Prevention, Eradication, and Containment of Invasive Species: Illustrations from Hawaii

Kimberly Burnett, Brooks Kaiser, Basharat A. Pitafi, and James Roumasset

Invasive species change ecosystems and the economic services such ecosystems provide. Optimal policy will minimize the expected damages and costs of prevention and control. We seek to explain policy outcomes as a function of biological and economic factors, using the case of Hawaii to illustrate. First, we consider an existing invader, *Miconia calvescens*, a plant with the potential to reduce biodiversity, soil cover, and water availability. We then examine an imminent threat, the potential arrival of the Brown treesnake (*Boiga irregularis*). The arrival of the snake in Guam has led to native bird extirpations, power outages, and health costs.

Key Words: invasive species, bioeconomics, optimal control, *Miconia calvescens*, *Boiga irregularis*

Invasive species change ecosystems and the economic services that ecosystems provide. Policy decisions must weigh anticipated costs and benefits before a new species becomes introduced, actual costs and benefits of any mitigating actions once a species has become established, anticipated costs and benefits of accommodating the change, and the comparative advantages of preventing and/or controlling one set of ecosystem changes over another. Optimal policy regarding invasive species will minimize the expected damages and costs of control within an ecosystem.

The existing literature on the economics of invasive species has taken several complementary approaches to evaluating policy options, but to date these efforts remain fragmented across the timeline of an invasion or set of invasions. Due in part to the complexity of modeling and the specificity of biological factors involved in creating ecosystem changes, most case studies focus on a single invading species (Eiswerth and Johnson 2002, Knowler 2005, Knowler and Barbier 2000, Settle and Shogren 2002, Kaiser and Roumasset 2002) or on damages at a particular location and time (Kasulo 2000, Turpie and Heydenrych 2000). Here we examine the case of ecologically unconnected invaders in Hawaii across time, investigating both existing invasions and potential invaders, since monies available for combating damages from both types of invasions are often expected to come from the same resource pools.

Theoretical treatment of invasions introduces conditions for eradication (Olson and Roy 2002), policy options under uncertainty (Horan et al. 2002, Eiswerth and van Kooten 2002, Olson and Roy 2002), and integrated prevention and control (Pitafi and Roumasset 2005). Tying invasions to trade (Costello and McAusland 2003), ecological-economic feedback loops (Settle, Crocker, and Shogren 2002), and capital growth (Barbier and Shogren 2004), authors highlight the connections between human decision making and invasive species impacts.

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The case of Hawaii is used to illustrate dynamic policy options for invasive species. Hawaii's ecosystems provide direct and indirect ecosystem services, with high expected value generated from the preservation of existing ecosystem conditions stemming from unique biodiversity assets. Invasive species concerns extend beyond biodiversity to include human health concerns and infrastructure for both power and water supply. Each invasion presents its own set of potential damages; we seek to aid policy determinations through methodology that assesses the present value of expected damages and costs given the likelihood of a successful invasion and the optimal control or accommodation of its associated expected damages. We examine the theoretical findings with data from two cases affecting Hawaii: the weedy shrub Miconia (*Miconia calvescens*), an existing invader, and the Brown treesnake (*Boiga irregularis*), a highly likely invader.

In most if not all cases, the most costly anticipated changes are irreversible. Due to the need to anticipate irreversible change, policy decisions may vary with the status of ecosystem health, i.e., the levels of invasion and the imminence of the threat. In this work, we seek to explain how biology and economics work together to determine policy outcomes, and introduce the possibility of integrating optimal policy across and among existing and potential invasive species. To improve results and avoid costly mistakes ranging from denying beneficial introductions to spending money on ecologically impossible control or eradication efforts, these policies must be seen as a continuous effort to manage ecosystems rather than separate decisions handled as emergencies as they arrive.

Case Overview

Hawaiian Ecosystems in Perspective

Hawaiian ecosystems provide excellent grounds for studying the economic consequences of ecological change due to invasive species for several reasons. These ecosystems developed in relative isolation, and before human-aided transport, species had to traverse approximately 3,000 miles of ocean desert in any direction before reaching the islands and becoming established. Human manipulations, including purposeful and accidental introductions, which began only with the arrival of Polynesians around the fourth century AD, are fairly well identified and understood. The limited geographical scale and incomplete biota (e.g., the only pre-contact mammal present in Hawaii was a bat, now extinct) render ecological changes both visible and potentially severe. Under such isolated conditions, adaptive radiation has generated unique biodiversity in the flora and fauna that is particularly fragile and susceptible to biological invasion.

The islands also display for researchers the importance of integrated ecosystem health. The introduction of cattle by Captain Vancouver in 1794 created such noticeable change that, within half a century, denuded hillsides caused concerns about dwindling water supplies. By the beginning of the twentieth century, large tracts of land, covering most of the mountainous areas, were placed into conservation districts with cattle removed in order to protect vital fresh water that was key not only to human survival but also to the economic prospects for agriculture, particularly sugar cane.

Hawaii's development as a tourist destination has been aided by its natural capital as well. While its unique plants and birds provide many interesting viewing opportunities, missing ecological niches, especially insects, diseases, and snakes, have increased tourism values as well as human health values for residents and tourists alike. The $10 billion tourist industry dominates the state's $44 billion (GSP) economy (DBEDT 2003) and threats to the industry can inflict significant damages if realized.

Miconia calvescens

One significant threat comes in the form of the woody shrub, *Miconia calvescens*. Miconia, a member of the Melastomataceae family from Central America, was purposefully introduced to Hawaii. Planted in a handful of backyards and arboretums four decades ago, it has been spreading with increasing rapidity on the islands of Maui and Hawaii. It is also present on Kauai and Oahu, though it has not claimed significant acreage in either location.

A model of its potential expansion and damages is available through comparison with Tahiti, where dense, monotypic stands of the tree now cover 65 percent or more of the main island of Tahiti after a single specimen was introduced to the Papeari Botanical Garden in 1937 (Medeiros et al. 1997). Miconia has earned itself descriptors like the "green cancer" of Tahiti and the "purple plague" of Hawaii. Vast tracts of Miconia have wiped out native forest and reduced forest cover, increasing the potential for soil erosion, landslides, and damages to nearshore resources. The explosive growth and potential damages were not appreciated in Tahiti until the 1980s, however. By that time, Miconia was already established in Hawaii.

The damages in Tahiti and the potential threats to Hawaiian biodiversity and watersheds have rendered Miconia a priority weed in Hawaii. Since the early 1990s, millions of dollars have been spent in the battle against its spread, though success at spatial containment on Hawaii and Maui and eradication on Oahu and Kauai remains elusive. We explore quantitatively the policy options and their economic consequences for the continued treatment of the invasion in the modeling and discussion sections below.

Brown Treesnake (Boiga irregularis)

Hawaii faces several other threats from invasive species, which all must be considered simultaneously for optimal prevention and control efforts to minimize expected damages to the state's ecological assets and economy. Perhaps the most dramatic pretender to Hawaii's list of top pests is the Brown treesnake (*Boiga irregularis*). This native of Australia and New Guinea, upon establishment in Hawaii, would introduce snakes[1] to the islands and create a list of damages that include direct economic impacts as well as widespread ecological disaster.

We know the potential of the damages because the snake was introduced to the previously snake-free island of Guam in the 1950s and has since reached high-density populations of 12,000 snakes per square mile, sending thousands to the hospital with venomous bites over the last 10 years, causing the extirpation of 10 of 13 bird species, generating power outages with increasing frequency (a snake-generated outage averaged 1.5 hours every other day in 2003, up from one every 3–4 days in 1997), and reducing poultry productivity. [See U.S. Geological Survey (2005) for a recent overview of damages. Power and medical data courtesy of Shwiff (2005).]

The snake is an imminent threat to Hawaii: eight interceptions of identifiable Brown treesnakes have occurred in the past two decades [see Rodda et al. (1999) for details]. Trade between Guam and Hawaii is extensive, and Hawaii now pays to support Guam's efforts to prevent the Brown treesnake from escaping the island. We use the considerable information from Guam's infestation and expenditures on prevention to model the optimally integrated prevention and control strategy for minimizing damages from the snake.

Methodology

Optimal Control of Existing Invader

Optimal control of an existing invader is related to the size of the initial population relative to the steady state population. If the population is currently at its steady state population, optimal control requires maintenance of the population, thus generating a stream of economic costs and damages indefinitely. If the population is currently above its steady state level, we expend control costs to reduce the population to its steady state and then maintain that population. If the population lies below the steady state population, we accumulate damages as the population grows until, at the steady state population, we initiate maintenance as just described.

Optimal control of an existing invader is similar to harvesting a beneficial species [see, e.g., Clark (1990)], except that the harvest yields no benefits, and the stock of the resource generates damages. Three steady states are plausible *a priori*: eradication, accommodation without control, and maintenance of a constant population with a constant control effort. Each steady state can be associated with an optimal approach path, as discussed below. The globally optimal approach path is the one with the minimum present value of harvesting costs and damages.

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[1] With the exception of *Ramphotyphlops braminus*, a harmless blind snake that may have arrived as a stowaway in plant material from the Philippines around 1930 (http://indaba.iucn.org/archives/aliens-1/2004-09/00006446.htm, accessed March 2006).

Formally, the social planner minimizes W, where

$$(1)\qquad W=\int_0^{\infty} e^{-rt}\left(\int_{n-x}^{n} c(\gamma)d\gamma + D(n)\right)dt\,,$$

subject to

$$(2)\qquad \dot{n} = g(n)-x \qquad 0 \le x \le n\,, \qquad n_0 \text{ given,}$$

where $c()$ is the cost of removals and $D()$ are the damages from a population of n originating from an initial population n_0.

Our solution algorithm searches for a minimum over the range of candidate steady state populations, N, where $0 \le N \le N_{\max}$, and $N \in \{1,2,3\ldots\}$. This will generate an automatic comparison of corner solutions and the internal steady state solution. Depending on whether our steady state N is greater than, equal to, or less than n_0, we require two procedures for determining the present value, V, of reaching and maintaining N. Therefore V will be dependent on the initial population, n_0, and is denoted as $V(n_0,N)$. The minimum present value costs of population reduction and expansion are described below.

Population reduction. If the candidate population, N, is less than n_0, the optimal approach path involves instantaneous adjustment to the target population. Intuitively, spreading out adjustment over several years will increase removal costs (since the pest keeps multiplying) and add to the damage stream during the intervening years of adjustment. By assumption, there are no gains of spreading out the removal because the cost of removal is dependent only on the pest population, not the rate of removal.[2] The cost of that population reduction, $x = N-n_0$, will be

$$\int_{n_0}^{N} c(n)dn\,.^{3}$$

In addition, there will be the cost of maintaining population at N by cutting $x = g(N)$ every year forever, and the cost of damages incurred by remaining at that population level forever. Therefore, we define the present value of the costs and damages of reducing the invasive stock from n_0 to N, and remaining at that level in perpetuity as

$$(3)\quad V(n_0, N \le n_0) = \int_{n_0}^{N} c(n)dn + \frac{c(N)g(N)}{r} + \frac{D(N)}{r}\,.$$

Population expansion. The optimal approach to stationary N's above n_0 involves allowing the population to grow at its natural rate until the target population is reached. The damages accrued during this interval are

$$\int_{t_0}^{T} e^{-r\tau}D(N)d\tau\,,$$

where T is the time period associated with arrival at the target population (determined by its natural rate of growth). We thus define the present value of the costs and damages from allowing the stock to grow from n_0 to N, and remaining at that level in perpetuity as

$$(4)\quad V(n_0, N \ge n_0) = \int_{t_0}^{T} e^{-r\tau}D(N)d\tau + e^{-r[T-t_0]}\left(\frac{c(N)g(N)}{r} + \frac{D(N)}{r}\right).$$

The procedures described above result in one V for each of the candidate stationary populations investigated. The next step in the algorithm is simply to search across the N's and find the value of N that minimizes $V(n_0,N)$, i.e.,

$$\underset{N}{\text{Min}}\quad V(n_0, N)\,,$$

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[2] Inasmuch as removal requires some fixed investments, there are likely to be economies of scale in the range of few removals and diseconomies of scale above some high rate of removals. Knowing the exact parameters of this U-shaped average cost function is beyond the scope of the current study, however. Accordingly, we approximate the cost function using the conventional assumption that the unit removal cost is independent of the rate of removal.

[3] Note that n is used to delineate a changing population whereas N is used to delineate a steady state population. Thus, the cost of removals changes as the population moves from the initial population to a steady state N.

where

$$V(n_0, N) = \begin{cases} \displaystyle\int_{n_0}^{N} c(n)dn + \frac{c(N)g(N)}{r} + \frac{D(N)}{r}, & 0 \le N \le n_0 \\ \displaystyle\int_{t_0}^{T} e^{-r\tau} D(n) d\tau + e^{-r(T-t_0)} \left(\frac{c(N)g(N)}{r} + \frac{D(N)}{r} \right), & \\ \dot{n} = g(n), \quad n_0 \le N \le N_{\text{MAX}}. & \end{cases}$$

The optimal population that results from the minimization problem is designated as N^*, i.e.,

$$(5) \qquad V(n_0, N^*) = \operatorname*{Min}_{N} V(n_0, N).$$

If the optimal population requires eradication ($N^* = 0$), or results in a population where new arrivals from outside the area will have noticeable impacts on growth, then prevention efforts must be considered.

Optimal Prevention of Potential Invader

The existing renewable resource literature does not allow for the possibility that the resource to be harvested optimally may not yet be present in the system. The optimal control analysis of a potential invader follows directly from that of the existing invader above, but with the additional concern that the invading population is not yet viable. As we include within the definition of "potential invader" the case where small numbers of species in a new area have not yet formed a viable population, we consolidate the probabilities of arrival and establishment into one concern: if a population exists which is large enough so that the impact of new arrivals on growth is negligible, the problem is the same as that of an existing invader. If not, we must consider continuing prevention as defined by efforts taken to keep the population below a critical population level through eradication.

Ideally, we would like to simultaneously solve for optimal prevention and control with a probabilistic arrival function. Unfortunately, this would involve an extremely complicated decision tree, with branches for arrival and non-arrival at each point in time that extend infinitely into the future. Moreover, we cannot solve for prevention and control today independently of prevention and control in future periods. We simplify this problem by assuming that there is some population, $\underline{N}$, beyond which prevention expenditures are negligibly effective, relative to pest removal.[4] Prevention is not necessary if $N^* \geq \underline{N}$, since a population of $\underline{N}$ specimens is considered viable in that additional arrivals do not significantly impact current levels of reproduction. If $N^* < \underline{N}$, however, we have the case where new entries will increase the population growth sufficiently and the first-best control solution may not be the first-best integrated prevention and control solution. To determine this, we must compare whether it is preferable to remain at a steady state population, $\underline{N}$, or whether it is better to maintain $N^* < \underline{N}$ and optimally invest in prevention.

In order to maintain population at $N^* < \underline{N}$, the resource manager optimally invests in prevention of new entrants as well as control of existing ones. Inasmuch as such investment reduces the probability of a new pest arrival to a lower probability and therefore only delays the ultimate arrival, maintaining N^* requires continuous removal of snakes that evade initial prevention in addition to prevention expenditures.[5] As shown in Pitafi and Roumasset (2005), the expected present value of this prevention and removal sums the prevention expenditures and expected value of introduction followed by entrant removal expenditures over time:

$$\begin{aligned} (6) \qquad Z = \; & y + \frac{[y + p(y)\,E]}{(1+r)} + \frac{[y + p(y)\,E]}{(1+r)^2} \\ & + \frac{[y + p(y)\,E]}{(1+r)^3} + \ldots \\ & = y + \sum_{t=1}^{\infty} \frac{1}{(1+r)^{t-1}} \frac{[y + p(y)\,E]}{(1+r)} \\ & = \frac{(1+r)y + p(y)\,E}{r}, \end{aligned}$$

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[4] The stark assumption of no prevention beyond a low population can be replaced by a function representing the gradually decreasing importance of new introductions as the stock grows larger. We leave this possibility for future work.

[5] For distinction, consider initial prevention expenditures those that intercept a species before it has left a confined port of entry.

where $p(y)$ is the probability of successful introduction with prevention expenditures y, and E is the cost of removing additional entrants, the incidence of which is decreasing in y.[6] E can be calculated as $V(e_0, N)$, where e_0 is the number of entrants that escape detection given prevention expenditures y. Note that eradication and single entry removal costs may be infinite, reflecting the imperfection of detection.

Minimizing Z with respect to prevention expenditures results in the following condition for optimal spending y:

$$-\frac{p'(y)E}{(1+r)} = 1 . \tag{7}$$

Denote the minimized value of Z by $Z^*(N)$. This is the cost of prevention and control when the steady state stock is less than $\underline{N}$. Our optimal population (N^*) and level of prevention expenditures (y^*) represent a choice between combined prevention and control ($y^* > 0, x > 0, N^* < \underline{N}$) or control alone ($y^* = 0, x > 0, N^* \geq \underline{N}$).[7] We select the total cost minimizing population from the following comparisons: (i) if the population that minimizes $V(n_0, N)$ is greater than or equal to $\underline{N}$, then N^* is that population, and V is minimized according to equation (5) as in the case of an existing invader. Additional arrivals do not add meaningfully to the minimization of total costs and damages. However, if (ii) the population minimizing $V(n_0, N)$ is less than $\underline{N}$, we must evaluate whether the additional costs of the prevention/entrant-removal cycle that will be incurred after reaching any population level below $\underline{N}$ are greater or less than maintaining a controlled steady state population greater than or equal to $\underline{N}$, where additional prevention would not reduce impacts and $y^* = 0$.

The optimal prevention expenditure from equation (7) is substituted into equation (6) to obtain the minimized value $Z^*(N)$ for each population level below $\underline{N}$; in this way we know the minimized prevention costs for any population for which new arrivals increase impacts. One of these populations, N^*, will minimize the sum of $Z^*(N)$ (preventing and removing new entrants) plus the control cost of achieving and maintaining (independent of the new entrants) for $N < \underline{N}$, or $Z^*(y, N^*) + V(n_0, N^*)$. The cost of this population is compared to $V(n_0, N \geq \underline{N})$ to determine whether the optimal policy for integrated prevention and control involves (i) staying at the current population level ($n_0 = N$), (ii) reducing stock and not spending on prevention ($n_0 > N \geq \underline{N}$), (iii) reducing stock and spending on prevention ($N < \underline{N}$), or (iv) allowing the stock to grow unabated until some maintained steady state greater than the current level ($N > \underline{N} > n_0$).

Empirical Investigation

We investigate empirically the cases of Miconia and the Brown treesnake, discussed above. For each case, we determine cost, damage, and growth function parameters with the help of scientists researching the species and resource managers actively pursuing prevention or control. Throughout, we use a 2 percent discount rate. The remaining parameters are discussed below, followed by results for each case.

Miconia Calvescens

Growth function. We utilize a standard logistic growth function, $g(n) = bn\,[1-(n/N_{\max})]$, to represent the spread of the invasive tree. The intrinsic growth rate, b, is estimated here to be 0.3, and the carrying capacity, $N_{\max}$, is estimated to be 100 trees per acre covering 1 million acres, or 100,000,000. The acreage of the carrying capacity is determined by the chief limiting factor for Miconia in Hawaii, precipitation, so the potential range indicates areas above the 1800 mm/yr rainfall line. The growth rate was determined by analyzing the spread of the tree on Hawaii, where we know the origins of the first population and its spread until harvesting began in the mid-1990s.

Damage (net benefit) function. We estimate damages from Miconia as evolving from indirect ecosystem services as well as non-market goods like biodiversity. The damages may occur as

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[6] We use discrete time here for clarity of exposition.

[7] Prevention alone will be insufficient in this model because the probability of arrival accumulates to one over time. We make this assumption to reflect the imperfection of prevention.

rapid ecological changes: the plant can invade native forest (Meyer 1998), and the seed bank is both long-lived and plentiful (Loope 1997, Duffy 2005).

Hawaii is home to a great percentage of the United States' and the world's identified endangered species. Changes in forest composition may threaten endangered plant species, bird species, and invertebrate species in particular. The wet, higher elevations of Maui and Hawaii contain most of the only remaining healthy native forests supporting such diversity in the state, which are now threatened by Miconia. For example, the upper Kipahulu Valley on Maui is a conservation district reserve containing stands of Ohia (*Metrosideros polymorphata*) and Koa (*Acacia koa*) that are the primary habitat for rare native Hawaiian birds and insects, and Miconia has been discovered in the lower valley (Anon. 2001).

In the federal register listing materials for the endangered Elepaio (*Chasiempis sandwichensis*) bird on Oahu, the main justification for protection is based on the bird's reliance on the current forest structure [see U.S. Fish and Wildlife Service (2001) for example]. Since Miconia poses a significant threat to that structure, the plant is listed directly as one of the concerns for the bird's survival. A set of studies indicates that, on average, each household would be willing to pay \$31 (95 percent confidence interval of \$16.66–\$48.92) per bird species per year to keep a species from extinction (Loomis and White 1996). This amounts to an annual value for Hawaii's residents of \$12.4 million per avian species preserved. As an approximation of the potential damages from Miconia, we estimate the full threat of loss in biodiversity on all islands to be equivalent to a loss of half the endangered bird species, or \$103–303 million per year (value per bird × 31 birds × 0.5). Note that this is expected to be a conservative estimate in that it includes benefits only to the state's residents, and that even though virtually all of the 31 species live in the same pristine habitat that the tree is likely to invade, we count at most half of the birds as threatened by the potential invasion. The uncertainty associated with this estimate is particularly high; we create a range of estimates using the 95 percent confidence interval to underscore a portion of this uncertainty.

Additional damages to watershed functions are also expected from dense stands of Miconia. The hydrological properties of Miconia suggest that there may be a significant change in the water balance, with an increase in runoff and a potential reduction in groundwater recharge.[8] Groundwater recharge is of significant consequence for Oahu but less important for Maui, Hawaii, and Kauai, which generate less of their fresh water supply from groundwater. Estimates of potential expected losses from an invasion of Miconia on Oahu to groundwater recharge may be as high as \$137 million per year (Kaiser and Roumasset 2002). Increased sedimentation will incur surface water quality damages on any infested island; costs for Oahu have been estimated to be at least \$4.84 million per year (Kaiser and Roumasset 2000). Extrapolating from this figure to Hawaii, Maui, Molokai, and Kauai by susceptible land area as a first approximation, damages for the state could increase approximately tenfold, to \$48.4 million per year. If the infestation takes hold only in the highly likely cases of Hawaii, Maui, and Oahu, then we estimate these damages at \$33.9 million per year.

If all damages occurred, then the total damages would range from \$273.9 million to \$488.4 million, with an estimated average of \$377.4 million per year. Assuming that any one tree should be equally responsible for its portion of damages, *ceteris paribus*, we determine a per-tree damage rate of \$3.77. Total expected damages for any given population are described by the function[9]

$$D(n) = 3.77n. \tag{9}$$

Control cost function. Control efforts began on Maui in 1991 and continue to expand on the four

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[8] The particular role of Miconia in groundwater recharge is uncertain; on the one hand, increased runoff suggests that there is less water available for recharge, but changes in evapotranspiration rates may counteract this loss. Surveys of forested watershed experts list Miconia as a very serious threat, however (see http://homepage.mac.com/ondinebak for survey results on threats to watershed quality on Oahu).

[9] For simplicity, we assume a uniform distribution function where any tree contributes to the loss equally, given the existing population level, and the cumulative distribution as the probability of total losses for any given population, n, is just $n/N_{\max}$. The marginal damages from loss of biodiversity and watershed quality are thus \$3.77 per tree, and the expected damages for any given population is \$3.77 × n.

To model a potentially more realistic situation where the damages are increasing at an increasing rate with population, the beta distribution might be preferred.

invaded islands. While each island has its own strategy for controlling the invader, we simplify control to represent operations on the two most heavily invaded islands of Maui and Hawaii. Control in these areas begins with reconnaissance in helicopters to identify infestations and is followed by either herbicide treatment from the helicopters themselves or by operations on the ground to treat or manually remove the trees. In any case, there are two separate functions that must occur—the trees must first be found, then treated.

We therefore define a cost function consisting loosely of two parts: the "search" component and the "treatment" component. While the unit cost of treating a tree with herbicide and/or cutting a tree may be constant across population levels, the cost of finding and gaining access to a tree is rapidly decreasing in population size (and density). That is, it is extremely expensive to find the last tree, but much less so to find one tree out of 100 million trees. In 2003, the total number of trees controlled on four islands was 72,339. Conversations with resource managers suggest that annual control expenditures for Miconia in 2003 were over \$1 million, with about \$1 million spent on Maui to remove 69,000 trees, \$321,000 spent on Oahu to remove 1,100 trees, and uncertain budgets on Hawaii and Kauai to remove the remainder. The 72,339 trees that were removed were thought to be less than one-quarter of the existing population. Using this data along with budgetary and removal data from operations on Maui, Oahu, and Hawaii sporadically available from 1996–2002, we approximate the following total marginal cost function for Miconia control across all islands:

$$s(n,x) = \frac{1{,}000{,}000{,}000}{n^{1.66}},$$

$$h(x) = 13.39,$$

where $s(x)$ represents the search cost component and $h(x)$ represents the cost of treatment (harvest), constant across all population levels. The full marginal cost is thus defined as

$$c(n,x) = s(n,x) + h(x) = \frac{1{,}000{,}000{,}000}{n^{1.66}} + 13.39,$$

so that the total cost of searching and treating x trees is

$$(10) \qquad C(n,x) = \left(\frac{1{,}000{,}000{,}000}{n^{1.66}} + 13.39 \right) * x.$$

This cost function does imply that eradication costs are infinite, which we accept as likely given both the longevity of the seed bank [now thought to be at least 10 years (Duffy 2005)] and the great difficulty of locating and accessing seedlings in remote areas.

Optimal control results. While the exact number is uncertain, a rough estimate of the current population level is 436,000. Using the above parameters and the assumption that the current stock of Miconia is 436,000 trees, we find that we should engage in population reduction and maintain an N^* of 31,295 trees. Regardless of the initial stock, we find that this internal steady state is optimal. Note that these trees will be the most expensive to locate and remove; as our cost function assumes increasing search costs as populations dwindle, our results implicitly factor in at least one aspect of the spatial concerns.

Because our initial population is greater than 31,295 trees, it is optimal to remove trees until we reach this population size. We should then remove the growth produced by this stock level each and every year in order to remain at this population level. This internal optimum of 31,295 trees is a consequence of very high search costs of finding very few remaining trees combined with the significant damage produced at high population levels. This case is illustrated in Figure 1 below.

While the results for Miconia suggest an internal optimum, a species with different bioeconomic parameters may require a corner solution of accommodation or eradication. For example, if we replace our constant marginal damage function with one that is two orders of magnitude smaller or with a rapidly decreasing marginal damage function, we find that accommodation is the optimal policy. Accommodation is also preferred if the removal cost function is significantly higher. Furthermore, if the removal cost function involves a low marginal cost at low populations and a high marginal cost at high populations, we find that eradication is optimal.

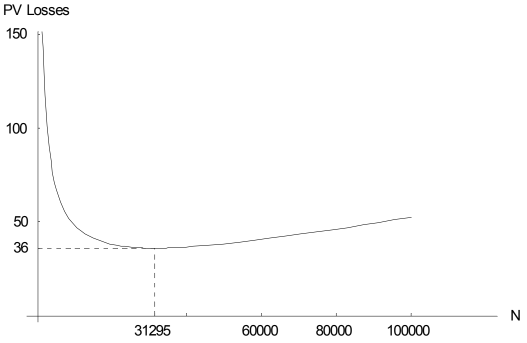

Figure 1. Present Value of Alternative Stationary Populations of Miconia ($N^* = 31{,}295$) (millions of dollars)

Status quo vs. optimal policy. Current Miconia policy in Hawaii entails spending approximately $1 million per year on control efforts. In 2003, approximately 72,000 trees were treated with this budget. As a final exercise for this species, we compare the consequences of status quo spending to those associated with the optimal policy program in order to investigate the extent to which status quo expenditures may be misaligned with optimal expenditures.

Table 1 highlights the cost savings from switching to the optimal program for Miconia. Society can either continue to spend $1 million per year on control indefinitely, which will result in a status quo steady state population near carrying capacity, or spend $6.27 million today to reduce the population to 31,295 and spend only $449,245 per year following this removal. The present value cost of the optimal program is $28.7 million, while the present value cost of the status quo regime is $50 million. Additionally, the optimal policy will reduce losses by over $12 billion compared to the status quo. Therefore, compared to the optimal plan, spending $1 million each period not only costs more but also yields a much lower return.

An increase in spending to $2 million per year on control, which is politically tractable under current conditions, leads to a steady state population of trees very close to the initial population. The present value cost of following this status quo policy is $100 million, producing present value damages of $90 million for a combined loss of $190 million, compared to the optimal program whose combined losses equal $36 million.

The large gap in damages between the $1 million and $2 million policies is due to the fact that, currently, $1 million per year in expenditures is essentially wasted, as it only slightly slows the rate of natural growth, while $2 million per year is sufficient to reach an internal steady state at a level close to the current level of infestation. Thus, a seemingly small relative increase in current expenditures can make a significant improvement in overall welfare if it is enough to move from a policy where growth continues to outpace removals to one where growth is contained.

Boiga irregularis

Growth function. Again, we utilize the logistic growth function to represent the potential spread of the snakes. In this case, $b = 0.6$, based on estimated population densities at different time periods on Guam (Rodda et al. 1999). The maximum elevation range of the snake may be as high as 1,400 m (Kraus and Cravalho 2001). We estimate that there are just over 777,000 hectares (3,000 square miles) of potential snake habitat on Hawaii, so that, extrapolating from Guam's maximum densities of 50 snakes per hectare (Rodda, Fritts, and Conry 1992), $N_{\max}$ for Hawaii is estimated at 38,850,000.

Damage function. Guam has a land area of approximately 53,900 hectares, with a maximum elevation of about 400 meters. With a population density of 50 snakes per hectare, we estimate a potential maximum population level ($N_{\max}$) for Guam to be 2.695 million snakes. With approximately 272 hours of power outages per year attributable to snakes, we estimate that there are 1.01×10^{-4} power outages per snake per year. Annual electricity generation capacity per capita in Guam is virtually the same as on Hawaii, at 2kW/capita. We estimate that an hour-long power outage on Oahu causes $1.2 million in lost productivity and damages (Fritts and Chiszar 1997). Positing a linear relationship between snake population and power outages, the expected damage per snake in Hawaii, in terms of power outage costs, is $121.11.

Guam has experienced a snakebite frequency average of 170 bites per year, at an average cost of $264.35 per hospital visit. Thus the expected

Table 1. Cost Savings under Optimal Miconia Policy

	First period removal cost	Annual removal cost	Annual damages at steady state	PV damages	PV costs	PV losses (damages plus costs)
Status quo	\$1 m	\$1 m	\$369.5 m	\$12.35 b	\$50 m	\$12.4 b
2 × status-quo spending	\$2 m	\$2 m	\$1.76 m	\$90 m	\$100 m	\$190 m
Optimal program	\$6.27 m	\$449,245	\$117,982	\$7.4 m	\$28.7 m	\$36.1 m

level of bites per snake per year is at least 6.31×10^{-5}, with an expected cost of \$0.02 per snake. Hawaii's population density below 1,400 m is approximately half that of Guam's. Snake-human interactions should occur less frequently per hectare. However, Hawaii's population is eight times greater than Guam's, so we adjust the expected costs for Hawaii to \$0.07 per snake.

The Brown treesnake has extirpated 77 percent (10 of 13 species) of Guam's native forest bird population since its arrival (Vice 2005). As discussed above, contingent valuation studies have estimated the average value of the continued existence of an endangered bird species at \$31 per household per year for Hawaii. There are 15 endangered bird species in Hawaii whose main habitat is below this level. Of these, three are native to small, unpopulated islands that are unlikely to experience the arrival of the snake, and four are water birds, also users of unlikely habitat for the arboreal snake. If we assume a 75 percent chance of losing each of the remaining eight species, the expected value of these damages to 403,240 households in Hawaii is \$75 million. If each snake is equally likely to contribute to the extirpation, the expected damages per snake are \$1.93. We consider these the base level for high damages possible from the presence of the snakes. If the snakes do not have the same success at extirpating bird populations, or if there is a bias in the contingent valuation estimates that does not account for the marginal benefit of saving an additional species as being potentially lower than \$31 per household per year (see Loomis and White 1996), then this estimate may be too high. We use the value to Hawaii residents of losing one species, or \$12.5 million, to estimate the base for low damages, with an expected per snake damage level of \$0.32.

Thus, expected damages from human health factors, power outages, and expected endangered species losses can be expressed as $D_H = 123.11 \times n_t$, and $D_L = 121.50 \times n_t$. The maximum annual damages that Hawaii faces without control efforts are therefore $N_{\max} \times 123.11$, which equals \$4.8 billion. We take the expectation of the high and low damages to get a damage function of

$$D = 122.31 \times n_t. \tag{11}$$

Control cost function. We assume that control costs are decreasing in n and linear in x. We choose the marginal cost function

$$c(n) = \frac{378{,}512}{n^{0.621}}, \tag{12}$$

based on the current expenditures for snake control on Guam and on attempts to capture suspected snakes in Hawaii. We fit our cost curve to match the expected costs to capture one snake out of the predicted current population and the first snake out of 39 million. Experts are uncertain about how many snakes might have arrived in Hawaii, but suggest there may be between zero and 100 already here. Searches occur when a positive sighting has taken place. Catching a single snake out of roughly 15 will be at least \$76,000, and catching one out of a population of one will cost just under \$1,000,000. Catching a single snake from a population at capacity costs about \$31 based on the cost of night traps and a study investigating the use of dogs in detecting planted snakes.

Probability of arrival (establishment). We assume that prevention expenditures buy a reduction in the probability that a snake will arrive and become established. The higher the probability of arrival, the easier it will be to capture any one entrant. Conversations with *Boiga irregularis* scientists and resource managers have led us to understand the following regarding the probability of arrival given different levels of funding. Under current prevention expenditures of \$2.6 million, Hawaii faces an approximate 90 percent probability that a single snake will arrive over a ten-year time horizon. If expenditures were increased to \$4.7 million, the probability of a single arrival decreases twofold, to about 45 percent. Finally, if we increase preventative spending to \$9 million per year, the probability of an arrival decreases another twofold, to about 20 percent. Based on these data, we choose the Weibull distribution to describe the probability of a single arrival over 10 years, given annual prevention spending y:

$$p(y) = e^{-0.2y^{0.6}}. \tag{13}$$

Figure 2 illustrates this function.

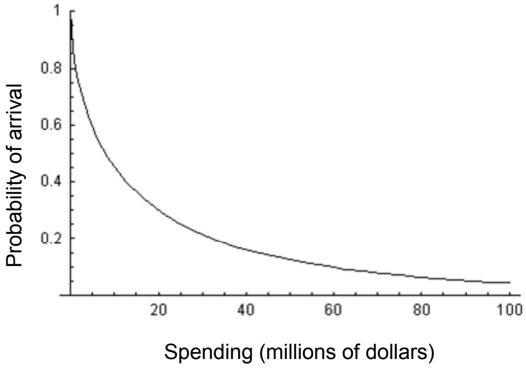

Figure 2. Probability of Brown Treesnake Arrival as a Function of Prevention Spending

Optimal prevention and control results. If prevention were costless, the optimal strategy would be immediate eradication. Using the parameterization described above and the assumption that there are seven snakes currently in Hawaii,[10] this would result in $V^0 = V^* = \$2.09$ million. With costly prevention, however, eradication requires repeated cycles of eradication followed by prevention. In order to maintain a zero population of snakes, we would need to spend \$1 million to eradicate any potential entrant. Given this cost, our minimized Z^0 is \$49.8 million. When this cost is added to the present value of costs and damages of achieving the zero population level (V^0), total losses[11] equal \$51.9 million. This turns out to be suboptimal.

As discussed earlier, we assume that optimal prevention expenditures are negligible once the pest reaches the threshold population $\underline{N}$. For illustrative purposes, we set $\underline{N}$ equal to three. The total losses associated with maintaining one snake are \$27.4 million; the total losses of a two-snake population are \$26.4 million. With three snakes, we no longer have to add the Z^* component of repeated entrant removals and prevention; optimal prevention is zero. Therefore, the total cost[12] associated with maintaining a population of three snakes is \$17.8 million. Figure 3 illustrates the four components of total losses: the perpetual prevention/entrant removal expenditures (if the candidate N is less than three), perpetual population maintenance costs, perpetual damages, and the instantaneous removal cost (if the candidate N is less than seven). The thick line sums these components and illustrates the conclusion that the optimal population is three. Inasmuch as damages and removal costs are comparatively low, the solution is primarily determined by the balance between prevention expenditures and perpetual population maintenance. The monotonically increasing total represents ever-increasing losses for populations above three snakes.

One is tempted to draw the general conclusion from this example that the steady state solution will always be $\underline{N}$ and that it should be attained as rapidly as possible. This turns out not to be the case, however. Figure 4 illustrates the case where prevention expenditures become negligibly effective, relative to killing or sterilizing the existing

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[10] As discussed earlier, the exact population of Brown treesnakes in Hawaii is not known, although experts suggest that the number may be between zero and 100. We use a conservative guess of seven for exposition.

[11] Total losses here are defined as $V^N + Z^N$ and include the cost to reduce population from seven to the desired level, the perpetual cost of catching the growth at the desired level, the perpetual damages incurred from maintaining this level of snakes, and the cost of the prevention/removal cycle Z^*.

[12] Now the total cost is everything mentioned in the above footnote, minus Z^*.

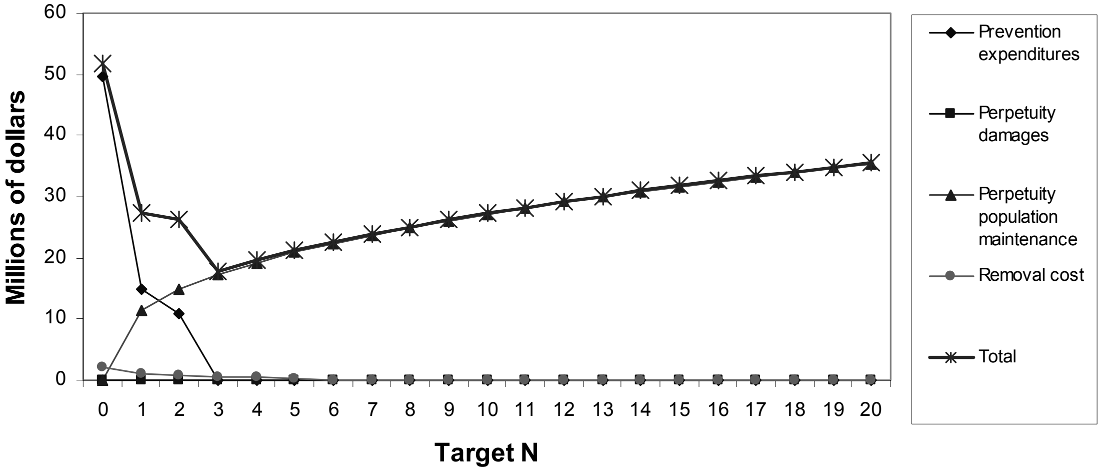

Figure 3. Optimal Population Reduction ($\underline{N}$ = 3, N^* = 3)

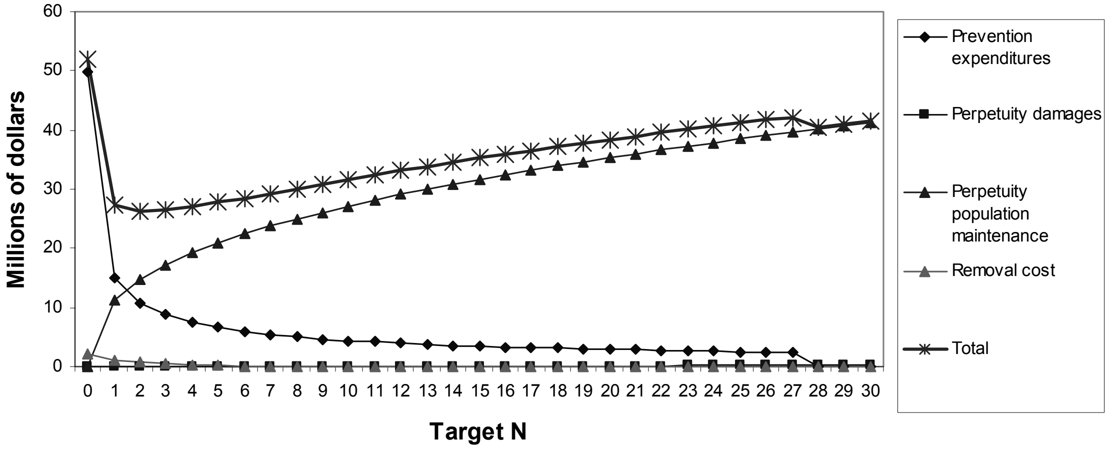

Figure 4. Optimal Population Reduction When $\underline{N}$ = 28, N^* = 2

population, only after the population reaches 28. In this case, the population should be immediately decreased from seven to two snakes and maintained at that steady state thereafter. As illustrated, the steady state occurs where the marginal present value costs of population maintenance costs (due to higher growth) equal the marginal benefits of reduced removal costs plus the reduced prevention costs. The result of a small stable population is robust to our choice of $\underline{N}$. For values of $\underline{N}$ greater than 28, N^* remains at 2.[13]

In summary, the optimal strategy calls for immediate reduction down to a very low population followed by maintenance at that population, possibly with continued prevention expenditures.

Status quo vs. optimal policy. As in the Miconia case, we now compare the status quo policy of spending \$76,000 annually on control and \$2.6 million on prevention to the optimal program of population reduction from seven to three and continued maintenance at three. First we compare the cost of both programs. The present value of the cost of the status quo policy is \$133.8 million and only \$17.8 million[14] if we were to follow the optimal policy. Table 2 below provides details on

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[13] The prevention cost curve is low and flat, while the population maintenance curve is high and monotonically increasing.

[14] The cost of reduction from seven to three snakes is \$573,510, and the cost and damages of maintaining the three-snake population is \$17.2 million.

Table 2. Cost Savings under Optimal Brown Treesnake Policy

	First period cost	Annual cost	PV costs	Annual damages	PV damages	PV losses
Status quo	\$2.676 m	\$2.676 m	\$133.8 m	\$4.5 b	\$138.5 b	\$138.6 b
Optimal program	\$573,510	\$344,397	\$17.79 m	\$363	\$19,000	\$17.81 m

benefits of switching to the optimal program.

While the status quo policy involves considerable spending, the policy does not have a significant impact on the growth of the snake population because it continues to focus on preventing new arrivals, when there may already be a minimum viable population in Hawaii.[15] With this level of spending, after 20 years the population of snakes will have already gone from seven to 709,615 snakes, and after about 35 years will have already reached its status quo steady state, very close to the carrying capacity of 38.8 million snakes.

This rapidly increasing population associated with status quo policy guarantees high levels of damage. If we instead spend more money up front (\$573,510 instead of \$76,000) to reduce the population from seven to three, we are able to spend less every year to remain at this level and can avoid the high levels of losses promised by following the status quo policy. As before, the optimal policy costs less in present value terms, while preventing over \$100 billion in losses as compared to the status quo.

Concluding Remarks

Using optimal control theory, we generate necessary conditions for efficient spending on population management of an invasive species that has already established itself in an environment, and prevention and control expenditures for a potential invader.

Current expenditures on Miconia removal are not containing expansion. If status quo spending is continued, expansion will continue until the population is at a much higher level, incurring both high damages and wasted control resources in the process. By spending more today on Miconia, the population can be contained at a relatively low level of sustained damages and subsequent control expenditures.

Consideration of our simplifying assumptions is suggestive of possible extensions. For example, we chose to model the seed bank by assuming that the removal of a single tree would require retreatment over time, in effect subsuming the removal of future growth from seeds into the cost of removing the parent tree. We also subsumed spatial considerations into the population growth function and the search and removal cost function. It may be illuminating to model these aspects explicitly. For example, it may be desirable to consider strategies such as containment of the core (dense, original infestations) or satellite (more sparse, spread out offspring) populations of Miconia. While spatial elements are implicitly represented in the search cost function, it would be useful to model these considerations explicitly.

In the case of the Brown treesnake, our results suggest that maintenance of a very low population is the optimal policy choice. If in fact a small number of snakes are already present, the mix of prevention and control must focus on both intercepting arrivals and catching existing snakes and maintaining a low population in perpetuity. At low populations, perpetuity damages and first-period removal costs become relatively unimportant, and the optimal number of snakes occurs where the marginal benefit of population reduction (due to lower costs of population maintenance) equals the marginal cost due to increased prevention expenditures.

Modeling simultaneous prevention and control in general is intractable due to dependence of optimal decisions today on optimal and interdependent decisions in the future, which depend in turn on the realization of stochastic variables. We have chosen one way to break through the com-

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[15] Close monitoring of the progress of the snake in Saipan (Commonwealth of the Northern Mariana Islands) should give a good indication of what is in store for Hawaii; though considerable efforts are being expended to forestall additional arrivals, there is agreement that an incipient population is in place. Efforts there continue to focus on interception rather than removal.

plexity; other methods will be the subject of future research. One complication avoided in the present paper regards the distinction between uncertain arrival and uncertain population establishment. This in turn suggests needed scientific knowledge. In particular, a better understanding of the probability of a snake mating and reproducing would enhance our ability to accurately determine the probability of establishment separate from the probability of entry, rather than the combined, average probability of entry and establishment used here. Sex ratios in arrivals may also be considered. The scientific evidence from Guam suggests that male-female ratios are not one-to-one, with perhaps many fewer females than males moving into transport zones (Rodda 2005). An extended model of the snakes would also consider the extent to which future introductions matter, with their importance decreasing with population size.

Another important source of uncertainty surrounding the snake results is the current population of snakes in Hawaii. While the actual number may be as low as zero or as much as 100, we illustrate the model using the subjective mode of seven. Because this type of information will likely never be understood with complete certainty, it would be useful to derive results for a range of initial populations. These may help counter the political tendency to ignore the possibility that a small snake population already exists.

A high priority for future research is to explicitly model the risk of establishment from both the existing population and potential new invaders. This line of research is likely to support the qualitative conclusion here that a balance between prevention and control is indicated. Clearly, spending nothing on prevention would result in population establishment. But establishment may also occur from snakes that are already present, especially if existing snakes have entered through the same or a small number of pathways.

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