambelluca 1998; Pouteau *et al.* 2010). The current upper limit of the distribution range of *Spathodea campanulata* and *Miconia calvescens* in Tahiti is about 1 400 m elevation (J.-Y. Meyer, personal observation), and may shift towards higher altitude, threatening the still relatively undisturbed montane cloud forests. Some ornamental aliens that currently depend on the artificial climate of gardens might become invasive if climates shift in their favour (Dukes and Mooney 1999).

Expansion of secondary forest is predicted, formed by fast-growing, relatively short-lived, second-growth species, such as the vines *Merremia peltata* and *Mikania micrantha* or the trees *Spathodea campanulata*, which are already spreading in abandoned agriculture lands, forest clearings and after cyclones (e.g. in Cook Islands or Samoa in the early 1990s). The importance of “novel” or “hybrid” ecosystems (Hobbs *et al.* 2006; 2009; Lugo 2009), where natural habitats have been partially or totally modified by humans, is increasing in the Pacific Islands, especially with demographic and economical development (Thaman 2014). For instance the small (140 km²) island of Moorea, French Polynesia, now an *île-dortoir* (bedroom-island) for many people working in the capital city of Papeete (Tahiti, located 20 km away) and a favourite destination for foreign and local tourists, has experienced recent rapid demographic growth, with a population increase of 35.5% between 1996 and 2007 to a current population of approximately 17 000 inhabitants (density of 126 per km²). Rapid growth has led to increasing anthropogenic impacts, such as urbanization, deforestation for agriculture, accidental fires, and biological invasions. A recent study indicates that only 6% of the island is now covered by natural forest (Pouteau *et al.* 2013), much of which is fragmented and restricted to high elevation site above 900 m, and that half of the rare, threatened and/or legally protected native and endemic plant species are currently found in “hybrid” habitats where native species co-occur with naturalized alien species, representing 45% of the island area (Meyer *et al.*, unpublished data). Management of these fragmented landscapes that incorporate dynamic shifts and sudden changes from climate impacts will require a coordinated, integrated approach (Jupiter *et al.* 2014).

CONCLUSIONS: NEEDS AND PROPOSED SOLUTIONS

Pacific islands are facing an irreversible increase of newly introduced and naturalized species leading to a progressive homogenization of their floras. Easter Island or Rapa Nui is the ultimate example, not only of past native forest past loss caused by humans leading to ecological collapse (Diamond 2005), but also of recent massive invasion by alien plants, with more than 180 naturalized alien species versus less than 50 native and endemic vascular plants (Table 1; Meyer 2008). The increasing rates of human movement and global trade will provide repeated exposure to a more diverse seed pool of alien species from different geographic areas (called propagule pressure or colonization pressure; Lockwood *et al.* 2009b; Simberloff 2009), some of them with unpredictable ecological behaviours because of the potential effects of climate changes. Therefore, we still need more knowledge and research on IAP of local and regional importance, focussing on those with strong impacts on biodiversity and/or ecosystems services (e.g., shade-tolerant forestry or ornamental tree species) in order to revise the lists of dominant invaders in Pacific Island countries and territories (and in global databases) for management prioritization.

The rapid influx and colonization of alien plant species has been shown to affect successional patterns of native forests and other vegetation types. To be able to efficiently manage and conserve these habitats, we also need a better understanding of island ecosystem functioning that includes native vegetation and forest dynamics, secondary plant succession, and plant community resilience in the face of increasing natural and anthropogenic disturbances. Establishment of permanent sampling plots in different vegetation types for monitoring of native and alien plant communities will be crucial to study the short-term and long-term changes in species richness, diversity and abundance/cover over-time, with or without disturbances. There is increasing evidence of invasive alien species facilitating the establishment of other non-native species (“invasional meltdown”, Simberloff and Von Holle 1999), including nitrogen-fixing species favouring other alien plants such as *Morella faya*, *Albizia moluccana*, and introduced frugivorous birds dispersing alien invasive plants (e.g., Spotswood *et al.* 2012). More studies should be conducted on native/alien species interactions, including dispersal networks, food chains and nutrient cycling.

Even if management of IAP is commonly carried out in Pacific islands with many control or eradication projects (Table 3), very few studies have evaluated the success of these projects in

light of the recovery of native vegetation. We need more projects on habitat restoration focussing on the re-establishment of native plant communities after weed control experiments. The study of the resilience of native ecosystems might be a key to a better management (Keppel *et al.* 2014). Some habitats might be more resistant to plant invasions, such as the native highland vegetation of the

Galápagos after manual control of *Cinchona pubescens* (Jäger and Kowarik 2010) or to anthropogenic disturbances such as fire in the subalpine vegetation of Tahiti (J.-Y. Meyer, personal observation). Long-term follow-up is required to assess success or failure of IAP eradication, mainly re-invasion by the same invasive species from the soil seed bank or the seed rain, or re-infestation by other alien plant

Table 3. Non-exhaustive list of documented IAP control or eradication programmes conducted in tropical Pacific Islands (excluding Hawai'i and Galápagos Islands), by alphabetical order of plant scientific names. Control method: MC = Manual control (e.g., uprooting, cutting, bark removal), CC = Chemical control (use of herbicides), BC = Biological control (release of natural enemies). 1: "Aliens. The Invasive Species Bulletin (1995–2014)" (<http://www.issg.org/publications.htm#aliens>), 2: "Vatthe Vine Control project (August 2010)" (www.forestandbird.org.nz), 3: "Forest Health highlights, Pacific Islands (2005–2011)" (www.fs.fed.us/foresthealth), 4: "PII (Pacific Invasives Initiative) News (2006–2014)" (www.pacificinvasivesinitiative.org), 5: PILN (Pacific Invasives Learning Network) Soundbites (2010–2014)" (<http://www.sprep.org/piln/soundbites-documents>), 6: Taputuarai *et al.* (2012)

Species (Family)	Habit	Island country or territory	Control method(s)	References
<i>Acacia crassicarpa</i> , <i>A. auriculiformis</i> (Fabaceae)	Tree	Mauke (Cook)	MC, CC	4
<i>Acacia farnesiana</i> (Fabaceae)	Shrub	Tahiti (French Polynesia)	MC, CC	5
<i>Adenanthera pavonina</i> (Mimosaceae)	Tree	American Samoa	Not reported	5
<i>Albizia moluccana</i> (syn. <i>Falcataria moluccana</i>) (Fabaceae)	Tree	American Samoa	MC, CC	3
<i>Antigonon leptopus</i> (Polygonaceae)	Vine	Niue, Pohnpei and Yap (FSM), Guam, Saipan and Tinian (Mariana Is.)	MC	3, 4
<i>Castilla elastica</i> (Moraceae)	Tree	American Samoa	MC, CC	3, 5
<i>Chromolaena odorata</i> (Asteraceae)	Herb	Guam, Saipan, Rota and Tinian (Mariana Is.), Marshall Is., Pohnpei and Yap (FSM)	MC, CC, BC	3
<i>Chrysobalanus icaco</i> (Chrysobalanaceae)	Shrub	Raiatea (French Polynesia)	MC, CC	6
<i>Clerodendron chinense</i> (Verbenaceae)	Shrub	Pohnpei (FSM)	MC	4
<i>Clidemia hirta</i> (Melastomataceae)	Shrub	American Samoa, Fiji, Palau	MC, BC	3, 4
<i>Coccinia grandis</i> (Cucurbitaceae)	Vine	Pohnpei (FSM), Guam and Saipan (Mariana Is.), Tutuila (American Samoa)	MC, CC, BC	3, 4
<i>Imperata cylindrica</i> (Poaceae)	Grass	Palau, Yap (FSM)	Not reported	3
<i>Leucaena leucocephala</i> (Mimosaceae)	Tree	Fiji	MC, CC	3
<i>Merremia peltata</i> (Convolvulaceae)	Liana	Palau, Vanuatu	MC, CC	2, 3
<i>Miconia calvescens</i> (Melastomataceae)	Tree	French Polynesia	MC, CC, BC	1
<i>Mikania micrantha</i> (syn. <i>M. scandens</i>) (Asteraceae)	Vine	Fiji, Palau, Yap (FSM)	BC	3, 4
<i>Mimosa diplotricha</i> (syn. <i>M. invisa</i>) (Mimosaceae)	Shrub	Guam, Niue, Palau, Pohnpei (FSM), Saipan and Tinian (Mariana Is.), 'Uvea (Wallis et Futuna)	MC, CC, BC	3
<i>Piper aduncum</i> (Piperaceae)	Shrub	Fiji	MC	4
<i>Piper auritum</i> (Piperaceae)	Shrub	Pohnpei (FSM)	MC	3, 4
<i>Psidium cattleianum</i> (Myrtaceae)	Tree	American Samoa, Rapa (French Polynesia)	MC	3
<i>Rhodomyrtus tomentosa</i> (Myrtaceae)	Shrub	Raiatea (French Polynesia)	MC, CC	6
<i>Robinia pseudo-acacia</i> (Fabaceae)	Tree	Rapa Nui	MC, CC	4
<i>Schefflera actinophylla</i> (Araliaceae)	Tree	Palau, Pohnpei (FSM), Tahiti (French Polynesia)	MC	3, 4
<i>Spathodea campanulata</i> (Bignoniaceae)	Tree	American Samoa, Palau, Pohnpei, Chuuk and Yap (FSM), Tahiti (French Polynesia)	MC, CC	3, 4
<i>Sphagneticola</i> (syn. <i>Wedelia</i>) <i>trilobata</i> (Asteraceae)	Herb	Fiji	MC	4
<i>Syzygium cumini</i> (Myrtaceae)	Tree	Mauke (Cook Is.)	MC, CC	4
<i>Syzygium jambos</i> (Myrtaceae)	Tree	Pitcairn (Pitcairn Is.)	MC, CC	1
<i>Thunbergia grandiflora</i> (Acanthaceae)	Vine	Pohnpei (FSM)	MC	4

species. For widespread dominant species, biological control should be carefully considered, recognizing that it is not as a miracle solution nor a high-risk method, but a tool for integrated management.

More IAP cost-benefits analysis should be realized to assess temporary *versus* long-term benefits of IAP (as well as the cost of inaction) for both convincing decision-makers and funders of the relevance of IAP management and reducing conflicts of interests between stakeholders. There is a general agreement that mitigation of the impacts of IAS is best coordinated regionally (Sherley 2000; Tye 2009), but the local context of Pacific Island countries and territories, including socio-economical context and cultural values, should be taken into account. The establishment of the Pacific Invasives Initiative (PII) and the Pacific Invasives Learning Network (PILN) have allowed for coordinated activities, shared skills and resources, linkages to technical expertise, increased information exchange, and accelerated ground-action.

But there is still a need for networks at local, regional and global level to improve the exchange of scientific information and to foster scientific training in order to have a bigger pool of invasion biologists and managers in the Pacific region, especially island-based. The Critical Ecosystems Partnership Fund for the Polynesia-Micronesia Biodiversity Hotspot has successfully funded locally-managed projects specifically dedicated to IAS in Pacific Island countries and territories between 2008 and 2012. The Pacific-Asia Biodiversity Transect (PABITRA) network or the Society for Conservation Biology Oceania section might also serve as platforms to address research and management questions on IAP. Academic training dedicated to biological invasions and IAP in Pacific islands in particular, such as the course recently developed in 2013 at the University of French Polynesia for MSc students in biology (“Master 2 Environnement Insulaire Océanien”), will contribute to the research effort.

Finally, it is recognized that strong legislation is essential for biodiversity conservation through invasive alien species prevention and control. Public support, awareness, participation and practice will also serve as the first line of defense against IAP in the Pacific Island countries and territories.

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