

Control of Invasive Species: Lessons from *Miconia* in Hawaii

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1. Introduction

Invasive species change ecosystems and the economic services that ecosystems provide. Policy decisions regarding existing invasive species must compare the anticipated and actual costs and benefits of any mitigating actions once a species has become established to the anticipated costs and benefits of accommodating the change (control). Optimal policy regarding invasive species will minimize the expected damages and costs of control within an ecosystem.

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). Here, we seek to apply the theory to the case of an invasive shrubby tree, *Miconia calvescens*, now present in Hawaii. By testing the theory with a real-world case, we intend to highlight both the usefulness of the theory and its limitations.

The case of *Miconia* is used to illustrate dynamic policy options for forest invasive species that are already present in an ecosystem. Hawaii's forest ecosystems provide direct and indirect ecosystem services, with high expected value generated from the preservation of existing ecosystem conditions stemming from unique biodiversity assets. *Miconia* generates concerns that extend from biodiversity to infrastructure for water supply as it threatens moist tropical island watersheds and forest ecosystems.

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. To improve outcomes and avoid costly mistakes ranging from denying beneficial introductions to spending good 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. A case study approach allows us to investigate how useful the existing economic theory of invasions will be at facilitating these efforts.

2. Case Overview: *Miconia Calvescens*

One significant threat to Hawaii's forest ecosystems comes in the form of the woody shrub, *Miconia calvescens*. A member of the Melastomataceae family from Central America, the plant was purposefully introduced to Hawaii; starting 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 yet 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 near-shore 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 forested 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.

3. Methodology: Optimal control of an Existing Invader

As the theoretical literature mentioned above demonstrates, optimal control theory provides an excellent methodology for considering economic policy toward invasive species. Using optimal control, we define our problem so that we minimize the expected costs and damages from the presence of and control activities undertaken against the invading species. Thus the objective function is:

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

subject to:

$$\dot{n} = g(n) - x \tag{1}$$

$$0 \leq x \leq n \tag{2}$$

$$n_0 \text{ given,} \tag{3}$$

Where n and \dot{n} are the population of the invasive species and its associated time derivative, $g()$ the growth function of the invasive, x represents the number of removals, $c()$ the marginal cost function for removals, which varies with population level, and $D()$ the damages incurred at population n .

3.1 Mitigation without Eradication: Internal Solutions

We first seek an internal solution for the choice of control level x in the standard manner (e.g. Clark, 1990), defining the current value Hamiltonian as:

$$H = - \int_{n-x}^n c(\gamma) d\gamma - D(n) + \lambda [g(n) - x].$$

Application of the Maximum Principle (assuming an interior solution for x) generates first order conditions:

$$\frac{\partial H}{\partial x} = -c(n-x) - \lambda = 0 \quad (4)$$

$$\frac{\partial H}{\partial n} = -[c(n) - c(n-x)] - D'(n) + \lambda g'(n) = r\lambda - \dot{\lambda} \quad (5)$$

$$\frac{\partial H}{\partial \lambda} = g(n) - x = \dot{n} \quad (6)$$

Taking time derivatives of (4) yields

$$\dot{\lambda} = -c'(n-x)(\dot{n} - \dot{x}) \quad (7)$$

Substituting (4), (6), and (7) into (5) yields

$$\dot{x} = g(n) - x + \frac{c(n) + D'(n) + [g'(n) - r - 1]c(n-x)}{c'(n-x)}. \quad (8)$$

Optimal harvest of the invasive species will result in a steady state population when

$\dot{x} = \dot{n} = g(n) - x$ and thus

$$\frac{c(n) + D'(n) + [g'(n) - r - 1]c(n-x)}{c'(n-x)} = 0, \quad (9)$$

or

$$-c(n) - D'(n) = [g'(n) - r - 1]c(n-x). \quad (10)$$

In other words, the marginal costs and damages of the steady state population (LHS) must be just equal to the marginal opportunity costs of maintaining that population (RHS). If the LHS is

greater than the RHS, we should be increasing the harvest rate, while if the LHS is less than the RHS, we should be decreasing the harvest rate.

3.2 Eradication or Accommodation: Corner Solutions

The internal solution must be compared to alternative policy options of eradication ($x^*=n$) or accommodation ($x^*=0$). Thus, we compare the present value of eradication and accommodation policies to the present value of the internal solution to determine if the internal solution is dominated by either alternative.

The optimal policy for an existing invader can then be summarized by considering first where the population is in relation to an 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, unless eradication or accommodation has a lower expected present value of costs and damages. 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, unless, again, a corner solution is preferable. 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.

4. Empirical Investigation

We investigate empirically the case of *Miconia calvescens*, discussed above. We determine cost, damage, and growth function parameters with the help of scientists researching the species and resource managers actively pursuing *Miconia* control. As potential habitat size, costs of control, and damages vary widely across space, we specify each by island. The parameters are discussed below, followed by results.

4.1 Growth Function

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

Thus:

$$g(n) = bn \left(1 - \frac{n}{K} \right), \quad (11)$$

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 1.2 million acres, or 120,000,000. 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 as delineated in state GIS data (DBEDT, 2005). 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. The standard logistic function is chosen because a single tree is capable of starting a population. The seed bank is not addressed directly by the growth function; instead, we rely on the need to return to a tree to treat its seed bank over time to build in the effects of the seed bank on reducing the effectiveness of control cost expenditures (see cost function discussion). Carrying capacities for each island are listed in Table 1.

Table 1. Carrying Capacity by island

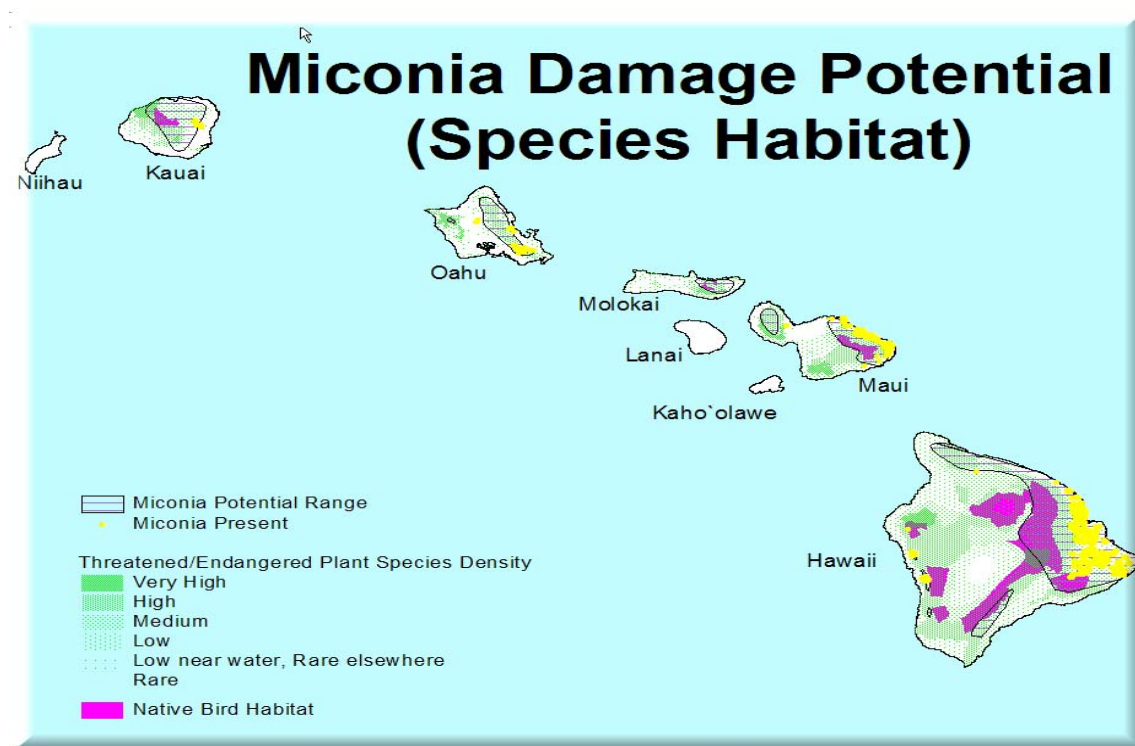
Island	K
Kauai	15,849,057
Oahu	8,713,551
Maui	14,133,791
Hawaii	78,216,124
Molokai	3,087,479

4.2 Damage (Net Benefit) Function

We estimate damages from *Miconia* as evolving from indirect ecosystem services as well as non-market goods like biodiversity. Particularly significant threats are a reduction in habitat for endangered species and a shift in the hydrological cycle that may reduce freshwater recharge and increase runoff and sedimentation.

Figure 1 illustrates the potential range for *Miconia* and the primary biological resources threatened by invasion.

Figure 1. *Miconia*'s Range and Damage Potential



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. For these reasons, we focus on bird losses to provide a lower bound estimate of expected damages.

Additionally, damages to watershed functions are 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 recharge¹. Estimates of potential expected losses from an invasion of *Miconia* on Oahu to groundwater recharge suggest that a loss of 41 million gallons per day (mgd) would generate economic losses of \$137 million per year (Kaiser and Roumasset 2002), or \$3.3 million per mgd. Additionally, increased surface water runoff is expected to increase damages by \$1.2 million per mgd reduction in groundwater due to increased sedimentation costs (Kaiser and Roumasset 2000). Figure 2 illustrates the groundwater assets at risk.

Figure 2. Groundwater Resources and *Miconia*

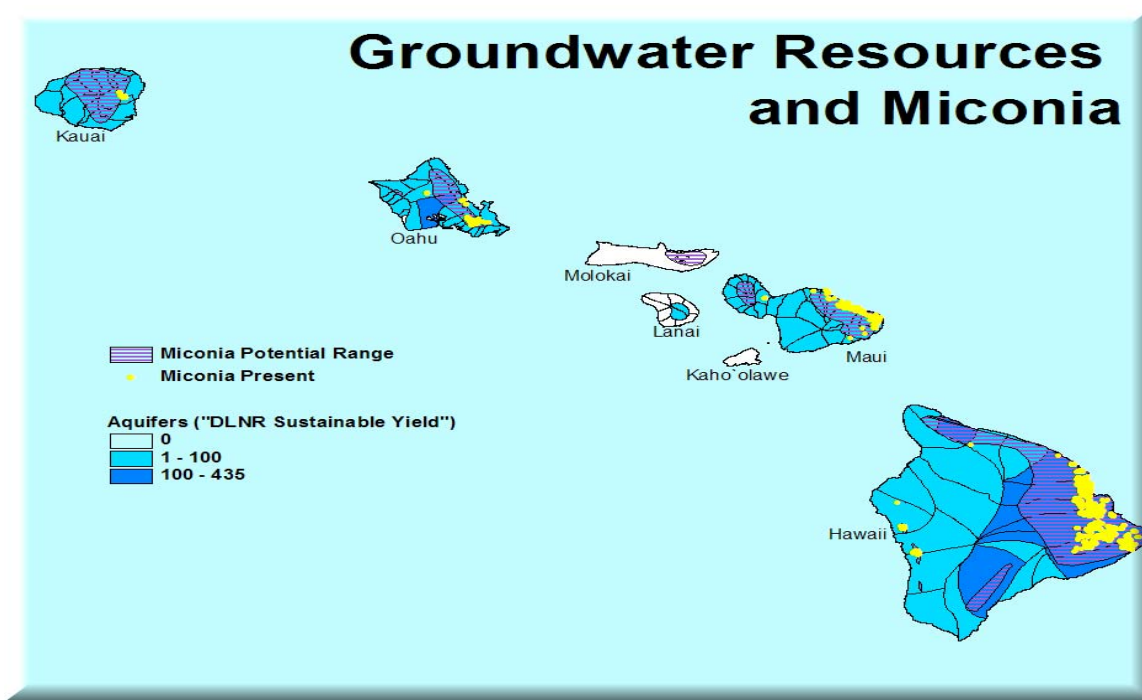


Table 2 shows the expected maximum damages (from a complete accommodation of the invasion) by island, giving a low, medium, and high estimate. We calculate damages by first compiling the assets at risk for each island in terms of threatened or endangered bird species and groundwater recharge to aquifers (mgd). Then the dollar figures described above are included accordingly.

The number of threatened or endangered bird species on the island is shown in column 2 of Table 2. We conservatively assume that $\frac{1}{2}$ of these birds is at risk from habitat loss to *Miconia*, (Figure 1 illustrates), and that Hawaiian households value birds equally regardless of whether they are on the same island as the household or not, while non-Hawaiian household values are not considered though they certainly would increase expected damages. We can then calculate a lower bound range of damages by using half the number of birds per island multiplied by the number of Hawaiian households (400,000) and the estimated loss to a household from a bird's extinction. Note again 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 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.

The potential loss to aquifers is calculated by first summing the sustainable yield estimates from the HI Department of Land and Natural Resources (shown in Figure 2) for aquifers that lie beneath potential *Miconia* habitat. Then, using estimates taken from survey data of forested watershed experts in Oahu regarding the expected increase in runoff that would occur with a shift in forest structure to one dominated by *Miconia* (Kaiser and Roumasset, 2005), we calculate the expected annual reduction in recharge for these areas. The surveys of experts indicate that a 0.85% to 1.04% annual decrease in runoff might occur. We determine a lower bound and an upper bound estimate for lost recharge damages by multiplying these decreases by the sustainable yield. A dollar value is attached by multiplying this groundwater loss by \$4.5 million (\$3.3 million in groundwater + \$1.2 million in surface water generated losses).

Table 2: Damage Estimate Characteristics by Island

Island	# birds	mgd lost recharge		Total annual damages by island (millions)		
		low	high	Low	Medium	High
Kauai	22	3.2	3.9	91.6	154.0	234.0
Oahu	13	3.2	3.9	61.4	98.5	145.0
Maui	17	3.7	4.6	77.8	126.0	187.0
Big Island	20	18.0	22.0	169.0	225.0	297.0
Molokai	11	NA	NA	36.9	68.2	108.0

Total expected damages for any given population are described by the function²:

$$D(n) = d_i n, \tag{12}$$

Where d_i is the linearized per-unit medium damage estimate as shown in Table 4 below.

4.3 Control Cost Function

Control efforts began on Maui in 1991 and continue to expand on the four invaded islands. 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 activities that must occur – the trees must first be found, then treated.

We therefore define a cost function consisting 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 a tree is rapidly decreasing in population size. That is, it is extremely expensive to find the last tree, but much less so to find one tree out of 120 million trees.

Because each island is unique in aspects such as topography, area, and vegetation, it is helpful to estimate separate cost of control functions for all five major islands. We determine the two components for each island in the following manner. The search component involves a fixed cost which depends on the island’s potential habitat acreage and which decreases with increased access to that habitat. Based on discussions with resource managers, searching one average acre for *Miconia* costs approximately \$1,000. Therefore, the numerator of the search component is obtained by multiplying the number of potential acres by \$1,000.

However, the ability to search an island’s habitat will depend on several characteristics of the surrounding area, such as density of vegetation, the steepness of the terrain, etc. One major determinant is ease of access into the potential habitat. We use the combined length of roads and trails as a proxy for this variable. The length of roads and trails as compared to Molokai, the most expensive island to search because it has the fewest roads and trails per acre of habitat, is used to determine the coefficient γ in the denominator of the search component. Higher values for γ imply greater ease of access, which translate into lower search costs.

$$\sigma(x) = \frac{\$1,000 * \text{potential acres}}{n^\gamma},$$

$$\tau(x) = 13.39,$$

where $\sigma(x)$ represents the search cost component and $\tau(x)$ represents the cost of treatment, constant across all population levels. Total marginal cost is thus defined as:

$$c(n, x) = \sigma(x) + \tau(x) = \frac{\$1,000 * \text{potential acres}}{n^\gamma} + 13.39, \text{ so that the total cost of searching and}$$

treating x trees is:

$$c(n, x) = \left(\frac{\$1,000 * \text{potential acres}}{n^\gamma} + 13.39 \right) * x \quad (13)$$

While search costs will differ across islands, we assume that treatment costs remain constant across islands. We begin by fitting these functions for the islands of Oahu and Maui. While the exact budget for all islands is uncertain, we do have a good understanding of control budgets on Oahu and Maui. Oahu Invasive Species Committee allocated \$321,000 to *Miconia* control in 2005 (Ryan Smith, personal communication) while Maui expends approximately \$1 million per year on *Miconia* related activities (Teya Penniman, personal communication). These status quo levels of expenditures, along with estimates of current populations, allow us to parameterize the control function appropriately.

Specifications for each island's control costs are given in Table 3.

Table 3: Cost of Control Function Parameters

Island	Search Constant (acres*1000)	Search Coefficient (γ)	Treatment Coefficient
Kauai	158,490,570	1.6095	13.39
Oahu	87,135,510	1.6258	
Maui	141,337,910	1.6089	
Big Island	782,161,240	1.6028	
Molokai	30,874,790	1.6	

4.4 Optimal Control Results

4.4.1. Expected Scenario

Using the parameters calculated above and the assumption that the current stock of *Miconia* in Hawaii is given by the initial populations listed with the summary of parameters in Table 4, we find the following. Optimal policy calls for population reduction on the islands of Oahu, Maui, and Hawaii, population expansion on Kauai, and continued prevention on Molokai.

Table 4: Summary of Bio-economic Parameters

Island	n_0	K	Coefficient on Damage function	Cost of control function
Kauai	1,540	15,849,057	9.74	$\left(\frac{158,490,570}{n^{1.6095}} + 13.39\right) * x$
Oahu	6,890	8,713,551	11.30	$\left(\frac{87,135,510}{n^{1.6258}} + 13.39\right) * x$
Maui	111,050	14,133,791	8.93	$\left(\frac{141,337,910}{n^{1.6089}} + 13.39\right) * x$
Hawaii	315,000	78,216,124	2.88	$\left(\frac{782,161,240}{n^{1.6028}} + 13.39\right) * x$
Molokai	0	3,087,479	22.09	$\left(\frac{30,874,790}{n^{1.6}} + 13.39\right) * x$

Differences in steady state populations are mainly the result of variations in search costs and potential habitats. For example, the population on Kauai is currently “too small,” that is, the high search cost calls for waiting until the population is larger to invest in harvesting. Although the damage per tree is significant, it does not outweigh the magnitude of the search component of control.

On the island of Oahu, however, ease of access to *Miconia* habitat is facilitated by the comparatively large amount of roads and trails on the island. Furthermore, the search cost on Oahu is almost half that of Kauai (due to fewer potential acres of habitat). Therefore, we find reduction of approximately 1,400 trees to be optimal.

Maui and Hawaii have lower per unit expected damages than Oahu, and higher search costs due to both greater amounts of habitat and more difficult access. While optimal populations are higher than Oahu and Kauai, significant reductions of the Maui and Hawaii populations are preferred to the current state.

For the island of Molokai, where no trees are currently believed present, annual monitoring expenditures of \$13,500 should be continued. If this prevention fails and a population establishes on the island, we find the steady state population occurs at 2,300 trees. At this level, annual costs of the invasion are minimized at \$149,000, significantly higher than prevention.

Table 5. Initial versus Optimal Populations for the Five Major Hawaiian Islands

Island	n_0	n^*
Kauai	1,540	9,171
Oahu	6,890	5,495
Maui	111,050	8,901
Hawaii	315,000	39,937
Molokai	0	0

While we conjecture that the above parameterization best describes the *Miconia* story in Hawaii, we repeat our analysis using 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. In particular, we examine changes in the cost function that allow for the possibility of eradication and changes in the damage function that are non-linear and/or allow for the possibility of accommodation.

4.4.2 Alternative Functional Forms for Damages: Control versus Accommodation

The parameters above assume constant marginal damages across population levels. If we instead use the increasing marginal damage functions³ shown in column 2 of Table 6, we find higher steady state equilibrium populations, since marginal damages at lower populations will be lower as the damages are delayed until the population is greater. Optimal policy still calls for population reduction on both Maui and the island of Hawaii due to their high initial number of trees and thus high damages, but now requires Oahu to engage in population expansion. This is a

reflection of the much lower damages⁴ incurred at low population levels under this damage function.

Table 6. Increasing Marginal Damages

Island	Damage function	n_0	n^*
Kauai	$D(n) = 8 * 10^{-2}n + 6 * 10^{-7}n^2$	1,540	20,009
Oahu	$D(n) = 2 * 10^{-2}n + 1.3 * 10^{-6}n^2$	6,890	12,833
Maui	$D(n) = 5 * 10^{-2}n + 6.28 * 10^{-7}n^2$	111,050	18,787
Hawaii	$D(n) = 1 * 10^{-2}n + 3.68 * 10^{-8}n^2$	315,000	56,895
Molokai	$D(n) = 5 * 10^{-2}n + 7.1 * 10^{-6}n^2$	0	0

If damages are sufficiently small or sufficiently delayed, accommodation may become more attractive than control. However, for this case, maximum damages would have to be over two orders of magnitude smaller than our minimum estimates to meet this condition. For expositional purposes, the linear coefficients on demand for each island for which the optimal internal solution and accommodation generate equivalent present value societal losses are provided in Table 7 below.

Table 7: Damage Specifications: Control versus Accommodation

Island	Control dominates accommodation		Control and accommodation equivalent		Maximum total damages below which accommodation becomes optimal ($d_i K_i$)
	d_i	n^*	d_i	n^*	
Kauai	9.74	9,171	0.01881	20,290	7.34×10^6
Oahu	11.30	5,495	0.02150	12,900	5.59×10^6
Maui	8.93	8,901	0.02090	18,951	1.07×10^7
Hawaii	2.88	39,974	0.01178	56,918	3.20×10^7
Molokai	22.09	0	0.01349	7.676	2.38×10^6

4.4.3 Lower Search Costs at Low Population Levels: A Case for Eradication

As a third case, we consider the possibility that the search costs for the last trees are not so extravagant, and that eradication costs are not infinite. This might correspond to a case where spatial containment is possible, so that search could be limited in area. With both wind and bird dispersal of *Miconia* seeds, this seems unlikely to be the case, but we consider it for expository purposes. We make the following adjustments in cost functions, shown in reducing both the fixed component of the search cost and the rate at which costs decline.

$$c(n, x) = \left(\frac{\text{potential acres}}{(n)^{\gamma'}} + 13.39 \right) * x,$$

where we assume that search costs average to \$1 per acre for the final tree on each island, or, alternatively, that the problem is spatially confined and we need only seek across about 1/1000 of the habitat. The coefficients on search are reduced as shown in Table 8, and treatment costs remain the same. The marginal costs at n_0 and K are shown for comparison to the existing data, which they match well. We use the linear demand functions whose coefficients are listed in column 2 of Table 7.

Table 8: Reduced Cost Functions and Optimal Populations of Zero

Island	Search Constant (potential acres)	Search Coefficient (γ')	Implied marginal cost at n_0	Implied marginal cost at K	n_0	n^*
Kauai	158,491	0.6095	\$1821.80	\$19.90	1,540	0
Oahu	87,136	0.6258	\$358.70	\$17.30	6,890	0
Maui	141,338	0.6089	\$133.10	\$19.60	111,050	0
Hawaii	782,161	0.6028	\$393.00	\$27.00	315,000	0
Molokai	30,875	0.6000	\$77,200.90	\$17.30	0	0

In this case, eradication is optimal on each island for each initial population. At the damage levels where there was before indifference between accommodation and an internal solution, however, accommodation is now the cost effective strategy. In other words, these flatter marginal costs, which impose slightly greater costs of removal at capacity (e.g. \$19.8 vs. \$13.4 for Kauai) generate sufficiently large present value costs so that accommodation is preferable.

4.5 Status Quo vs. Optimal Policy for Expected Scenario

Current *Miconia* policy in Hawaii entails spending different amounts on control efforts on each island. As a final exercise for this species, we compare the consequences of status quo spending to those associated with the optimal policy program for Oahu and Maui in order to investigate the extent to which status quo expenditures may be misaligned with optimal expenditures. We also highlight consequences of two other potential policies: doing nothing, or spending on control in order to remain at the same initial population forever. Comparisons are drawn in Table 9 below.

Table 9. Present Value Policy Comparisons for Oahu and Maui, Expected Scenario

Island	Policy			
	Do nothing	Remain at current population forever	Status quo spending	Optimal policy of population reduction
Oahu	\$3.08 billion	\$10.5 million	\$16.9 million	\$10.4 million
Maui	\$4.6 billion	\$73.5 million	\$51.7 million	\$17.2 million

From Table 9, we see that if Oahu switches to the optimal policy of population reduction instead of spending \$321,000 per year from today into the future, a present value benefit of \$6.5 million can be realized. Likewise, if Maui switches to their optimal policy of population reduction, a net present value benefit of \$34.5 million is possible. The difference is more significant for Maui due to their larger current population and higher cost of control.

5. Limitations and Directions for Further Research

Using optimal control theory, we generate appropriate comparisons for policy options concerning an existing invasive species. In the cases above, we show that the status quo policy for *Miconia* is woefully inefficient and improving the policy would provide additional resources to combat other ecological and economic concerns. We find that eradication, internal controlled steady state populations, and accommodation might all be optimal outcomes depending on the forms and parameterization of the cost, damage, growth functions and initial level of the invasion. We argue that the parameterization most accurately reflecting the current situation requires removals and maintenance of steady state populations on Oahu, Maui, and Hawaii, whereas Kauai's population could grow slightly before reaching a cost-effective steady state.

Molokai should continue their current levels of spending to prevent invasion of *Miconia* to their island. Additionally, as the optimal policies can be traced out across time, we can compare the costs and benefits of deviating from optimal policies if budgets are restricted below the point where the optimal policy can be implemented in the present.

We conclude with an acknowledgment of the limitations of our ability to analytically determine optimal policy for prevention and control and a discussion of directions for further research. In the simple cases described above, we encounter both quantitative limitations and theoretical ones.

In computing the optimal outcomes for *Miconia calvescens*, we encountered quantitative challenges regarding the specification of functional forms for all three essential components: costs, damages, and growth. In particular, choosing functional forms that both accurately reflected our understanding of the biological and economic processes and resulted in computationally feasible equations required several simplifications upon which further research might improve.

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 sublimating the removal of future growth from seeds into the cost of removing the parent tree so that we did not need to invoke delay-differential equations, which would render computations extremely complex if even solvable. We also abstracted away from important spatial considerations of the problem. 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 with search cost functions that differ across islands and include considerations for access, and are discussed as potential shifts in the cost curves, it would be useful to model these considerations more explicitly.

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 the results. For *Miconia*, we find that at the state's current population level, investing in the optimal program of population reduction where appropriate is welfare-improving for a range of assumptions concerning costs of control and damages. In particular, damages could be more than 2 orders of magnitude smaller and the steady state populations would still remain closer to the estimated optimal population at the higher damages than they would to the current population (e.g. $N^*(\text{expected damages}) = 39,974$, $N^*(\text{low damages}) = 56,919$, $N_0 = 315,000$ for Hawaii). However, damages below this threshold make accommodation preferable.

The status quo policy will result in the stock growing to almost reach capacity. It is only because we continue to spend an annual \$1 million on Maui and \$321,000 on Oahu that keeps the population from reaching its carrying capacity on each island. 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.

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¹ 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).

² 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 120,000,000 plants, based on a density of 100 plants per acre and 1,200,000 potential acres of habitat. To model a potentially more realistic situation where the damages are increasing at an increasing rate with population, the beta distribution might be preferred.

³ This damage function was fitted such that total damages at the island capacities match estimated damages, and are equal to the linear damage values at $n=10\%$ of capacity.

⁴ Smaller damages at lower population levels is a reasonable assumption, in that most watershed and biodiversity damage is expected to be borne under dense stands of the trees, rather than with a sparse number of trees spread out around the island.