Invasive Plant Science and Management 2012 5:37–46

Management of the Invasive Hill Raspberry (*Rubus niveus*) on Santiago Island, Galapagos: Eradication or Indefinite Control?

Jorge Luis Renteria, Mark R. Gardener, F. Dane Panetta, and Mick J. Crawley*

The eradication of an invasive plant species can provide substantial ecological and economic benefits by eliminating completely the negative effects of the weed and reducing the high cost of continuing control. A 5-yr program toward the eradication of hill raspberry (*Rubus niveus* Thunb.) in Santiago Island is evaluated using delimitation and extirpation criteria, as well as assessment of the ecological community response to management techniques. Currently, hill raspberry is located in the humid zone of Santiago island. It is distributed over three main infestations, small patches, and many scattered individuals within an area of approximately 1,000 ha. New infestations are constantly being found; every year, new detections add an area of approximately 175 ha. Adult and juvenile individuals are still found, both beyond and within known infestations. Both plant and seed bank density of hill raspberry decreased over time where infestations were controlled. Species composition in the seed bank and existing vegetation were significantly different between areas under intensive control and adjacent uninvaded forest. After 5 yr of intensive management, delimitation of hill raspberry has not been achieved; new populations are found every year, increasing the infested area that requires management. Off-target effects on native species resulting from control efforts seem to be substantial. Although a vast increase in economic investment would allow intensive searching that might enable all individuals to be found and controlled, the resultant disturbance and off-targets effects could outweigh the conservation benefits of eradication.

Nomenclature: Hill raspberry, *Rubus niveus* Thunb.

Key words: Eradication, delimitation, extirpation, invasive, weed.

Eradication is defined as the elimination of every individual of a species (including propagules) from an area in which recolonization is unlikely to occur (Myers et al. 1998). When managing invasives, eradication is the preferred course of action because other alternatives (such as containment or control to a level below a determined threshold) require ongoing investment of resources, unless an effective biological control can be found (Cunningham et al. 2004; Panetta and Timmins 2004; Wittenberg and Cock 2001; Zavaleta 2000). For natural ecosystems, the ultimate goal of an eradication program is either to prevent negative effects on diversity and ecosystem function or to reverse such effects once they have occurred (Zavaleta et al. 2001). However, a careful analysis of eradication costs and likelihood of success must be made and adequate resources mobilized before eradication is attempted (Panetta 2009; Wittenberg and Cock 2001). It has been suggested that only the eradication of weeds occupying less than 1,000 ha is likely to be realistic (Rejmánek and Pitcairn 2002), although eradications over larger areas have been successful (Woldendorp et al. 2004). Most of the successfully completed eradication programs were inexpensive because the infestations were small or some other element (e.g., limited seed viability) contributed to the feasibility of the effort (Buddenhagen 2006; Tye 2007; Woldendorp et al. 2004).

DOI: 10.1614/IPSM-D-11-00043.1

* First and fourth authors: Graduate Student and Professor, Department of Biology, Imperial College London, Silwood Park Campus, Ascot, Berkshire SL5 7PY, UK; second author: Charles Darwin Foundation, Puerto Ayora, Galapagos Islands, Ecuador and Research Institute for Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia; third author: Principal Scientist/Professional Leader, Invasive Plant Science, Biosecurity Queensland, Department of Employment, Economic Development and Innovation, Ecosciences Precinct, GPO Box 267, Brisbane Qld 4001, Australia. Corresponding author's E-mail: j.renteria07@imperial.ac.uk

Management Implications

During the last two decades, the Galapagos National Park Directorate and the Charles Darwin Foundation have been carrying out control and eradication programs to restore natural areas that have been degraded by invasive plant species. Control has proved to be costly and perennial and at best has slowed the spread of invasives. Similarly, eradication has had limited success, with only four (targeting weeds with small distributions and transient seed banks) out of 30 programs reaching the goal. Hence, there is an obvious need to evaluate the current 5-yr eradication program targeting hill raspberry in Santiago Island. Eradication does not seem feasible with the current search methodology, primarily because of the failure to find all plants before they fruit. Therefore, one of the fundamental requirements for successful eradication is not being met. Increase in the frequency and extent of search and control operations would be needed to meet this requirement, with all known and potential sites being visited at 4- to 6-month intervals to prevent fruit production. Although larger infestations can be seen from a helicopter, the discovery of all individual plants would require cutting a very closely spaced network of paths through nearly impenetrable low spiny forest. This more effective search methodology would need to be implemented and maintained until the elimination of the seed bank has been achieved (a minimum of 4 yr and up to 10 yr). It is estimated that it would cost USD 10 million (USD 1 million yr^{-1}) over 10 yr to achieve eradication, a 6.7-fold increase in investment from the current level. Additionally, although herbicide control successfully kills individual plants, it has also affected natural vegetation, and opening a tight network of tracks would also cause concomitant disturbance. If the increased investment is not available or the off-target effects are considered greater than the benefits of eradication, or both, alternative objectives and methodologies (e.g., biocontrol) must be considered for the management of hill raspberry.

Eradication of invasive species depends on the interplay of biological, operational, socio-political, and economic factors (Cacho et al. 2007; Gardener et al. 2010a; Simberloff 2003). Once the first three factors have been addressed, the required investment will ultimately determine whether eradication is feasible, a factor that affects all control programs, even when eradication is not the goal (Panetta 2009; Simberloff et al. 2005). Some other complementary factors to be considered are the prevention of reinvasion and effects on off-target species (Myers et al. 1998; Panetta 2009).

In contrast to eradication efforts that target pest animals, plant eradication program can be protracted owing to the presence of persistent seed banks and difficulties in detecting the organisms (seedlings are often small and indistinguishable) (Gardener et al. 2010b; Panetta 2004; Simberloff 2003). Plant eradication programs often require 10 yr or more to complete (Cacho et al. 2006; Mack and Lonsdale 2002). Knowledge of the extent of a weed incursion (the "delimitation" criterion) and detectability are considered fundamental for eradication success because ongoing spread and reinvasion of treated areas will occur where any infestations remain undetected and thus uncontrolled (Cacho et al. 2007; Panetta and Lawes 2005).

When eradication is not considered feasible with available resources, the next in the sequence of management options is containment. As for eradication, containment requires delimitation of extent, but its main objective is to prevent or reduce further spread. This is a challenge, particularly if the plant has effective dispersal mechanisms (Panetta 2009; Wittenberg and Cock 2001). The final option is sustained control. The objective here is to reduce the density of an invasive organism to below an acceptable threshold of impact (Wittenberg and Cock 2001; Zavaleta et al. 2001). This strategy is typically implemented in sites where their effect is particularly unacceptable because of important values such as biodiversity and signifies an ongoing investment (i.e., site-led control sensu Timmins and McAlpine 2008).

When managing invasive plants in natural ecosystems, the most common objectives are the enhancement of wildlife habitat and the restoration and maintenance of native plant communities (Rice and Toney 1998). Assessment of the germinable seed bank could provide an indication of which species could colonize areas after control of the weed (Panetta 1982, 2004; Rice and Toney 1998; Turner et al. 2008). Composition of the seed bank in controlled areas can also be used as a measure of both eradication and restoration progress (Panetta 1982, 2004; Panetta and Groves 1990).

Because eradication programs for plants generally require relatively long-term funding and institutional commitment (Panetta and Lawes 2007; Panetta and Timmins 2004; Simberloff 2003), it is important that progress toward the eradication objective be evaluated so that potentially successful programs can be distinguished from those that are destined to become indefinite control efforts (Bomford and O'Brien 1995; Panetta and Lawes 2007; Panetta et al. 2011). Such evaluations allow managers to change objectives, investment, and planning. A 5-yr program to eradicate hill raspberry (*Rubus niveus* Thunb.) from the uninhabited island of Santiago in the Galapagos National Park has reached the stage where evaluation is necessary. Some unique elements of this program offer an opportunity to learn more about the eradication process. It is unclear when hill raspberry arrived in Santiago, but it has been present at least since 2001, when it was first discovered. Hill raspberry originally had a limited distribution, but its spread has become more obvious after goats were eradicated from the island in 2006 (Atkinson et al. 2008). Since 2006, an intensive control project has been carried out with the goal of eradication. Good records have been kept on search history, treatment, spatial location of controlled plants, and costs (transport, material, and labor).

The objective of this study was to evaluate the effectiveness of the current project targeting hill raspberry on Santiago Island. Using a series of criteria developed by Panetta and colleagues, we investigated, in greater detail, the effectiveness of the program. We then assessed the community response to management action by comparing the composition of the seed bank and vegetation where hill raspberry had been controlled with that in adjacent uninvaded vegetation.

Materials and Methods

Site Description. Santiago Island is an uninhabited island in the center of the Galapagos archipelago, large (58,465 ha) and high enough (the highest point is 908 m above sea level [asl]) to support many vegetation types, and contains a rich biodiversity, including single-island endemics (Atkinson et al. 2008; Carrion et al. 2007). For more than 100 yr, the unique flora and fauna of Santiago were highly affected by introduced large herbivores (Atkinson et al. 2008; Hamann 1981; Tye 2003). The eradication of these herbivores has led to the rapid recovery of the native vegetation, including population increases of threatened single-island endemics (Atkinson et al. 2008; Lavoie et al. 2007). However, the release from herbivory has also increased the abundance and distribution of introduced plants, including one of the worst weeds on the archipelago, hill raspberry (Atkinson et al. 2008).

Hill Raspberry Biology. This perennial shrub is native to India, southeastern Asia, the Philippines, and Indonesia. It was cultivated throughout the world owing to its heavy fruit production (Morton 1987), although there is little evidence of its value as a commercial crop. Seedlings are shade tolerant and can reach maturity at 6 to 8 mo. Hill raspberry can form a large seed bank, with up to 7,000 seeds m^{-2}, that persists for several years (Landázuri 2002; Ruiz Cevallos 1992; Soria 2006). It grows rapidly from seed and produces root suckers and daughter plants from stem tips. It is known to be dispersed by rats and both introduced and native birds in Galapagos (Buddenhagen and Jewell 2006; Guerrero and Tye 2009; Landázuri 2002; Soria 2006). Hill raspberry has been reported as an invasive species in the Galapagos Islands and Hawaii (Itow 2003; Motooka et al. 2003) and is an emerging threat in Central America, the United States, Australia, and South Africa (FCD and DPNG 2009). In Galapagos, hill raspberry grows in a wide range of habitats, mainly in the humid highlands. It forms dense thickets, replacing native vegetation and threatening many rare endemic plants. Hill raspberry renders farmland useless and is difficult and expensive to control. It is estimated that 30,000 ha are already invaded by this weed, and its potential distribution in the Galapagos archipelago could be as large as 90,000 ha (Atkinson et al. 2008).

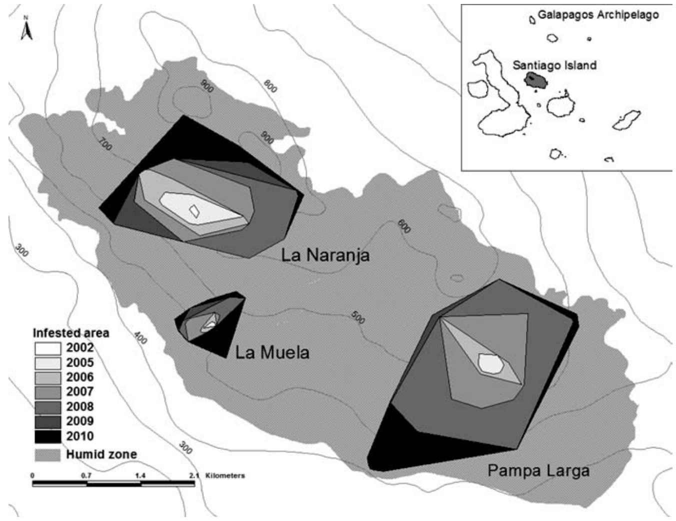


Figure 1. Spatial distribution of hill raspberry on Santiago from 2002 to 2010. Shading shows new areas of infestation found each year.

Search and Control Strategies. Control of hill raspberry on Santiago Island began in 2001 when some plants were discovered by goat hunters in the site "La Naranja" (Figure 1). It is not known when hill raspberry arrived in Santiago, but it is likely that the species had been present at low density for some years before being detected. The mechanism of its arrival is debated; it could be the result of inadvertent introduction by conservation managers or scientists who visit the island or by bird dispersal from nearby inhabited islands.

From 2001 to 2005, the control of hill raspberry was carried out as a complementary activity to goat hunting; plants found while searching for goats were marked by Global Positioning System (GPS) and then controlled manually or with the herbicide glyphosate (Rentería et al. 2006). Plants were discovered at "La Muela" in 2003 and "Pampa Larga" in 2005 (Figure 1). During 2005 and 2006, the Charles Darwin Foundation (CDF) and the Galapagos National Park Directorate (GNPD) undertook some trips specifically for the control of hill raspberry in Santiago Island, officially starting the campaign for the eradication of this species. Because of its reputation as a serious invader, it was immediately concluded that eradication should be attempted. It was not until 2007 that a project involving systematic control of the known infestations and surveillance for new infestations was established. Intensive surveillance, using equidistant points at a spacing of 5 m, was carried out in the main infestations. Since 2006, information about patch size, location, reproductive status, search area, control history, and seedling recruitment has been collected and stored in a database. Known infestations have been treated every 3 to 4 mo.

Surveillance. In 2007, a buffer of 200 m around each of the main infestations was included in the search area. Since 2007, field surveys have been conducted on foot by a field team of six people walking along pre-established parallel transects guided by GPS units. As a result of goat eradication, the density of the five dominant native woody species, especially the spiny shrub *Zanthoxylum fagara* (L.) Sarg., has markedly increased, making movement and effective searching difficult. Because of this difficulty, the search method was changed in 2008. The new method involved cutting paths at 50-m intervals across the entire width of the humid zone and searching the surrounding area from horseback (giving greater visibility). A total of 180 new paths (each approximately 2 km long) have been cut, covering an area of 1,800 ha in 2010. A complete search of the entire area is carried out over a period of 12 mo. In addition to the ground survey, three systematic helicopter searches (2007, 2009, and 2010) were completed over the total area. This method has been very effective for detecting isolated adult plants beyond the established search areas. The plants located using the helicopter were subsequently controlled, and these new areas were included in the systematic search.

Control Method. Small plants were removed by hand, and adult plants were subjected to a foliar spray. Until 2007, Roundup (glyphosate) at 2% was the main herbicide used. In 2008, a switch was made to a chemical with a longer residual time in the soil, thus aiding in the control of plants arising from the seed bank (Truper [picloram] at 1%) (Santos et al. 1991).

Data Acquisition. Information on the date of discovery, location, area of each infestation and management were acquired for each infestation from 2006 to 2010 and were sourced from notes, field records, and reports from the GNPD and CDF. Management of hill raspberry in Santiago has been carried out for 6 yr at a cost of approximately USD 582,000 (G. Garcia, personal communication).

Seed Bank Sampling. The seed bank is a key factor for persistence of hill raspberry. To develop a quantitative measure of success, we used presence of seed bank (detectable seed bank) vs. time since control. We assumed all patches had the same age (i.e., same initial seed bank) and seed input had not occurred since first control. Soil samples were taken from sites where intensive control had been carried out for different periods of time: 5, 4, 3, 1, and 0 yr of control (n = 8, 6, 7, 8, and 6, respectively). In each control site, paired soil samples were taken from the adjacent uninvaded forest to assess the seed bank composition. Five samples (4.5 cm in diameter by 5 cm deep) were collected within a 1-m^2 quadrat using a metal cylinder and then combined. Soil was put in trays, watered regularly, and not allowed to dry out in a shade house to maximize germination. Species and numbers of seedlings were recorded. To determine the effect of control measures on vegetation composition and structure, a percent cover estimation of all abundant species was made in a plot 3 by 3 m in the same invaded and uninvaded plots.

Assessment against Eradication Criteria. Infested areas (''net area'') were defined spatially by the convex polygons generated from all recorded GPS points that incorporated the outermost plants in an infestation. ''Gross area'' was defined as the area covered for monitoring purposes; it incorporated the limits of the search effort in each year (Gardener et al. 2010b; Panetta and Lawes 2005; Panetta and Timmins 2004).

Delimitation (D) was assessed in any given year using the formula from Panetta and Lawes (2007):

$$D_n = A_d / [P_n + \log(A_s + 1)] \quad [1]$$

where A_d represents the area of infestation newly detected in year n, P_n represents the proportional change in total infested area between year $n - 1$ and year n, and A_s represents the area that is searched in year n. D trends to zero as delimitation approaches 100% (Brooks et al. 2009). Conformity with the extirpation criterion was assessed through examination of the trends in the numbers of controlled individuals (adult and juvenile plants) over time (Brooks et al. 2009; Panetta 2007). Information from the database was manipulated in ArcGIS. To calculate control efficacy, we selected 16 monitoring sites where control started in 2006. In each site, a quadrat (30 by 30 m) was marked. The total number of points/plants controlled within each quadrat per year was recorded.

Linear models (lm), generalized linear models (glm), and principal coordinate analysis (PCO) were used to determine patterns to assist in the evaluation of the eradication program.

Results and Discussion

Delimitation. In 2010, hill raspberry was located in three major infestations covering an approximate net area of 920 ha. Its distribution extends from 400 m asl to the highest part of the island (Figure 1). Worldwide, hill raspberry has an extensive climatic range; it is found as high as 3,000 m in its native range and as low as 100 m in the tropics (FCD and DPNG 2009; Morton 1987; Weber 2003). In Galapagos, however, it seems to be limited to the humid and very humid zones, where edaphic conditions (especially soil depth, moisture-holding capacity, and fertility) may be more suitable (Atkinson et al. 2008; Hamann 2001; Itow 1995; Rentería and Buddenhagen 2006).

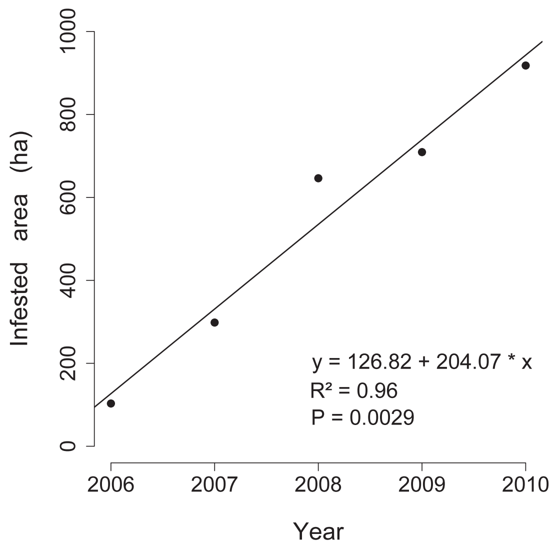


Figure 2. Cumulative net infested area (from 2006 to 2010) of hill raspberry on Santiago using minimum convex polygons.

The area known to be infested by hill raspberry has increased at a constant rate over time despite management actions (Figure 2). In 2008, the known net area increased twofold from the previous year. By 2010, the net area reached 920 ha, which represents 30% of its potential distribution in the humid zone. Assuming a conservative linear relationship between time and area of spread (Figure 2), at the current rate, it will take approximately 14 more years to reach its ecological maximum of 3,000 ha (approximate area of humid highlands) (Figure 1). New infestations were constantly being found over time. Hill raspberry net infested area is increasing at a rate of 175 ha yr^{-1}. The spatial pattern of hill raspberry within this area is not uniform, manifesting as scattered individuals, clumps, and dense patches. In 2008, infestation increased significantly as a result of additional plants found within and around the known area of infestation (Figure 3a).

In 2007, systematic searching on foot commenced over an area of 260 ha. In 2008, searching on horseback was started, covering 250 ha. Although this new surveillance strategy is time consuming to set up (necessitating wider paths), the advantage of searching from horseback is significant because the observer is above the vegetation and can detect plants readily. By 2010, the area searched (1,800 ha) increased sevenfold and represented about 60% of the humid zone of Santiago Island (Figure 3a). Helicopters have been used successfully to locate distant or large patches of mature of hill raspberry. However, because of the height from which observations are made, helicopter searches have proved less effective in locating smaller patches and individual plants. In 2010, the full extent of the planned search area was completed for the first time (1,800 ha) and resulted in the detection of 210 ha, representing three main infestations. The D values

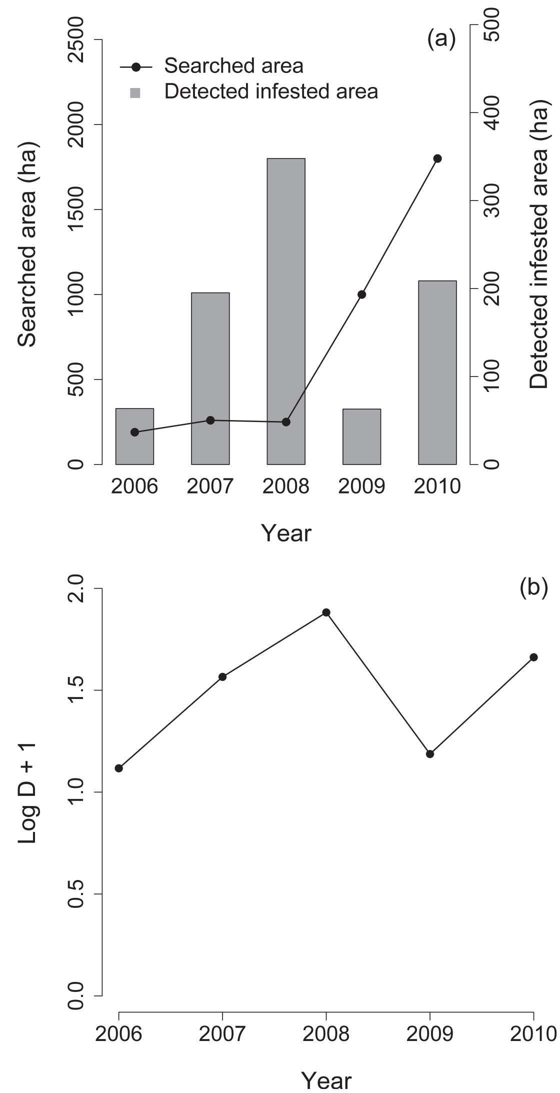
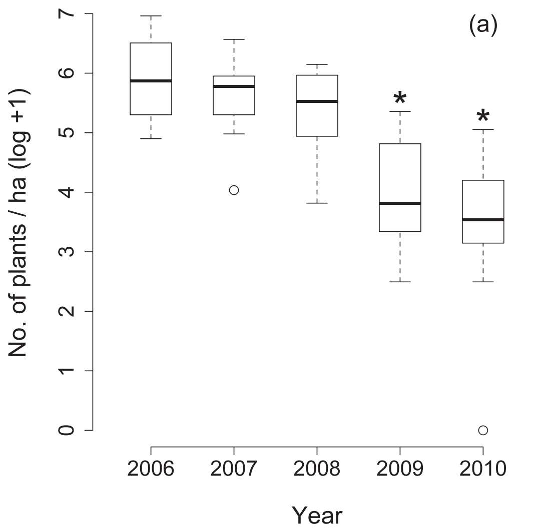


Figure 3. (a) Newly detected area infested by hill raspberry and searched area on Santiago Island from 2006 to 2010. (b) Temporal trends in the delimitation measure (D) for hill raspberry.

(Equation 1) increased because the search area continued to increase as new infestations were found (Figure 3b).

New hill raspberry populations were continually being found despite a systematic, expensive, and well-designed surveillance and control effort. Owing to thick native vegetation, distance between searchers, rapid maturation of the species, and search frequency, not all hill raspberry plants can be found before they produce fruits. Furthermore, birds continue to disperse seeds (Buddenhagen and Jewell 2006; Guerrero and Tye 2009; Landázuri 2002; Soria 2006), which makes it difficult to prevent dispersal from undiscovered fruiting plants beyond the search areas.

Extirpation Criterion. The ratio of juvenile to adult plants stayed roughly constant over time. An average of 2,700 ±



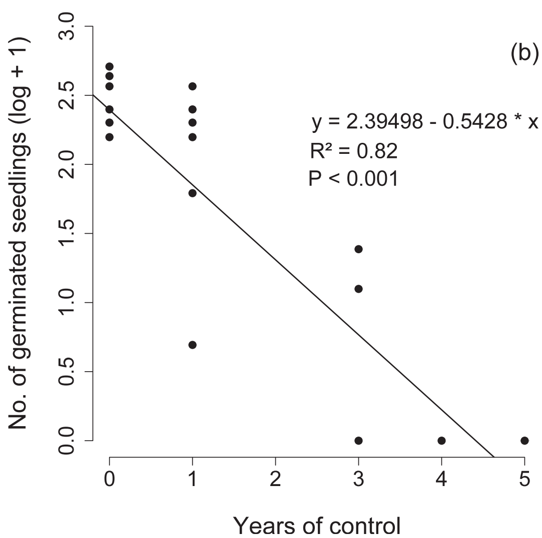


Figure 4. (a) Number of plants controlled within the known infestations over time (* lme statistically significant, df = 4, F = 48.8775, P $\leq$ 0.0001), (b) Number of hill raspberry seedlings germinated from soil samples in areas under intensive control. In year 0, soil samples were taken from recently discovered sites in which control was carried out only once. Number of sites where control had been undertaken for 2 yr was insufficient to sample.

229 juvenile and 180 $\pm$ 69 adult plants were controlled each year from 2006 to 2010. This constant number is because new infestations of adult plants were found as the search area increased. However, the density of plants declined within the area of known distribution over the last 2 yr, indicating that the rate of removal exceeded the rate of seedling emergence (Figure 4a). The mixed-effects model (lme) showed that the density of controlled plants in 2009 and 2010 was significantly less than that of previous years.

The soil seed bank of hill raspberry seems to have declined considerably in areas where systematic and intensive control has been carried out, but we cannot rule out the possibility that a few dormant seeds could remain in the soil seed bank. No hill raspberry emerged from soil taken from sites where control has been carried out for more than 4 yr (Figure 4b). This is because most plants were removed before reaching maturity and hence there was no opportunity to replenish the seed bank, which appears to become exhausted after 4 yr. According to Landázuri (2002), hill raspberry seeds buried in the soil for 1 yr still had 80% germinability, whereas Panetta (1982) found a more rapid loss in seed viability of European blackberry (*Rubus fruticosus* L.). However, evidence suggests that seed banks of some *Rubus* spp. can persist for more than 10 yr (Graber and Thompson 1978; Olmsted and Curtis 1947; Oosting and Humphreys 1940; Whitney 1986), so we cannot rule out the possibility that dormant seeds could remain for a considerable time.

Plant Community Response to Management. Composition and abundance of plant species between the area under control and the adjacent natural forest (areas without hill raspberry) were significantly different, with greater herbaceous species abundance in the controlled area. Similarly, analysis (MANOVA using two-axis scores generated by the Principal Coordinate) revealed differences between sites (controlled area and the adjacent natural forest) in the composition of the seed bank and standing vegetation (Figure 5a and 5b).

Furthermore, seed banks in the controlled area and adjacent natural forest were clearly different; there were almost no woody plants in the seed bank of the controlled area, in which annual species such as *Hyptis* spp., *Kyllinga brevifolia* Rottb., and grasses (*Paspalum conjugatum* P.J. Bergius) predominated. Although it is possible that the residual herbicide picloram influenced the emergence of seeds in controlled areas, in most cases, the herbicide had been applied more than 6 mo before (the approximate residual time in the soil), so we believe this is unlikely.

Both areas had a similar vegetation composition, with woody species (*Zanthoxylum fagara, Iochroma ellipticus* (Hook. f.) Hunz., *Tournefortia rufo-sericea* Hook. f., *Psidium galapageium* Hook. f., and *Psychotria rufipes* Hook. f.) slightly more abundant in uncontrolled areas. Similarly, the herbaceous layer was made up of the same species but *Paspalum conjugatum*, *Pteris quadriaurita* auct. non Retz., *Hyptis* spp., and hill raspberry were more abundant in the controlled areas. The fern *Ctenitis sloanei* (Poepp. ex Spreng.) Morton and herbs *Pleuropetalum darwinii* Hook., *Commelina diffusa* Brum. f., *Alternanthera halimifolia* (Lam.) Standl., and *Blechum pyramidatum* (Lam.) Urb. were more abundant in the adjacent natural forest.

Although vascular plant diversity was similar in both areas, management activity appears to have altered community dynamics; controlled areas are apparently on different trajectories (Crone et al. 2009; Marrs 1985; Rice

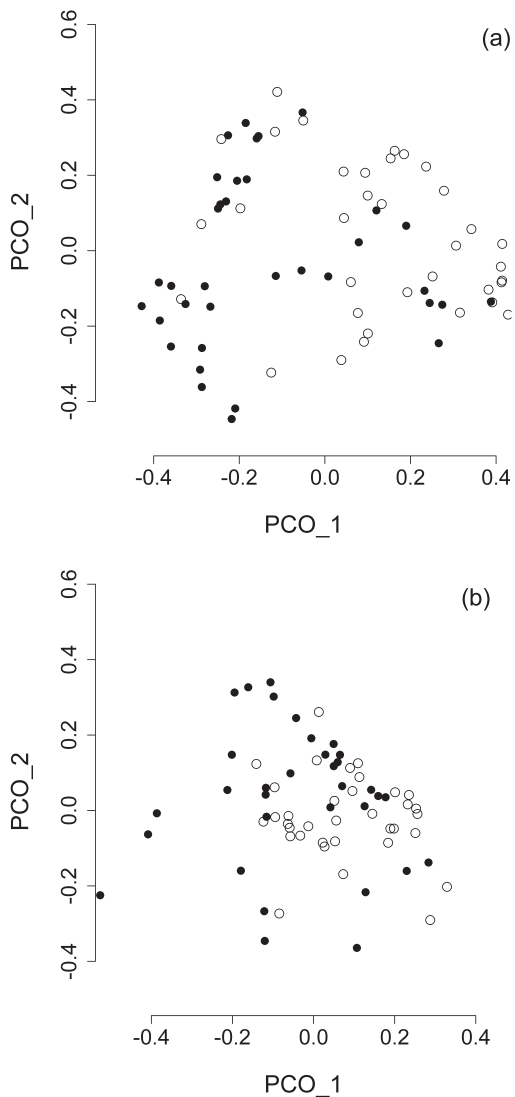


Figure 5. Principal Coordinate Analysis (PCO, $k = 2$) describing vascular plant species composition and abundance in the area where hill raspberry was controlled (filled circles) and the adjacent natural forest (empty circles). (a) Seedling species abundance (MANOVA: df = 1, F = 14.001, P = 8.31e-06); (b) species presence as determined from the vegetation survey (MANOVA: df = 1, F = 6.6917, P = 0.002241).

et al. 1997). The woody vegetation in controlled areas remained from precontrol times, having mostly survived the off-target effects of herbicide application. In general, long-lived plants often have traits enabling local populations to persist, even when habitat quality deteriorates (Eriksson and Ehrlén 2001).

Soil seed bank assessment did not detect woody species in controlled areas. The continued use of herbicides might have resulted in reduced seed production of the remnant woody individuals, and also could have affected directly the seeds in the soil (Rice et al. 1997). Nevertheless, the soil germination method might not be the most appropriate to detect the presence of woody species.

Differences in cover within strata between natural forest and controlled areas were statistically significant. Vegetation structure was clearly different between controlled infestations and adjacent natural forest, with trees and shrubs being dominant in the natural forest, and herbs (and grasses) dominating the vegetation in controlled areas (Figure 6).

After eradication of herbivores from Santiago Island, the vegetation in the humid zone has been recovering (Atkinson et al. 2008; Carrion et al. 2007; Lavoie et al. 2007; Tye 2003). However, in areas where intensive weed control has been carried out for 5 yr, the cover of trees and shrubs has been reduced considerably; the disturbance and off-target effect of intensive herbicide use has resulted in a transition from vegetation dominated by hill raspberry to grassland.

Recruitment in controlled areas comprised principally the grass *Paspalum conjugatum* and annual herbs, including the exotic *Hyptis* spp. (*H. rhomboidea* M. Martens & Galeotti and *H. pectinata* (L.) Poit.). The exotic *Kyllinga brevifolia* was abundant in the seed bank but not in vegetation (sampling was probably undertaken at the wrong time of year because *Kyllinga brevifolia*, an annual, germinates during the wet season). Disturbance from control actions facilitated the spread of several ephemeral weed species such as *Hyptis* spp. and *Bidens pilosa* L. Hence, the controlled areas appeared to be on a trajectory toward areas dominated by grass and herbs with increased exotic species abundance (Carlson and Gorchov 2004; Endress 2008; Rice et al. 1997) (Figure 6). It is unknown whether this is the end of the trajectory or whether, over the longer term, these areas might be colonized by native trees and shrubs.

Where the forest is more intact, low light conditions prevail, which suits many species, particularly ferns. Only the pioneer ferns such as *Pteridium arachnoideum* (Kaulf.) Maxon and *Pteris quadriaurita* were found in the open controlled areas. However, the light conditions in the natural forest were still not low enough to prevent the recruitment and growth of hill raspberry. The only vegetation that appears to be dense enough to prevent hill raspberry recruitment is that dominated by *Paspalum conjugatum*, but this would also prevent recruitment of almost all other plants (Endress 2008; Rice et al. 1997; Rice and Toney 1998).

After 5 yr of intensive management, delimitation of hill raspberry in Santiago Island has yet to be achieved. New populations continue to be found throughout the island on an annual basis. The surveillance techniques used thus far have not been a completely effective way of locating all individuals within the dense vegetation before maturity and prevent subsequent seed production; hence, seed dispersal

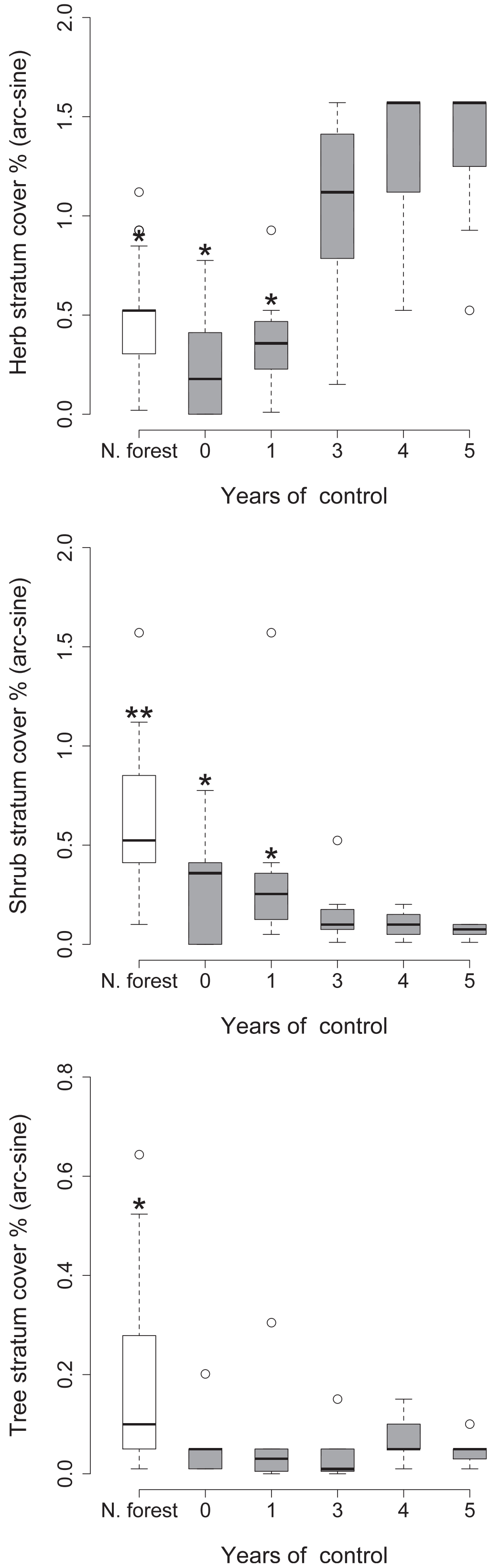


Figure 6. Vegetation cover in areas under control and adjacent natural forest (* glm statistically significant. Herbs: df = 1, F = 45.802 , P = 3.708×10^{-9}; shrubs: df = 2, F = 29.194, P = 7.62×10^{-10}; trees: df = 1 , F = 21.116, P = 1.930×10^{-5}).

appears to be ongoing. On a more positive note, most of the sites where the plant has been located are readily accessible, and control has been very effective in reducing plant density and depleting the seed bank.

Current search and control actions have not been adequate to achieve eradication. To achieve this goal, a more effective search method that covers every square meter of potential distribution in the humid highlands of Santiago (3,000 ha) would be required. This search and control operation would need to be continued for a minimum of 4 (and up to 10) yr to exhaust the seed bank and would need to guarantee that not a single plant reached maturity in this time frame. The economic and environmental costs of such an operation are difficult to estimate. Based on a net area of 28 ha, Atkinson et al. (2008) estimated it would take 15 yr at cost of USD 150,000 per year (USD 2.25 million) to reach the target of eradication. However, this was based on cutting paths at 50-m intervals to detect new infestations, not individual plants. Since this decision was made, the native vegetation has grown even denser and can be described as an impenetrable thorn thicket (dominated by *Zanthoxylum fagara*) in which an individual large plant could be seen on horseback from a maximum distance of about 10 m. Thus, for the objective of total detectability to be met, we would have to search the whole potential range of hill raspberry with search paths cut every 10 m, resulting in a total of about 3,000 km of paths. Considering the cost in 2010 was USD 154,000 to cut, maintain, search, and control 360 km of paths, we estimate that cutting 3,000 km of paths would cost USD 1.3 million dollars. Although the cost of maintenance would decrease over time, the total eradication cost could easily sum to USD 10 million over 10 yr.

The environmental impact of turning 10% of the highlands of Santiago into paths, and the need for a 10-fold increase in team size and number of horses is even more difficult to estimate. In this paper, we have already shown that control operations have increased the abundance of other weeds such as *Hyptis spp.* and *Bidens pilosa,* and it is likely that such increased disturbance and traffic would result in other well-known human-mediated introductions, such as *Desmodium* spp. and *Cleome viscosa* L. Other unforeseen consequences are also likely to occur (Zavaleta et al. 2001).

Furthermore, given that hill raspberry is distributed by birds, some of which are known to be able to fly between islands, even if eradication were possible, the possibility of reintroduction from the neighboring island of Santa Cruz (c. 40 km away), which is heavily infested with hill raspberry, still remains high.

Although the authorities are willing to manage this weed and they regard it as one of the top priorities in Galapagos, continued funding for the project at current levels is not forthcoming at present. This study has shown that the

https://doi.org/10.1614/IPSM-D-11-00043.1 Published online by Cambridge University Press

current level of investment does not prevent the further spread of the population because delimitation criteria have not been met, but it is it is unlikely that the Galapagos National Park would increase financial support for this project to the level required (which would amount to 5% of the annual budget), or have sufficient trained personnel available for a single project. However, the current level of funding (approximately USD 150,000 per year) would be sufficient to support investigation of a more long term option: biological control. Biocontrol is a conceivable long-term option in Galapagos because the weed is widespread and specificity requirements would be less restrictive than in other locations because there are no native *Rubus* or even Rosaceae. The intention would be to reduce the density and vigor of the invasive over time. However, the establishment, build-up, and spread of a biocontrol control agent usually requires years, so all the available and affordable measures should be used as an integrated management strategy. The total cost to develop a biocontrol agent has been estimated at around USD 800,000 (FCD and DPNG 2009)

A further management option would be to identify and define endemic biodiversity hotspots in the highlands, focusing on the distribution of the 24 threatened plant species as well as other taxa and to carry out intensive management in these areas. Although this might prevent species extinctions, it would not protect the unique highland ecosystem that is found in Santiago, would cause disturbance, and entail a perennial cost of management.

Acknowledgments

We thank Galapagos National Park Directorate, the managers of this project, particularly Victor Carrion and Wilson Cabrera, for supplying spatial and temporal location and control data. Rachel Atkinson designed much of the monitoring and control protocols. Linblad Expeditions paid for the control and evaluation of this work. We are grateful for the invaluable contribution to this manuscript made by Rachel Atkinson, Reitumetse Molotsane, Chris Buddenhagen, and Alan Tye; we also thank Cristina Banks and Bernardo Carreras, who provided assistance with graphics and data analysis. Claudio Crespo helped with the seed germination trials. This study is part of a Ph.D funded by the Galapagos Conservation Trust.

Literature Cited

Atkinson, R., J. Rentería, and W. Simbaña. 2008. The consequences of herbivore eradication on Santiago: are we in time to prevent ecosystem degradation again? Page 121–124 *in* L. J. Cayot and M. V. Toral, eds. Galapagos Report 2007–2008. Puerto Ayora, Galapagos, Ecuador: Charles Darwin Foundation, Galapagos National Park, INGALA.

Bomford, M. and P. O'Brien. 1995. Eradication or control for vertebrate pests? Wildlife Soc. Bull. 23:249–255.

Brooks, S., F. D. Panetta, and T. Sydes. 2009. Progress towards the eradication of three melastome shrub species from northern Australian rainforests. Plant Prot. Q. 24:71–78.

Buddenhagen, C. E. 2006. The successful eradication of two blackberry species *Rubus megalococcus* and *R. adenotrichos* (Rosaceae) from Santa Cruz Island, Galapagos, Ecuador. Pac. Conserv. Biol. 12:272–278.

Buddenhagen, C. E. and K. J. Jewell. 2006. Invasive plant seed viability after processing by some endemic Galapagos birds. Ornitol. Neotrop. 17:73–80.

Cacho, O., D. Spring, P. C. Pheloung, and S. Hester. 2006. Evaluating the feasibility of eradicating an invasion. Biol. Invasions 8:903–913.

Cacho, O. J., S. Hester, and D. Spring. 2007. Applying search theory to determine the feasibility of eradicating an invasive population in natural environments. Aust. J. Agric. Res. Econ. 51:425–443.

Carlson, A. M. and D. L. Gorchov. 2004. Effects of herbicide on the invasive biennial *Alliaria petiolata* (garlic mustard) and initial responses of native plants in a southwestern Ohio forest. Restor. Ecol. 12:559–567.

Carrion, V., C. J. Donlan, K. Campbell, C. Lavoie, and F. Cruz. 2007. Feral donkey (*Equus asinus*) eradications in the Galapagos. Biodivers. Conserv. 16:437–445.

Crone, E. E., M. Marler, and D. E. Pearson. 2009. Non-target effects of broadleaf herbicide on a native perennial forb: a demographic framework for assessing and minimizing impacts. J. Appl. Ecol. 46: 673–682.

Cunningham, D. C., S. C. Barry, G. Woldendorp, and M. B. Burgess. 2004. A framework for prioritizing sleeper weeds for eradication. Weed Technol. 18:1189–1193.

Endress, B. A. 2008. Herbicide and native grass seeding effects on sulfur cinquefoil (*Potentilla recta*)–infested grasslands. Invasive Plant Sci. Manag. 1:50.

Eriksson, O. and J. Ehrlén. 2001. Landscape fragmentation and the viability of plant populations. Page 157–175 *in* J. Silvertown and J. Antonovics, eds. Integrating Ecology and Evolution in a Spatial Context. Oxford: Blackwell Science.

FCD and DPNG. 2009. Diagnóstico y planificación para el desarrollo de un agente de control biológico para Rubus niveus en las islas Galápagos. Puerto Ayora, Galápagos, Ecuador: Fundación Charles Darwin–Dirección del Parque Nacional Galápagos. 166 p.

Gardener, M. R., R. Atkinson, and J. L. Renteria. 2010a. Eradications and people: lessons from the plant eradication program in Galapagos. Restor. Ecol. 18:20–29.

Gardener, M. R., S. Cordell, M. Anderson, and R. D. Tunnicliffe. 2010b. Evaluating the long-term project to eradicate the rangeland weed *Martynia annua* L.: linking community with conservation. Rangeland J. 32:407–417.

Graber, R. E. and D. F. Thompson. 1978. Seeds in the organic layers and soil of four beech-birch-maple stands. USDA Forest Service, Research Paper NE-401.

Guerrero, A. and A. Tye. 2009. Darwin's finches as seed predators and dispersers. Wilson J. Ornithol. 121:752–764.

Hamann, O. 1981. Plant communities of the Galapagos Islands. Dan. Bot. Ark. 34:1–163.

Hamann, O. 2001. Demographic studies of three indigenous stand-forming plant taxa (*Scalesia*, *Opuntia*, and *Bursera*) in the Galápagos Islands, Ecuador. Biodivers. Conserv. 10:223–250.

Itow, S. 1995. Phytogeography and ecology of *Scalesia* (Compositae) endemic to the Galápagos Islands. Pac. Sci. 49:17–30.

Itow, S. 2003. Zonation pattern, succession process and invasion by aliens in species poor insular vegetation of the Galápagos Islands. Global Environ. Res. 7:39–58.

Landázuri, O.d.P. 2002. Distribución, fenología reproductiva y dynámica del banco de semillas de mora (*Rubus niveus* Thunb) en la parte alta de la isla Santa Cruz, Galápagos. Undergraduate dissertation. Quito: Universidad Central del Ecuador. 184 p.

Lavoie, C., F. Cruz, G. V. Carrion, K. Campbell, C. J. Donland, S. Harcourt, and M. Moya. 2007. The thematic atlas of Project Isabela: an illustrative document describing, step-by-step, the biggest successful goat eradication project on the Galapagos Islands 1998–2006. Puerto Ayora, Galapagos, Ecuador: Charles Darwin Foundation. 58 p.

Mack, R. N. and W. M. Lonsdale. 2002. Eradicating invasive plants: hard-won lessons for islands. Page 164–172 *in* D. Veitch and M. Clout, eds. Turning the Tide: The Eradication of Invasive Species. Auckland, New Zealand: Invasive Species Specialty Group of the World Conservation Union (IUCN).

Marrs, R. H. 1985. The effects of potential bracken and scrub control herbicides on lowland Calluna and grass heath communities in East Anglia, UK. Biol. Conserv. 32:13–32.

Morton, J. 1987. Mysore Raspberry. Page 109–110 *in* J. F. Morton, ed. Fruits of Warm Climates. Miami: Florida Flair Books.

Motooka, P., L. Castro, D. Nelson, G. Nagai, and L. Ching. 2003. Weeds of Hawaii's Pastures and Natural Areas: An Identification and Management Guide. Honolulu: College of Tropical Agriculture and Human Resources, University of Hawaii.

Myers, J. H., A. Savoie, and E. v. Randen. 1998. Eradication and pest management. Annu. Rev. Entomol. 43:471–491.

Olmsted, N. W. and J. D. Curtis. 1947. Seeds of the forest floor. Ecology 28:49–52.

Oosting, H. J. and M. E. Humphreys. 1940. Buried viable seeds in a successional series of old field and forest soils. Bull. Torrey Bot. Club 67:253–273.

Panetta, F. D. 1982. In situ potential for rehabilitation of sites dominated by blackberry (*Rubus polyanthemos* Lindeb.). Weed Res. 22:1–6.

Panetta, F. D. 2004. Seed banks: the bane of the weed eradicator. Page 523–526 *in* Proceedings of the 14th Australian Weeds Conference. Sydney: Weed Society of New South Wales.

Panetta, F. D. 2007. Evaluation of weed eradication programs: containment and extirpation. Divers. Distrib. 13:33–41.

Panetta, F. D. 2009. Weed eradication—an economic perspective. Invasive Plant Sci. Manag. 2:360–368.

Panetta, F. D., O. Cacho, S. Hester, N. Sims-Chilton, and S. Brooks. 2011. Estimating and influencing the duration of weed eradication programmes. J. Appl. Ecol. 48:980–988.

Panetta, F. D. and R. H. Groves. 1990. Weed management and revegetation programs. Proc. Ecol. Soc. Aust. 16:545–549.

Panetta, F. D. and R. Lawes. 2005. Evaluation of weed eradication programs: the delimitation of extent. Divers. Distrib. 11:435–442.

Panetta, F. D. and R. Lawes. 2007. Evaluation of the Australian branched broomrape (*Orobanche ramosa*) eradication program. Weed Sci. 55:644–651.

Panetta, F. D. and S. Timmins. 2004. Evaluating the feasibility of eradication for terrestrial weed incursions. Plant Prot. Q. 19:5–11.

Rejmánek, M. and M. J. Pitcairn. 2002. When is eradication of pest plants a realistic goal? Page 249–253 *in* D. Veitch and M. Clout, eds. Turning the Tide: The Eradication of Invasive Species. Auckland, New Zealand: Invasive Species Specialty Group of the World Conservation Union (IUCN).

Rentería, J. L. and C. E. Buddenhagen. 2006. Invasive plants in the *Scalesia pedunculata* forest at Los Gemelos, Santa Cruz, Galápagos. Not. Galápagos 64:31–35.

Renteria, J. L., R. Atkinson, A. M. Guerrero, and J. Mader. 2006. Manual de identificacion y manejo de malezas en las Islas Galápagos. Second ed. Santa Cruz, Galápagos: Fundación Charles Darwin.

Rice, P. M. and J. C. Toney. 1998. Exotic weed control treatments for conservation of fescue grassland in Montana. Biol. Conserv. 85:83–95.

Rice, P. M., J. C. Toney, D. J. Bedunah, and E. C. Clinton. 1997. Plant community diversity and growth form responses to herbicide applications for control of *Centaurea maculosa*. J. Appl. Ecol. 34:1397–1412.

Ruiz Cevallos, J. R. 1992. Estudio fenologico y metodos de control de las especies: *Lantana camara* L. (''Supirrosa'') Verbenaceae y *Rubus niveus* Thunberg (''Mora'') Rosacea. Undergraduate dissertation. Quito: Universidad Central del Ecuador. 185 p.

Santos, G. L., L. W. Cuddihy, and C. P. Stone. 1991. Control of yellow Himalayan raspberry (*Rubus ellipticus* Sm.) with cut stump herbicide treatments in Hawaii Volcanoes National Park. Honolulu (HI): Cooperative National Park Resources Studies Unit, University of Hawaii at Manoa, Department of Botany. 25 p.

Simberloff, D. 2003. Eradication—preventing invasions at the outset. Weed Sci. 51:247–253.

Simberloff, D., I. M. Parker, and P. N. Windle. 2005. Introduced species policy, management, and future research needs. Front. Ecol. Environ. 3:12–20.

Soria, M. 2006. Avian seed dispersers of the invasive *Rubus niveus* (Rosaceae) in Santa Cruz Island, Galapagos, Ecuador. Master's Thesis. St. Louis: University of Missouri–St. Louis. 52 p.

Timmins, S. M. and K. G. McAlpine. 2008. Do weed managers need to know about weed impacts? Page 26 *in* Proceedings of the 16th Australian Weeds Conference. Brisbane: Queensland Weed Society.

Turner, P. J., J. K. Scott, and H. Spafford. 2008. The ecological barriers to the recovery of bridal creeper (*Asparagus asparagoides* (L.) Druce) infested sites: impacts on vegetation and the potential increase in other exotic species. Austral Ecol. 33:713–722.

Tye, A. 2003. Plant research for conservation in Galapagos. Report for the years 1998–2003. Puerto Ayora, Galapagos, Ecuador: Charles Darwin Foundation for the Galapagos Islands. 51 p.

Tye, A. 2007. Cost of rapid-response eradication of a recently introduced plant, tropical kudzu (*Pueraria phaseoloides*), from Santa Cruz Island, Galapagos. Plant Prot. Q. 22:33–34.

Weber, E. 2003. Invasive Plant Species of the World: A Reference Guide to Environmental Weeds. Wallingford, UK: CABI Publishing. 560 p.

Whitney, G. 1986. A demographic analysis of *Rubus idaeus* L. and *Rubus pubescens* Can. J. Bot. 64:2916–2921.

Wittenberg, R. and M.J.W. Cock. 2001. Invasive Alien Species: A Toolkit of Best Prevention and Management Practices. Wallingford, UK: CABI Publishing. 228 p.

Woldendorp, G., M. Bomford, S. Barry, F. D. Panetta, and D. Cunningham. 2004. Development of strategies for eradication of selected agricultural sleeper weeds. Page 492–495 *in* Proceedings of the 14th Australian Weeds Conference. Sydney: Weed Society of New South Wales.

Zavaleta, E. 2000. The economic value of controlling an invasive shrub. Ambio 29:462–467.

Zavaleta, E. S., R. J. Hobbs, and H. A. Mooney. 2001. Viewing invasive species removal in a whole-ecosystem context. Trends Ecol. Evol. 16: 454–459.

Received June 7, 2011, and approved November 30, 2011.