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REVIEW PAPER

Woody plant invasions and restoration in forests of island ecosystems: lessons from Robinson Crusoe Island, Chile

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Abstract Islands are susceptible to exotic plant invasion, and Robinson Crusoe Island (RCI), Juan Fernandez Archipelago (33°S, 78°7′W, Chile) is no exception. Through a literature review, we assessed plant invasion and compared it to other oceanic islands worldwide. Here, we discuss measures to enhance forest recovery on RCI based on knowledge accumulated from studies on RCI and other islands. Although these findings are designed to halt the progress of invasion on RCI, they could also be applied to other insular ecosystems. We addressed the following questions: (1) What is the plant invasion status on RCI in relation to other islands worldwide? (2) How imminent is biodiversity loss by plant invasion on RCI? (3) How is woody plant invasion taking place on RCI? (4) What methods are effective in controlling invasive woody species on islands worldwide? (5) What is the ability of natural forests to recover after controlling invasive plants on RCI? We found that (1) RCI is globally the fourth most invaded island for woody species. (2) Invasive woody species expansion is estimated at 4.3 ha annually. (3) Some invasive species establish under forest canopy gaps, out-competing native species. (4) Control of invasive plant species

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should focus on small gaps, and restoration should promote plant cover and soil protection. Mechanical and chemical control of invasive species seemed to be insufficient to prevent biodiversity loss. Developing alternatives like biological control are indispensable on RCI. (5) Six years after invasive species control, floristic composition tended to recover.

Keywords Chemical treatment of plant control · Invasive plant control · Juan Fernández Archipelago · Restoration · *Rubus ulmifolius* · *Aristotelia chilensis*

Introduction

Island ecosystems cover less than 5% of the world's surface but are home to around 25% of the vascular plant species on the planet (Kier et al. 2009). Since the 1600 s, most reported plant extinctions have occurred on islands (extinction is probably higher on tropic mainland, but that flora is less known and, thus more difficult to report). Twenty percent of the global plant species extinctions have occurred on islands in the Pacific Ocean (Smith et al. 1993). Considering that islands are key habitat for endemic and endangered plant species and avifauna species at local (Hahn et al. 2009; Vargas et al. 2011), regional and global scales (Myers et al. 2000; Kier et al. 2009), focusing on conserving and restoring degraded forest ecosystems on islands may prevent species extinctions. Biodiversity loss and ecosystem degradation on islands is mainly driven by direct human impact and by invasive exotic species (Kawakami and Okochi 2010; Carrion et al. 2011). Currently, management and control of exotic species is one of the biggest challenges for biodiversity conservation on islands. While hundreds of positive examples of animal control and eradication efforts for conservation have been reported on islands (Donlan and Wilcox 2007, http://diise.islandconservation.org/), almost no success has been reached for invasive plant species (Vila et al. 2001; Tassin et al. 2006). Invasive alien woody species can decrease native forest diversity and interfere with natural regeneration dynamics (Vargas et al. 2013a). The strategy of the invasive species is to progressively expand into the forests, especially when dispersed by birds (Baret et al. 2004; Smith-Ramírez et al. 2013). When the seeds of invasive plants reach a canopy gap, they can establish inside the inner non-invaded forest. Displacement of native plants by the invasive plant *Miconia calvescens*, *Bischofia javanica* and *Rubus alceifolius*, and has been reported in canopy gaps on Hawaii and Ogasawara Archipelagos and Réunion Island, respectively (Baret et al. 2004; 2008; Medeiros et al. 1997; Tanaka et al. 2010; Kawakami and Okochi 2010). The same process has been thoroughly described for *Rubus ulmifolius* and *Aristotelia chilensis* on Robinson Crusoe Island (RCI) (Vargas et al. 2013a, b; Arellano-Cataldo and Smith-Ramírez 2016). It is frequently observed that after the control of exotic plant species, the same controlled species, or other exotic plants, reestablish faster than natives, making continuous treatment, monitoring and restoration efforts necessary (Loh and Daehler 2008; Vargas et al. 2013a; Meyer 2014).

RCI is part of the Juan Fernández Archipelago National Park and World Biosphere Reserve, located about 670 km away from continental Chile at 33°S, 78°7′W. The forests on RCI are closely related to the rainforests of southern Chile, and exhibit common characteristics with the *Syzygium* communities of the subtropical islands of New Zealand, and with the Hawaiian Metrosideros-forest (Skottsberg 1953; Mueller-Dombois and Fosberg 1998). Ever since humans began to occupy RCI, disturbances like fires, selective cuttings, and the introduction of exotic animals and plants severely affected the island (Woodward 1969; Dirnböck et al. 2003). Natural abiotic disturbances such as erosion, landslides, rock falls, torrential rains, strong winds (Castro et al. 1995) earthquakes, volcanic eruptions and even tsunamis (Skottsberg 1920–1956, p. 403) have also shaped the landscape of the island. RCI has an area of 4,794 ha, of which 805.9 ha are covered by the alien invasive association formed by *Rubus ulmifolius,* a shrub, (Rosacea, 2–4 m high) and the small tree *Aristotelia chilensis* (Elaeocarpaceae, 8–10 m high, Díaz 2013; Smith-Ramírez et al. 2013). The *Rubus*-*Aristotelia* association is extremely aggressive, occupying nearly 100% of the invaded areas (Arellano 2012). In the case of trees, after disturbances the successional endemic forest recovering occurs in about 20–30 years (Vargas et al. 2010); in the case of forest ferns it could be less, probably 6–10 years (based on the growth of Crusoe ferns in the Royal Botanic Garden, Edinburgh, author's per. obs.). In the case of invasive species the successional recovering in forest gaps occurs after 5–8 years (Arellano-Cataldo and Smith-Ramírez 2016). Only three forest fragments on RCI present an extension of more than 40 ha, totaling 290 ha (Smith-Ramírez and Arellano 2013). There are about 703 ha that are patches of endemic forest $\geq$5 ha. Individual endemic trees inside an invasive matrix of *Rubus*-*Aristotelia* association constitute nearly 156 ha. In total, the remnant native forest covers only 1014.8 ha. The highly threatened biodiversity of RCI and of the others two islands that make up Juan Fernández Archipelago, has led to consider this archipelago as the number one priority to conserve worldwide (Durrell 2011).

The main goal of this research is to review worldwide findings about invaded forests on islands in order to propose restoration tasks for the endemic forests of RCI. First, in order to prioritize the global necessity for restoration, we reviewed the effect of invasive species on RCI's forest and other island forests worldwide. Then, we analyzed the capacity of the forest on RCI to resist plant invasion and retain biodiversity, considering which control and restoration techniques have been most effective. While we mostly reviewed previous findings, we also aimed at contributing with new data that can help the conservation and management decision making on RCI and other islands. Even though this is mainly a review, some unpublished information gathered by us is shown in tables and figures, and methods are briefly explained in the text. We structured this document tackling a sequence of questions from basic to applied knowledge. Specifically, we sought to answer the following questions: (1) What is the plant invasion status on RCI in relation to other islands worldwide? (2) Is loss of biodiversity imminent on RCI given the expansion of invasive plant species? To answer this question we consider the forest invasion rate on RCI. (3) How does plant invasion dynamics work on RCI? This information includes knowledge about succession, sprouting capacity, seed bank longevity of invasive plants, seed dispersal dynamics and whether expansion of invasive species continues over time. (4) What would be the best control methods to develop on RCI against invasive woody species? This question includes information regarding whether mechanical, chemical or biological methods are more appropriate for controlling invasive plants. (5) What is the recovery capacity of the forest after invasive plant control? We discuss restoration actions that could be undertaken in forest areas where the invasive plants are controlled.

What is the plant invasion status on RCI in relation to other islands worldwide?

In order to contextualize RCI's case worldwide, we conducted a literature review of natural forests on island ecosystems, which are prone to be highly invaded by woody species. We reviewed scientific articles based on Thomson Reuters Web of Science (Web of Science

2014) searching for the keywords: "island" + "invasion" + "forest" +"invasive plant control". We also included reports found in Google scholars and Google (i.e., gray literature). A remarkable species in relation to the invaded area on islands, is *Miconia calvescens* which in Tahiti occupies ca. 80.000 ha, but is dominant in only 25% of this area (76.6% of the island's total area, dominant in 19.1% of the area) and 60.000 ha in Moorea island (44.8% of the island's total area; Meyer et al. 2015). In addition, *Morella faya* invaded 30.500 ha on Big Island, Hawaii (2.9% of the island's total area) (Benitez et al. 2012), among others (see Table 1). We found that, according to what has been reported, RCI is worldwide the fourth island forest with most invaded area, with 21.5% of its territory invaded by woody species. The alien *Rubus*-*Aristotelia* association accounts for ca. 17.8% of the island's area. Another highly invasive woody species on RCI worth noting is *Ugni molinae* (Myrtaceae) that occurs on 2.6% of RCI and has increased around 3.4 ha between 2003 and 2010 (Díaz 2013; Table 1).

Table 1 The most massive invasions of woody species on islands worldwide

Islands	Invasive plant species	Invaded area (Km^2)	Invasion proportion (%)	Author
Tahiti	*Miconia calvescens*	800	76.6	Meyer and Fourdrigniez (2011)
Moorea	*Miconia calvescens*	60	44.8	Meyer et al. (2015)
Pico	*Pittosporum undulatum*	445.2	26.3	Lourenco et al. (2011)
Robinson Crusoe	*Rubus ulmifolius-Aristotelia chilensis*	8.06	16.8	Díaz (2013)
Santa Cruz	*Cinchona pubescens*	160	16.5	García and Gardener (2012)
São Miguel	*Pittosporum undulatum*	117.1	15.7	Lourenco et al. (2011)
Hahajima	*Bischofia javanica*	3.0	14.7	Tanaka et al. (2010)
Flores	*Pittosporum undulatum*	18,1	12.9	Lourenco et al. (2011)
Santa Maria	*Pittosporum undulatum*	11.7	12.0	Lourenco et al. (2011)
Azores	*Pittosporum undulatum*	238.9	10.3	Lourenco et al. (2011)
Faial	*Pittosporum undulatum*	17.6	10.1	Lourenco et al. (2011)
São Jorge	*Pittosporum undulatum*	20.2	8.3	Lourenco et al. (2011)
S. Miguel	*Pittosporum undulatum*	37.0	5.0	Lourenco et al. (2011)
Graciosa	*Pittosporum undulatum*	3,4	5.6	Lourenco et al. (2011)
Pico	*Pittosporum undulatum*	20.21	4.5	Lourenco et al. (2011)
Terceira	*Pittosporum undulatum*	13.5	3.4	Lourenco et al. (2011)
Hawaii	*Myrica faya*	304.9	2.9	Benitez et al. (2012)
Robinson Crusoe	*Ugni molinae*	1.16	2.4	Díaz (2013)
Robinson Crusoe	Exotic-trees escaped from plantations	1.11	2.3	Díaz (2013)
Chichijima	*Bischofia javanica*	50.9	2.1	Tanaka et al. (2010)

The quantified areas were determined with different methods of remote sensing, specified in the cited works. The invasion proportion (%) was calculated as the ratio of the invaded area and the total area of each island

Is loss of biodiversity on RCI imminent given the expansion of invasive plant species?

This is a fundamental question to be asked before deciding to undertake control and restoration management actions. In the case of RCÍs forests, one study based on a comparison of historical and recent records of exotic invasive plants showed that the expansion of *R. ulmifolius, A. chilensis* and other invasive woody species, such as *U. molinae*, are highly likely to reduce the endemic forest area by half in the next 80 years (Dirnböck et al. 2003). Moreover, based on a comparison of two satellite images (2003–2010), Díaz (2013) found that the expansion rate of the *Rubus*-*Aristotelia* association, *U. molinae* and other invasive trees from forest plantations (i.e., genera *Acacia*, *Cupressus*, *Eucalyptus* and *Pinus*) is annually about 0.53 ha. However, Díaz (2013) calculated only the expansion of these invasive species outside or along the forest borders, but not the invasion processes occurring inside the forest. The low resolution on QuickBird images (from 2003) made it impossible to estimate the invasion rate, through canopy gap dynamics within the forest, which is the main form that *R. ulmifolius* and *A. chilensis* use to invade the forest (Díaz 2013). Nevertheless, by using field-collected data, we were able to generate a rough estimation of the area of endemic forest likely to be invaded annually through gap dynamics. At Plazoleta El Yunque forest stand (PEY), in 7.5 ha of relatively flat ground (<20° slope), we recorded each new forest canopy gap formed during a 5-year period (2009–2014). Gap size was calculated with the ellipse formula using the longest and shortest diameters measured from the canopy gap border tree (Runkle 1982). During the 5 years, 16 canopy gaps were created either by fallen trees or broken branches (Table 2). On average, 359.14 $m^2 \pm 21.4$ (standard error) of forest gaps were created annually in the 7.5 ha stand (Table 2). If we roughly extrapolate this value to the remnant forest >1.1 ha (794 ha; Smith-Ramírez and Arellano 2013) we can conclude that 3.8 ha will be transformed annually into a *Rubus*-*Aristotelia* association. Assuming that all new gaps will be invaded, which is likely to occur (Arellano 2012), Díaz (2013) established that the third most important woody invasive species, *U. molinae*, expands at a rate of 0.4 ha annually; while trees escaping from forest plantations expanded around 0.1 ha annually. In total, the annual expansion of invasive woody species is estimated in about 4.3 ha. However, in the case of *Ugni* and plantations not all their expansion occurs by displacing native forests. The annual expansion of invasive species could be even larger if we consider all native forest remnants patches (1014.9 ha) and not only those >1.1 ha.

Considering the high rate of invasion of woody species on RCI, this ecosystem is highly likely to change into invasive scrublands over the next years, as predicted by Dirnböck et al. (2003). Since the highest amount of Critically Endangered and Endangered species of flora and avifauna are found in the forest (Vargas et al. 2011), we predict that invasive plant species will continue reducing biodiversity on RCI. This predicament has implications at species level of endemic animals: Hahn et al. (2005, 2011) and Hagen et al. (2010) have found that endemic birds and arthropods, notably prefer the endemic forests, being almost absent in the invasive scrublands of *Rubus*-*Aristotelia*.

How does plant invasion dynamics work on RCI?

If the idea is to protect RCI's biodiversity, forest restoration and invasive plant species control seem indispensable. Thus, analyzing the colonization dynamics and the seed viability of invasive species is essential in order to point proper management measures.

Table 2 Forest gaps generated in Plazoleta El Yunque stand (PEY), Robinson Crusoe, Chile (2009–2014)

Period	Gap creator species	Area (m^2)	Gap formation characteristics
2009–2010	*Fagara mayu*	159.3	Tree
2010–2011	*Nothomyrcia fernandeziana*	239.2	Tree
2010–2011	*Drimys confertifolia*	35.2	Branch
2010–2011	*Nothomyrcia fernandeziana*	132.8	Tree
2010–2011	*Nothomyrcia fernandeziana*	132.5	Tree
2010–2011	*Nothomyrcia fernandeziana*	207.4	Two trees
2010–2011	*Fagara mayu*	214.3	Branch
2010–2011	*Fagara mayu*	2.8	Branch, gap expansion
2010–2011	*Fagara mayu*	1.3	Branch, gap expansion
2011–2012	*Nothomyrcia fernandeziana*	67.7	Tree
2011–2012	*Nothomyrcia fernandeziana*	3.5	Branch
2011–2012	*Bohemeria excelsa*	2.4	Branch, gap expansion
2012–2013	*Nothomyrcia fernandeziana*	84.8	Tree
2012–2013	*Fagara mayu*	122.5	Branch
2012–2013	*Fagara mayu*	180.8	Tree
2013–2014	*Bohemeria excelsa*	209.2	Tree
Total gaps area (n = 16 gaps)		1795.7	
Average annual gap formation area (n = 5 years)		359.14	

The stand has a surface of 7.5 ha

The colonization dynamics of *Rubus-Aristotelia* association occurs through two independent processes: i) forest canopy gaps created by fallen trees and by landslides; ii) edges of forest fragments in contact with *Rubus-Aristotelia* scrublands. Forest gaps are a common disturbance in RCI forests, with sizes ranging from 13 to 350 m^2, and an average of about 160 m^2, constituting 0–45% of the native forest landscape (average 22 ± 15.8%; Vargas et al. 2010, 2013a; Arellano 2012; Vargas et al. unpublished data). As stated previously, the annual rate of canopy gap creation is estimated in 3.8 ha/year, but the annual landslide rate is unknown. In Fig. 1 we show an invasion dynamic model based on data collected in two forest stands of RCI: PEY (stand 1) and Villagra (stand 2). The endemic tree seedlings and sapling species are dominant after the first year of the canopy gap creation, but five years later, juveniles of *Rubus* and *Aristotelia* out-compete the endemic species, ultimately excluding them from the gap (Arellano-Cataldo and Smith-Ramírez 2016; Fig. 1). Vegetative expansion of invasive species by stolons occurs rapidly from the few individuals established in the gaps by seeds. In addition, the mound and pit parts of the gap-maker trees have been found to favor the establishment of invasive species (Fig. 1; Arellano-Cataldo and Smith-Ramírez 2016). Several studies have considered *Aristotelia* and *Rubus* as shade intolerant species both in continental Chile and on RCI (e.g., Donoso 2006; Vargas et al. 2010; Arellano-Cataldo and Smith-Ramírez 2016). The studies that measured light o transmittance in the forest understory and canopy gaps on RCI forests, have determined that canopy gap areas perceived significantly more lighter than ecotone and closed forest areas. Thus, invasive species take advantage easily of gap expansions, which are frequent on RCI [i.e., gap enlargement due to the falling of gap bordering trees (Vargas et al.

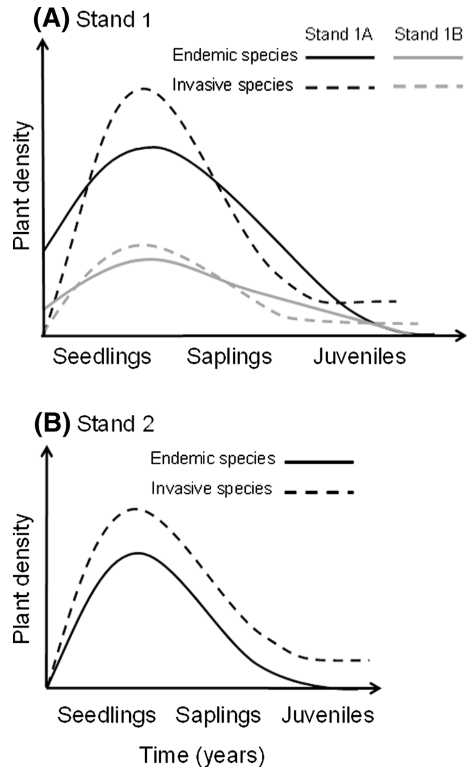

Fig. 1 Invasion models in Plazoleta El Yunque (*stand 1*) and Villagra (*stand 2*). *Stand 1A* and *Stand 2* model represent forest gaps invaded by *Rubus* and *Aristotelia*. *Stand 1B* model represents invasion of these species in mound and pit zones generated by a gap maker tree

2013a, b)]. In the case of *Ugni*, this species needs even more light than *Aristotelia* and *Rubus*, growing only in the ecotone areas or in the middle of fern formations that have low canopy (no more than 1.5 m), rocks, pastures, but not in forest gaps or under forests. To analyze whether seeds or sprouts are more important for the expansion of *Rubus,* in areas bordering treated patches of *Rubus-Aristotelia*, we sampled 16 plots (1 m^2) counting *Rubus* seedlings after one year of treatment. Seedlings produced by seeds were less important ($6.3 \pm 0.1/m^2$), than those produced by sprouts ($7.4 \pm 0.2/m^2$; $t = 2.12$; $p < 0.05$).

A major challenge for the control and eradication of invasive plants is the formation of persistent seed banks (Vivian-Smith and Panetta 2009). Efforts to eradicate species that do not form persistent seed banks as is the case of *Morella faya*, are more achievable than those involving species of long persistence, since removal might take several years (Walker 1990). For example, it took 4 years to stop the emergence of seedlings from *R. niveus* stored in the soil in certain places on Santiago Island in the Galapagos (Renteria et al. 2012). A similar control time may be required for *M. calvescens* whose seed viability is estimated between 2 and 4 years (Medeiros et al. 1997). One of the longest eradication periods has been shown by *Lantana camara* and some species of the genus *Rubus*, whose seeds can persist between 3 and 11 years (Vivian-Smith and Panetta 2009), or even for 50 years (Clark and Moore 1993).

In order to know the viability of *R. ulmifolius* seeds beneath the invasive scrubland, and its potential to form seed banks, in February 2011 we collected 400 *Rubus* seeds from the soil surface (5–7 cm depth) in three different areas. Two months after seed collection, we carried out germination experiments, finding out that all seeds were dead. The same low viability in *R. ulmifolius* seeds was found in soil samples collected in Navarra, Spain (Piudo and Cavero 2007). We then aimed at knowing the viability of fresh *Rubus* seeds in January 2010. For this we collected 400 seeds directly from fruits on RCI. A tetrazolium test was applied to a sample of 60 seeds. Almost all seeds (92%) from the sample were stained red, indicating that they were mainly alive. To test dormancy, the rest of the seeds (340) were kept at 4 °C. One year later, the tetrazolium test was repeated with a second random sample of 60 seeds, and we found that almost all seeds (90%) were dead.

Based on these findings, we suspect that *Rubus* seedlings that germinate after the invasive parent plants were removed, might be mainly fresh seeds dispersed after the area had been cleaned. We believe the seed bank is not a big issue in the case of *Rubus*, nor in the case of *Aristotelia*, which has recalcitrant seeds, and does not form seed banks (Smith-Ramírez et al. 2013).

What would be the best control methods to develop on RCI against invasive woody species?

We reviewed the scientific literature about the most effective control methods on invasive plant species (chemical, mechanical and biological) used on islands worldwide (Web of Science 2014). In the case of chemical control we were interested in the type of herbicides and doses used. We found published information and reports about control of woody plants for 47 species on islands (six of them were *R. ulmifolius* species; Table 3). The Archipelagos where the control has mainly been reported are: Hawaii, Galapagos, Micronesia, American Samoa, French Polynesia, Seychelles and Juan Fernández, among others. The mechanical and chemical control has been applied in almost all the species mentioned in Table 3. *Rubus alceifolius* on Réunion has one of the longest histories of woody plant management on islands, and the widest mechanical and chemical management (ca 240 ha) (Hivert 2003; Kueffer and Lavergne 2004). Another example of invasive plant management is *Albizia moluccana*, of which 6000 adult individuals were removed from the Samoan forests (Hughes et al. 2012). The chemical and mechanical control has been reported to be useful but insufficient to control *Rubus* in Galápagos, Rèunion and on RCI (Atkinson et al. 2010; Le Bourgeois et al. 2013; Vargas et al. 2013a).

We found that the chemical control of invasive woody species included the use of diesel (highly pollutant) and the herbicides glyphosate, metsulfuron metil, picloram and triclopyr, either pure or mixed, and in different dosages, applied by contact or aspersion. In Hawaii, after trying different methods to control invasive woody plants, it was concluded that the best method was the use of herbicide injections, which was successful against 16 different invasive woody species (Loope et al. 2013). Biological control has been used to control and even eradicate 19 woody species on 11 islands from the Pacific and Indian Ocean (Table 3). One Aracnidae, 8 fungus and 106 Insecta species (Table 3) have been released on these islands, but control success has been reported in only few cases (Meyer and Fourdrigniez 2011). One of the reported successes was the release of the fungus *Colletotrichum gloeosporioid*es for the control of the invasive Melasttaceae, *Clidemia hirta* and *Melanostoma calvescens* in Hawaii and Taiwan, respectively (Meyer and Fourdrigniez 2011).

On RCI the chemical control of *Rubus* and *Aristotelia* was used from 1999 to 2012. A mixture of Triclopyr at 1% mixed with equal parts of diesel and water was applied to control these plants. Currently diesel has been replaced by vegetable oil. The invasive

Table 3 Invasive woody species and type of control applied found in the literature review

Invasive spp.	Island	Applied control	Author
Acacia crassicarpa	Cook Islands	Mechanic NE, chemical NE	Meyer (2014)
Acacia farnesiana	French Polynesia	Mechanic NE, chemical NE	Meyer (2014)
Ageratina adenophora	Hawaii	Insecta NE (1), Insecta S (1)	Conant et al. (2013)
Albizia moluccana (syn. Falcataria moluccana)	American Samoa	Mechanic NE, chemical NE	Meyer (2014)
	American Samoa	Mechanic S	Hughes et al. (2012)
	Hawaii,	Mechanic and chemical (T) S	Hughes et al. (2012)
	Seychelles	Mechanic PS	Wiederkehr and Anderegg (2001)
Antigonon leptopus	Micronesia	Mechanic NS	Muniappan et al. (2002)
	Micronesia, Mariana Islands	Mechanic NE	Meyer (2014)
Aristotelia chilensis	Robinson Crusoe	Mechanic and chemical (T) S, control of bird seed disperser of invasive species	Vargas et al. (2011), Smith-Ramírez et al. (2013)
Bischofia javanica	Ogasawara Islands	Mechanic NS, mechanic (sapling remotion), chemical (G) S	Tanaka et al. (2010)
Castilla elastica	American Samoa	Mechanic NE, chemical NE	Meyer (2014)
Cederela odorata	Galapagos	Mechanic, chemical (P + M) NE	García and Gardener (2012)
Cestrum auriculatum	Galapagos	Mechanic, chemical (P + M, D) NE	García and Gardener (2012)
Chromoleana odorata	Micronesia	Insecta (1) PS	Muniappan et al. (2002)
Cinchona pubescens	Galapagos	Mechanic, chemical (P + M) S	Buddenhagen et al. (2004)
		Mechanic, chemical (P + M) NE	García and Gardener (2012)
Cinnamomum verum	Seychelles	Mechanic NS, chemical (G)	Beaver and Mougal (2009), Meyer (2014)
Clidemia hirta	American Samoa, Fiji, Palau	Mechanic NE, biological control NE	
	Hawaii	Insecta PS (2), insecta NS (3), Insecta NE (1), Fungi PS (1)	Conant et al. (2013), Trujillo (2005)
	Micronesia	Insecta (2) NE	Muniappan et al. (2002)
	Seychelles	Mechanic NS	Beaver and Mougal (2009)
Coccinia grandis	Hawaii	Insecta S (1), Insecta PS (1), Insecta NE (1)	Conant et al. (2013)
	Micronesia, Mariana Islands, American Samoa	Mechanic NE, chemical NE, biological control NE	Meyer (2014)
Cordia curassavica	Mauritius	Insecta S (2 in conjunction), Insecta NS (1)	Fowler et al. (2000)
Datura metel	Kiribati	Mechanic PS	Space and Imada (2004)

Table 3 continued

Invasive spp.	Island	Applied control	Author
Hypericum perforatum	Hawaii	Insecta S (2)	Conant et al. (2013)
Lantana camara	Galapagos	Mechanic, chemical (P + M) NE	García and Gardener (2012)
	Hawaii	Fungi S (1), Insecta NS (33)	Trujillo (2005)
	Mauritius	Insecta S (1), Insecta NE (2)	Fowler et al. (2000)
	Several Islands	Insecta S(2), Insecta PS (7), Insecta NS (6) Insecta NE (8), Fungi NE (1)	Conant et al. (2013)
Leucaena leucocephala	Fiji	Mechanic NE, chemical NE	Meyer (2014)
	Galapagos	Mechanic and chemical (P + M) NE	García and Gardener (2012)
Ligustrum robustum	Réunion	Mechanic and chemical (G) S	Kueffer and Lavergne (2004)
Melastoma septemnervium	Hawaii	Insecta PS (1), Insecta NS (2)	Conant et al. (2013)
Miconia calvescens	French Polynesia	Fungi S (1)	Meyer and Fourdrigniez (2011)
	French Polynesia	Mechanic NE, chemical NE, biological control NE	Meyer (2014)
	Hawaii	Mechanic S, chemical (T) S, Fungi PS (1)	Conant et al. (2013), Medeiros et al. (1997)
Mikania micrantha	Taiwan	Mechanic PS	Kuo (2003)
Mimosa diplotricha (syn. M. invisa)	Micronesia	Insecta NE (1)	Muniappan et al. (2002)
	Micronesia, Mariana Islands, Wallis, Futuna	Mechanic NE, chemical NE, biological control NE	Meyer (2014)
Myrica faya	Hawaii	Mechanic NS, chemical (P) PS, grazing (goats) NS, Insecta S (1), Insecta NS (3), Fungi NE (1)	Conant et al. (2013), Loope et al. (2013), Lutzow-Felling et al. (1995)
Passiflora edulis	Galapagos	Mechanic, chemical (P + M, G) NE	García and Gardener (2012)
Passiflora tarminiana (syn. P. tripartita, P. mollissima)	Hawaii	Insecta NS (1), Insecta NE (1), Fungi PS (1)	Conant et al.(2013), Trujillo (2005)
Piper auritum	Micronesia	Mechanic, chemical NE	Muniappan et al. (2002)
	Micronesia	Mechanic NE	Meyer (2014)
Pluchea carolinensis	Hawaii	Insecta NS (2)	Conant et al. (2013)
Psidium cattleianum	American Samoa, French Polynesia	Mechanic NE	Meyer (2014)
	Seychelles	Mechanic NS, chemical (G) PS	Beaver and Mougal (2009)
Psidium guayaba	Galapagos	Mechanic, chemical (P + M, G) NE	García and Gardener (2012)

Table 3 continued

Invasive spp.	Island	Applied control	Author
Rhodomyrtus tomentosa	French Polynesia	Mechanic NE, chemical NE	Meyer (2014)
Ricinus communis	Galapagos	Mechanic, chemical (G)	García and Gardener (2012)
Rubus adenotrichos	Galapagos	Chemical (P + M, G) S	Buddenhagen (2006)
Rubus alceifolius	Réunion	Mechanic, chemical PS, Insecta (1) S	Kueffer and Lavergne (2004), Le Bourgeois et al. (2013)
Rubus argutus	Hawaii	Insecta PS (2), Insecta NS (1), Insecta NE (2)	Conant et al. (2013)
Rubus megaloccocus	Galapagos	Chemical (P + M, G) S	Buddenhagen (2006)
Rubus niveus	Galapagos	Mechanic, chemical (G) NE	García and Gardener (2012)
Rubus ulmifolius	Robinson Crusoe	Mechanic, chemical (T) S, control of seed dispersal bird of invasive species	Smith-Ramírez et al. (2013), Vargas et al. (2013a)
Schefflera actinophylla	Micronesia, French Polynesia	Mechanic NE	Meyer (2014)
Schinus terebinthifolius	Hawaii	Insecta NS (2), Insecta NE (1)	Conant et al. (2013)
Senna surattensis	Hawaii	Fungi S (1)	Trujillo (2005)
Spathodea campanulata	American Samoa, Micronesia, French Polynesia	Mechanic NE, chemical NE	Meyer (2014)
Syzygium cumini	Cook Islands	Mechanic NE, chemical NE	Meyer (2014)
Syzygium jambos	Galapagos	Mechanic, chemical (P + M) NE	García and Gardener (2012)
	Pitcairn	Mechanic NE, chemical NE	Meyer (2014)
Thunbergia grandiflora	Micronesia	Mechanic NE	Meyer (2014)
Ulex europaeus	Hawaii	Insecta PS (2), Insecta NE (4), Arachnida PS (1), Fungi NE (1)	Conant et al. (2013)

(The number of species used for biological control is shown in brackets). S: Successful, PS: Partially Successful NS: Not Successful, NE: Succeed not established, G = Glyphosate, T = Triclopyr, P = Picloram, M = Metsulfuron metyl, D = Diesel

plants are cut and piled together to be sundried. The sprouts produced in the next season are controlled mechanically cutting them up-to 30–25 cm high, and brushing them with the same chemical mixture (Hagen et al. 2005). One of the problems with this type of control is that the selection of herbicides, dosages and application applied has not been based on a scientific study and, furthermore, after the invasive plants are removed, the soil is not covered by vegetation, catalyzing, in some cases, severe erosion processes (Castillo and Smith-Ramírez in review).

What is the recovery capacity of the forest after invasive plant control?

Once canopy gaps are invaded by the *Rubus*-*Aristotelia* association, it can cover up to 100% of the gap in a few seasons (Arellano 2012). Comparisons of natural non-invaded forest gaps (those that present <3% cover of *Rubus*-*Aristotelia*) with invaded gaps (>10% invasive cover) and treated ones (invasive removed) has made it possible to evaluate the impact of invasive species over the natives, as well as the recovery capacity of RCI forests after restoration activities (Vargas et al. 2013a, b). Invasion of gaps turned into a reduction of endemic tree species (Table 4). Control and a proper active monitoring (chemical application of Triclopyr to the sprouts after control) influenced the recovery of *Drimys confertifolia,* one of the endemic tree species. After control, species composition of ferns was similar to that previous to invasion (Table 4). This is particularly positive, considering that fern cover seems to facilitate the regeneration of the main forest species (Vargas et al. 2013b; Bastias 2014). Actually, after 2–6 years of treatment, tree regeneration represent 66% of that of non- invaded gaps, being not significantly different (Table 4). Nevertheless, the vascular species richness in RCI changes significantly after invasion and treatment (Fig. 2). Currently, after control, it seems unrealistic to expect a return to a floristic composition similar to pre-invasion. Although the data of species recovery after removal of invasive species suggests a trend towards a floristic recovery, the process is still incomplete after 6 years. Recovery is maximized in smaller gaps (<200 m^2) that present only native tree species as canopy- gap border trees (Vargas et al. 2013a). Treated gaps present a different floristic composition given the persistence of the invasive species due to seed rain and vegetative reproduction, as well as invasions by new exotic species. These are common problems experienced after attempts at controlling invasive plant species in other islands such as Galapagos and Hawaii (Jäger and Kowarik 2010; Loh and Daehler 2008). Non-treated and non-invaded gaps have a smaller proportion of exotic over native species. Exotic species appeared to take advantage of the space, lack competition and have increased resources available following gap treatment. Promoting native fern cover and limiting invasive species, in particular the cover of *Rubus* to <10% seem to be key factors for gap restoration on RCI (Vargas et al. 2013b). Large interventions (>200–300 m^2, depending on the slope) are more difficult to restore, considering that native tree species do not perform well in exposed areas, and that large interventions may also be associated with higher rates of erosion.

Discussion and management recommendations

Protection and conservation of remnant native forests on oceanic islands is a worldwide problem. The aggressiveness of some invasive species demand enormous efforts to conserve part of the threatened biodiversity on islands. In several cases where mechanical and chemical treatments were used to control woody species, these methods were either inefficient, pollutant (e.g. glyphosate) or too expensive (García and Gardener 2012). When they were environmentally successful, the restoration costs were usually extremely high (Meyer 2014). On Santiago Island (Galapagos, 585 km^2) Rentería et al. (2012*a*) estimated the cost of controlling *R. niveus* in around 10 million USD (1 million USD year − 1) over a 10 year period to achieve eradication. In RCI we estimate the control cost for 1 ha of *Rubus*-*Aristotelia* near the town (<5 km away) to be around 7612-8198 USD. This amount

Table 4 Natural regeneration of native tree species sampled in 30 canopy gaps of RCI (Plazoleta El Yunque forest)

	Extended canopy gap (n = 489 fine scale plots/ 60 regeneration plots)			Border (n = 89 fine scale plots/30 regeneration plots)			Forest (n = 56 fine scale plots/30 regeneration plots)		
	Natural (n = 160)	Invaded (n = 141)	Treated (n = 188)	Natural (n = 28)	Invaded (n = 29)	Treated (n = 32)	Natural (n = 24)	Invaded (n = 16)	Treated (n = 16)
Cover ferns (%)	34.3 ± 0.7a	8.3 ± 0.3b	35.7 ± 1.0a	48.8 ± 0.7a	19.6 ± 0.5b	22.6 ± 0.6b	27.4 ± 0.6a	25.6 ± 0.6a	26.4 ± 0.3a
Cover *Rubus* (%)	0.4 ± 0.0a	54.3 ± 1.0b	5.3 ± 0.3c	0.2 ± 0.0a	26.9 ± 0.8b	0.8 ± 0.1a	0.0 ± 0.0a	0.5 ± 0.1a	0.5 ± 0.1a
Cover *Aristotelia* (%)	0.5 ± 0.1a	16.8 ± 0.6b	5.3 ± 0.3c	0.3 ± 0.0a	9.8 ± 0.5b	2.0 ± 0.1ab	0.4 ± 0.0a	5.9 ± 0.3a	0.0 ± 0.0a
Regeneration of native tree species	(n = 20)	(n = 18)	(n = 22)	(n = 10)	(n = 9)	(n = 11)	(n = 10)	(n = 9)	(n = 11)
Nothomyrcia	5215.6 ± 93.5a	1860 ± 80.6b	3211.2 ± 132.8b	6731.2 ± 78.4a	7480.8 ± 163.8a	6768.4 ± 117.4a	3024.4 ± 59.6a	3216.8 ± 79.6a	3455.2 ± 59.5a
Fagara	650.4 ± 38.9a	118 ± 14.9a	432.4 ± 61.6a	420 ± 27.6a	0 ± 0.0a	0 ± 0.0a	123.2 ± 13.5a	0 ± 0.0a	0 ± 0.0a
Drimys	200 ± 18.4a	20 ± 2.8a	199.2 ± 18.1a	0 ± 0.0a	0 ± 0.0a	239.2 ± 19.4a	0 ± 0.0a	96 ± 8.2a	0 ± 0.0a
Rhaphithamnus	40 ± 3.8a	0 ± 0.0a	65.6 ± 5.8a	0 ± 0.0a	0 ± 0.0a	146 ± 16.5a	0 ± 0.0a	0 ± 0.0a	0 ± 0.0a
Bohemeria	0 ± 0.0a	48.8 ± 6.1a	131.2 ± 20.1a	0 ± 0.0a	0 ± 0.0a	0 ± 0.0a	0 ± 0.0a	0 ± 0.0a	0 ± 0.0a
Regeneration ≤ 0.5 m	1019.2 ± 28.7a	724 ± 32.2a	1061.6 ± 35.2a	1462 ± 34.3a	1882.8 ± 45.4a	1916 ± 48.7a	566.8 ± 25.3a	914.8 ± 18.1a	1128.8 ± 10.4a
Regeneration 0.5-2 m	2120.4 ± 44.5a	527.2 ± 31.7b	1584 ± 57.5ab	2747.6 ± 41.9a	3248.4 ± 85.2a	3006.8 ± 68.9a	1306.8 ± 28.4a	1170 ± 38.6a	1425.6 ± 32.0a
Regeneration >2 m	2966.4 ± 71.0a	795.6 ± 37.9b	1394 ± 64.0ab	2941.6 ± 37.1a	2349.6 ± 46.0a	2230.8 ± 30.1a	1274 ± 22.6a	1228 ± 41.8a	900.8 ± 29.1a
Total regeneration	6106 ± 113.2a	2046.8 ± 91.7b	4039.6 ± 141.7ab	7151.2 ± 81.7a	7480.8 ± 163.8a	7153.6 ± 132.6a	3147.6 ± 59.0a	3312.8 ± 76.1a	3455.2 ± 59.5a

We present cover and regeneration density (individuals <5 cm dbh/ha) comparing natural, invaded and treated areas in a cline gap-forest. Canopy gaps, border zones (southern border) and below forest cover zones are compared independently. Significant differences are shown with different letters (Wilcoxon test, $P < 0.05$). Sampling effort was stabilized through 1000 random samples based on 634 fine scale plots (4 m^2) and 120 regeneration plots (25 m^2). Adapted from Vargas et al. (2013b)

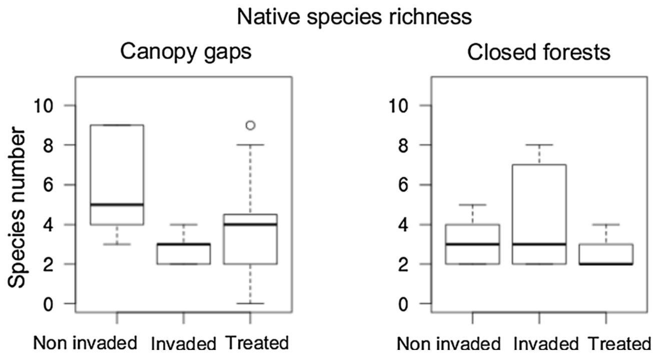

Fig. 2 Native plant species richness in 30 canopy gaps (>25 m^2) and surrounding forest areas on RCI (Plazoleta El Yunque forest). Not invaded (<5% cover of *Rubus-Aristotelia*, n = 10), invaded (>30% cover of *Rubus-Aristotelia*, n = 10) and treated gaps (sampled 2–6 years after mechanic- chemical removal of *Rubus-Aristotelia*, n = 10) are shown based on 5 × 5 m plots set either in the gap center, or inside the closed forest, south of the canopy gaps (~ 20 m)

includes costs of manpower and fungible materials, but it does not include restoration planning, organization costs or forestry tools.

In the case of RCI, future treatments of *Rubus*-*Aristotelia* should explore methods that mimic natural microsites required for the establishment of endemic forest tree species, for example, treatments with herbicides or girdling (not the removal of mature *Aristotelia)*, in combination with treatment and removal of *Rubus*. We propose not to remove dead *Aristotelia* trees since this invasive woody species has deep roots that protect the soil; they must be removed carefully, with well-planned procedures. Allowing dead adult *Aristotelia* to remain can protect the soil while the active restoration and self-regeneration proceeds. At the same time, it is necessary to cover the soil, with species such as *Gunnera* and ferns to avoid erosion. *Gunnera* is a tall forb with thick, big leaves (around 1 m diameter) that can protect the soil during several months of the year, by creating an umbrella that avoids light entrance completely, thus excluding the shade-intolerant invasive species altogether (Bastias 2014). Direct planting of endemic trees is necessary, but it should not be the predominant strategy, considering tree seedlings are browsed by rabbits and present a very slow growth rate. Moreover, on an annual basis seedlings and juvenile trees develop a lower cover for protecting the soil, compared with that of *Gunnera* or fern species (pers. observation). There are experiences in the Royal Botanical Garden of Edinburgh in which the big ferns of RCI developed large leaves (>40 cm) in only four years (pers. observation). Vargas et al. (2010) also mention that *Gunnera* may protect endemic species by shading trees, promoting their establishment. Future studies are necessary to monitor invasive species exclusion capability of *Gunnera* and ferns.

At the same time, it is necessary to look for a biological control of *Rubus, Aristotelia* and the other invasive woody species, such as *Ugni molinae*. Based on methods from other islands, Australia and continental Chile, the best invasive plant to begin with, looking for a biological control on RCI seems to be *Rubus*, using the fungus *Phragmidium violaceum* (Morin et al. 2006). In addition, it has been proposed by the National Forestry Committee (CONAF) and Smith-Ramírez et al. (2013) that, controlling the population of the only seed disperser bird of *Rubus, Aristoteli*a and *Ugni (Turdus falcklandii*, a common native bird from Chile and Argentina) could be an effective method for restricting future dispersion and expansion of invasive species. Another management action could be to allow the use of hemorrhagic diseases (such as myxomatosis or caliciviruses, and other methods) in order to eradicate rabbits on Crusoe Island. The Chilean Agricultural Law protects the introduced rabbits from these diseases, but a change in this law in the case of Crusoe Island is not only necessary but urgent.

However, how can remote and lightly populated islands such as RCI be restored, especially when it is fairly expensive to travel there? In the case of RCI, we believe that it is possible only with the support of the Chilean Government. At present, the National Park and Juan Fernandez Biosphere Reserve have an annual budget of ca. 26,000 USD for conservation issues, which is clearly insufficient for an effective management (Valladares 2015). Just as an example, the costs of eradication of invasive mammals (rats, rabbits, coatis, feral cats and goats) are estimated in 17 million USD (Saunders et al. 2011). We believe that, with the help of three Ministries: Army, Environment and Agriculture, it is possible to conserve and restore the remnant and threatened endemic forest of RCI and Juan Fernandez archipelago (Smith-Ramírez and Arellano 2013). The same strategy could be followed on other islands worldwide with the civil volunteer work, and the help of the Army to reduce the high, and sometimes unaffordable costs of mechanical control. Without the political commitment of the Chilean Government, RCI will follow the same unfortunate path as has Easter Island, another Chilean Island that suffered severe biodiversity loss and is now covered mainly by introduced plant species.

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References

Arellano GS (2012) Evaluación de la dinámica de invasión de *Aristotelia chilensis* (Elaeocarpaceae) y *Rubus ulmifolius* (Rosaceae) en claros de dosel en un bosque de la isla Robinson Crusoe, Archipiélago de Juan Fernández, Chile. Magister's Thesis, Universidad de Chile, Chile

Arellano-Cataldo G, Smith-Ramírez C (2016) Effects of small and micro-scale natural disturbances on the successful establishment of alien plant species in Robinson Crusoe's- island forest. Plant Ecol 217:289–302

Atkinson R, Guezou A, Guzman JG, Llerena A, Gardener M, Rentería JL (2010) Diagnóstico y planificación para el desarrollo de un agente de control biológico para *Rubus niveus* en las islas Galápagos. Charles Darwin Foundation and Galapagos National Park Service, Puerto Ayora

Baret S, Maurice S, Le Bourgeois T, Strasberg D (2004) Altitudinal variation in fertility and vegetative growth in the invasive plant *Rubus alceifolius* Poiret (Rosaceae), on Réunion island. Plant Ecol 172:265–273

Baret S, Cournac L, Thébaud T, Edwards P, Strasberg D (2008) Effects of canopy gap size on recruitment and invasion of the non-indigenous *Rubus alceifolius* in lowland tropical rain forest on Réunion. J Trop Ecol 24:337–345

Bastias I (2014) El rol de los helechos en la sucesión secundaria y su potencial en restauración ecológica en la isla Robinson Crusoe. Undergraduate Thesis, Universidad de Chile, Chile

Beaver K, Mougal J (2009) Review and evaluation of invasive alien species (IAS) control and eradication activities in Seychelles and development of a field guide on IAS management. Plant Conservation Action Group, Noordhoek

Benitez DM, Loh R, Tunison T, Zimmer NG, Makaike J, Mattos R, Casali M (2012) The distribution of invasive plant species of concern in the Kīlauea and Mauna loa strip areas of Hawai'i Volcanoes

National Park, 2000–2010. Technical Report No. 179. Honolulu, Hawai'i: The Hawai'i-Pacific Islands Cooperative Ecosystem Studies Unit and Pacific Cooperative Studies Unit, University of Hawai'i

Buddenhagen CE (2006) The successful eradication of two blackberry species *Rubus megalococcus* and *R. adenotrichos* (Rosaceae) from Santa Cruz Island, Galapagos, Ecuador. Pacific Conserv Biol 4:272–278

Buddenhagen CE, Renteria JL, Gardener M, Wilkinson SR, Soria M, Yánez P, Tye A, Valle R (2004) The control of a highly invasive tree *Cinchona pubescens* in Galapagos. Weed Technol 18:1194–1202

Carrion V, Donlan CJ, Campbell KJ, Lavoie C, Cruz F (2011) Archipelago-wide island restoration in the Galápagos Islands: reducing costs of invasive mammal eradication programs and reinvasion risk. PLoS ONE 6(5):e18835

Castro C, Brignardello L, Cereceda P (1995) Determinación de áreas con riesgo morfodinámico en San Juan Bautista, Isla Robinson Crusoe, comuna de Juan Fernández, V Región. Revista Geográfica de Chile Terra Australis. Revista Geográfica de Chile Terra Australis 40:43–61

Clark JR, Moore JN (1993) Longevity of *Rubus* Seeds after Long-term Cold Storage. HortScience 28:929–930

Conant P, Garcia JN, Johnson MT, Nagamine WT, Hirayama CK, Markin GP, Hill RL (2013) Releases of natural enemies in Hawaii since 1980 for classical biological control of weeds. In: Wu Y, Johnson MT, Sing S, Raghu S, Wheeler G, Pratt P, Warner K, Center T, Goolsby J, Reardon R (eds) Proceedings of the XIII International Symposium on Biological Control of Weeds. USDA Forest Service, Waikoloa, pp 224–230

Díaz I (2013) Análisis y modelación de la evolución espacio-temporal de la invasión de *Rubus ulmifolius*, *Aristotelia chilensis* y *Ugni molinae*, en la Isla Robinson Crusoe. Undergaduate Thesis, Universidad de Chile, Chile

Dirnböck T, Greimler J, Lopez P, Stuessy TF (2003) Predicting future threats to the native vegetation of Robinson Crusoe Island, Juan Fernandez Archipelago, Chile. Conserv Biol 17:1650–1659

Donlan CJ, Wilcox C (2007) Complexities of costing eradications. Anim Conserv 10:154–156

Donoso C (2006) Las especies arbóreas de los bosques templados de Chile y Argentina. Autoecología, Marisa Cúneo Ediciones, Valdivia

Durrell Wildlife (2011) Durrell Wildlife Conservation Trust. http://www.durrell.org/. Accessed 14 May 2016

Fowler SV, Ganeshan S, Mauremootoo J, Mungroo Y (2000) Biological control of weeds in Mauritius: past successes revisited and present challenges. In: Spencer NR (ed) Proceedings of the X International Symposium on Biological Control of Weeds, 4–14 July 1999. Montana State University, Bozeman, pp 43–50

García G, Gardener M (2012) Evaluación de proyectos de control de plantas transformadoras y reforestación de sitios de alto valor ecológico en Galápagos. Dirección del Parque Nacional Galápagos y Fundación Charles Darwin, Galápagos

Hagen E, Odum P, Johow F, Wainstein M (2005) Conservación del picaflor de Juan Fernández (*Sephanoides fernandensis*), especie endémica en peligro de extinción. Informe Parque Nacional Juan Fernández

Hagen E, Barker J, Gara R (2010) Aerial Arthropod Communities of Native and Invaded Forests, Robinson Crusoe Island, Chile. Environ Entomol 39:1159–1164

Hahn I, Römer U, Schlatter RP (2005) Distribution, habitat use, and abundance patterns of landbird communities on the Juan Fernández Islands, Chile. Ornit Neotr 16:371–385

Hahn I, Römer U, Vergara P, Walter H (2009) Biogeography, diversity, and conservation of the birds of the Juan Fernández Islands, Chile. Vertebr Zool 59:103–114

Hahn I, Vergara P, Römer U (2011) Habitat selection and population trends in terrestrial bird species of Robinson Crusoe Island: habitat generalists versus forest specialists. Biodivers Conserv 20(12):2797–2813

Hivert J (2003) Plantes exotiques envahissantes: état des methods de lutte mises en oeuvre par l'Office National des Forêts à La Réunion. ONF-Réunion, Réunion

Hughes RF, Uowolo AL, Togia TP (2012) Recovery of native forest after removal of an invasive tree, *Falcataria moluccana*, in American Samoa. Biol Invasions 14:1393–1413

Jäger J, Kowarik I (2010) Resilience of native plant community following manual control of invasive *Cinchona pubescens* in Galapagos. Restor Ecol 18:103–112

Kawakami K, Okochi I (eds) (2010) Restoring the oceanic island ecosystem: impact and management of invasive alien species in the Bonin Islands. Springer, Tokyo

Kier G, Kreft H, Lee TM, Jetz W, Ibisch PL, Nowicki C, Mutke J, Barthlott W (2009) A global assessment of endemism and species richness across island and mainland regions. Proc Natl Acad Sci 106:9322–9327

Kueffer C, Lavergne C (2004) Case Studies on the status of invasive woody plant species in the Western Indian Ocean, 4. Réunion. Forestry Department, Food and Agriculture Organization of the United Nations

Kuo YL (2003) Ecological characteristics of three invasive plants (*Leucaena leucocephala, Mikania micrantha,* and *Stachytarpheta urticaefolia*) in Southern Taiwan. Bulletins of Food & Fertilizer Technology Center for the Asian and Pacific Region, no. 541

Le Bourgeois T, Baret S, de Chenon RD (2013) Biological control of *Rubus alceifolius* (Rosaceae) in La Réunion Island (Indian Ocean): from investigations on the plant to the release of the biological control agent *Cibdela janthina* (Argidae). *In*: Wu Y, Johnson MT, Sing S, Raghu S, Wheeler G, Pratt P, Warner K, Center T, Goolsby J, Reardon R, eds. Proceedings of the XIII international symposium on biological control of weeds Waikoloa. USDA Forest Service, Hawaii, pp 153–160

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

Loope LL, Hughes LR, Meyer JY (2013) Plant invasions in protected areas of tropical Pacific Islands, with special reference to Hawaii. In: Foxcroft LC, Pyšek P, Richardson DM, Genovesi P (eds) Plant invasions in protected areas: patterns, problems and challenges, invading nature. Springer, Dordrecht, pp 313–348

Lourenco P, Medeiros V, Gil A, Silva I (2011) Distribution habitat and biomass of *Pittosporum undulatum*, the most important woody plant invader in the Azores Archipelago. For Ecol Manag 262:178–187

Lutzow-Felling CJ, Gardner DE, Marking GP, Smith CW (1995) Technical report 94: *Myrica faya*: Review of the biology, ecology, distribution, and control, including an annotated bibliography. University of Hawaii at Manoa Park Service, Hawaii

Medeiros AC, Loope LL, Conant P, McElvaney S (1997) Status, ecology, and management of the invasive plant, *Miconia calvescens* DC (Melastomataceae) in the Hawaiian Islands. Mus Occas Pap 48:23–36

Meyer JY (2014) Critical issues and new challenges for research and management of invasive plants in the Pacific Islands. Pacific Conserv Biol 20:146–164

Meyer JY, Fourdrigniez M (2011) Conservation benefits of biological control: the recovery of a threatened plant subsequent to the introduction of a pathogen to contain an invasive tree species. Biol Conserv 144:106–113

Meyer JY, Pouteau R, Spotswood E, Taputuarai R, Fourdrigniez M (2015) The importance of novel and hybrid habitats for plant conservation on islands: a case study from Moorea (South Pacific). Biodivers Conserv 24:83–101

Morin L, Evans K, Sheppard A (2006) Selection of pathogen agents in weed biological control: critical issues and peculiarities in relation to arthropod agents. Aust J Entomol 45:349–365

Mueller-Dombois D, Fosberg FR (1998) Vegetation of the Tropical Pacific Islands. Ecol Studies 132:773

Muniappan R, Cruz J, Bamba J (2002) Invasive plants and their control in Micronesia. Micronesica 6(Suppl.):85–92

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GA, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Piudo MJ, Cavero RY (2007) Banco de semillas del suelo y composición florística de un carrascal de Navarra (España). Bull Soc Hist Nate (Toulouse) 141:215–220

Renteria JL, Gardener MR, Panetta FD, Crawley MJ (2012) Management of the Invasive Hill Raspberry (*Rubus niveus*) on Santiago Island, Galapagos: eradication or Indefinite Control? Invasive Plant Sci Manag 5:37–46

Runkle JR (1982) Patterns of disturbance in some old-growth mesic forests of Eastern North America. Ecology 63:1533–1546

Saunders A, Glen A, Campbell K, Atkinson R, Sawyer J, Hagen E, Torres H (2011) Feasibility of managing invasive species in the Juan Fernández Archipelago, Chile. Invasive Species International, Lincoln

Skottsberg C (1920–1956) The natural history of Juan Fernandez and Easter Island. Almqvist & Wiksells Boktrickeri AB, Uppsala

Skottsberg C (1953) The vegetation of Juan Fernández islands. In: The natural history of Juan Fernández and Easter island, Uppsala, pp 793–960

Smith FD, May RM, Pellew R, Johnson TH, Walter KR (1993) How much do we know about the current extinction rate? Trends Ecol Evol 8:375–378

Smith-Ramírez C, Arellano G (2013) Necesidad de que las Fuerzas Armadas de Chile realicen control de plantas invasoras en el archipiélago de Juan Fernández, Chile. Bosque 34:3–6

Smith-Ramirez C, Arellano G, Hagen E, Vargas G, Castillo J, Miranda A (2013) El rol de *Turdus falcklandii* (Ave: Passeriforme) como dispersor de plantas invasoras en el archipiélago de Juan Fernández. Revista Chilena de Historia Natural 86:33–48

Space JC, Imada CT (2004) Report to the Republic of Kiribati on invasive plant species on the islands of Tarawa, Abemama, Butaritari and Maiana. U.S. Forest Service and Bishop Museum, Honolulu

Tanaka N, Fukasawa K, Otsu K, Noguchi E, Koike F (2010) Eradication of the invasive tree species *Bischofia javanica* and restoration of native forests on the Ogasawara Islands. In: Kawakami K, Okachi I (eds) Restoring the oceanic island ecosystem. Springer, New York, pp 161–171

Tassin J, Rivière JN, Cazanove M, Bruzzese E (2006) Ranking of invasive woody plant species for management on Réunion Island. Weed Res 46:388–403

Trujillo EE (2005) History and success of plant pathogens for biological control of introduced weeds in Hawaii. Biol Control 33:113–122

Valladares A (2015) Análisis estratégico de la labor de CONAF en la conservación de la biodiversidad del Archipiélago Juan Fernández. Undergraduate Thesis. Universidad de Chile, Chile

Vargas R, Cuevas JG, Le Quesne C, Reif A, Bannister J (2010) Spatial distribution and regeneration strategies of the main forest species on Robinson Crusoe Island. Revista Chilena de Historia Natural 83:349–363

Vargas R, Reif A, Faúndez MJ (2011) The forest of the Robinson Crusoe island, Chile: an endemism hotspot in danger. Bosque 32:61–70

Vargas R, Gärtner S, Alvarez M, Hagen E, Reif A (2013a) Does restoration help the conservation of the threatened forest of Robinson Crusoe Island? The impact of forest gap attributes on endemic plant species richness and exotic invasions. Biodivers Conserv 22:1283

Vargas R, Gärtner SM, Hagen E, Reif A (2013b) Tree regeneration in the threatened forest of Robinson Crusoe Island, Chile: the role of small-scale disturbances on microsite conditions and invasive species. For Ecol Manag 307:255–265

Vila M, Garcia-Bberthou E, Sol D, Pinto J (2001) Survey of the naturalised plants and vertebrates in peninsular Spain. Ecologia Mediterranea 27:55–67

Vivian-Smith G, Panetta F (2009) Lantana (*Lantana camara*) seed bank dynamics: seedling emergence and seed survival. Invasive Plant Sci Manag 2:141–150

Walker L (1990) Germination of an invading tree species (Myrica faya) in Hawaii. Biotropica 22:140–145

Wiederkehr F, Anderegg M (2001) Problems with *Paraserianthes falcataria* on Mahé, *Seychelles*. Master's thesis, Geobotanical Institute ETH, Switzerland

Woodward RL (1969) Robinson Crusoe's island: a history of the Juan Fernández Islands. University of North Carolina Press, Chapel Hill