

# Models of Spatial and Intertemporal Invasive Species Management

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## 1. Introduction

Damages from invasive species are spatially and intertemporally variable. We define invasive species as those which have negative net benefits to society when introduced to an area in which they are non-native.<sup>1</sup> Valuation of these damages is often the first uncertain step in determining policy responses to invasive species problems.

As an invasive species spreads and increases in density across a landscape over time, the costs of locating and controlling it also change. Human intervention must therefore be spatially and temporally sensitive if it is to achieve the goal of minimizing net losses from the spread of invasive species. The three main, interdependent policy interventions are prevention, early detection and rapid response (EDRR), and control.

For clear guidance on optimal responses, all three policies require information on the likelihood of arrivals and establishment or re-establishment of an invasion, expected growth (spread), control costs, and expected damages. This is due to the recursive nature of the problem; spending large amounts of money to prevent a species that can be cheaply controlled at levels where it causes little actual damage if it establishes is a waste, while spending large amounts of money to control for a species that is likely to re-invade without integrated prevention decisions may also be a waste. Unfortunately, due in part to this recursivity but also due to the generally nonlinear nature of biological growth and spread, analytical solutions to a fully integrated, spatially and temporally explicit prevention and control problem even for a single species are generally intractable (see Smith et al. 2007, Burnett 2007).

Numerical solutions, with caveats and assumptions about transference of biological growth, expected costs, and damages from other locales, are possible for species with sufficient information about these parameters and the likelihood of invasion. A small but growing set of such case studies is evolving both in the economics and the ecological literatures, though few to date tackle both spatial and temporal issues together (Rejmanek and Richardson 1996, With 2002, Eiswerth and Johnson 2002, Burnett et al. 2006, Kaiser and Burnett 2007, Burnett et al. 2007). Certain locations encourage and facilitate analysis; the Hawaiian islands, the Cape of

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<sup>1</sup> Many introductions are purposeful as they convey anticipated net benefits for those responsible for the introductions; in these cases there is the additional complication of unaligned incentives and distributional considerations in policy. We abstract from these considerations here, but mention them to highlight the fact that many of the consequences of invasive species are inflicted upon ecosystems (and their ecological benefits) rather than markets.

South Africa, Australia, and New Zealand, for example, all have fragile, isolated ecosystems where the rate of change in species introduction has rapidly increased with increased global integration over the past 400 (or fewer) years. These locations generate valuable benefits from biodiversity and are also, as they try to develop diversified global economies supporting growing populations and/or tourism, dependent on ecosystems for services like water quantity and quality, agricultural production, and aesthetics or other environmental factors that create a general satisfaction with life.

Due to the visible and significant threats these localities face, they are understandably at the forefront of efforts to manage invasive species problems. We focus on the Hawaiian Islands as a representation of the broader threat because in Hawaii the full problem, from establishment to eradication and back again, is writ large. We present three cases, described below, as analyzed independently in previous research, and draw comparisons and generalizations as possible from them.

First, we focus on measuring damages from an invasive species. This analysis does not inform policy decisions regarding prevention, EDRR, or control directly. Rather it demonstrates the first step in determining the expected damages. We examine the costs from frogs (*Eleutherodactylus coqui*) on the Big Island in Hawaii in terms of noise pollution effects on property values (Kaiser and Burnett 2006). We recap it and add it to the discussion here because it captures an essential consideration for ecosystem valuation and the threat from invasive species. In the words of Joni Mitchell: “you don’t know what you’ve got till it’s gone.” The ability to value the anticipated losses from the frog depends on the losses that have already occurred due to the early stages of the invasion.

Second, we investigate optimal control of a species with a limited presence already, the shrubby tree miconia (*Miconia calvescens*) (Burnett et al. 2007). Significant ecological damages are anticipated from the continued spread of the tree. Some of these damages have market connections, in particular ground water quantity, while others do not, in particular biodiversity.

Third, we investigate optimal EDRR of a species with a possible, currently undetected, presence, the brown treesnake (*Boiga irregularis*) (Kaiser and Burnett 2007). Significant economic and ecological damages are anticipated from the snake’s presence in Hawaii. The snake threatens some of the same ecosystem benefits as the tree (biodiversity) as well as the power supply and human health. Several specimens have been intercepted between Guam and Hawaii in the past 20 years, and it is possible that others have gone undetected. The appropriate policy tool in this case, EDRR, is explicitly spatial, as the searches, and their costs, are location specific.

From these three cases, we cull findings on the sensitivity of policy decisions to the parameters of import outlined above, namely arrivals, biological growth, control costs, and damages.

## 2. Case Studies

### 2.1 Coqui Frogs

*Eleutherodactylus coqui*, a small frog native to Puerto Rico, was introduced to Hawaii in the late 1980s, presumably as a hitchhiker on plant material from the Caribbean or Florida. The frogs are present on the four main islands of Kauai, Oahu, Maui, and the island of Hawaii, although the populations are limited to specific areas on each island.

The primary economic effect of the frog is noise pollution. The combined lack of predation and competition for resources has resulted in densities reaching 55,000 frogs per hectare,<sup>2</sup> more than double the highest densities in the frog's native Puerto Rico (Beard and Pitt 2005). The males' calls, which are individually between 80-90 dBA at 0.5 m, now extend from an hour before sunset until dawn. The Hawaii Department of Health sets the threshold for minimizing impacts to human health and welfare at only 70 dBA (Department of Health, Hawaii Revised Statutes Section 324F-1). We concentrate on elucidating these damages through changes in property values. Economic theory suggests that property values for locations with noise pollution should be lower than comparable properties without. Since the frog's calls reach approximately 500 to 800 meters, we investigate whether properties within this range of a registered coqui complaint trade at lower prices than those beyond that perimeter.

We use a standard hedonic pricing model to evaluate the effect of registered coqui complaints on property values. Using this theory and a of real estate transactions from 1995 to 2005 for Hawaii county, we consider that individuals buy and sell properties as bundles of characteristics: here, the relevant characteristics for the properties are proximity of frog complaints, district, acreage, year of transaction, presence of housing structures, broad zoning class, and finely gradated neighborhoods as defined by the tax authority.<sup>3</sup> Our reduced form price function is:

$$P_i = f(D_i, F5_i, F8_i, A_i, M_t, L_i, Y_i, Z_i, N_i), \quad (1)$$

Where  $P_i$  = natural log of sales price of transaction  $i$ ,

$D_i$  = district (Puna, South Hilo, North Hilo, Hamakua, North Kohala, South Kohala, North Kona, South Kona, Kau),

$F5_i$  = indicator variable for frog complaint within 500 m previous to sale,

$F8_i$  = indicator variable for frog complaint between 500-800 m previous to sale,

$A_i$  = natural log of acres for property  $i$ ,

$M_t$  = natural log of average mortgage rate for month of transaction,

$L_i$  = indicator variable for housing structures on property,

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<sup>2</sup> Densities of up to 133,000 per ha have been recorded on the island of Hawaii.

<sup>3</sup> Ideally, we would wish to include housing stock to control for effects of changes in supply. Unfortunately, this data is not available. The best we can do is use this time trend to broadly capture such differences. The neighborhood variables also help control for supply shifts, however these cannot be isolated as instruments so a two-stage estimation procedure is not possible.

$Y_i$  = year of transaction,  
 $Z_i$  = zoning class (agriculture, apartment, unimproved residential, improved residential, conservation, industrial, resort, commercial)  
 $N_i$  = tax assessor's neighborhood classification (1736 groupings).

We have data from the Hawaii County Tax Assessor's office on 50,033 real estate transactions and properties from 1995-2005, shown in Figure 1. We omit unvalidated sales and sales that fall within the lowest 1% or highest 1% of prices to eliminate outliers and pricing irregularities. This results in 37,228 properties, each of which changes hands between 1 and 6 times (average 1.2 times), for a total of 46,405 transactions.

Table 1. District Level Summary Statistics

District	Number of transactions	Mean Transaction Price (\$) (standard error)	Mean fraction of properties within 500m of frog complaints (standard error)
Puna	20,914	25,912 (40,177)	0.17 (0.38)
South Hilo	4,163	99,130 (81,389)	0.37 (0.48)
North Hilo	412	128,321 (110,007)	0.00 (0)
Hamakua	683	123,091 (109,196)	0.02 (0.13)
North Kohala	1,452	179,028 (153,884)	0.01 (0.09)
South Kohala	4,595	197,095 (176,779)	0.21 (0.41)
North Kona	7,871	187,438 (150,954)	0.33 (0.47)
South Kona	1,427	124,315 (154,234)	0.14 (0.35)
Kau	5,049	23,362 (43,874)	0.00 (0.04)

We expect that frog complaints cause a greater reduction in property values the closer they are. Currently, we have frog complaints reported to USDA/APHIS or the Big Island Invasive Species Committee (BIISC) from 1997-2001. We use geographical information systems (GIS) software (ArcView) to match the verified frog complaints to property transactions, and generate indicator variables for whether a property is within 500m of a previous complaint and whether it is within 800m of a previous complaint. We then generate an indicator variable for whether a property is between 500-800m of a previous complaint. Incentives of both buyers and sellers are such that properties with frogs should trade at prices lower than properties without frogs, and our reduced form estimates include loss in value to sellers as well as the lower willingness to pay of buyers.

The remaining variables control for other characteristics of properties affecting their value, and more detailed discussion can be found in Kaiser and Burnett (2006).

Table 2 shows the results for the regression including all of the districts (neighborhood controls not reported). Note that Puna is the omitted district and agriculture is the omitted zoning, so that the interpretation of the dummy variables is relative to the constant term representing Puna agricultural land transactions. Since we have transformed the continuous variables into logs, the results of our analysis will estimate elasticities. Thus, a one percent change in acreage, for example, will generate an estimated percent change in price indicated by the coefficient in column 2, Table 2, or 0.43 percent.

Table 2. Regression Results (dependent variable: Log Price)

Variable	Coefficient	Standard error	P-value
Frog500m	-0.16	0.01	0.00
Frog800m	-0.12	0.01	0.00
Log Acres	0.43	0.02	0.00
S_hilo_acres	-0.12	0.04	0.00
N_Hilo_acres	-0.15	0.07	0.03
Hamakua_acres	-0.06	0.04	0.09
N Kohala_acres	-0.08	0.03	0.01
S Kohala_acres	-0.24	0.03	0.00
N Kona_acres	-0.26	0.03	0.00
S Kona_acres	-0.31	0.05	0.00
Kau_acres	0.17	0.07	0.02
Log mortgage rate	-0.45	0.04	0.00
Residential structure	1.27	0.01	0.00
Year of sale	0.07	0.00	0.00
Improved Residential	0.23	0.14	0.10
Apartment	0.31	0.17	0.07
Commercial	0.14	0.26	0.58
Industrial	1.98	0.17	0.00
Conservation	-0.19	0.20	0.34
Resort	0.32	0.19	0.09
Unimproved Residential	0.53	0.33	0.11
Constant	-139.57	4.51	0.00

From the table, we see that most variables have the expected sign and influence on price. Virtually all variables are significantly different than zero at the 99% level (P-value < 0.01) (Huber-White robust errors correcting for heteroskedasticity due to the wide variation across districts). The overall fit of the regression is quite good, with an  $R^2$  of 0.86.

The net impacts are in general fairly small, with only the residential structure and industrial property indicators, in addition to some neighborhood indicators (not reported), generating impacts on price greater than 1%.

The presence of frogs, however, does have a significant negative impact on property values. For properties within 500 meters of a complaint, property values decline 0.16%, or about 1/3 as much as values decline from a 1% increase in mortgage rates (-0.45%). For properties within 800m but not within 500 meters, property values decline less severely, at 0.12%. This is about 1/4 of the drop from a 1% increase in mortgage rates.

Thus we have an estimate of net marginal damages from the spatial spread of the frog as a function of the properties in an invaded location. We could use this estimate, with additional

estimates of damages to the floriculture industry, in conjunction with estimates of the cost of spread and the costs of capture to generate control policies for the frog. Misaligned incentives and missing information hinder this analysis, however. The floriculture industry, for example, is reluctant to share information on the frog's effects on their business.<sup>4</sup> The spread of the frog has been much faster and at a higher density than its behavior in its native range would suggest and so it has been underestimated over the last fifteen years. Early control techniques (e.g. spray caffeine) resulted in significant external costs to ecosystem health and had to be abandoned; new techniques (e.g. direct application of hydrated lime) are costly and not as effective. Hand capture is often possible for individual males because they can be located by their call, but female frogs do not call and also are believed to spend the days in the forest canopy, making them difficult control targets.

There is some risk that the frog is reducing native arthropod populations, but the science regarding the extent of this possibility remains unclear, as the frogs exhibit quite generalist eating behavior. While some might argue that the damages from the frog are not communal and that the frog should be treated like any household pest, left to the individual owners to treat or not treat, large source populations exist on public land. Control of these populations as well as prevention of the spread of the frog to new areas is clearly within the scope of public policy. The rule of thumb for such invasions has generally been that the quicker one acts the lower the overall costs. We examine this belief by examining the cases of miconia and the Brown treesnake, below.

## 2.2 *Miconia calvescens*

One well known 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. *Miconia* is not thought to be present on the island of Molokai. The length of time from the initial invasion and the considerable efforts that have been expended in controlling and surveilling for the tree's expansion over the last two decades mean that there is sufficient data to generate estimates of growth and control costs. Extracting this data from the resource managers and processing it into a useable form is a challenge we discuss at greater length in section 3.

When considering optimal management of *Miconia*, two spatial considerations matter. First, the likelihood and magnitude of the invasion (as measured by population growth over time) will vary spatially according to the current population and dynamics of growth. Second, the natural capital assets may be unevenly distributed across space.

We use Geographical Information Systems (GIS) to map the current and future populations of *miconia* on the island of Oahu, Hawaii, and the potential damages to water quantity, water

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<sup>4</sup> The frog is transported in nursery stock and the risk of its presence lowers willingness to pay and the costs of selling un-infested plants, because they do not want to admit the presence of the frog and incur these losses.

quality, endangered bird habitat, and native habitat housing endangered plants, snails, and insects. We develop a control cost function that includes locating and treating miconia plants. Using optimal control theory, we find the spatially dependent optimal population levels of miconia and the paths to these populations over time.

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. In an advance over the current literature, we allow costs and damages to vary spatially as well as temporally. Thus the objective function is:

$$\text{Max}_{x_{it}} \int_0^{\infty} -e^{-rt} \sum_i \left( \int_0^{x_{it}} c(n_{it}) d\gamma + D(n_{it}) \right) dt \quad (2)$$

subject to:

$$\dot{n}_i = g(n_i, n) - x_i \quad (3)$$

$$0 \leq x_{it} \leq n_{it} \quad (4)$$

$$n_0 = n(0), \quad (5)$$

where  $i$  denotes the spatial location (grid cell),  $t$  represents the time period,  $n_{it}$  and  $\dot{n}_{it}$  are the population of the invasive species in a given location and its associated time derivative,  $n_t$  is the total population at  $t$ ,  $g(n_{it}, n_t)$  the growth function of the invasive,  $x_{it}$  represents the number of removals,  $c(n_{it})$  the marginal cost function for removals, which varies with population level, and  $D(n_{it})$  the damages incurred at population  $n_{it}$ . In the following, we drop the time subscripts for ease of notation.

Defining the current value Hamiltonian for each location as:

$$H_i = - \int_0^{x_i} c(n_i) d\gamma - D(n_i) + \lambda_i [g(n, n_i) - x_i]. \quad (6)$$

Applying the Maximum principle and rearranging the subsequent first order conditions, we find

$$D'(n_i) = rc(n_i) - c(n_i)g'(n, n_i) - c'(n_i)g(n, n_i) \quad (7)$$

In this way, we see that at an optimal population, the marginal damages should be equated with the costs of maintaining that population for the location. Were marginal damages to be higher

(lower), additional (fewer) trees could be removed, reducing the overall losses. Areas with higher marginal damages, then, will have more trees removed.

We divide Oahu into 16 ha plots, or cells, to analyze the optimal management of miconia for the island over space and time. Each cell contains information on habitat quality and the current presence of the invading plant. We assume that the current invasion has already been underway for 37 years, and was initiated by purposeful individual plantings.

### 2.2.1 Miconia Growth

Invasive species managers on the heavily invaded island of Hawaii estimate that the densest areas contain approximately 100 trees per acre. Our spatial cells are 16 hectares each. Carrying capacity per cell is thus 3,952 trees.

In theory, we can calculate the spread of the current population using an estimated diffusion rate of  $0.208 \text{ km}^2$  and the distance from the current population, based on Fisher and Skellam (Shigesada and Kawasaki 1997), to determine the expected Miconia population in a given cell at a given time period. We assume the population changes as a function of both diffusion and internal growth:

$$\dot{n}_i = D \left( \frac{\partial^2 n_i}{\partial x_i^2} + \frac{\partial^2 n_i}{\partial y_i^2} \right) + (b - \mu n_i) n_i \quad (8)$$

Where  $n(x,y,t)$  is population at time  $t$  in spatial coordinate  $(x,y)$  as measured from the original specimen's location,  $D$  is the diffusion rate,  $b$  is the intrinsic growth rate,  $\mu > 0$  captures intraspecific competition,  $x$  and  $y$  are spatial coordinates, and the radial distance,  $r$ , is determined by  $r^2 = x^2 + y^2$ . The first term captures the rate of spread, the second captures population growth within the given coordinates. We estimate that the maximum carrying capacity in any habitat compatible cell ( $K$ ) is 3,952 trees, while it is zero in any non-compatible cells.

In practice, there is no explicit solution to the non-linear problem of equation (8) unless growth is exponential and  $\mu = 0$ . The assumption of unlimited exponential growth on a small island is not tenable. In order to create a tractable model that incorporates both spread and internal growth, we use the explicitly solvable Skellam model for spread with exponential growth until the population of the cell reaches the point where it diverges significantly from a logistic growth function, which occurs at approximately 20 trees. From that point, we use a logistic growth function to determine population in an area. Thus the population in any given cell will be driven first by arrivals through the Skellam expansion model, allowing internal growth, and then only by internal growth from the logistic function. We do not simply use the logistic function because it does not allow for radial spread to and from other cells.

Assuming an initial distribution where  $n_0$  individuals invade the origin at  $t=0$ , we have untreated populations

$$n(r,t) = \frac{n_0}{4\pi Dt} \exp\left(bt - \frac{r^2}{4Dt}\right) \quad (9)$$

until  $n(r,t) \geq 20$ . After this point,

$$n(r,t) = n_r \left( \frac{Ke^{bt}}{K + n_r(e^{bt} - 1)} \right) \quad (10)$$

where  $n(r)$  is the population of the cell (here, 20) when the growth function changes.

### 2.2.2 Miconia Damages

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. Details of the damage estimates are available in Burnett et al. 2007. In short, marginal damages for any given location will be calculated according to:

$$d_{it} = d_{bird\ habitat\ or\ range} + d_{water} + d_{native\ habitat} \quad (11)$$

Because not all locations will have all of these characteristics and because water damages will vary by aquifer, marginal damages will vary spatially. We find that in our analysis marginal damages range from \$0.22 per tree to \$19.06 per tree. Marginal damages from bird habitat losses range from \$0.00 to \$6.34 per tree; damages from watershed losses range from \$0.22 to \$0.70 per tree; damages from native habitat losses range from \$0.00 to \$12.02 per tree.

### 2.2.3 Control Costs

The marginal cost of searching and treating  $x$  trees is:

$$c(n_i) = \left( \frac{\$39,520}{n_i^{1.6258}} + 13.39 \right) \quad (12)$$

There are two separate activities that must occur – the trees must first be found, then treated, so that the cost function consists 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.

We determine the two components for Oahu 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. The numerator of the search component for each spatial cell on Oahu is \$1,000 per potential acres, or \$39,520 per 16 ha cell.

The ability to search an island's habitat will also 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 exponent on population in the denominator of the search component. Higher values imply greater ease of access, which translate into lower search costs. Due to the number of well maintained roads and trails throughout Oahu's forests, Oahu has the highest search coefficient of all islands, at 1.6258. Additional details on the specification of the cost function are in Burnett et al. 2007.

#### **2.2.4 Miconia Results**

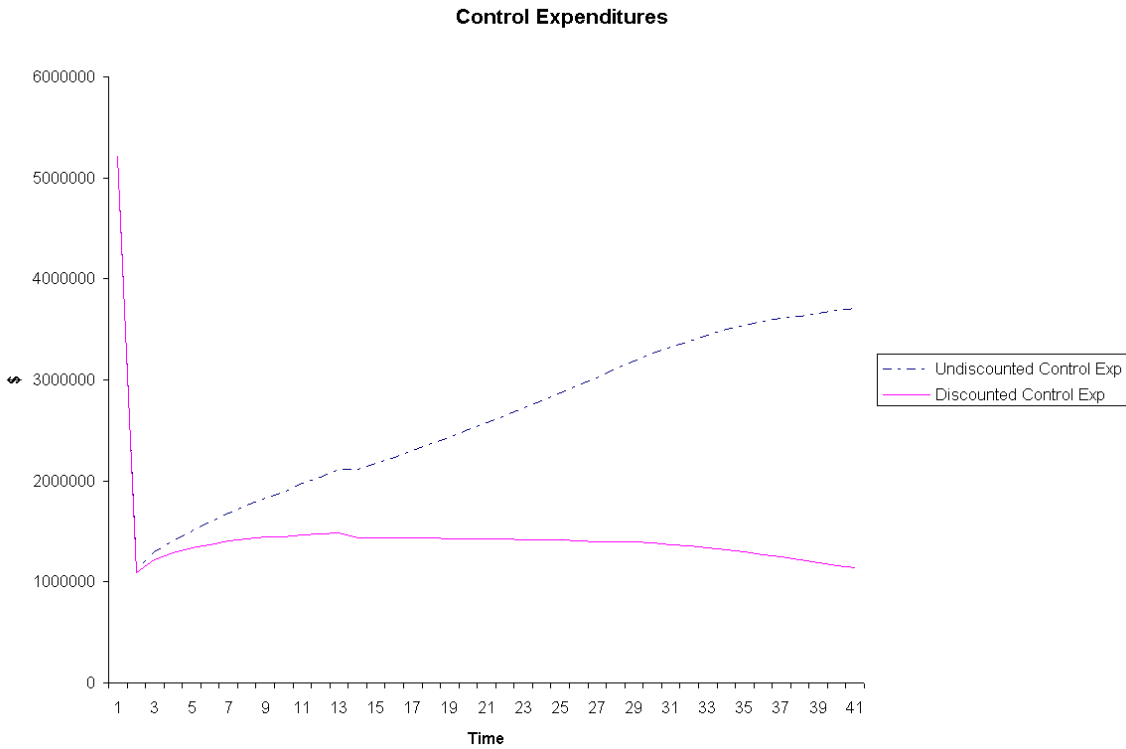
If left untreated, the damages from *miconia* will grow at an increasing rate into the foreseeable future. Unchecked damages over the next 40 years have a present value of approximately \$627 million dollars using a 3% discount rate.<sup>5</sup> This is the cost of doing nothing.

Using the parameterization described above, we solve for the optimal populations in each spatial location over time. We find that 9616 ha need immediate treatment at an expected cost of \$5.21 million dollars. This should be followed by spending that keeps the population in each location cell somewhere between 43 and 705 trees per 16 ha plot. Over 40 years, this cost will increase from \$1.12 million per year to \$3.71 million per year. The total present value of control costs from now until 40 years into the future should be \$54.5 million, using a 3% discount rate.

Figure 1: *Miconia* control costs over time

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<sup>5</sup> Under our parameterization of the spread, it will take approximately 80 more years for *miconia* to blanket its potential habitat on Oahu in the way that it now covers Tahiti. In part because planning horizons are short and in part because new treatment technologies are likely to evolve in the long run that will change control costs, we focus on the more immediate future and investigate the benefits of management over a forty year time horizon. In particular, remote sensing technology already can identify large stands of *Miconia*, and improvement in this technology may allow for quick identification of smaller *Miconia* populations. Additionally, since the loss of an endangered species is irreversible and the demand for groundwater is likely to change over time as well, damages may not be constant over the long run either.



As shown in Figure 1, the initial large immediate outlay of \$5.21 million should be followed by continuous control expenditures. Note that while these expenditures are increasing in current dollars, after year 12 they are decreasing in present value. We therefore emphasize that long run planning is essential to optimal management; it will become increasingly difficult to find new funds for management, so that setting aside funds for future management so that they can keep pace with the discount rate will be helpful to achieving optimal management goals.

Figure 2: Optimal control vs. no control over time

### Optimal vs. No Control

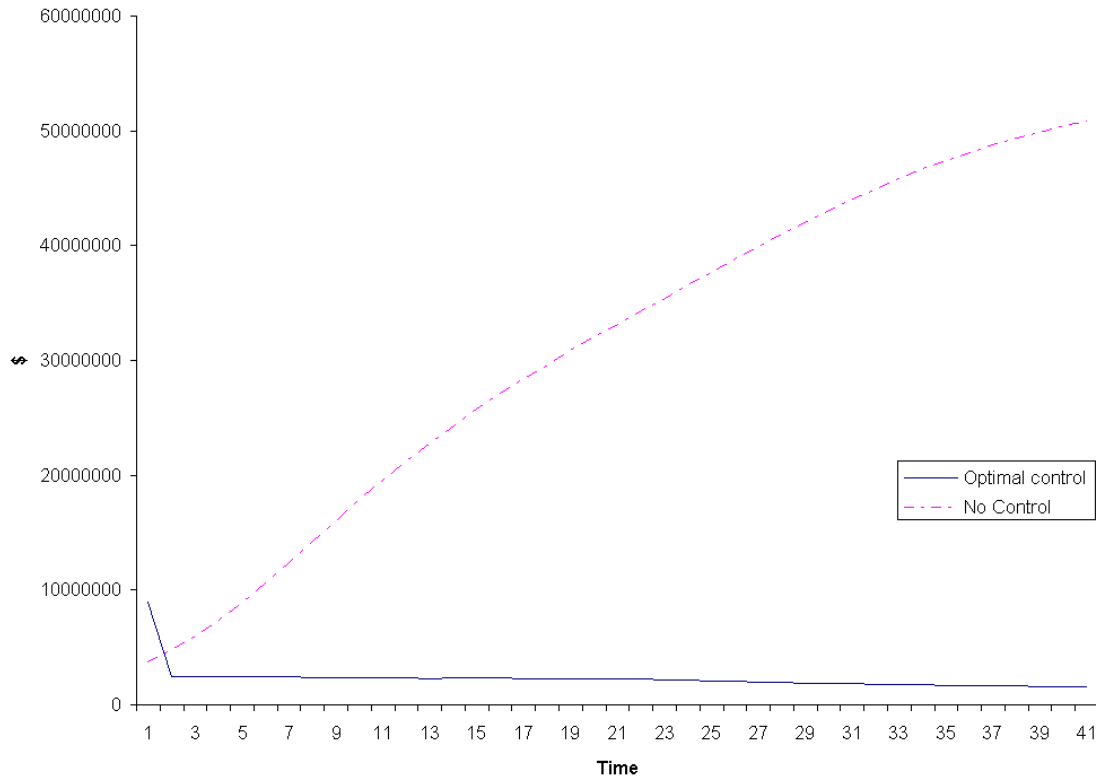


Figure 2 shows the comparison outcomes of no control, as measured by damages, to those of optimal control, as measured by damages from untreated trees plus the control costs for treated trees. We find that the returns to control grow in present value over time. In the first year of management, current expenditures and damages (\$9.0 m) are more than current untreated damages (\$3.8 m) by \$5.2 million. By the second year, however, optimal management costs \$2.2 million less than untreated outcomes and the benefit: cost ratio increases to just over 10:1 by year 40, with annual present value net benefits between \$14 m - \$17 m beginning in year 12. Net benefits over the forty year period from optimal control are \$534 million.

### 2.3 Brown Treesnake

In this section, we address EDRR as an explicitly spatial policy instrument using the case study of the Brown treesnake on the island of Oahu, Hawaii. The brown treesnake is another well known potential invader of Hawaii and much effort has been expended to study the potential effects of an invasion to Hawaii. (Savidge 1987, Fritts et al. 1987, 1990, 1994, Burnett et al, 2006, Burnett, 2007, Burnett et al, 2007). There have been eight brown treesnakes captured at the ports on the island of Oahu and hundreds of other sightings reported throughout the island. EDRR technology has been developed in the form of specially trained teams based throughout the Pacific who are immediately deployed following a credible sighting of a Brown treesnake on Oahu or on other at-risk islands. Two such deployments have occurred in Hawaii in the last two years, one on the island of Maui and the other on Oahu, although neither effort produced a snake.

Using Geographical Information Systems (GIS) software, we analyze spatially-explicit EDRR policies given the reality that prevention of the snake's entry may already have failed or will eventually fail at least one of the most likely entry points, regardless of budget (Burnett et al. 2006, Olson and Roy 2005). EDRR policies comprise of search and destroy activities that occur beyond incoming crafts at points of entry (prevention) to target removal of uncertain but likely specimens throughout the potential habitat range that have evaded detection. Intertemporal and spatial differences in policies are compared given varying assumptions about planning and management horizons and the arrival of the snake.

As in the case of miconia, we divide Oahu into a grid, but we use a finer subdivision for the case of the snake and each grid cell measures only 4 ha each. The choice of grid cell has potentially large effects that we discuss in section 3. Each cell is assigned initial properties that include currently existent data on likelihood of snake presence (distance from points of entry, proximity to roads<sup>6</sup>), resource assets at risk (bird habitat, presence of power transmission lines, human population density) and accessibility of treatment (proximity of roads and trails, slope, and land ownership).

From these initial conditions, we estimate expected snake populations for each cell across a thirty year period based on the likelihood of the presence of snakes, the expected marginal damages (per snake) as a function of the resources at risk and the marginal costs (per 4 ha area) as a function of accessibility and terrain.

Using this information, we build a spatial-intertemporal model that minimizes the expected net damages from the brown treesnake on Oahu. Since treatment decisions are EDRR search decisions, the unit of decision is the spatial cell rather than the snake population directly. Net expected damages are calculated for each cell by assuming that treatment clears an area of snakes for that time period, so that population-based damages are avoided.

The theoretical model is formalized as:

$$\min_{x_{it}} \sum_i \sum_{t=0}^T \beta^t \left( d_i \left( n_{it} \left( x_{it}, \sum_i x_{it} \right) \right) + C_i(x_{it}) \right) \quad (13)$$

$$n_{it} = n(r, t, x) = \frac{\sum_i x_{i,t-1}}{I} g(n_{t-1}, r) * (1 - x_{it}) \quad (14)$$

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<sup>6</sup> We have more specific information about habitat than distance from points of entry, but after extended discussions with several Brown treesnake scientists it has become clear that the main limiting factor in Hawaii will be the availability of prey, for which we do not have specific densities. Fortunately for our analysis though unfortunately for avoiding the spread of the snake, the one point of agreement between all of the scientists on this matter is that they believe there exists sufficient prey base for snake expansion in all habitats present on Oahu for a population explosion comparable to the one on Guam after its arrival. Thus, since there exists no scientific evidence or theoretical model to credibly believe that forest habitat is more amenable than urban, for example, we accept that there will be abundant prey in every habitat and that differences for the snake will be minimal.

$$\sum_i C_{it} \leq A_t \tag{15}$$

Where  $d_i$  is the expected damage for cell  $i$ ,  $n_{it}$  is the population of the cell at time  $t$  as a function of own-cell ( $x_{it}$ ) and other-cell ( $x_{jt}$ ) EDRR treatments,  $C_i$  is the cost of EDRR for cell  $i$ ,  $I$  is the total number of cells,  $g$  is the biological growth function which depends spatially on the distance from the expected start of the invasive population,  $\beta$  represents the discount factor, and  $A_t$  represents a temporally constrained appropriations budget for EDRR.

Spending  $C_i$  brings the population for period  $t$  to zero for an area, but invasion from other parts of the island, or anew from off-island, re-initiates growth in the next period. The larger the

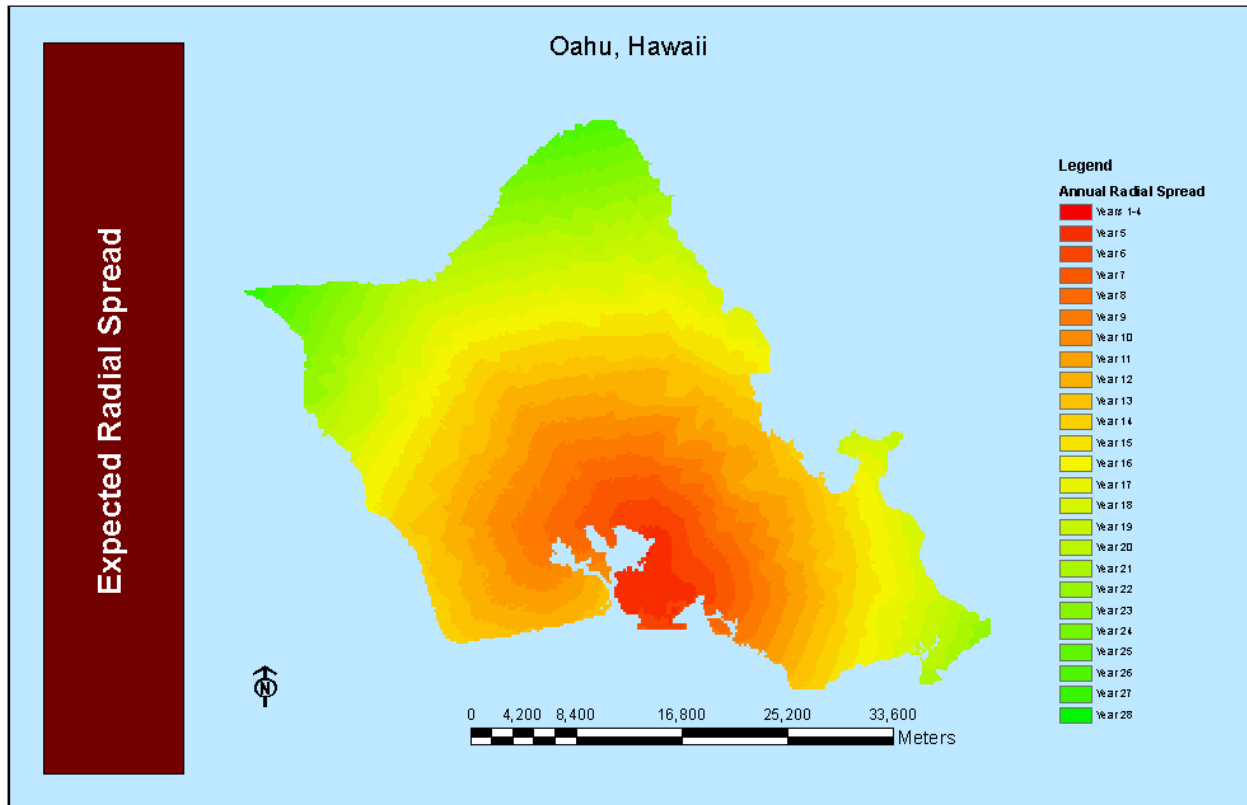
proportion of treated cells  $\left( \frac{\sum_{i=1}^I x_{i,t-1}}{I} \right)$ , the lower the rate of re-growth.

### 2.3.1 Snake Growth

The expansion path without intervention is based on the estimated expansion rate of 1.6 km/yr (Wiles et al. 2003) from the expected origins of the airport runways and Schofield facilities and the terrain through which the snakes must pass (Fig. 1d). Expected origins were weighted by capture experience on Oahu to date, with HNL being the most likely port of entry. Roads and trails are expected to provide the most rapid expansion paths (Timmins 2006); distance from roads and trails slows the radial spread.

Figure 3 illustrates the expected spread, with each change in color shade indicating another year's expansion of territory, from red to green. While there is a positive probability that the snake may appear in any cell at any time, the range is determined by the expected presence of at least one full snake.

Figure 3: Expected annual snake expansion



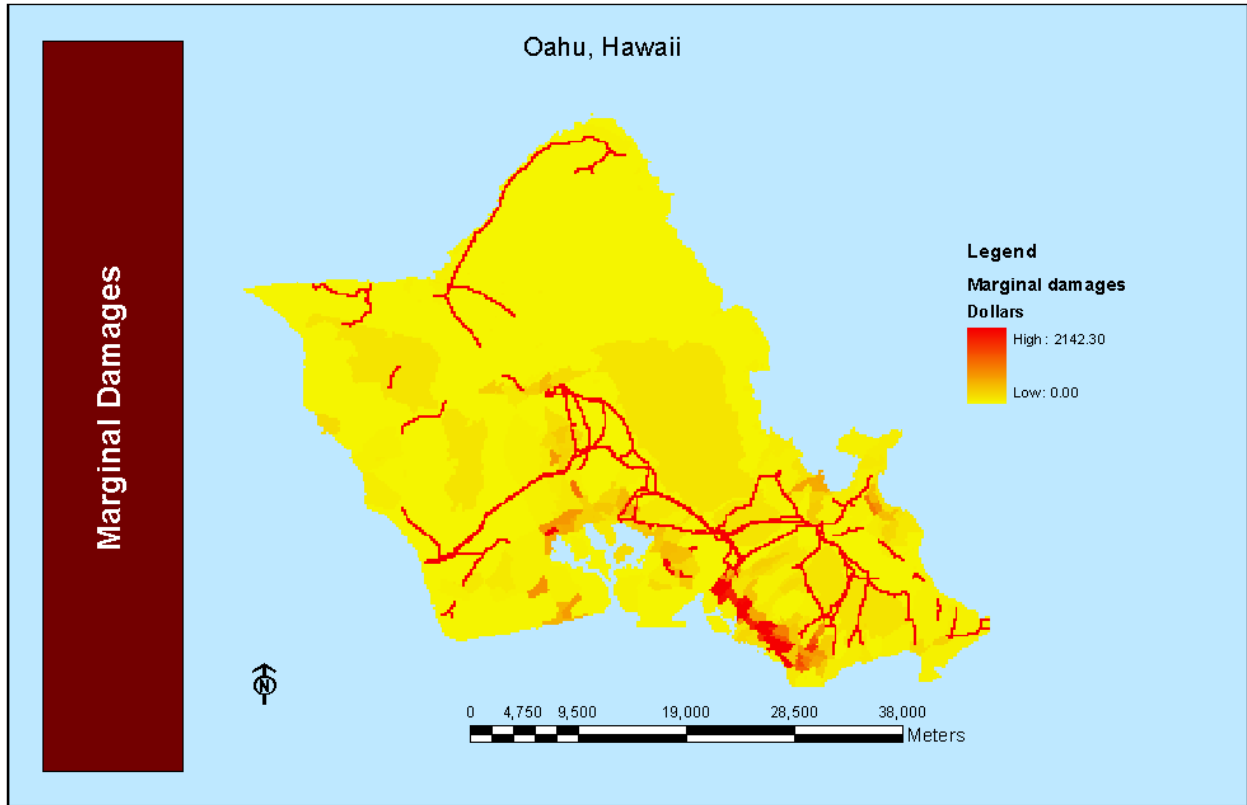
Note: Expected entry at Honolulu Airport (HNL) or the adjacent Hickam Air Force Base Airport (3/4 weight), Barber's Point Air Station (1/8 weight) or Schofield Barracks (1/8 weight)

For population, we use the same functional form expressed for miconia in equations (9) and (10). In the case of the snake, however, the transition between the exponential growth and spread and internal logistic growth occurs at 40 snakes in a cell with a maximum snake carrying capacity in any cell ( $K$ ) of 200 snakes. Further details are available in Kaiser and Burnett 2007.

### 2.3.2 Damages

Figure 4 illustrates the range of damages across Oahu. Damages are calculated using a per snake linear coefficient that varies from a minimum of \$0 and a maximum of \$2143 (Fig. 4). Damages consist of three potential impacts: power outages, medical costs and human-snake interactions, and biodiversity losses. Details are available in Kaiser and Burnett 2007.

Figure 4: Total Damages



### 2.3.3 Snake Control Costs

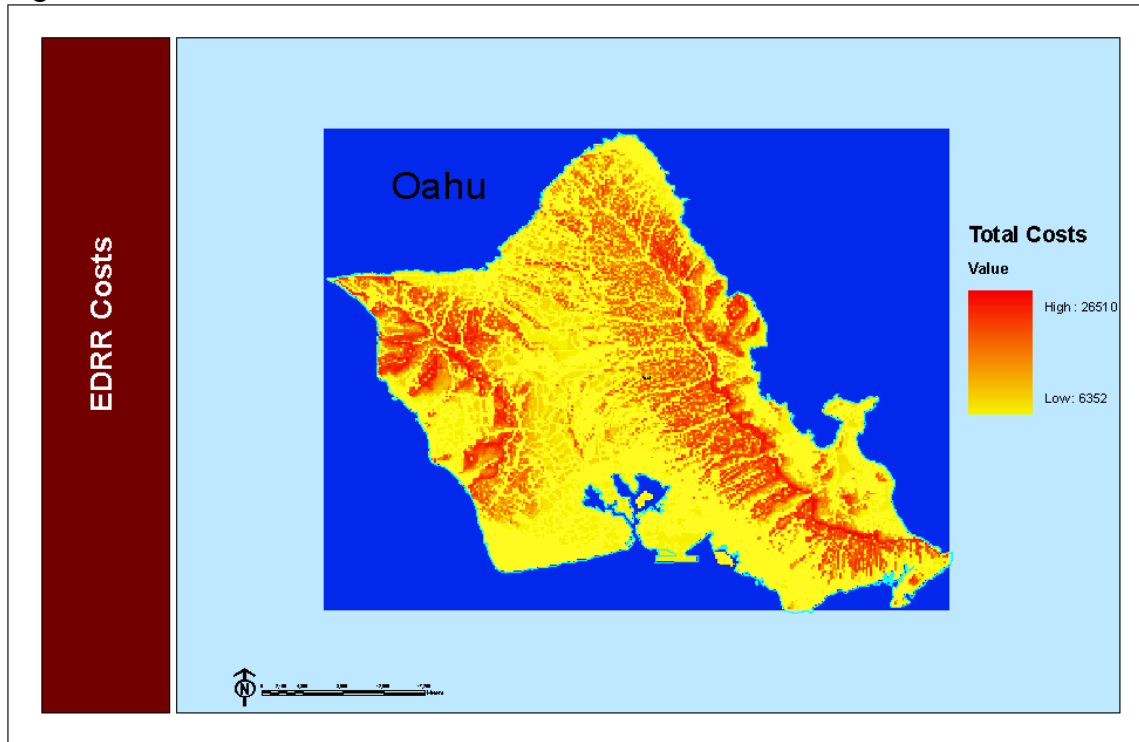
As discussed, a particular distinction between EDRR and other discussions of invasive species control is that with EDRR it is not known with certainty that there exists a population, while with control one generally assumes one can “harvest” a known population of the invasive species. Costs are therefore allocated spatially rather than as a function of population.

We describe EDRR treatment as consisting of preventative search, trapping and hand-removal (the only way to currently remove snakes too small to be trapped). Costs vary with terrain. Records on the costs of clearing an enclosed 5 ha plot on Guam (Rodda, personal communication) provide a least cost estimate of removing snakes from an area. Costs are scaled up from this base cost of \$6,352 per 4 ha cell to account for slope of the terrain and distance from a road. The steeper the grade, the more energy required to search the area. Since the cost of searching is a labor cost, we use a model of the American College of Sports Medicine to translate grade into energy expenditure, and then increase costs proportionally to the increase in effort. The energy expenditure rate (EER) is estimated to be:

$$EER = 0.1v + 1.8v \cdot a + 3.5 \quad (16)$$

Where  $v$  is the speed of walking and  $a$  is the percent grade (Sabatini et al. 2004). We assume a constant slow rate of walking at 0.5 km/hour to accommodate searching (Rodda, personal communication, Lardner, personal communication). Average slope for each cell is calculated from hillshade projections of Oahu in ArcGIS 9.1. Figure 3 illustrates total costs.

Figure 3: Snake Control Costs



For each cell, we first calculate the energy expenditure rate, EER. We then generate an energy expenditure ratio where we divide the cell's EER by the EER when the slope is zero, which provides an estimate of how much more difficult clearing the cell is than clearing the 5 ha test plot (which was on level ground) cost. This ratio is therefore multiplied by the base cost of \$6352.<sup>7</sup>

Costs also increase with the distance of the cell needing treatment from accessible roads. We use analogous methodology to determine distance costs from roads by using ArcView Spatial Analyst to calculate the least cost distance path. First, based on the EER from the nearest road to the cell, we determine the least cost EER path from the nearest road to the cell. Then we create a ratio of this distance cost to the linear distance from the road. We then multiply this ratio by the labor cost of reaching the cell, estimated at \$60 per unit. The maximum access cost is approximately \$3420, while the average is approximately \$540. The total cell cost is then the sum of the in-cell treatment cost and the distance (access) cost.<sup>8</sup>

<sup>7</sup> The maximum cost for thoroughly searching a cell for EDRR purposes using this formula is approximately \$27,500, while the average cost is \$11,700.

<sup>8</sup> Note this does not allow for treatment in multiple adjacent cells at discounted distance cost. However, since this method also assumes only one treatment time necessary (rather than repeated nights of search) the net effect is unclear. We leave this for later modeling. We also delay modeling of any external cost to accessing private land. One possibility is to assume that gaining access to private land and/or convincing private landowners to engage in search activities themselves is one of the main purposes of awareness campaigns, and that expenditures targeting awareness of a species can be considered additional costs of treating private land. In the case of the Brown treesnake in Oahu, this amounts to only about \$3 per cell of private land, thus we have ignored this cost for now.

### 2.3.4 Snake Results

Currently, no known snake populations exist on Oahu, but there is general agreement amongst the scientific community that there may be between 0 and 100. We begin our analysis with  $n_0=1$ .<sup>9</sup> Thus, our initial application is for search only. Current search on Oahu occurs only after a suspected sighting, while all other funds are expended on Guam and are targeted at preventing snake arrival at defined points of entry. Previous research (Burnett et al. 2006) indicates that this may actually focus too much on the points of entry if snakes have already evaded detection there. Our results concur.

We calculate the spatial-temporal treatment schedule that minimizes the overall net damages and costs in present value terms for a thirty year period.

We find a present value of expected damages of \$371 million accumulated over 30 years from an initial invasion of a single snake at one of three possible entry locations with no EDRR action. We start the optimization with treatments indicated for all cells when and where the current year damages exceed the current year costs, treatment of which will certainly reduce the social welfare losses. We then test whether treating these cells or neighboring cells before the damages exceed the current year costs reduces the present value of net damages by reducing the future populations and their damages. We find that under our parameters for the discount rate, growth, costs and damages, it does not.

We find that treatment reduces social welfare losses to \$101 million dollars. Over the thirty year period, we find the need to treat just over 3000 cells, or 8% of the island. The treatment plan also delays any search until the 12<sup>th</sup> year after an invasion. This result is driven by the interplay between the discount rate and the growth function; the chances of finding snakes when they are spreading out across the potential habitat and are at low densities, and causing low damages, mean that waiting discounts the costs more than the growth in the damages.

The hazard rate (the probability of arrival during the intervals between arrivals) should affect these results in two ways. We have used a thirty year time frame in part because this is the time, given the growth parameters, that it should take for the entire island to have snake populations. In this time frame, damages have just grown to exceed the present value of costs for an entire-island sweep (which occurs in year 28, see section 3), which suggests that is the appropriate time to switch from an EDRR policy to a control policy, where removal of the snake population is undertaken directly.

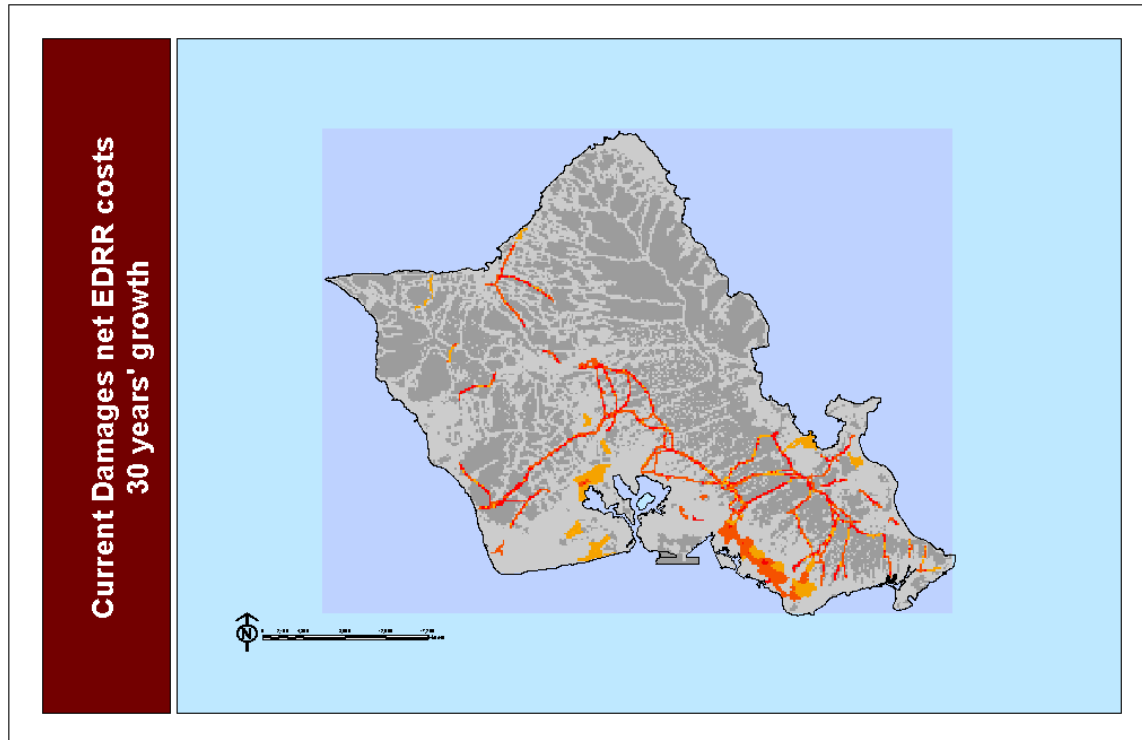
Figure 5 shows a snapshot of the net current damages (i.e. only the damages in that year) that would occur if all cells were treated in the last year of invasion. In a significant majority of cells, the current damages are below the current EDRR costs (shown in grayscale), and intervention cannot be justified on the basis of current damages alone. The area for which damages do exceed costs (shown increasingly from orange to red), so that EDRR treatment is cost-effective

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<sup>9</sup> Mitochondrial DNA evidence suggests that the entire population of snakes on Guam may have originated from a single female.

in this single period, are obviously also the areas where optimal EDRR should be targeted. One can see that these cells integrate damages, costs, and the biological spread in such a way that EDRR treatment, when there is only funding for sporadic and incomplete treatment, should focus on not just the areas closest to the most likely point of entry (HNL airport) but also along roadways with major power lines adjacent and in locations where human-snake interactions would be high (the orange areas along the southeastern coast in Figure 5 are the densely populated Honolulu and Waikiki areas).<sup>10</sup>

Figure 5: Current net damages across first 30 years of invasion



### 3. Discussion

#### 3.1 Parameter choices

##### 3.1.1 Grid cell size

Grid cell size matters in determining optimal policy for several reasons. Foremost, because the cost of search exhibits economies of scale that are spatially dependent, the finer the gradation is, the flatter the marginal costs as a function of population will be. Flatter marginal costs tend to increase the optimal population level. In the case of miconia, we see that including finer

<sup>10</sup> In spite of the level of urbanization, scientists assure us there is plenty of prey available, and as the snake is nocturnal and reclusive snake, it is likely to do well in an urban environment with many places to hide.

gradation in the analysis significantly increases the population of trees. When we analyze the entire Hawaiian islands as one continuous habitat, the optimal population of trees is 31,295 (Burnett et al. 2006). When we subdivide the analysis by island, we see that the optimal population of trees for Oahu is 5495 and the optimal population for the state is 63,504 (Kaiser et al., 2007). Finally, when we subdivide Oahu into 16 ha plots, the optimal population for the island will eventually reach almost 1 million trees and seedlings (52 years from the present), though the population in each 16 ha plot will range from only 40 to 705 trees and seedlings. (Burnett et al., 2007).

Additional considerations include the availability of reliable GIS data at finer resolutions and the computational limitations of perhaps millions of choices across cells, even if the choices are binary. In the case of miconia, habitat cells could not be reliably determined at any smaller resolution. In addition, there was little benefit from smaller units of analysis because helicopter searches can cover several acres in one pass. In the case of the snake, search is time-consuming and only small areas can be searched in any one night. Since the island of Oahu is considered all potential snake habitat, the resolution did not affect this parameter. Finally, since treatment was a binary decision to search or not search, having over 1 million cells, though cumbersome, was not impossible with the application of constraints from theory. In the miconia case, while theory guides the population levels in the cells, the populations are continuous and the reduction in cell numbers dramatically increases the ability to solve the problem.

### **3.1.2 Growth**

Both the internal growth parameters, here 0.3 for miconia and 0.6 for snakes, and the diffusion rates, here 0.208 km<sup>2</sup> for miconia and 1.067 km<sup>2</sup> for Brown treesnakes, are important factors in determining optimal policy. Combined with marginal damages, faster growth will increase the need for immediate treatment and increase the probability that delaying efforts will result in having to choose accommodation of the invasion over eradication or control at a small population. Faster growth will also lower marginal costs of treatment more quickly so that delay again is less beneficial.

Not only is delay more costly, inadequate control efforts are more wasteful. If control is applied at levels where growth continues to expand within a cell, the benefit of that control effort is lost to future damages. The faster the growth rate, the greater the penalty will be.

### **3.1.3. Costs**

For most invasive species, detecting the species is a significant portion of costs, at least at low densities. The area for which search costs are defined, then, will affect the marginal costs as a function of population and the optimal population, as described above. When costs are determined spatially, however, this concern is alleviated. If the species is known to be present in an area, like miconia is, then it is inappropriate to apply costs spatially since it does not allow the optimal population to vary within the area. Optimal population may indeed vary as a function of damages, growth, and costs. If there is no known population, however, and if the optimal population were its presence detected was low or zero, as it is in the case of the brown treesnake

(Burnett et al. 2006, Burnett 2007), then costs can be applied spatially. EDRR is a valuable and distinct management tool that needs greater analytical attention.

### **3.1.4 Damages**

Though damages for ecological benefits are often very uncertain, in all of our cases we have at least one market good to which we can tie damages, providing lower bound estimates. Thus we need not fear that our damage estimates are too high and that accommodation of these invasive species is actually the optimal policy. We may need to be concerned with upper bounds, however. If damages are significantly underestimated, it may be that eradication is the optimal policy in spite of high search costs or the inability to prevent future entries. This inability to prevent future entries, however, requires prevention and possibly EDRR activities be considered in this optimal policy decision.

EDRR is again an appropriate policy for considering the range of damages that might matter. Under our parameterization, we find that spending on EDRR for the snake should occur when the present expected damages exceed the present costs. Thus if one fears that an endangered species is undervalued, for example, then cells in which the species is present may deserve more EDRR effort. The essential finding that management activities must simultaneously incorporate expected costs, damages, and growth does not change.

## **3.2 Temporal Application of Policy**

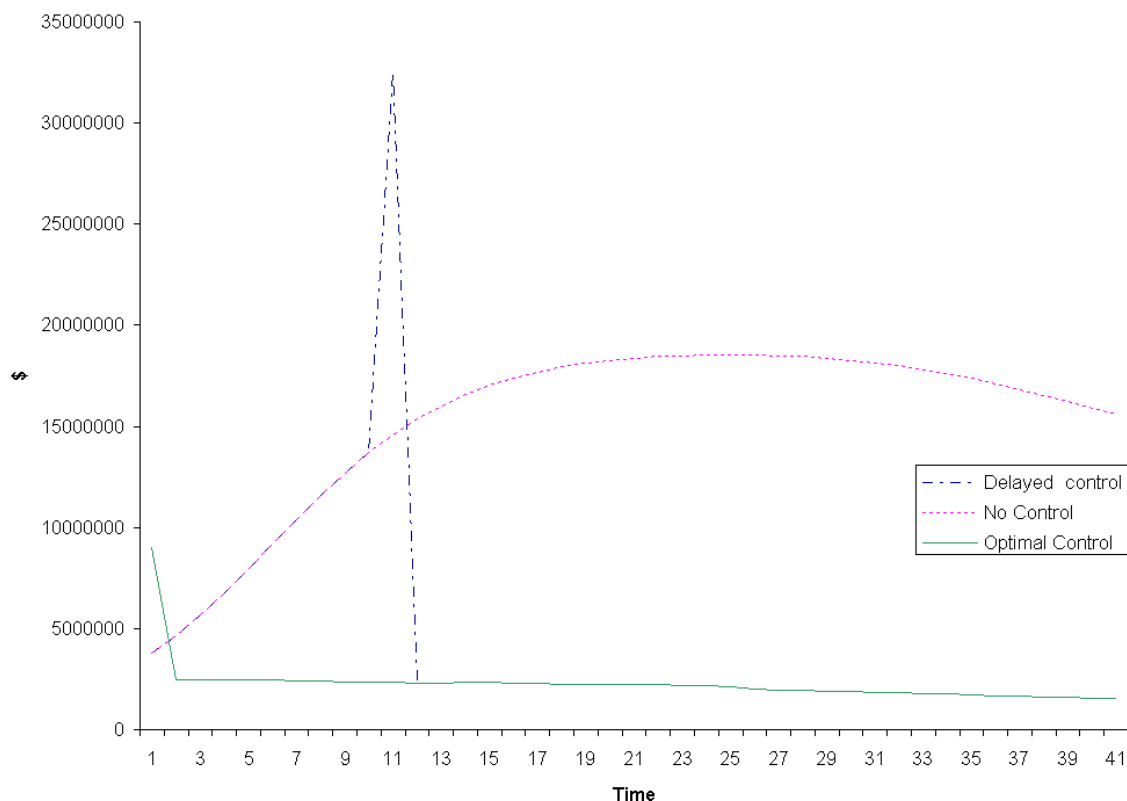
### **3.2.1. Delays in Policy Initiation**

The annual expenditures that maintain the optimal populations of miconia are, as figure 1 shows, around one to 1.5 million dollars. (Note that a steady state population is not reached in 40 years because the population of trees has not reached all habitat in that time.) Delaying the start of treatment will increase the need for current outlays when treatment does begin, as more trees will need to be removed. Figure 3 illustrates. In spite of the large returns that can be gained from delayed control, it is evident from figure three that there may well be a point at which it is too late, and accommodation should be favored over expensive removals and permanent control because the present value of uncontrolled net damages will be lower than that of controlled damages and costs.

The specific time at which this switch would occur, however, is a decreasing function of the discount rate and an increasing function the time horizon under consideration. An infinite time horizon is preferable to our current short term analysis, especially given the irreversible nature of many of the ecosystem benefits, and we do not seek to calculate this. In the case presented here, though net benefits of control fall from \$534 m to \$448 million, a loss of \$86 million over just ten years, it is still worthwhile to initiate delayed control.

Figure 3: Cost of Delayed Control

### Optimal vs. No Control



In the case of the snake, though delaying initial search until the 12<sup>th</sup> year after an invasion appears optimal, two caveats are offered that suggest additional benefits to earlier search. First, in an island-wide sweep, scientists may become confident that an early eradication is complete at a lower total cost than \$447 million as they gain evidence from the search experience. Second, our damage function is not currently applicable to extension beyond thirty years because of the expected irreversible loss of the elepaio bird species. The 11 bird species extirpated on Guam were lost in fewer than 40 years, and a similar time frame for Hawaii can be expected. Thus if eradication efforts are deferred, the irreversible loss of the species imposes a dramatic threshold damage penalty and reduces the expected benefits of further action, which will then only serve to reduce human-snake interactions and electrical supply damages.

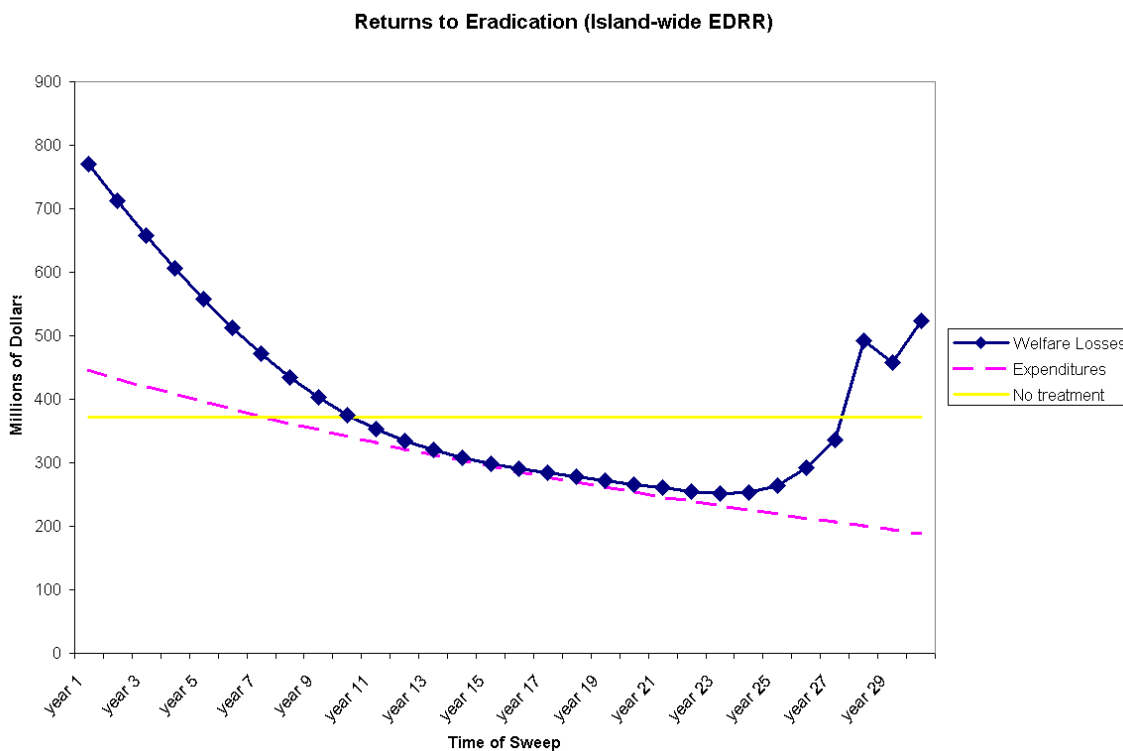
### 3.2.2 Application of inappropriate policy for the level of invasion

Eradication is almost always a stated goal for agencies charged with managing invasive species. The snake may be a case where eradication or at least low populations are optimal (Burnett et al. 2006, Burnett 2007). Under our specifications, a full island search would rid the island until the next arrival, therefore there may be some benefit, though not economically optimal, to periodic island-wide sweeps. We investigate the returns to island-wide sweeps at various stages to highlight these tradeoffs. The cost of a complete island search is estimated at just under \$447 million. In the worst case scenario, if an island-wide search is conducted, and then another snake

enters in the following year with no follow-up treatments, the total social welfare losses are \$771 million, far more than never conducting the search.

However, if a single island-wide search is conducted between years 11 and 27, the net benefits of the search are positive, even with re-infestation the next year. Social savings range from \$18 million to a peak of \$120 million before they begin to fall again and become negative after year 27. This is due to the fact that the damages grow exponentially with the expansion of the snake, so that while the present value of the costs is constantly falling, the damages from the spread of the snake outpace the discounting of the future damages. Waiting until year 30, for example, will have total social losses of \$523 million. Thus, the use of a lower discount rate might actually deter EDRR activities because the costs will appear higher for a longer period; using a 3% discount rate, the damages do not start to grow rather than decrease until year 16.<sup>11</sup> Figure 4 illustrates.

Figure 4: Returns to Eradication (Island-wide EDRR)



Extensive but random search, however, is likely to raise costs more than reduce damages, unless it is comprehensive (island-wide) and occurs between the 11<sup>th</sup> and 27<sup>th</sup> year of a successful invasion. When search is random but incomplete, the present value of social costs regularly lies between \$450 and \$750 million. Successful damage-minimizing EDRR activities target areas that have high expected net damages, either because they have a combination of high expected populations, high asset values, and low search costs. Small changes in treatment allocations that explicitly weigh expected damages, population growth, and treatment costs can dramatically

<sup>11</sup> At year 15, even with exponential growth, no cell has more than 28 snakes, just over 10% of carrying capacity. This begins to change rapidly in years 15 to 30.

improve random solutions. Thus, random or incomplete efforts may not be better than doing nothing, but strategic action can dramatically improve outcomes.

### **3.2.3 Changes in optimal policy when funds are uncertain**

In the case of a potential invader like the brown treesnake, we determined that the optimal policy for EDRR is not to search until populations are high enough that there is a chance to find them at a reasonable cost, here in the 12<sup>th</sup> year of an invasion.

A likely restriction for managers, however, is the inability to plan for EDRR funds over a long period of time. We investigate what the optimal policy should be if funding can only be secured in 5 year increments. In this case, we find that at the end of the first 5 years, if there is uncertainty regarding future funding, one is best off treating a small number of cells with high net expected damages, reducing the overall expected cost by about \$150 m to \$227m. Treating a slightly larger group of high expected damage cells after another five years reduces damages to \$142 m, while additional treatments at years 15 and 20 reduce the damages to \$126 m. Compared to the periodic island-wide sweeps, this targeted EDRR activity is preferable, in spite of the fact there may still be snakes present. Furthermore, it suggests that taking decisive and targeted EDRR action, even though it may not be the optimal action, is more likely to reduce overall damages than to increase expenditures, especially when those expenditures are large.

## **4. Conclusions**

Optimal management of invasive species will minimize total losses from invasion, including ecological damages, economic damages, and the costs of managing these invasions. The primary instruments for managing invasive species are prevention, early detection/rapid response, and control. Efficient management programs will vary across time and landscapes. In this paper we explore efficient spatial and intertemporal management for three invasive species in Hawaii, the coqui frog, miconia, and the brown treesnake.

We begin by considering economic damages from the coqui frog. We find that the presence of the frogs has a significant negative impact on property values. For properties within 500 meters of an official coqui complaint, property values decline 0.16%. While we do not explicitly model efficient management of the frog in this work, we produce an estimate of net marginal damages from the spatial spread of the frog as a function of the properties in an invaded location. In future work, this estimate will be used in conjunction with spread and capture cost estimates to generate optimal management policies for the frog.

For miconia, we find that optimal control entails treating immediately treating approximately 9,616 hectares on the island of Oahu, at an expected cost of \$5.21 million. This should be followed by spending that keeps the population in each location cell between 43 and 705 trees per 16 ha plot, depending on the spatial location of each plot, across an eventual total of about 53,000 ha.

In the case of the potential invader, the brown treesnake, we find that the optimal management program entails EDRR on less than 10% of the island of Oahu over a thirty year period. While the cost of inaction is approximately \$371 million, optimal treatment reduces social welfare losses to \$101 million dollars. This analysis confirms that search and removal should be focused not only on likely areas of entry, but around potentially high damage areas as well. We further find that after approximately 30 years, the benefits of EDRR should begin to be supplemented by direct control.

We conclude by investigating the sensitivity of policy decisions to key model components. We find that results are sensitive to grid cell size, as this affects the steepness of the marginal cost function and the resolution at which other parameters can be applied. Rate of growth will also influence the optimal program. Faster growth will increase the need for immediate treatment and will lower marginal costs of treatment more quickly. Specification of growth will also be related to the adequacy of management levels. Inadequate control efforts are found to be wasteful. If control is applied at levels where growth continues to expand within a cell, the benefit of that control effort is lost to future damages. The faster the growth rate, the greater these losses will be.

Deliberation consideration of space in the model improved our understanding and ability to model costs of control and damages from miconia and the brown treesnake. Temporal insights were advanced from previous work as well. For miconia, despite the large returns that can be gained from delayed control, we find a point at which it is too late, and accommodation should be favored over expensive removals and permanent control. For the brown treesnake, it appears that delaying initial search until the 12<sup>th</sup> year after an invasion is preferred to initiating search immediately.

Current policy regimes often tout eradication as the most favorable management option. Under our parameterization, we are not able to find any case in which full eradication and maintenance of a zero population is optimal. We also find that random or incomplete efforts may not be better than doing nothing, although strategic, efficient action can obviously improve outcomes.

Finally, because the dedication of future funding to invasive species efforts is often unknown or extremely limited, we investigate optimal brown treesnake policy under funding that can only be secured in 5 year increments. In this case, we find that treating cells with the highest expected damage first will reduce total losses by the most. This is an important result for policymakers in Hawaii and the Pacific, as limited brown treesnake funds are currently focused on searching around likely points of entry, rather than around high-valued assets at risk.

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