

Title: Prevention, Eradication, and Containment of Invasive Species: Illustrations from Hawaii¹

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Abstract

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 invasion, *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 to Guam has led to native bird extirpations, power outages, and health costs.

Keywords: Invasive species, bioeconomic, optimal control, *Miconia calvescens*, *Boiga irregularis*

1. Introduction

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 (prevention), actual costs and benefits of any mitigating actions once a species has become established compared to anticipated costs and benefits of accommodating the change (control), and the comparative advantages of preventing and/or controlling one set of ecosystem changes over another (relative risk analysis/damage assessment). 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 rather 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; Eiswaerth and van Kooten, 2002; Olson and Roy, 2002), and integrated prevention and control (Pitafi and Roumasset, 2005). Tying invasions to trade (Costello and McCausland, 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.

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 combination of the likelihood of a successful invasion and the optimal control or accommodation of its associated expected damages, from the current state of nature. We seek to examine the theoretical findings with data from two cases affecting Hawaii: the weedy shrub *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.

2. Case Overview

2.1. 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 3000 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 4th 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 20th 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 sugarcane.

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 dollar tourist industry dominates the state's \$44 billion dollar (GSP) economy (DBEDT, 2003) and threats to the industry can inflict significant damages if realized.

2.2. *Miconia Calvescens*

One significant threat comes in the form of the woody shrub, *Miconia Calvescens*. *Miconia calvescens*, a member of the Melastomataceae family from Central America, was purposefully introduced to Hawaii. Planted in a handful of back yards 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% or more of the main island of Tahiti after a single specimen was introduced to the Papeari Botanical Garden in 1937 (Medeiros, Loope 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 in Tahiti was not obvious, however, until aided by two successive hurricanes in the 1980s, at which point the ornamental plant 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 at bay. We explore quantitatively the policy options and their economic consequences for the continued treatment of the invasion in the modeling and discussion sections below.

2.3. 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 snakes¹ 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 USGS, 2005 for recent overview of damages; detailed power and medical data courtesy of Stephanie Shwiff, USDA).

The snake is an imminent threat to Hawaii; 8 interceptions of identifiable Brown Treesnakes have occurred in the past 2 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.

3. Methodology

3.1. Optimal control of existing invader

Optimal control of an existing invader can be summarized in most cases to considering where the population is in relation to its optimal steady state population, as determined by minimizing the present value of damages and control costs across an infinite time horizon. If the population is currently at this steady state population, then we continue to harvest new growth at the steady state, generating a stream of minimized economic costs and damages indefinitely. If the population is currently above the steady state population, 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 which are lower than the costs of maintaining these lower populations, until at the steady state population we initiate maintenance as described above. The relevant literature pertaining to using optimal control for harvesting a species in general is presented well in Clark (1990). The problem of an invasive species already present in an area might be written as a relatively standard optimal control problem:

$$\text{MAX} \int_0^{\infty} -e^{-rt} \left(\int_{n-x}^n c(\gamma) d\gamma + D(n) \right) dt \quad (1)$$

subject to:

$$\begin{aligned} \dot{n} &= g(n) - x \\ 0 &\leq x \leq n, \quad n_0 \text{ given,} \end{aligned} \quad (2)$$

Where $c()$ is the cost of removals, $D()$ are the damages from a population of n originating from an initial population n_0 and growing according to $g()$ net removals, x , if we assume an interior steady state solution. As the model is linear in control, any adjustments will be instantaneous (bang-bang) solutions. This remains true in our solution mechanism.

Invasive species, however, have two characteristics that may make it preferable to analyze the problem with a slightly different approach. These are: (1) the increased interest in understanding the corner solutions of eradication ($x=n$) and accommodation ($x=0$), and (2) the prevention of the introduction of a species not yet present. We choose therefore to use an algorithm which searches over all stationary population N 's. This will generate an automatic comparison of corner solutions and the identical steady state solution to that of the optimal control problem defined by equations (1) and (2), as well as improving the transparency of the additional prevention problem.

Since the optimal approach path will differ depending on if our target steady state N is greater than, equal to, or less than n_0 , we require two procedures for determining the present value of reaching and maintaining N , which we label V , for each of the candidates N 's. Therefore V will be dependent on the initial population n_0 , which we will denote with $V(n_0, N)$. These procedures are described below.

3.1.1. Population reduction

If the candidate population, N , is greater than n_0 , the optimal approach path involves instantaneous adjustment to the target population. The cost of that population reduction,

$x=N-n_0$,² 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

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

3.1.2. 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

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

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 which minimizes $V(n_0, N)$, i.e.,

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

where

$$V(n_0, N) = \begin{cases} \int_0^N c(n)dn + \frac{c(N)g(N)}{r} + \frac{D(N)}{r}, & 0 \leq N \leq n_0 \\ \int_{t_0}^T e^{-r\tau} D(n) d\tau + e^{-r(T-t_0)} \left(\frac{c(N)g(N)}{r} + \frac{D(N)}{r} \right), & \dot{n} = g(n), n_0 \leq N \leq N_{MAX} \end{cases}$$

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

$$\text{i.e., } V(n_0, N^*) = \text{Min}_N V(n_0, N). \quad (5)$$

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.

3.2 Optimal prevention of potential invader

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 are 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.

Prevention is also indicated when an initial pest population has been reduced below the minimum level required for the population to grow without further introductions. That is, prevention is necessary when the population falls below \underline{N} individuals⁴. 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} , where prevention no longer reduces expected welfare loss and should be zero, or whether it is better to maintain $N^* < \underline{N}$ and

optimally invest in prevention. Thus, if V^* involves $N^* < \underline{N}$, we must then consider the additional costs of removal upon re-entry, E .

In order to maintain population at $N^* < \underline{N}$, the resource manager optimally invests in prevention. 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 prevention in addition to prevention. As shown in Pitafi and Roumasset (2005), the expected present value of prevention and removal sums the prevention expenditures and expected value of introduction followed by removal expenditures over time:

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

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 .⁵

Note that eradication costs ($V(n_0, 0)$) may be infinite, which serves as a representation of the imperfection of detection, as evidenced in general by the costly search cost component of control. In this case, a unique level of prevention is not calculable for eradication at $N=0$. However, a valid comparison between prevention and control (at a population below which additional arrivals continue to matter) and control alone (at a population above which additional arrivals cease to have significant consequence) can be made. In this case, $E(N)$ is represented by $V(n_0, N)$.

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

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

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^*) represents a choice between combined prevention and control ($y^* > 0, x > 0, N^* < \underline{N}$) or control alone ($y^* = 0, x > 0, N^* \geq \underline{N}$)⁶. We select the total cost minimizing population from the following comparisons. (1) 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 (2), 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 $y^* = 0$. We generate $Z(y, N)$ so that prevention expenditures minimize the expected costs of maintaining $N < \underline{N}$. $Z^*(N)$ then represents the prevention and entrant-removal levels satisfying equation (7) for each (integer) population $N < \underline{N}$ and $E = V(n_0, N)$.

The optimal prevention expenditure from (7) is substituted into (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 $Z^*(y, N^*) + V(n_0, N^*)$, is compared to $V(n_0, N \geq \underline{N})$ to determine whether the optimal policy for integrated prevention and control involves staying at the current population level ($n_0 = N$), reducing stock and not spending on prevention ($n_0 > N \geq \underline{N}$), reducing stock and spending on prevention ($N < \underline{N}$), or allowing the stock to grow unabated until some maintained steady state greater than the current level ($N > \underline{N} > n_0$).

4. 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% discount rate. The remaining parameters are discussed below, followed by results for each case.

4.1 *Miconia calvescens*

4.1.1 Growth function

We utilize a standard logistic growth function to represent the spread of the invasive tree.

Thus:

$$g(n) = bn \left(1 - \frac{n}{K} \right), \quad (8)$$

Where b is the intrinsic growth rate, estimated here to be 0.3, and K is the carrying capacity, estimated to be 100 trees per acre over 1 million acres, or 100,000,000. 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.

4.1.2 Damage (net benefit) function

We estimate damages from *Miconia* as evolving from indirect ecosystem services as well as non-market goods like biodiversity. Figures 1 and 2 illustrate the potential range for *Miconia* and the primary biological and hydrological resources threatened by invasion.

<Figure 1 here>

<Figure 2 here>

The characteristics of the species that have branded it a nuisance species have three major dimensions for potential damages. First, it is an aggressive invader that appears to invade healthy native forest with success (Meyer 1998). Native forest and its biodiversity are replaced with dense, monotypic stands of *Miconia* that shade out all undergrowth and may change soil chemistry. Second, the seed bank develops quickly once the tree reaches flowering and fruiting size of 4-5 meters (4 cm dbh; at least 4-5 years of age) as a single tree can flower and fruit 2-3 times a year, with a typical fruiting event producing 3 million seeds (Loope 1997). Third, the seed bank has some longevity. It is known to last over 2 years, and may be as long as 10 years (Loope 1997; D. Duffy personal communication 2005). Canopy openings are quickly taken advantage of by new seedlings. Wind dispersal appears most prevalent, though birds are also dispersal agents. With sufficient rainfall (greater than 1800 mm / yr) and canopy openings, a single specimen may, in 5-15 years, start a stand that covers several hundred acres.

Hawaii is home to a great percentage of the United States' and the world's identified endangered species. Changes in forest composition as described may threaten endangered plant species, bird species, and invertebrate species in particular. Hawaii's evolutionary isolation has led to much adaptive radiation of species, where a single ancestor has generated a set of species that each depend on new and different types of habitat. The state is considered to house the most unique and diverse snail population in

the world despite the limitation that only 15% of snail families are represented (Asquith 1995). The wet, higher elevations of Maui and Hawaii contain most of the only healthy remaining native forest supporting such diversity in the state, and 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 (Staff 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 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% 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 state residents of \$12.4 million per avian species preserved. From the confidence interval, we assume the damages would lie between \$6.7m and \$19.6m.

Economic theory and research predicts that households will value invertebrates and plants at lower levels (Loomis and White 1996), and that non-residents will also have lower aggregate values. Thus, 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 this is expected to be a conservative estimate in that it only includes benefits to the state 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% 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 recharge⁷. 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 ground water. 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 only takes hold 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 \$273.9m to \$488.4m, 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 function⁸:

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

Though a linear model for damages may seem simplistic, we argue that the outcomes of choosing either an increasing function (where damages are delayed until the trees form dense stands or cover large amounts of habitat) or decreasing function (where damages occur quickly and additional population simply exacerbates the problem slightly) for marginal damages do not match our understanding of the potential problem any better than the linear model. For completeness, however, we try alternative specifications of the damage function in the results section below.

4.1.3 Control cost function

Control efforts began on Maui in 1991 and continue to grow on the four 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 pull 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 1100 trees, and uncertain budgets on Hawaii and Kauai to remove the remainder⁹. 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 removals data from operations on Maui, Oahu and Hawaii sporadically available from 1996-2002,¹⁰ 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:

$$c(n, x) = s(n, x) + h(x) = \frac{1,000,000,000}{n^{1.66}} + 13.39, \text{ so that the total cost of searching and}$$

treating x trees is:

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

This cost function does imply that eradication costs are infinite, which we accept as likely given both the longevity of the seed bank (now known to be at least 10 years (pers. comm., D. Duffy, 2005) and the great difficulty of locating and accessing seedlings in remote areas. As an extension we consider what costs could look like to make eradication possible and optimal.

4.1.4 Optimal control results

4.1.4.1 Specified Model

Using the above parameters and the assumption that the current stock of *Miconia* in Hawaii is approximately 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 should go and stay at a stock of 31,295 in the steady state, by the fastest path possible.

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 significantly damage produced at high population levels. This case is illustrated in Figure 3 below.

<Figure 3 here>

4.1.4.2 Alternative Damage and cost functions

While we conjecture that the above parameterization best describes the Miconia story in Hawaii, we follow this analysis using slightly different functional forms and parameters for the damages and costs to give an indication of the sensitivity of the results and to explicate potential changes in optimal policy as bio-economic conditions change. Table 1 summarizes the cases discussed below.

<Table 1 here>

The parameters above assume constant marginal damages across population levels. If we instead use the slightly increasing marginal damage function¹¹ of

$$D(n) = 3.38n + 4.83 * 10^{-7} n^2,$$

we again find a low internal steady state, now at 32,222 trees.

For perspective, we calculate the linear damage function that provides the “break-point” between an interior solution and accommodation ($x = 0$). We find that a damage function where $D(n) = 7.67 * 10^{-3} n$ results in indifference between an interior steady state population of 47,557 trees and accommodation of the 100,000,000 tree population with no control. At this level, the undiscounted damages from accommodation are \$767,000, and the present value cost of either accommodation or the interior steady state is \$2.63 million.

This damage function suggests that damages would have to be over two orders of magnitude smaller than expected at the capacity population in order to prefer accommodation to some level of control (at a fairly small population). This result gives us confidence that a model in which damages grow slowly until some later, discounted time period will not result in accommodation unless the discounted value of the damages is also significantly smaller than the current estimate. We consider, for example, the case where the damage function slows its rate of growth considerably after reaching some high population level: $D(n) = 3.77n^{0.7}$. Here, the maximum undiscounted damages are $1.50 * 10^6$, and the internal steady state, if applicable, would occur at 46,830 trees. The solution, however, depends on the initial population. If the initial population is above 2.924 million trees, then accommodation becomes preferable to scaling back the

population and containing it thereafter at 46,830. We see that non-linear damages make the optimal policy dependent on the initial population.

As a final set of cases, we consider conditions under which the cost function allows for eradication. One possibility is that the search component of costs does not decline so rapidly as population increases, so that the exponent on population is less than one. A reduction in this exponent below 1 makes it possible to compare the costs of eradicating the last unit to the costs of maintaining a population of $N=1$ or greater. If the cost

function becomes $C(n, x) = \left(\frac{1000000000}{n^{0.96}} + 13.39 \right) x$, it becomes preferable to maintain a

population of zero (eradicate) than to maintain a population of 1 specimen. With this cost function, however, costs do not fall as rapidly, so that the cost of removal of a tree at maximum population is \$46.80. This leads to accommodation always being preferred to both an internal solution and to eradication in this case. With damages at $D(n) = 3.77n$, the present value cost of accommodation is $\$1.3 \times 10^{10}$ compared to eradication with a cost of 3.5×10^{10} . In order for eradication to become just preferred to accommodation in this case, the search component of cost will have to fall by almost $2/3$, to an average of \$370

per acre, or $C(n, x) = \left(\frac{370000000}{n^{0.96}} + 13.39 \right) x$.

We also investigate the case where marginal damages slow, as described above, so that $D(n) = 3.77n^{0.7}$. Here, from an initial population of 436,000, eradication would cost approximately $\$3.5 \times 10^{10}$ whereas accommodation would incur only $\$5.5 \times 10^7$ in present value damages. It is particularly disconcerting that the local minimum of eradication is

dominated by accommodation. The flatter the cost curve becomes, the more accommodation is favored, as well.

We consider then the case where the cost of finding and removing the final tree is low enough that eradication is preferable, even with relatively low total damages. This will occur if the search component is reduced significantly. If $C(n, x) = \left(\frac{1385000}{n^{0.96}} + 13.39 \right) x$, then we are approximately indifferent between eradication and accommodation from an initial population of 436,000 at a cost of $\$5.5 \times 10^7$. Thus, if search can be accomplished at an average of \$1.39 per acre rather than \$1000, or where search can be limited to a portion of the viable habitat, eradication becomes optimal. Though this may not be likely for wind and bird dispersed species like *Miconia*, where the entire range of habitat is likely to become infested and evidence to date suggests that the plant continues to elude searchers and surprise them with its ability to travel to new locations, it is encouraging for species where containment is more feasible.

4.1.5 Status quo vs. optimal policy

Current *Miconia* policy in Hawaii entails spending approximately \$1 million per year on control efforts. While the exact number is uncertain, a rough estimate of the current population level is 436,000. 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 for two of our cases in

order to investigate the extent to which status quo expenditures may be misaligned with optimal expenditures.

<Table 2 here>

From Table 2, we see that if Hawaii continues to spend between \$1–\$2 million per year from today into the future, welfare will be lower in both of our illustrative cases. In our first case, starting from our current population level of around 436,000 trees, the present value of the cost of continuing to spend \$1-\$2 million every period is \$50-\$100 million, while the present value of the cost of the optimal policy (reducing stock to 31,295 and staying there) is less than \$30 million. Not only is the present value of the cost of the status quo policy higher, but it yields a lower social welfare value. That is, if we continue status quo spending, society faces a loss of \$190 million-\$12.4 billion, while reducing the population quickly and maintaining it at that level produces a significantly lower loss of \$36.1 million, resulting in a net benefit from a switch to the optimal program of \$154.2 million-12.3 billion dollars.

If damages are increasing with higher stock levels as in our second case, a switch to the optimal program may yield between \$20.4 million and \$158 billion in savings. This substantial savings results from the rapidly increasing damages after about 1 million trees. Reducing the population to 32,222 and maintaining this level allows us to avoid these steep damages.

<Table 3 here>

Table 3 above highlights the cost savings from switching to the optimal program under the first case 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.

Spending \$2 million per year on control indefinitely 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.

4.2 *Boiga irregularis*

4.2.1 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, Fritts et al. 1996). The maximum elevation range of the snake

may be as high as 1,200 m (Fritts 1988). We estimate that there are just over 777,000 hectares (3000 square miles) of potential snake habitat on Hawaii (Figure 5), so that extrapolating from Guam maximum densities of 50 snakes per hectare (Rodda et al. 1992), N_{max} for Hawaii is estimated at 38,850,000.

4.2.2. 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/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 snake-bite frequency average of 170 bites per year, at an average cost of \$264.35 per hospital visit. Thus the expected 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 1200 m is approximately half that of Guam's. Snake/human interactions should

occur less frequently per hectare. However, Hawaii's population is 8 times greater than Guam's, so we adjust the expected costs for Hawaii to \$0.07 per snake.

The Brown Treesnake has extirpated 77% (10 of 13) of Guam's native bird population since its arrival (USGS 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 (www.hear.org). Of these, 3 are native to small, unpopulated islands that are unlikely to experience the arrival of the snake and 4 are water birds, also users of unlikely habitat for the arboreal snake. If we assume a 75% chance of losing each of the remaining 8 species, the expected value of these damages to 403,240 Hawaiian households 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 Hawaiians 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. Figure 4 illustrates the overlap between snake habitat and bird habitat.

<Figure 4 here>

Thus, expected damages from human health factors, power outages, and expected endangered species losses can be expressed as:

$$D_H = 123.11 \cdot n_t, \text{ and } D_L = 121.50 \cdot n_t.$$

The maximum annual damages that Hawaii faces without control efforts are therefore $N_{\max} * 123.11$, which equals \$4.8 billion. We take the expectation of the high and low damages to get a damage function of:

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

4.2.3 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 in 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 how many snakes might have arrived in Hawaii, but suggest there may be between 0 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 \$76,000¹² and catching one out of one snakes will cost just under \$1,000,000. Catching a single snake from a population at capacity costs about \$31, based on single night trap costs (\$6-9) at an estimated 7-28% efficacy rate, (Rodda, Fritts et al. 1996), and a study investigating the use of dogs (\$100,000/year) in detecting planted snakes estimates their efficacy rate at 70% (Engeman 1997).¹³ Thus the capture of the last few specimens should be very expensive.¹⁴

This cost function describes costs of searching out snakes and capturing them as they arrive through a limited number of entryways. If the final search must include all potential snake habitat, i.e. if we are at a high population and considering the cost of reducing the population, we expect that the cost function would actually shift up substantially. This (somewhat less interesting) case is described below as well.

4.2.4 Probability of arrival (establishment)

We assume that prevention costs 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 expert *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% 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 two-fold, to about 45%. Finally if we increase preventative spending to \$9 million per year, the probability of an arrival decreases another two-fold, to about 20%. 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 :

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

Figure 5 illustrates this function.

<Figure 5 here>

4.2.5 Optimal prevention and control results

4.2.5.1 Expected parameterization

Using the parameterization described above and the assumption that there are 7 snakes currently residing in Hawaii, we obtain the following results. If it was not necessary to spend on prevention, and we could simply reduce the population from seven snakes, the optimal program involves reducing the snake population to zero. We find that population reduction is the preferred policy, with an optimal N^* of zero snakes, and $V^0 = V^* = \$2.09$ million.

We now seek to determine whether this remains the optimal population if we need to return again to prevention expenditures after eradication. The cost of the repeated

prevention and removal cycle at lower population levels is large, as one expects. 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 losses¹⁵ equal \$51.9 million.

Alternatively, we might not remove all snakes below \underline{N} . When comparing the cost of maintaining zero population to the cost of maintaining 1, 2, 3 or more snakes, we find that it is best to allow for a small positive population level of three snakes, where 3 is considered \underline{N} for the snake. The total losses associated with maintaining 1 snake are \$27.4 million, the total losses of a 2-snake population are \$26.4 million. However at 3 snakes, we no longer have to add the Z^* component of repeated entrant removals and prevention, since we no longer spend on prevention at a population of 3. Therefore, the total cost¹⁶ associated with maintaining a population of 3 snakes is \$17.8 million. This result is driven by our assumption that at 3 or more snakes, prevention expenditures are no longer effective in that additional arrivals will not outweigh the growth being produced by the reproducing population of 3 or more. Figure 6 below illustrates the optimality of a 3-snake population. The four black dots represent the present value of losses for a zero-snake population, a one-snake population, a two-snake population, and a three-snake population, respectively. The monotonically decreasing line represents ever-increasing losses for populations above 3 snakes.

<Figure 6 here>

We see then that including prevention expenditures for $N < \underline{N}$ renders $N=0$ less optimal than $N = \underline{N}$, with an associated increase in welfare loss for the control portion. However, integration of prevention and control saves resources, in this case by reducing total costs and damages from $V^0 + Z^0$ to $V(\underline{N})$.

4.2.5.2 Alternative cost structure

We have estimated the cost of capturing a single snake based on expenditures for low-population cases where searches are contained based on probable sightings of snakes. If we must search the entire potential snake habitat to eradicate the snakes, we estimate that the cost function would shift to $c(n) = \frac{1.92 * 10^9}{n^{1.18}}$, based on search and capture costs of approximately 1,000 per acre of habitat for an individual snake. Here, the cost of eradicating the last snake is infinite, the cost of removing one of 15 snakes is approximately 82 million, and the cost of removing one of 38.85 million snakes is still \$32. The internal steady state solution is a controlled population of 223,678 snakes at a social cost of approximately \$7.8 billion. Prevention expenditures are ineffectual at this internal steady state, so comparisons need not be undertaken. Clearly, keeping the population contained so that the originally described cost function is accurate is preferred.

We have assumed here that one way to model the spatial spread of the invasives is to describe one cost function according to costs of low population searches differently from

costs of high population searches. Though this is somewhat crude, it highlights the purpose of thinking of Early-Detection and Rapid-Response (EDRR) policies as cost-effective compared to control policies alone. By noting that the entire cost structure will increase if the spread is not contained, we include the spatial concerns without complicating the modeling process unnecessarily. This is particularly important in light of the dangers described in the next section, comparing the prevention-preferred status-quo to an optimal policy, in that it underscores potential additional costs of under-spending on low-population control.

4.2.6 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 on prevention to the optimal program of reduction from 7 to 3 and continued maintenance at 3. 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 million¹⁷ if we were to follow the optimal policy. Table 4 below provides details on benefits of switching to the optimal program.

<Table 4 here>

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 scientists believe there may already be a minimum viable

population in Hawaii¹⁸. With this level of spending, after 20 years the population of snakes will have already gone from 7 to 709,615 snakes, and after about 35 years will have already reached its status quo steady state, very close to carrying capacity of 38.8 million snakes. See Figure 7 below for an illustration of the current policy's impact on snake population.

<Figure 7 here>

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 7 to 3, we are able to spend less every year to remain at this level and can avoid high levels of losses promised by following the status quo. As before, the optimal policy costs less in present value terms while preventing over a \$100 billion in losses as compared to the status quo.

5. Concluding remarks

Using optimal control theory, we generate appropriate comparisons for policy options concerning existing and potential invasive species. Not only does the theory allow us to compare options within species, but we can also compare across species, given their current population levels, by comparing optimal policy for each case. In the cases above, we show that the status quo policies for both Miconia and the Brown Treesnake appear to be inefficient. Spending approximately \$1 million per year on Miconia removal only

slows its growth rate, leading to massive damages if the policy is continued and wasting resources on control that are ineffectual in the long term. By spending only \$6 million in one year, the population can be contained at a relatively low level of sustained damages and subsequent control expenditures of only half a million dollars annually. Similarly, if we redirect funds away from prevention of Brown Treesnakes, initially spend \$500,000 for population reduction followed by annual control costs of \$350,000, billions of dollars in damages may be avoided.

These results are primarily illustrative. In the case of the weedy plant, *Miconia calvescens*, for example, we chose to model the seed bank by assuming that the removal of a single tree would require re-treatment over time, in effect subsuming the removal of future growth from seeds into the cost of removing the parent tree. We also abstracted away from important spatial considerations of the problem. 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.

We investigate alternative cost and damage structures and find that results in this case are less sensitive to the marginal damages than to the marginal costs. Damages must be considerably lower to generate accommodation as the optimal policy. We find eradication may be a local optimum, even with high costs of removing the last specimen, but not a global one if unit costs do not quickly fall.

With respect to the snakes, additional scientific information might also improve the ability to estimate the probabilities associated with successful establishment. 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. The scientific evidence from Guam does suggest that male-female ratios are not one-to-one, with perhaps many fewer females than males moving into transport zones (Rodda 2005, personal communication). An extended model of the snakes would also consider the extent to which future introductions matter, which should be rapidly decreasing with population size. While we have approximated this consideration with a threshold population of three, a more detailed model would directly account for the increasing importance of limiting new introductions in the lower population levels. 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 potentially higher than 100, we illustrate the model using a guess of 7. 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. Regardless of the actual population, if we believe that the population may be higher than the officially stated case (that there are no snakes in Hawaii), increasing removal efforts now is likely to decrease long term costs more than increasing prevention efforts.

The linearity of the harvest control also limits the range of potential solutions to moving to eradication, accommodation, or an internal steady state by the fastest possible path. Adding a maximum annual removal capacity where the upper bound on x (\bar{x}), was less than carrying capacity, K , would allow for an additional case of contained population with continuous removal such that the steady state occurred where the growth rate equaled the removal capacity. The cost of attaining and maintaining this steady state would then be compared to accommodation, eradication, and any other internal steady states that might differ from the case where $\bar{x} = g(n)$. Nonlinearity in the control cost as a function of removals also adds the possibility that there will be times where it is worth it to wait to harvest until growth has slowed, even though population is greater.

Further policy implications may be drawn from both sets of results. For Miconia, we find that at Hawaii's current population level, investing in the optimal program of population reduction is welfare-improving for a range of assumptions concerning costs of control and damages. The status quo policy will result in the stock growing to reach almost the maximum at 99.8 million trees. It is only because we continue to spend the \$1 million that keeps the population from reaching its carrying capacity. Surely it is not the intention of resource managers to spend significant funds to sustain a plant population to be practically equal to its natural carrying capacity. Rather, optimal policy calls for comparisons of the time paths of costs and damages, and how various population levels influence these paths.

For the Brown Treesnake, results suggest that maintenance of a very low population is the optimal policy choice. The status quo policy of spending \$2.6 million per year on intercepting arrivals without increased efforts to prevent escapees from establishing the minimum viable population will likely lead to a rapid steady state population near carrying capacity. Since we are not certain about the true costs of controlling snakes that are already present, we may be underestimating the benefits of prevention. Our results suggest, however, that it is more advantageous to spend money finding the small population of snakes as they occur than attempting to prevent all future introductions. There may be significant payoffs from investing in more control in Hawaii. In the unlikely event that there is a large population of snakes in Hawaii, it would be useful to learn this early, rather than waiting until the population is unmistakable. We imagine that under other assumptions regarding initial populations and N the optimal policy will involve both prevention and control expenditures. To the extent that these biological parameters will always be uncertain, it is reasonable to believe that diversification of prevention and control strategies may be desirable. In future work we plan on exploring the conditions under which spending more on prevention will lead to the optimal result.

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Figure 1. Miconia's Range and Damage Potential

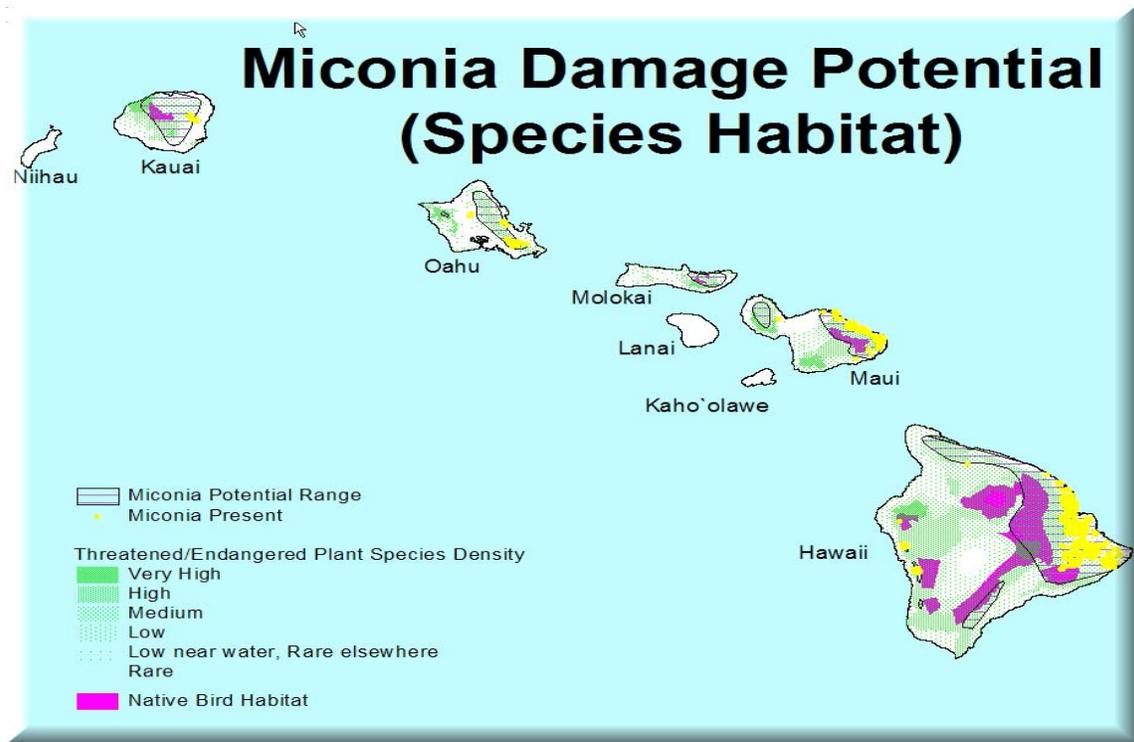


Figure 2. Groundwater Resources and Miconia

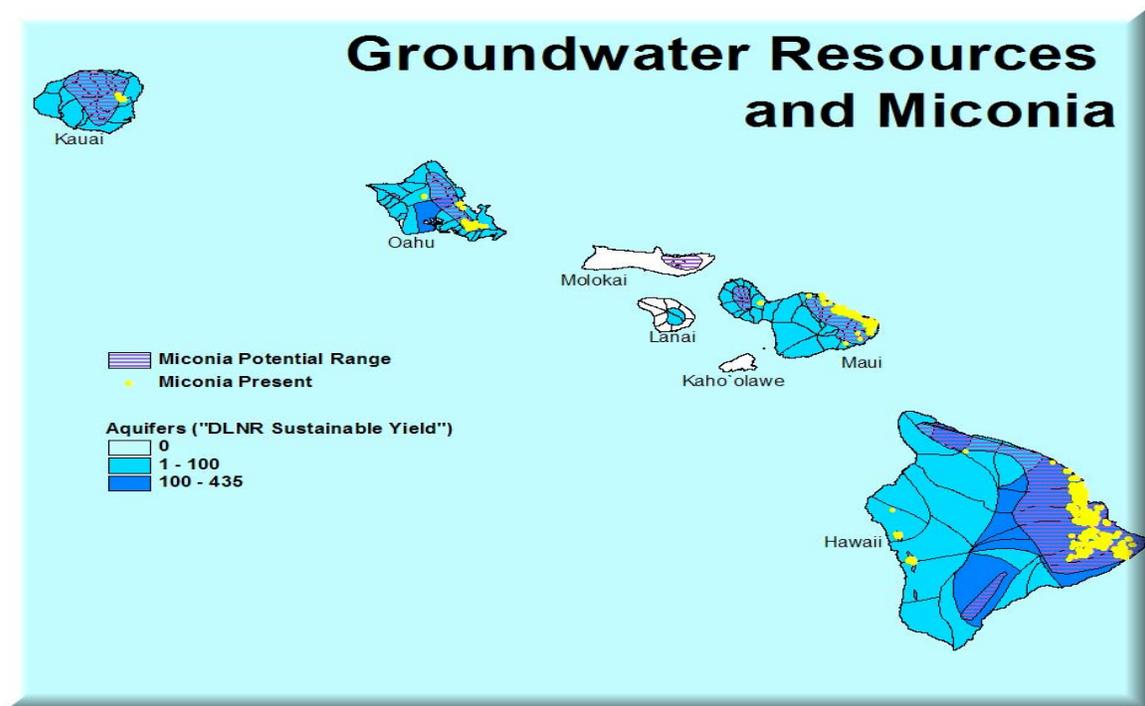


Figure 3. PV of Alternatives Stationary Populations (case 1)

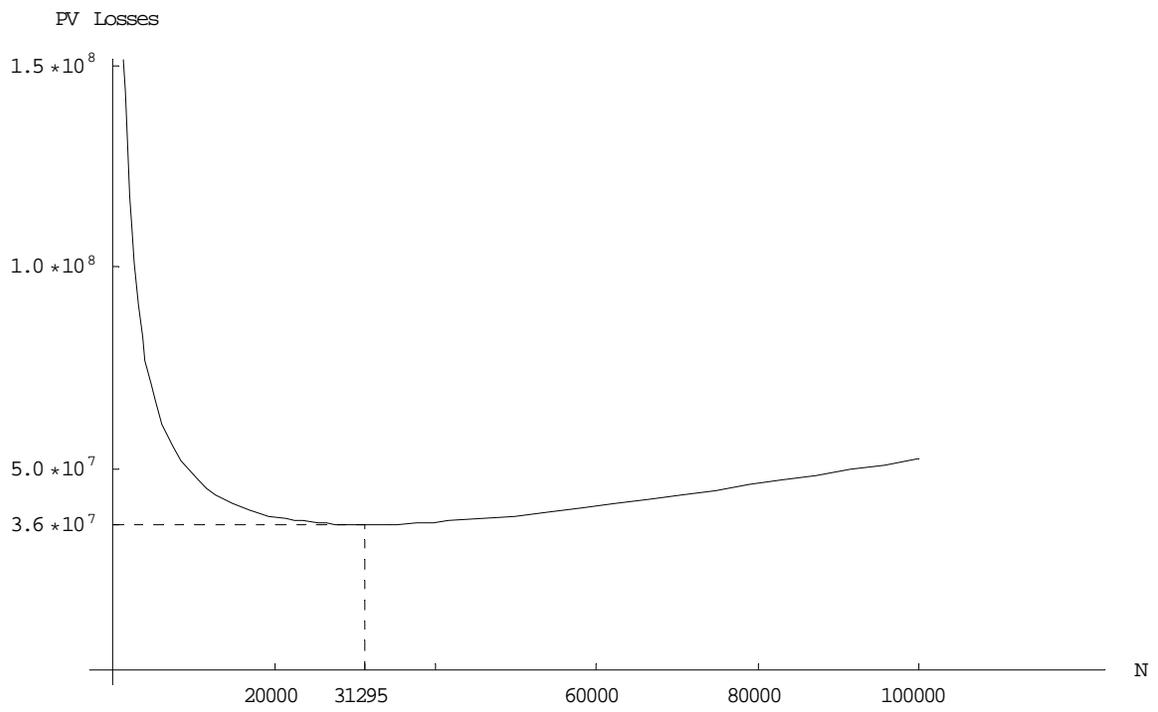


Figure 4: Brown Treesnake and Native Bird Habitat in Hawaii

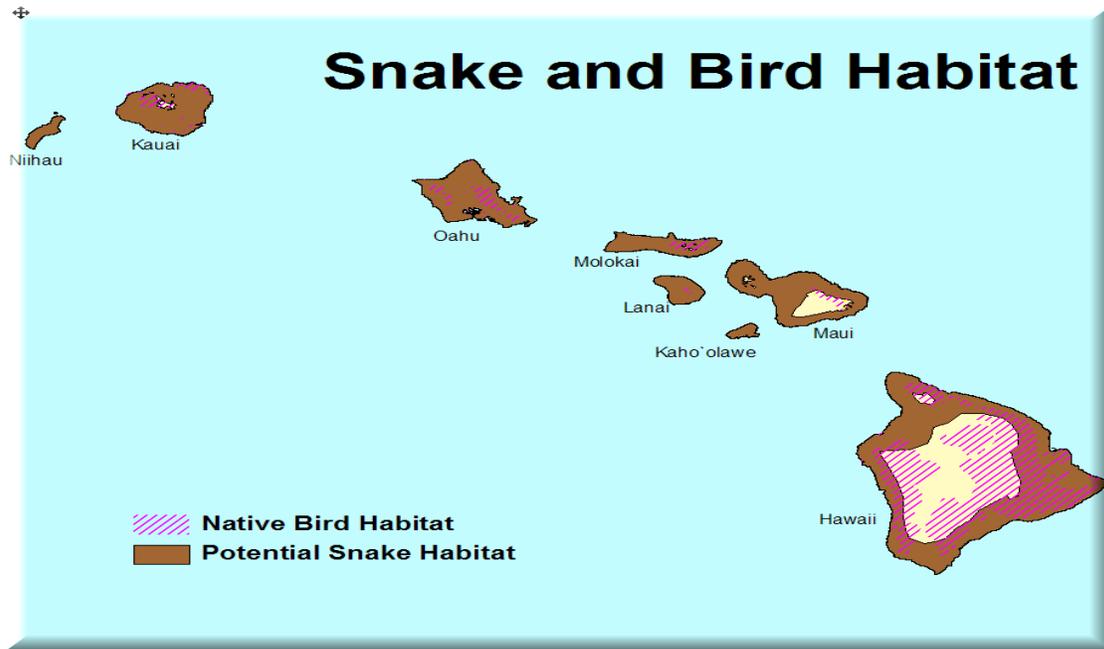


Figure 5. Probability of Arrival Given Prevention Spending

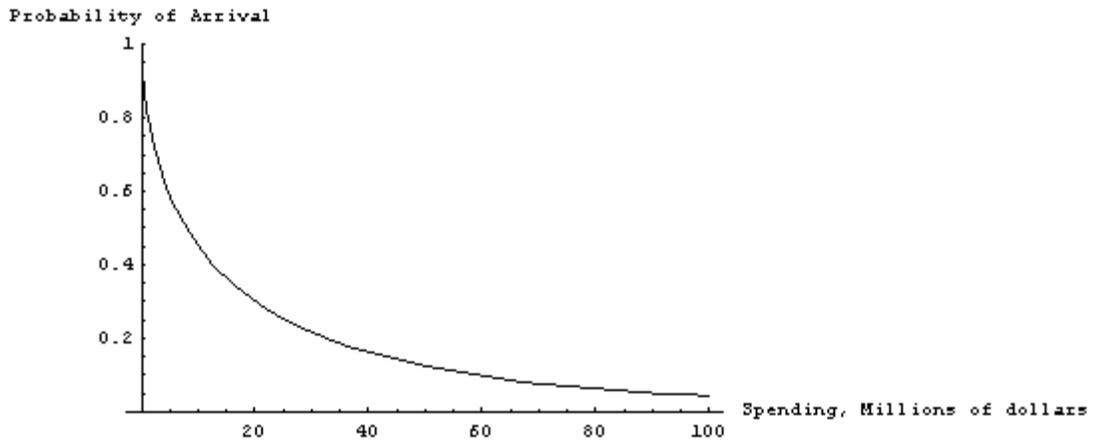


Figure 6. Present Value of Net Benefits for n Snakes

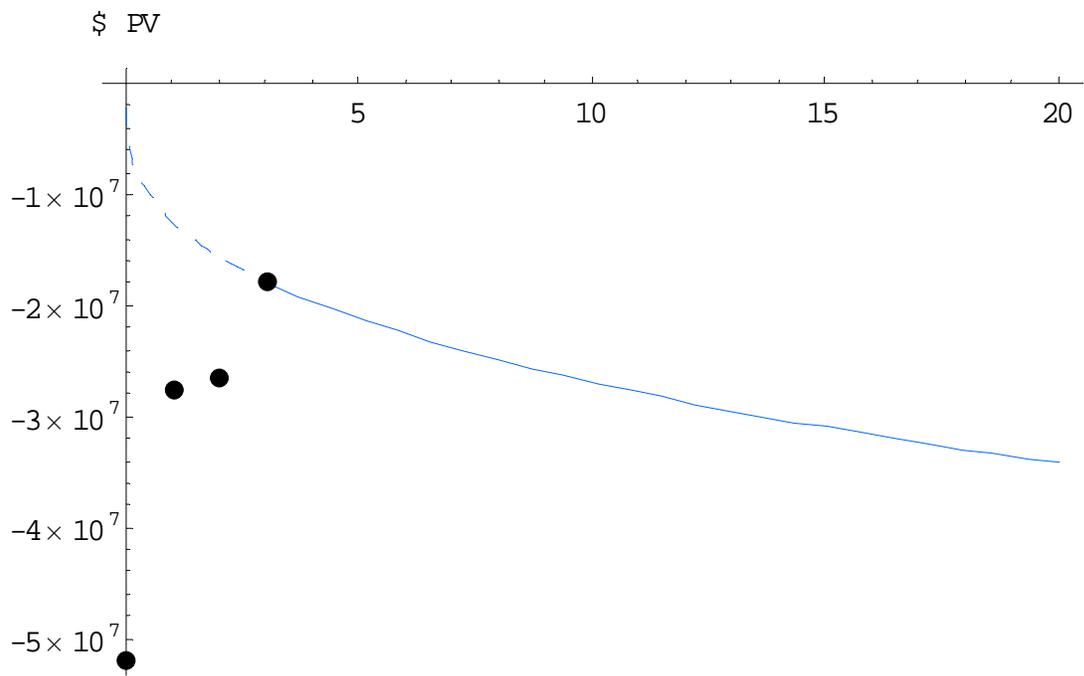


Figure 7. Population Growth Under Status Quo Control

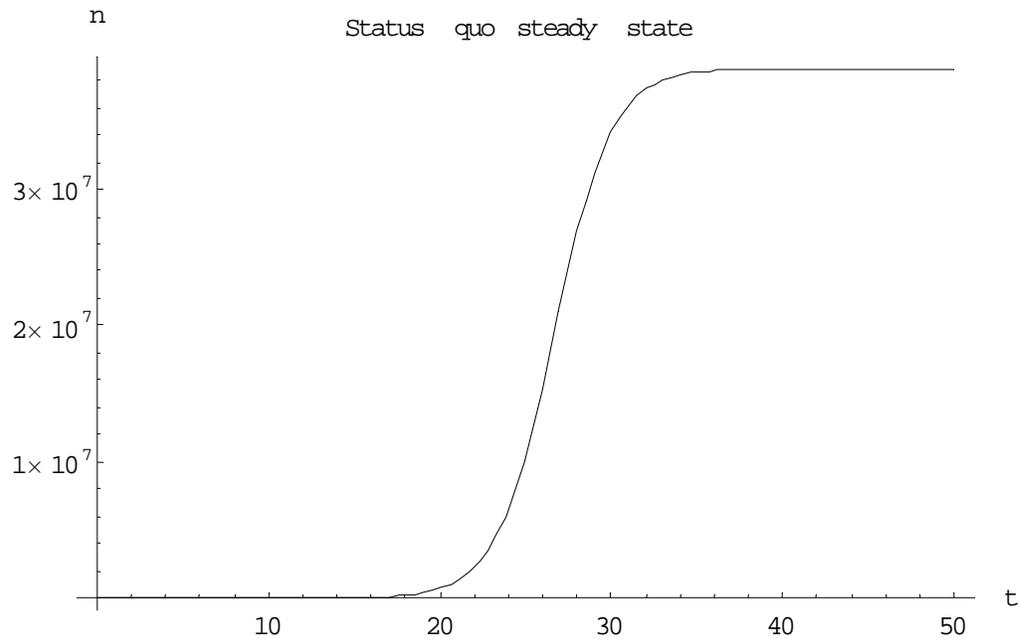


Table 1. Miconia Specifications

Case	Cost function	Damage function	Steady state	Policy
1	$\left(\frac{1,000,000,000}{n^{1.66}} + 13.39 \right) * x$	$3.77n$	31,295	Reduction
2		$3.38n + 4.83 * 10^{-7} n^2$	32,222	Reduction
3		$7.67 * 10^{-3} n$	47,557 100,000,000	Reduction or Accommodation (indifference)
4		$3.77n^{0.7}$	46,830 if $N_o < 2.9$ m 100,000,000 if $N_o > 2.9$ m	Reduction or Accommodation (threshold of 2.9 m)
5	$\left(\frac{1000000000}{n^{0.96}} + 13.39 \right) x$	$3.77n$	100,000,000	Accommodation
6	$\left(\frac{370000000}{n^{0.96}} + 13.39 \right) x$		0	Eradication
7	$\left(\frac{1385000}{n^{0.96}} + 13.39 \right) x$		100,000,000 0	Accommodation or eradication (indifference)

Table 2. Status Quo Policy vs. Optimal Policy, Assuming Current Population of 436,000

	Optimal Steady State Population	PV Losses, Optimal Program	PV Losses, Status Quo (spending between \$1-2 m)	Benefit of switching to Optimal Program
<p>CASE 1</p> $c(n, x) = \left(\frac{1,000,000,000}{n^{1.66}} + 13.39 \right) * x$ $D(n) = 3.77n$	31,295	\$-36.1 m	\$190.4 m-\$12.4 b	\$154.2 m-\$12.3 b
<p>CASE 2</p> $c(n, x) = \left(\frac{1,000,000,000}{n^{1.66}} + 13.39 \right) * x$ $D(n) = 3.38n + 4.83 * 10^{-7} n^2$	32,222	\$-35.5 m	\$55.9 m-\$158 b	\$20.4 m-\$157.9 b

Table 3. Cost Savings for Linear Damages (Case 1)

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

Table 4. Cost Savings for Brown Treesnake Optimal Policy

	First period cost	Annual cost	PV costs	Annual damages	NPV 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

¹ With the exception of *Ramphotyphlops braminus*, a harmless blind snake present in Hawai`i since 1930.

² Instantaneous adjustment to the target population is optimal because the cost of population reduction does not increase with the speed of removal and delays in reduction would only imply further growth and unnecessary harvesting and damage costs. See also Pitafi and Roumasset 2005.

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 .

⁴ Most successful invasions in Hawaii require very small initial populations, due in particular to the favorable climatic conditions and fragility of existing ecosystems. For both the examples presented here, for example, the minimum viable population may be

realistically as low as a single specimen. 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 We use discrete time here for clarity of exposition.

⁶ 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.

⁷ The particular role of *Miconia* in groundwater recharge is uncertain; on the one hand, increased runoff suggests 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).

⁸ 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} . We assume n_{\max} is 100,000,000 plants, based on a density of 100 plants per acre and 1,000,000 potential acres of habitat. 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.

⁹ The recent Hawaii efforts are known to be focused on considerable restructuring costs and very little ground management (pers. comm., CGAPS). These data are therefore not helpful in determining an accurate cost function for harvest.

¹⁰ Though data on number of removals is readily available, budget expenditures on these removals, accurate estimates of the current population, and fixed cost expenditures are extremely poorly reported. We have some evidence, including: (1) A \$500,000 make-work project on the Big Island in 2002 employed 64 persons at \$9.96 for 5 months, resulting in destruction of 337,382 plants. Assuming that these areas will need similar treatment at least three more times over the next decade, and adding in costs of equipment and supplies from estimated expenditures on Hawaii (2002) and Oahu (2004), the resulting average cost per plant is estimated at \$12 for plants in dense, easy-to-access locations. This is approximately the “total treatment cost” of an average tree in a known location. (2) Additionally, from improved budget keeping on Oahu in the last year we infer that costs at their current population of approximately 6890 trees for removing 1100 trees was \$323 per tree (pers. comm., OISC). (3) Maui’s expenditures in 2004 were approximately \$1 million and resulted in a removal of 69,000 trees (pers. comm., MISC). Search costs reflect the cost of locating and accessing trees; as population decreases search and access costs will increase. Due to both wind and bird dispersal of seeds, trees may be spread across all appropriate habitat. Helicopter search costs are estimated at \$1000/acre.

¹¹ This damage function was fitted such that marginal damages at the current population level of 436,000 trees are \$3.77 and \$100 at 1 million trees.

¹² There have been two recent combined attempts by state and federal agencies to catch snakes following single sighting events. One program was executed near Hana, Maui and the other in Kona, Hawaii. Each program lasted 3 weeks and entailed paying trained personnel overtime wages and flying specialized searchers out from Guam and the Northern Mariana Islands. We estimate each program to have cost around \$76,000.

¹³ The cost of catching 1 snake out of a population of 1 is vastly more expensive than catching 1 snake out of 14, or 1 snake out of 38 million. Integrating $c(n)$ between 0 and 1 gives us \$1 million; integrating $c(n)$ between 14 and 15 gives us approximately 76,000; integrating between 38,849,999 and 38,850,000 gives us approximately \$31.00.

¹⁴ They may not, however, be as costly as finding and removing the last *Miconia* tree, due to the abilities to attract the snake with bait or seek it with trained dogs, the lack of wind and bird dispersal of offspring, or a lingering seed bank. The snakes are estimated to move a maximum of 50 m a day so that search areas should be containable, lowering costs.

¹⁵ Total losses here are defined as $V^N + Z^N$ and include the cost to reduce population from 7 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^* .

¹⁶ Now the total cost is everything in mentioned in the above footnote, minus Z^* .

¹⁷ The cost of reduction from 7 to 3 snakes is \$573,510, and the cost and damages of maintaining the 3-snake population is \$17.2 million.

¹⁸ Close monitoring of the progress of the snake in Saipan (CNMI) should give 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.