Spatial Economic Analysis of Early Detection and Rapid Response Strategies for an Invasive Species

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Abstract

Economic impacts from invasive species, conveyed as expected damages to assets from invasion and expected costs of successful prevention and/or removal, may vary significantly across spatially differentiated landscapes. We develop a spatial-dynamic model for optimal early detection and rapid response (EDRR) policies, commonly exploited in the management of potential invaders around the world, and apply it to the case of the Brown treesnake (*Boiga irregularis*) in Oahu, Hawaii. EDRR consists of search activities beyond the ports of entry, where search (and potentially removal) efforts are targeted toward areas where credible evidence suggests the presence of an invader. EDRR costs are a spatially dependent variable related to the ease or difficulty of searching an area, while damages are assumed to be a population dependent variable. A myopic strategy in which search only occurs when and where current expected net returns are positive is attractive to managers, and, we find, significantly lowers present value losses (by $270m over 30 years). We find further that in the tradeoff between search costs and damages avoided, early and aggressive measures that search some high priority areas beyond points of entry even when current costs of search exceed current damages can save the island more ($295m over 30 years). Extensive or non-targeted search is not advised however.

1. Introduction

Management of invasive species presents spatial and temporal analytical challenges that require integrated biological and economic modeling. An invasive species may be intercepted or treated across the invasion timeline: before entry to a new location (prevention), shortly after introduction but before establishment (early detection and rapid response, EDRR), by restricting it to a location smaller than the potential host range after establishment (containment), by restricting a population through harvest (control), or by allowing it to become part of the ecosystem (adaptation). Clearly, there are certain complementary and substitutable activities across this spectrum, but hitherto, applied research integrating optimal management of invasive species generally has not accounted for EDRR or spatial variation. This has kept the focus clearly on the important intertemporal tradeoffs in invasive species management (Burnett et al, 2008; Burnett et al, 2006; Finnoff et al, 2007, Olson and Roy, 2005). However the addition of a spatial dimension has been shown to change optimal policy for control efforts when marginal costs of control include a cost of search, increasing the steady state level of a controlled invader as the spatial unit of analysis decreases (Burnett and Kaiser, 2007), and therefore must be explicitly included in any EDRR analysis.

In this paper, we exploit the significant biological and economic research to date on the potential ecological and economic damages and costs of a particularly well studied species of significant concern, the Brown treesnake (*Boiga irregularis*)[1]. The Brown

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[1] The Brown treesnake is one of fewer than ten species causing damage outside its home range that has specific US Federal Legislation passed to fund efforts to prevent its spread (Public Law 108–384, the

treesnake's imminent arrival in Hawaii (Rodda et al. 1992; Rodda et al. 1999; Burnett et al. 2006; Shwiff et al. 2010) provides an excellent case study to develop a spatially explicit, comprehensive dynamic EDRR management strategy to minimize the expected impacts of a potential invader. We contribute to the current literature in several respects, with a chief goal being to evaluate real-world invasive species management decisions in a bio-economic framework. First, we consider EDRR, a real-world policy instrument commonly exploited in the management of potential invaders around the world, although not explicitly analyzed as a policy option in the literature to date. Second, we attempt to mimic real decisions facing managers with long run dynamic consequences by examining decisions made across brief time horizons and assessing the impact of this constraint. Finally, our work expands, using real world data, the findings of Finnoff et al (2007) that managers should prefer prevention to control even when their risk preferences lead them to wait for an invasion before treating. To do this, we ask whether a myopic policy under which only locations where the expectation is that current benefits outweigh current costs are searched is preferable to a strategy where more aggressive EDRR occurs so that certain locations are searched even when current net benefits may be negative.

EDRR, defined here as intervention that occurs shortly after introduction but before there is a known population in a new location, consists of search activities beyond the ports of entry, where search (and potentially removal) efforts are targeted toward areas where credible evidence suggests the presence of an invader. EDRR should not be simply considered either ex-post prevention[2] or low-population control, (though both are components of EDRR) and deserves much greater analytical attention. This is due to the need to make decisions based on the possibility that a specimen is present across many possible locations.

Our paper is a first step in formally modeling EDRR as an invasive species management tool. In order to concentrate on the combination of spatial and temporal components and the comparison of real-world myopic policy to optimal policy, we set aside virtually all uncertainty, using a deterministic representation of expected outcomes of a new invasion based on estimated population growth, costs of search treatments, and damages. As such, we can concentrate on the benefit EDRR adds to the management strategies of prevention and/or control, given a new invasion. Inspection, barriers around ports of entry, or any other action taken to avoid invasion is typically thought of as prevention. Prevention differs from EDRR particularly as the opportunities for reaping high returns are foregone once a species has successfully passed any bottleneck entry conditions where intervention could occur. After a new species has arrived and is established, attempts to reduce the new population can be categorized as control. Control differs from EDRR particularly as control can be considered harvest of an unwanted species and planning can compare population-dependent harvest costs with population-dependent damages.

Brown Tree Snake Control and Eradication Act of 2004). The focused nature of its current 70-year invasion in Guam and the Commonwealth of the Northern Mariana Islands (CNMI), and more than ten years of interdisciplinary conversations between economists, ecologists, herpetologists and land managers provides a wealth of scientific data about potential damages and control methods.
[2] We use the term ex-post prevention to describe interception of newly arrived specimens outside the purview of standard interceptions at points of entry.

When discussing EDRR, the time frame is generally considered the time between when a species is likely to have first arrived based on knowledge of entry vectors and the time elapsed from that point until the new species' presence should be noticeable without directed search. For the Brown treesnake, we estimate this to be about 30 years, which is a relatively long planning horizon for land managers to consider. Currently EDRR is a response driven policy tool that is myopic, in particular in the sense that it is not applied based on the very real likelihood that prevention has at some point failed (Burnett et al, 2008), but rather when a credible sighting of the species occurs. It has also been rather haphazardly funded. To analyze the implications of this on decision-making, we examine the impact of restricted planning horizons, under which managers cannot expect funds to be available over the entire 30-year period. Previous work focusing on long-run intertemporal tradeoffs has rightly been criticized for its lack of a spatial dimension, thus by including shorter time horizons we seek to identify how the time frame influences actual EDRR decisions (and avoid the reverse pitfall).

2. Background and literature

Some previous analyses have focused on the dynamic progress of an invasion from inception until capacity and provided opportunities for intervention across time (Burnett et al. 2006, Kaiser et al. 2006, 2007, Eiswerth and Johnson 2002), while others have looked at spatial decisions to treat for invasive species over shorter time frames (Wainger 2006). Our analysis, however, stresses the importance of simultaneously analyzing both spatial and intertemporal dimensions.

Though widely recognized as important, few studies have integrated spatial considerations into damage assessment, spread of invasion, and determination of optimal management strategies. Economic impacts from invasive species, conveyed as expected damages to assets from invasion and expected costs of successful prevention and/or removal, may vary significantly across spatially differentiated landscapes. Species often exhibit both positive and negative attributes or otherwise vary spatially in the level of net damages (Horsch and Lewis 2009; Kaiser, 2006; Nicholson, 2006; Paynter et al, 2006; Zivin et al, 2000).

The most basic spatial consideration is whether an ecosystem has a physical niche that can be exploited spatially by a new species; establishment success and invasion magnitude are often dependent upon an invasion occurring where ecosystem resiliency is low(Elmqvist et al., 2003; Walker, 1995). Neubert and Parker (2004) discuss modeling the spread of invasive species using spatial models that include population structure and environmental variability. In our example, the Hawaiian islands have no snake species, and prey is sufficient throughout the islands, thus the threat extends across all land on the island.

Timar and Phaneuf (2009) develop a spatial and temporal model of spread for an aquatic invasive. Some aquatic invasives are transported by visitors rather than natural dispersion, suggesting human intervention must be built in to spatial-dynamic analyses of invasive species spread. Further evidence of the spatial role of humans comes from Koch et al. (2009) and Yemshanov (2009), who use data on tonnage of imports into the United States and Canada, respectively, to develop "risk maps" predicting the likelihood of a wasp invasion into these two countries.

Heterogeneous habitats, ecosystems, and human population densities will affect the magnitude of economic damages inflicted by a nuisance species, rendering damages borne by an invasive species unequal across a landscape. For example, human related damages will be higher in more densely populated areas, while biodiversity losses may be more pronounced in forested, less inhabited areas. When net damages from an invasive species are spatially separable like this, policy decisions should incorporate this additional information to improve asset allocation in the search to minimize the total cost of the damages and management activities. All other conditions equal, we should target mitigation activities in areas where the return, in damages avoided, is highest, and these activities should continue as long as the present value returns to the endeavor are at least as high as the returns from other investment opportunities.

All other conditions for mitigation activities are rarely equal, however. Policy must also incorporate variations in the present value expected costs of these efforts. The magnitude of the effect of costs on optimal management of invasive species may be significant, given the findings of Ando et al. (1998) with regard to the impact of costs on optimal conservation of land for endangered species.

Management of invasive species is further complicated by the political expediency of ignoring the problem until it grows into a visible and expensive threat. In the face of limited and ever-shrinking budgets, resource managers are forced to allocate funds to the most immediate and obvious problems. Spending money to search for what may or may not be present in an environment, regardless of the potential for costly damages in the future, is less popular with constituent groups than actively attacking presently visible problems. This is in part because with typical discounting, pushing costs into the future reduces their present value, so that reducing prevention efforts may seem cost effective *a priori*. Additionally, in the presence of uncertainty about the need for such control activities, uncertain prevention efforts may similarly be inefficiently delayed in favor of certain, but more costly, control efforts (Finnoff et al, 2006).

Prevention targets species that may or may not ultimately cause economic or ecological disruption, while control efforts target reductions in existing disruptors. These policy instruments are also distinguishable by the fact that prevention activities vary spatially by economic activity, while control activities will vary spatially depending on a combination of both economic and biological factors. For example, entry pathways for new invaders to remote islands are almost entirely human-assisted and will occur at sea or air ports of entry. Spread of an invasive species from that entry point, however, will be a species-dependent function of both local economic transport routes (e.g. Timmins, 2006) and biological growth (Christen and Matlack, 2006; Shigesada and Kawasaki, 1997).

In this light, prevention of invasion and removal of successful entrant species may be viewed as both complementary and competing technologies for mitigation of invasive species threats. The boundary between complementarity and competition is particularly clear in the case of EDRR efforts, where costly searches for possible invading populations must often extend far beyond the ports of entry targeted by prevention, yet may result in no captures (FICMNEW 2003, NISC 2003, Meacham 2001). EDRR is a distinct management instrument and should not be considered a typical prevention or control effort. The success or failure to adequately fund EDRR efforts is frequently cited as a major contributing factor to mitigating impacts of invasive species (e.g. New

Zealand successfully intercepted *Solenopsis invicta* and *Lymantria dispar* at the border; while the lack of appropriate EDRR systems has been blamed for the spread of *Avian Influenza* globally (Harris 2001, Pascoe 2001, Pitt et al. 2007, Elbers et al. 2005)) so the ability to more economically deploy these resources should have a significant effect on the success of invasive species policy.

In this paper, we address the effect of spatial variation in damages, costs, and biological growth on policy instrument choices over time using the case study of the Brown treesnake on the island of Oahu, Hawaii. The Brown treesnake has caused significant economic and ecological damages on Guam in the form of power outages, biodiversity losses, and medical costs related to snake bites (Savidge 1987, Fritts et al. 1987, 1990, 1994). 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. Three such deployments have occurred in Hawaii in the last few years, one on the island of Maui, one on the island of Hawaii, and the other on Oahu, away from initial ports of entry, although none of these efforts produced a snake.

Optimal policy generally seeks to minimize the net damages from such a threat as a whole, which will require spatially dependent, and therefore nonlinear, objective functions. Using economic theory and 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 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 likely specimens throughout the potential habitat range that have evaded detection. Dynamic and spatial differences in policies are compared given varying assumptions about the manager's planning and management horizons and the arrival of the snake.

3. Model and Implementation

3.1. Ecological and Economic Components of the Problem

The search decision is a function of the damages and costs anticipated across time and space from an invasion emanating from a likely point of entry. We model growth as spatially and population-dependent, with the specific form dependent on the bio-economic properties of its spread. In our case we use a diffusion model, discussed here. We model costs as an area-specific function of search effort, and marginal damages (per specimen) as area-specific functions of the resources threatened by the species' presence. Details of the growth, cost, and damage functions are described here to contextualize the problem.

We do not formally model uncertainty beyond that regarding the point of entry, using expected values for marginal damages, marginal costs, and an area's invasion population at a given time.

We parameterize the model using data from Hawaii and Guam, and investigate across thirty years of potential snake presence on the island of Oahu to identify the net benefits of EDRR. We initially assume that decisions are made without concern for a budget constraint in order to identify the net-damage-minimizing search strategy.[3] We later investigate the impact of land managers' inability to expect ready funds by imposing a temporal constraint on the availability of funds for search, mimicking the very real possibility of searches not made for lack of funding.

3.1.1. Expected Populations and Growth

We model the spread and population of the invader using a Skellam-Fisher diffusion process common in biological models (Shigesaga and Kawasaki, 1997), and now growing in use in bioeconomic models (Clark, 2005; Wilen, 2007). The diffusion process has been modified in our numerical solutions to accommodate discrete time and space as well as human intervention both within and across cells. Other choices, such as metapopulation models, are potentially appropriate but the biological research on the snake suggests diffusion is the best choice for our application.

In our diffusion process, populations increase by (weighted) radial expansion and by internal growth.[4] Assuming an initial distribution where n_0 individuals invade the origin at $t=0$, each cell $i = 1,…,I$ contains untreated populations from diffused growth:

$$n_D(i,t) = \frac{n_0}{4\pi Dt}\exp\left(bt - \frac{r(i)^2}{4Dt}\right),\qquad(1)$$

until $n_D(i,t) = n_\gamma$, where n_γ is a threshold population level at which the growth function changes, after which growth is represented by a logistic function. Here, $r(i)$ is the discrete, weighted radial distance from the origin to the center of the cell i, D is the

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[3] Expenditures are generally funded by a combination of allocated resources from land management agencies (local, state, and federal) as well as contingency funds awarded as "emergency funds" or otherwise appropriated as expenditures occur, so that budget constraints should be considered politically fungible in any case.

[4] Our discrete population model is adapted from a theoretical model (Shigesaga and Kawasaki, 1997) where population changes as a function of both dispersal and internal growth:

$$\dot{n} = D\left(\frac{\partial^2 n}{\partial x^2} + \frac{\partial^2 n}{\partial y^2}\right) + \left(b - \mu n\right)n$$

In which $n(x,y,t)$ 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, $\mu \geq 0$ captures intraspecific competition, and x and y are spatial coordinates, and the radial distance, r, is determined by $r^2 = x^2 + y^2$. The first term above captures the rate of spread, the second captures population growth within the given coordinates.

Because there is no explicit solution to this non-linear problem unless $\mu=0$, in order to create a tractable model that incorporates both spread and internal growth, we use the solution to the Skellam model for exponential growth and spread (i.e. $\mu=0$) (Eqn. 1) until the population of the cell reaches the point where it diverges significantly from a logistic growth function with a capacity of K snakes and an intrinsic growth rate of b, which occurs at n_γ snakes. From that point, we use a logistic growth function (Eqn. 2) to determine population in an area. We do not simply use the logistic function because it does not allow for radial spread to and from other cells; we do not simply use the diffusion with $\mu=0$ because this would imply no biological limit to population (which is certainly not appropriate at the spatial scale used here).

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diffusion coefficient, and *b* is the intrinsic growth rate (values for n_γ, *D*, and *b* are discussed below).

The radial distance, *r(i)*, is weighted for terrain accessibility, using Geographical Information Systems software (ArcGIS 9), by distance from roads. Roads are expected to provide the most rapid expansion paths (Timmins 2006); distance from roads slows the radial spread. This can be seen in the uneven concentric expansion across the island in Figure 1, in which we map the expected radial spread, and is calculated in ArcView 9 by increasing the radii proportionally as accessibility decreases according to the findings in Wiles et al. (2003).

Figure 1 shows expected expansion patterns of the snakes from an initial invasion at the Honolulu Airport (HNL) or the adjacent Hickam Air Force Base Airport, from Schofield Barracks, or from Barber's Point Air Station, weighted by historical interceptions.[5] These are selected as the most likely points of entry because of the eight Brown treesnakes that have been intercepted on Oahu, five were at HNL, two were at Schofield (with one of these known to have been transported directly from HNL), and one was discovered at Barber's Point.

The rate of expansion (without intervention) is a function of *D*, the diffusion coefficient, which we estimate to be 1.067 km^2/yr (Shigesada and Kawasaki 1997: 51, from an associated expansion rate of 1.6 km/yr (Wiles et al. 2003)), and the intrinsic growth rate, *b*, which we estimate to be 0.6, as well as time *t* and the initial population, n_0.

<<Figure 1 here>>

The expansion path illustrated in Figure 1 translates directly to population when population is low, but as population increases in a cell, intraspecific competition is accounted for by switching to a logistic growth function. Thus, n_γ is the population at which the growth paths for diffusive, exponential growth and logistic growth diverge, given carrying capacity, here $n_\gamma=40$. (Shigesaga and Kawasaki, 1997; Kaiser and Burnett, 2007) After this point, internal growth dominates and untreated populations grow according to a logistic function, n_l, where:

$$n_l(i,t) = n_\gamma \left(\frac{K_i e^{bt}}{K_i + n_\gamma (e^{bt} - 1)} \right),$$ (2)

In which K_i is the carrying capacity of the cell. We estimate from maximum densities experienced on Guam that the maximum snake carrying capacity in any cell (so $K_i=K$) is 200 snakes.

Human intervention affects the population of a cell directly, as we assume that search of an area is efficient in that any specimens located in a searched area are caught and removed. The result is that damages are not incurred in that time period and the population, if any, is reduced to zero, but restarted with diffused growth. We refer to this as 'treatment' of an area. Furthermore, search of any area today changes the expected radial spread of populations across the entire habitat for the next period, so the greater the

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[5] Individual maps for each of the three locations are available upon request.

search effort, the slower the expected diffused growth across the entire habitat.[6] If a cell is already expected to be driven by internal growth, changes in the population of other cells do not affect a cell's expected population.

Thus the cell populations are:

$$n_{it} = n\left(i, t, t_{\xi_i}, \xi_{it}, \sum_i \xi_{it-1}\right) = \begin{cases} \left(\dfrac{\sum_i \xi_{it-1}}{I}\right) * n_{Dit-t_{\xi_i}} * (1-\xi_{it}), & \text{if} \leq n_r \\ \\ n_{Iit-t_{\xi_i}} * (1-\xi_{it}), & \text{otherwise} \end{cases}$$

(3)

where t_{ξ_i} is the time period of the last search treatment for cell i, and $\xi_{it}=1$ if a cell is searched at time t and $\xi_{it}=0$ otherwise. The larger the proportion of treated cells in the previous period $\left(\dfrac{\sum_i \xi_{it-1}}{I}\right)$, the lower the rate of expected population expansion, or

alternatively the lower the expected population that has escaped notice island-wide. If the entire island were to be searched then the population would fall to zero and no damages would occur until another invasion. We simplify by assuming that a new snake arrives in the next year (alternatively, that a snake is missed). This resetting of n_D is accounted for by the time lag $t_{\xi i}$.

3.1.2. Damages

Figure 2 illustrates the range of marginal (per snake) damages across Oahu. Damages are calculated using a per snake linear coefficient that varies from a minimum of $0 and a maximum of $2143. Damages consist of three potential impacts: power outages, medical costs and human-snake interactions, and biodiversity losses.[7]

<<Figure 2 here>

First, Brown treesnakes are known to cause frequent power outages by damaging power transformers and power lines. Localized hour-long power outages on Guam occur on average every other day, with longer but less frequent wide-scale power outages when snakes interact with main power lines or transformers. We anticipate that an average snake will cause 1.01×10^{-4} power outages per year, that per-person power outage damages are $1.50, and that an island-wide power outage causes $1.2 m in damages (Burnett et al. 2006, Fritts 1998, personal communication). Thus, cells where only localized power outages are expected will have expected damages equal to the population

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[6] Though search might be expected to reduce the populations of neighboring areas more than more distant ones, we do not weight the effect of search by distance from the area searched. Our diffusion model places a positive population in any given area at all times that radiates from the expected points of entry, and if search reveals no specimens, as is likely with EDRR, then that information should affect expected populations island-wide and not just in neighboring areas. In our case study, it is also true that specimens are capable of travelling across wide ranges.

[7] Maps of these individual components are available upon request.

in the cell (determined from census 2000 census block population density figures) times per-person damages times the per-snake rate of outages, or

$$\delta_{pl} = popn * 1.5 * 1.01 \times 10^{-4}.$$

Cells where main power lines are located have expected damages per-snake of:

$$\delta_{ph} = 1.2 \times 10^{6} * 1.01 \times 10^{-4} * power = 121.2,$$

where power is an indicator variable for whether main power lines are present in the cell.

Second, Brown treesnakes are venomous and Guam's infestation has generated an average of 170 bites per year with average costs of \$264.35 per bite. This is an expected per-snake bite rate of 6.31×10^{-5} for Guam. We generate a bite rate for each Oahu cell by multiplying the Guam per-snake bite rate by the population density of the cell (again from the census block 2000 data) adjusted by the population density on Guam (United Nations 2001). Using this bite rate, we then calculate the expected direct medical damages as the bite rate*average costs.

Additionally, Hawaii has no snakes at present; human-snake interactions are expected to reduce tourism and local quality of life significantly (Shwiff et al. 2010). Following conservative estimates for patterns of legal settlements of compensatory versus punitive damages for psychological suffering and other intangible losses (Eisenberg et al. 1997), and other evidence on the ratio of physical damages to psychic ones, including U.S. congressional recommendations for limiting punitive damages to no more than three times the physical damages (Kahneman et. al 1998), we use the anticipated physical medical damages as a minimum per-person estimate in the loss of well-being generated by the snakes to generate an indirect 'medical cost', so that total human-snake interaction damages for each snake for each location are estimated at:

$$\delta_{m} = biterate * popn * 264.35 .[8]$$

Finally, adult Brown treesnakes prey upon birds, and Oahu's endangered birds, which have not co-evolved with snakes, are likely to fare as badly as Guam's indigenous birds have. On Guam, 11 of 18 indigenous species (Vice 2006, personal communication) have been extirpated. On Oahu, one native bird species occurs on island (Elepaio, *Chasiempis sandwichensis ibidis*) and its habitat covers only about 26,400 ha of the 1,500 square kilometers. Other birds at risk have habitat on other islands and are less likely to be rendered extinct through the presence of the snake, though the population losses will increase the probability of extirpation and extinction. Cells with habitat for the elepaio are estimated to have per snake damages of \$6.52, based on willingness-to-pay estimates determined for households in Hawaii for the existence of bird species, and cells with other native bird habitat are estimated to have per snake damages of \$1.13 (Loomis and White 1996, Burnett et al. 2006).[9] Biodiversity losses per snake per cell are thus

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[8] Note that if a human were actually bitten by a snake this would certainly confirm the presence of the snake (but not the population). Since populations are in expectation of an arrival at a given time at one of three potential locations, and thus damages are also expected values, we would still benefit from EDRR policy in targeting the move to control of the incipient population. We leave modeling this update in the information structure for future work.

[9] The expected per snake damage level for bird habitat other than the elepaio is estimated by assuming that at carrying capacity, there is roughly a 98% chance of losing a single species and a 5% chance of losing a second bird species. With an expected value to 280,000 Oahu households of losing one species of \$8.68 million (Loomis and White 1996), the expected per snake damage level is \$1.13 per year, assuming that

$$\delta_b = 6.52 * elepaio + 1.13 * otherbird,$$

where *elepaio* and *otherbird* are indicator variables for the presence of elepaio and other native bird habitats, respectively.

The per-snake damage function for each cell is therefore, $\delta_i = \delta_{pl} + \delta_{ph} + \delta_m + \delta_b$ so that the damage function is:

$$d_i(n_i) = \delta_i * n_{it} = \left(\delta_{pl} + \delta_{ph} + \delta_m + \delta_b\right) * n_{it}.$$

3.1.3   Costs

As discussed, a particular distinction between EDRR and other discussions of invasive species control is that with EDRR one is searching thoroughly, as if trapping the last snake in a population control situation, while with control one generally assumes one can "harvest" a known population of the invasive species, balancing costs of the marginal specimen captured with damages from the remaining population directly. We describe Brown treesnake 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 from 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:

$$EER = 0.1v + 1.8v \cdot a + 3.5 \tag{4}$$

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. Rugged terrain (steep or highly variable slopes) and high elevation are the most costly to search, while cells near roads, flatter and closer to sea level require less energy and effort and are therefore less expensive to employ EDRR.

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 $6,352.[10]

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each snake is equally likely to contribute to the extirpation.   The damage estimate for the elepaio is similarly calculated by assuming that the presence of the snake at capacity level in every habitat location will result in extinction, thus the per snake damages are $8.68 million/(6654 cells*200 snakes), or $6.52.

[10] 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.

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 cost for cell i, C_i ,is then the sum of the in-cell treatment cost and the distance (access) cost.[11,12]

3.2 Bioeconomic Model

A manager who expects there has been an invasion that has escaped detection at a port of entry can choose whether to initiate an EDRR response and search, or wait until the invasion is confirmed without search costs, at which point population control might begin. Because low population control will suffer some of the same difficulties as EDRR in terms of determining where and when to best to target removals, our deterministic model of an unfolding invasion in effect addresses both policies. As EDRR bridges the gap between prevention at bottleneck entry points and control, the best places to search will also be the first places one will attempt removals if snake populations unfold as expected. While more elaborate modeling would incorporate the exact number of snakes removed upon a search and using that to update expectations or allow marginal tradeoffs between snakes controlled today and those left for the future within an area, the complexity of such modeling is not necessary to highlight the importance of EDRR as a management option when populations or their associated damages may be masked for a period before becoming known. For additional insights into the role of uncertainty in EDRR (without explicit spatial considerations), see Haight and Polasky (this issue).

Thus our theoretical model is formalized as:

$$\underset{\xi_{it}}{\mathrm{Min}} \sum_{i}^{I} \sum_{t=0}^{T} \beta^{t} \left(d_i(n_{it}) + C_i(\xi_{it}) \right) \tag{5}$$

s.t.

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[11] 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.
[12] Figures illustrating these total costs are available upon request.

$$n_{it} = n(i, t, t_{\xi_i}, \xi_{it}, \sum_i \xi_{it-1}) \qquad\qquad (6)$$

and

$$\xi_{it} \in \{0,1\}, \quad \sum_i C_{it} \leq A_t \ \forall t \qquad\qquad \text{(7) and (8)}$$

where n_{it} is the population of the cell at time t as a function of biological growth (determined by diffusive and internal growth and a function over time of the cell's location, i), EDRR search (treatments) in the cell today (ξ_{it}) and all other cells $\left(\sum_i \xi_{it-1}\right)$ in the previous period, as well as the time since the last search treatment in the cell, t-$t_{\xi i}$, as described above in section 3.1.1 and stated in Eqn (3). Expected damages, d_i, are the total expected damages for area (cell) i as a function of population n_{it}, as described above (section 3.1.2). C_i is the cost of EDRR for cell i, also as determined above (section 3.1.3). I is the total number of cells, β represents the discount factor, and A_t represents a temporally constrained appropriations budget for EDRR, should such a consideration arise.

Spending C_i brings the expected 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.

Because the control variable ξ is a binary integer, 1 if removal that period and 0 otherwise, standard first order conditions are not possible. In order to understand the nature of our solution, we write discrete versions of the necessary conditions. Taking the difference between the Lagrangian with ξ_{it}=1 and with ξ_{it}=0, this condition is:

$$\Delta L = L(\xi_{it} = 1) - L(\xi_{it} = 0) = \sum_t \beta^t \left[\sum_i \left(d_i(n_{it}[1]) + C_i(1) + \lambda_{it} \left(n\left(i, t, t_{\xi_i}, \xi_{it}, \sum_i \xi_{it-1}\right) - n_{it}[1] \right) \right) \right]$$

$$- \sum_t \beta^t \left[\sum_i \left(d_i(n_{it}[0]) + C_i(0) + \lambda_{it} \left(n\left(i, t, t_{\xi_i}, \xi_{it}, \sum_i \xi_{it-1}\right) - n_{it}[0] \right) \right) \right]$$

$$= \beta^t \left(d_i\left(n_{it}[1]\right) - d_i\left(n_{it}[0]\right) + C_i(1) - C_i(0) + \lambda_{it}\left[n_{it}[1] - n_{it}[0]\right] \right) \leq 0 \qquad (9)$$

$$\Rightarrow C_i(1) \leq d_i\left(n_{it}[0]\right) + \lambda_{it} n_{it}[0], \qquad\qquad (10)$$

$$\Rightarrow \left(C_i(1) - d_i\left(n_{it}[0]\right)\right) / n_{it}[0] \leq \lambda_{it}, \quad \text{or} \quad \frac{C_i(1)}{n_{it}[0]} - \frac{d_i\left(n_{it}[0]\right)}{n_{it}[0]} \leq \lambda_{it} \qquad (11)$$

where λ_{it} is the (approximate) shadow value of specimens in cell i at time t. From equation 10, the cost of control should not exceed the imputed damages from specimens in the cell, where these costs reflect both current and future values. From equation 11, the average cost of control net of the average damage of a specimen not removed should not exceed the shadow value of the current population for control to be worthwhile.

3.3 Implementation

Consider that Oahu's 1,500 square kilometers are divisible into a grid of approximately 38,000 cells measuring 4 ha each. Using Geographical Information Systems (GIS) software (ArcView 9), each cell is assigned initial properties that include currently existent data on likelihood of snake presence (distance from points of entry, proximity to roads[13]), resource assets at risk (bird habitat, presence of power transmission lines, human population density) and accessibility of treatment (proximity of roads and trails, slope). We use this information to determine explicit spatially dependent values for $r(i)$, $d_i(n_{it})$, and C_i as described in section 3.

The objective will be to minimize the present value of costs and damages over a 30 year time horizon for EDRR as described in our model. Since treatment decisions are EDRR search decisions, the unit of decision is the spatial cell rather than the snake population directly.

We use Microsoft Excel and Palisade's Evolver Industrial Version, a genetic algorithm solver add-on to Microsoft Excel, for analysis.[14] We program MS Excel using the expected values for costs, damages, and population spread and growth so that we have three interconnected matrices that together can be used for dynamic analysis and optimization. The first of our three matrices generates population values, as described above in section 3.1.1. For each spatial cell, the population is calculated for 30 years from an initial expected invasion and varies with EDRR treatment of the cell.

If no treatment occurs, the population grows according to its biological limitations and location on the island. For all other treatment levels, the diffused growth population of each cell is reduced by the percentage of treated cells on the island.[15] This is

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[13] 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. 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 assume 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.

[14] As computing power increases, policy managers will be able to integrate simple software like MS Excel into their decision-making tools. By using this program, we hope to provide a blueprint, in anticipation of the rapid pace of technological change, of how such analysis, using widely available data as in this case, can become a more standard management tool.

[15] Though this does not treat neighbors of treated cells differently, the Skellam growth model already incorporates the distance from the initial invasion and thus treatment will reduce growth proportionally. The simplification is necessary for computational feasibility. The population matrix includes, for each cell, a decision regarding whether logistic, local growth dominates or dispersal growth does, as discussed in section 3.1.1.

considered roughly equivalent to searching an area that should have n_{it} snakes, finding it does not, and reducing expected populations island-wide. The more cells that are treated in any time period, the smaller the remaining population available to spread across the island, or the fewer hiding places for a very small population.

Treatment of a cell ($\xi_{it} = 1$) returns the base population for the subsequent time period to n_{i1}, the expected re-infestation from diffused growth, at a cost of C_i, and resets the expected population from that point. Therefore our second matrix consists of the binary treatment decisions, ξ_{it}. This is the (binary integer) control variable in the optimization.

Net damages are calculated for each cell as a function of whether it is treated or not and its population; in our third linked matrix, net damages for a cell at time t are equal to either untreated damages or the cost of treatment, in other words $C_i\left(\xi_{it}\right) + d_i\left(n_{it}\left[\xi_{it}\right]\right)$. The present value of these net damages over all cells for thirty years is summed and forms the object of minimization.

To solve this optimization, we begin by bounding the problem to treat cells as soon as the current benefits of doing so outweigh the current costs. We consider this to be a myopic solution in that it misses opportunities for aggressive treatments that reduce the invasive population. It is, however, a solution that managers find appealing as it avoids searching for infestations that do not exist. We label an EDRR solution that searches a cell only when the current benefits outweigh the current costs as the "myopic policy solution."

From this initial benchmark, we take three steps to improve upon our solution. First, we set Evolver to search for mutations from this myopic solution that improve upon the minimization. Over 50,000 trials of mutations initiated from the myopic solutions as well as 20,000 trials of random mutations across space and time resulted in zero cases of improved outcomes. We interpret this as confidence that the myopic solution provides a reasonable upper bound for minimization.

Our theoretical results suggest that the myopic solution is incomplete in that it neglects the final term of equation (6), as the shadow values are not included. As a next step, then, we identify cells for treatment that lower the total net damages over time even when their current costs outweigh current damages. We begin to do this by applying the myopic solutions (where current damages outweigh current costs) generated for different growth rates. This systematically captures the cells where expected damages over time are most different from current damages, and therefore the most likely cases to benefit from targeted search even when current costs are less than current damages.

Finally, following this, we sort the cell locations by their treatment costs and marginal damages, and use brute force to test whether the new solution can be improved upon for cells where damages are relatively high or fast growing or costs are relatively low. These are identified as locations where at least one treatment occurred over the 30 years under the myopic solution where growth was assumed to be 0.7. By assuming that growth is somewhat higher than our estimated value of 0.6, we can systematically identify cells with more rapidly growing damages toward the end of the 30 year period, which will benefit from treatment but are not identified under the myopic policy solution. We tested whether using an even higher growth rate to identify potential cells for search would lead

to any new net-damage-minimizing treatments; it does not. Targeting these cells would be too aggressive.

This combination of steps resulted in our optimal solution, described next.

4   Results

We present results on the optimal search, alternative search strategies, and discuss policy implications. Results are summarized in Table 1. As we assume an invasion at *t=0*, the overall cost of doing nothing, in accumulated damages, is estimated to be $371 million over 30 years.

In the case of a mixed integer non-linear programming problem (MINLP) like this, determining the optimal treatment pattern across time and space numerically is computationally challenging, given the large number of choice cells that result from considering each 4 ha plot every year for thirty years (approximately 1.1 million cells). Thus after finding an optimal solution using the steps described above, we try a series of treatment options meant to mimic real policy decisions and draw from these treatment options policy lessons that will enable improved searches following relatively straightforward managerial advice that relies on the interplay of marginal costs, marginal damages, the growth rate and carrying capacity of the invasive species.

4.1 Optimal Search

In our baseline case, we find that optimal search can reduce social welfare losses to $77 million, a savings over a no-action policy of $294 million.[16] This treatment strategy leads to search of 10% of the potential habitat (approximately 0.3% of all cells across space and time), with 1.3% of the island receiving more than one treatment.

The first treatments begin at t=9, and occur in areas that would soon have positive current benefits from treatment. The area treated is small, with only one 4 ha cell searched at t=9 and an additional 75 cells searched over the first five years of searching. Search then becomes more rapid, with between 85 and 510 cells searched each year. Several cells are treated more than once; one cell near Honolulu International Airport should be searched and cleared 4 times. These cells integrate damages, costs, and the biological spread in such a way that EDRR 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.[17]

4.2 Alternative strategies

To illustrate several key policy findings of our analysis, we consider alternative strategies that highlight certain aspects of EDRR that are different from other invasive species solutions or that have been missed by skipping over EDRR in analysis of the dynamics of invasions. We first consider myopic searches, where current benefits must

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[16] Note that this solution, like all "optimal" solutions in MINLP, is an approximation. Other savings may also be possible, but individual inspection of the cells that have the lowest cost: damage ratio as a function of population cannot improve upon this outcome (brute force), and random trials from this starting point do not improve it either.

[17] 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.

outweigh current costs in order to instigate action. We then consider a no action alternative, the possibility of periodic island-wide sweeps, and searches where timing is thwarted by budget fears.

4.2.1 Myopic search

Figures 3-5 present temporal snapshots of the current net damages (i.e. only the damages net costs in that year) that would occur if all cells were treated just once, at that time. Figure 3 shows treatment outcomes 15 years out. Here, only a handful of cells have current damages less than current costs. Just 5 years later, shown in Figure 4, we see the number of cells benefiting directly from treatment is significantly greater. Figure 5 shows locations with positive immediate returns to treatment after 30 years, our full time frame. 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 areas for which damages do exceed costs (shown increasingly from orange to red), so that EDRR treatment is cost-effective in this single period, are obviously also the areas where optimal EDRR should be targeted.

<<Figures 3-5 here>>

Figures 3 through 5 highlight a key role for EDRR beyond prevention. If search is likely to be as productive or more so at key locations beyond expected ports of entry, where net damages are expected to be high from ignoring a new invasion, EDRR should be undertaken. In cases of species like the Brown treesnake, where the number of interceptions and the propagation pressure from the source (here Guam) is high enough that it is quite likely that prevention will soon fail, if it has not already, EDRR activities, optimal or not, are vitally important. (See Burnett et al., 2008 for more on the snake's probability of arrival).

Recall that the first step in solving for our optimum was a myopic treatment strategy where only cells where the current costs were less than the current benefits received treatment. This myopic strategy only reduces social welfare losses to $101 million dollars, rather than the $77 million losses under the optimal solution. Over the thirty-year period, just over 3000 cells were identified for treatment using this strategy, 8% of the island, or 0.2% of all possible searches across the thirty year period. This treatment plan also delays any search until the 12th 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.

Our optimal solution is more aggressive. It treats a bit more of the island, earlier. In particular, as with the myopic solution, it targets search in areas with high net returns, whether they are due to low costs, high expected marginal damages, or high population growth and thus high total damages.

Below we consider other alternative strategies.

4.3. No Action

We identify the "do nothing" case – a present value of expected damages of $371 million accumulated over 30 years from an initial invasion of a single snake, spreading with a growth rate of 0.6, from one of three possible entry locations with no EDRR action.

4.4. Island-wide Searches

Though random search is counterproductive and targeted search treats only a very small amount of the island, there may be some benefit to periodic island-wide sweeps, in which the entire island is searched at one time. We investigate the returns to island-wide sweeps at various stages to highlight these tradeoffs. The cost of an island-wide search is estimated at just under $447 million. In the worst case scenario (island-wide search + new snake), 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 for this timed island-wide search 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 for a late island-wide search. 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 start to grow only after year 16.[18]

4.5 Restricted planning horizons

Another likely restriction for managers is the inability to plan for EDRR funds over a long period of time. We investigate the effects on the various strategies if funding can only be secured in 5 year increments. No plans are made for spending after each five year increment due to the lack of funds. The strategy we take here is to start with the optimal policy, but then evaluate how the funding timeline affects the overall outcome. On the one hand, if the funding comes, say, only in the first 5 year period of the invasion, it may be worth moving up the search timeline and searching some locations in the 5th year (even though optimal policy would delay until the 9th year, and even then be sparse). The present value losses of this case ("5 year plan") are $227m.

Then, if funding is again renewed, the evaluation can be reproduced for the next 5 year period, again with most search choices being whether to search earlier than is optimal given the funds are in-hand.

At this point, however, it also becomes necessary to evaluate whether some of the treatments that would be optimal if retreatment could happen later, might be better off being delayed to the end of the 5 year period, trading off a bit more current growth (and

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[18] At year 15, even with exponential growth, no cell has greater than 28 snakes, just over 10% of carrying capacity. This begins to change rapidly in years 15 to 30.

damages) for slower growth for a few extra years into the unsearched future. Treating a slightly larger group of high expected net damage cells after another five years ("5 and 10 year plan") reduces damages to $142 m, while additional treatments at year 15 reduces the damages to $126 m ("5, 10, and 15 year plan"). These significant savings over doing nothing come even when our earlier analysis shows most of the growth and damages are not occurring until after year 15, which highlights the value of EDRR even when long term planning is not feasible and when benefits may seem far removed from search efforts.

Extensive but random search, however, is likely to raise costs more than reduce damages, unless it is comprehensive (island-wide) and occurs between the 11th and 27th year of a successful invasion, when the present value of costs is reduced and growth and associated damages are starting to take off in many cells. When search is random but incomplete, the present value of social costs regularly (across 50,000 trials) lays 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 improve random solutions. Thus, random or incomplete efforts may not be better than doing nothing, but strategic action can dramatically improve outcomes.

5. Conclusions

We develop a spatial dynamic model and solve for the optimal search strategy for a location needing EDRR. Such cases exist when prevention has a high likelihood of failing and/or incipient populations are very difficult to detect; these are common conditions for invaders. We then apply this model to the case of the Brown treesnake in Hawaii.

We find that effective EDRR search targets limited areas of high expected net damages. Only 10% of the island needs treatment in a thirty-year period, if it is applied efficiently, with 1.3% of the island receiving more than one treatment. A myopic strategy where cells are only treated when current benefits (damages foregone) exceed current costs also significantly reduces social welfare losses. In comparing the optimal policy to the myopic policy, we see that more aggressive, yet carefully targeted, search is preferred to a point, but that aggressive policies that are random in nature are more costly than either the optimal or the myopic strategies.

Our application technique also allows examination of several simpler strategies that policy managers might adapt, given temporal and/or financial budgets, and identifies where search should be currently occurring, across the landscape, for maximum benefit under these conditions. A significant debate regarding EDRR hinges on whether overall spending is too little or too much – too little if the budget is spread too thin to do any permanent good; too much if we are searching for highly unlikely snakes now. Our results indicate that under virtually all expected conditions, any targeted search in high impact areas (where damages from the species would be relatively high and search costs are relatively low) will increase social welfare by reducing damages from the snake[19].

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[19] This is true even if we overestimate the biological growth rate and underestimate the costs of search substantially. Details available from authors upon request.

This strong result should encourage Hawaii and other locations threatened by such stealthy invaders to increase EDRR efforts strategically, mapping out expected net damages and engaging in EDRR search in high impact areas, which may differ from ports of entry where prevention activities are focused. Spatial analysis using geographical information systems (GIS) software and integrating biological parameters with economic ones can assist in developing optimal prevention and EDRR policies for invasive species as layers of information regarding damages, costs, and biological growth can be coalesced for analysis.

This result also should strengthen the generalized debate, ranging from climate change to prevention screening of all sorts, for example, over whether early action, when environmental threats may seem small or have low expected value, is worthwhile. The spatial component is particularly relevant to the case of wildfire monitoring, where early fires, detected in remote areas, are expensive to detect but can be extinguished rather easily, while once the fire is easily visible (say, by satellite imaging), it is often too late to intervene and damages are much greater than they could be.

Currently, no known snake populations exist on Oahu, but there is general agreement amongst the scientific community that there may be a small population. We begin our analysis with $n_0=1$.[20] 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.

Inefficient search, on the other hand, can be extremely costly, if it is random or incomplete. However comprehensive island-wide searches can reduce social welfare damages and may have additional external benefits, especially if prevention at entry points is highly effective at reducing the hazard rate.

While the new evidence compiled for this study presents a myriad of opportunities for conclusions and controversy, we focus on four main findings we believe will stand the test of time and that extend beyond the case study presented here.

First, treating EDRR as a separate but vital link between prevention at points of entry and control of known populations allows for insights into the costs of delay at low invasion population levels. In particular, in the absence of sufficient funding to assure eradication, within 5 years of an invasion it becomes preferable to pursue limited EDRR, in most cases even if it is not optimally allocated, as compared to doing nothing. This is true even though there are no areas after only five years where current damages exceed current costs of search – in fact there are no such areas until after 12 years. The benefits of imperfectly administering EDRR increase for the first 20-25 years, and then begin to decrease as damages accumulate. Given the handful of captures and the large number of credible snake sightings on Oahu, these results inform us that if the snake did in fact arrive on island even as far back as the 1980's, it is not too late to prevent significant losses in the future by acting today, though the window of opportunity is rapidly shrinking.

Second, getting ahead of the problem pays off. The pattern for optimal targeted search is more aggressive than a myopic strategy that delays until current damages

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[20] Mitochondrial DNA evidence suggests that the entire population of snakes on Guam may have originated from a single female.

outweigh current costs. Furthermore, limited targeted search based on bio-economic considerations of costs, damages, and growth (covering perhaps 10% of the island) outperforms random search significantly.

Third, in spite of the fact that eradication through concerted island-wide sweeps can be profitable, it is never optimal in our model. Thus, though managers often tout eradication as the best management policy, few economic studies to date, including this one, have been able to verify this. While economic models have proposed eradication as a possible corner solution (Kaiser et al. 2007, Olson and Roy 2005), few case studies have found that it is preferred over some sort of interior solution. With budget constraints and myopic decision-making opportunities, it is beneficial to know that positive and significant returns can be generated by sporadic and incomplete treatments, especially if they are targeted to areas of high net expected damages. The threat of re-invasion reduces the advantages of eradication, but still eradication could be preferred to unmitigated growth.

Finally, incomplete treatments should not be simple extensions of prevention that focus efforts solely on areas adjacent to points of entry. The simple analytical tools and widely available data used here can be tuned to reduce search costs and increase the reduction in damages. In the case of the Brown treesnake, EDRR should be applied to high population density areas as well as areas that serve both as conduits to new territory (roads) and areas that would experience particularly high damages from high snake populations in them. Certainly areas closer to points of entry are likely to have higher invasion populations, however optimal search efforts will also weigh the net expected damages to locate the most efficient search locations and times.

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Tables

Table 1: Summary of Search Scenarios when g=0.6, Original Costs, Damages, and K

Scenario	Description	Present Value Losses
Optimal (baseline) case	Treat approximately 4300 cells, or 10% of the island	$77 million
Alternative Strategies		
Myopic Strategy		
Myopic Strategy (treat when $C_{it}<D_{it}$)	Treat approximately 3000 cells, or 8% of the island	$101 million
No Action		
Do Nothing	30 years of unmitigated snake spread and damages	$371 million
Island-wide Searches		
Island-wide search	Treating each and every cell following initial snake entry	$447 million
Island-wide search + new snake	Treating each and every cell following initial snake entry, followed by another entry	$771 million
Timed island-wide search	Treating each and every cell once between years 11 and 27	$327-$429 million
Late island-wide search	Treating each and every cell in year 30	$523 million
Restricted-Planning Horizons		
5 year plan	Treat few cells with high net expected damages at end of 5 years	$227 million
5 and 10 year plan	Treat few cells with high net expected damages at end of 5 years, then again (not necessarily same cells) in year 10	$142 million
5, 10, and 15 year plan	Treat few cells with high net expected damages at end of 5 years, again in year 10, and again in year 15 (not necessarily same cells).	$126 million

Figures

Figure 1: Expected annual snake expansion, entry at HNL (3/4 weight), Barber's Point (1/8 weight) or Schofield (1/8 weight)
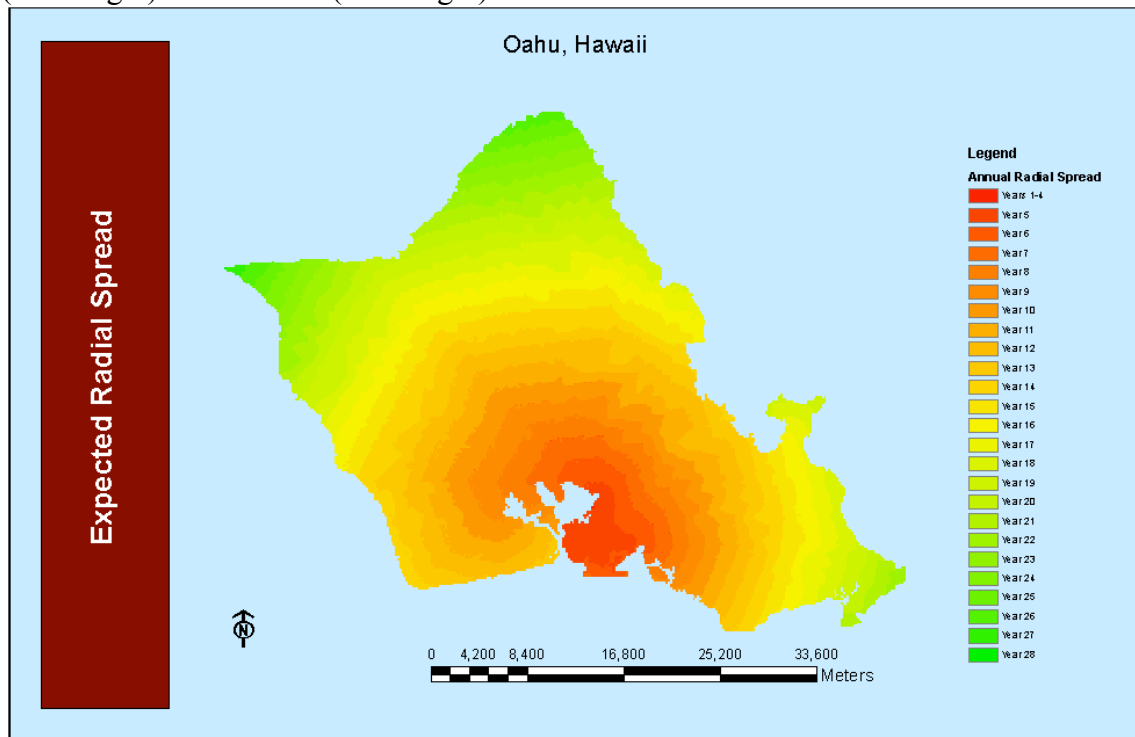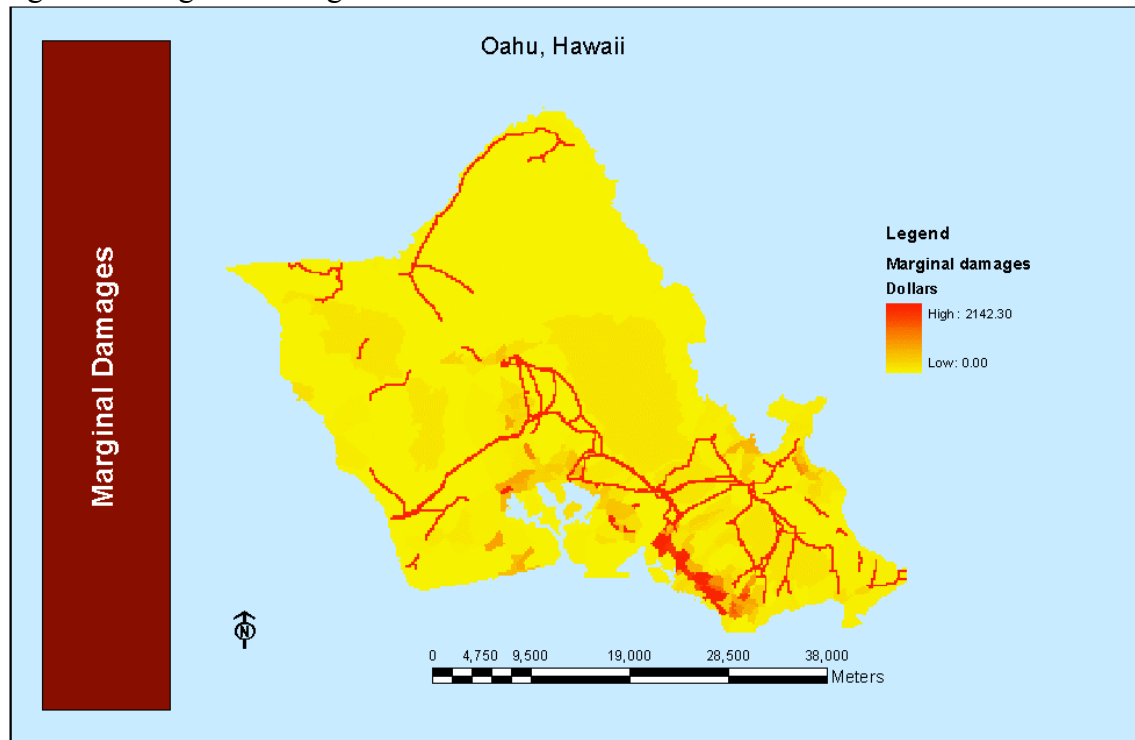
Figure 2: Marginal Damages

Oahu, Hawaii

Marginal Damages

Legend
Marginal damages
Dollars
High : 2142.30
Low: 0.00

0   4,750 9,500   19,000   28,500   38,000
Meters

Figure 3: Current net damages with treatment after first 15 years of invasion

Oahu, Hawaii

Net Current Damages, if Treated, Year 15.
(Static View of Myopic Policy)

Schofield Barracks

Barbers Point   HIA

Current Net Damages,
with treatment
(neg. indicates benefit)

26,508 - 6,367
6,367 - 0
0 - -25,000
-25,000- -38,569

0   4,200   8,400   16,800   25,200   33,600
Meters
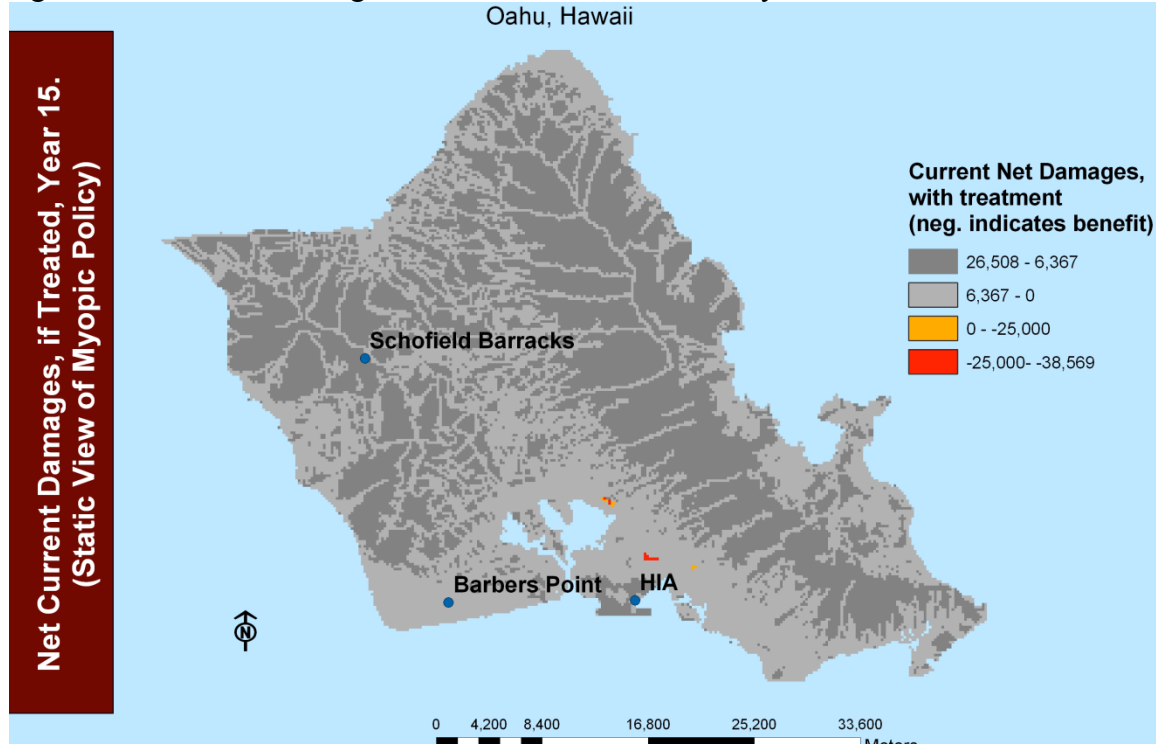
Figure 4: Current net damages with treatment first 20 years of invasion

Oahu, Hawaii

Net Current Damages, if Treated, Year 20.
(Static View of Myopic Policy)

Schofield Barracks

Barbers Point   HIA

Current Net Damages,
with treatment
(neg. indicates benefit)

26,431 - 6,367
6,367 - 0
0 - -25,000
-25,000- --183,201

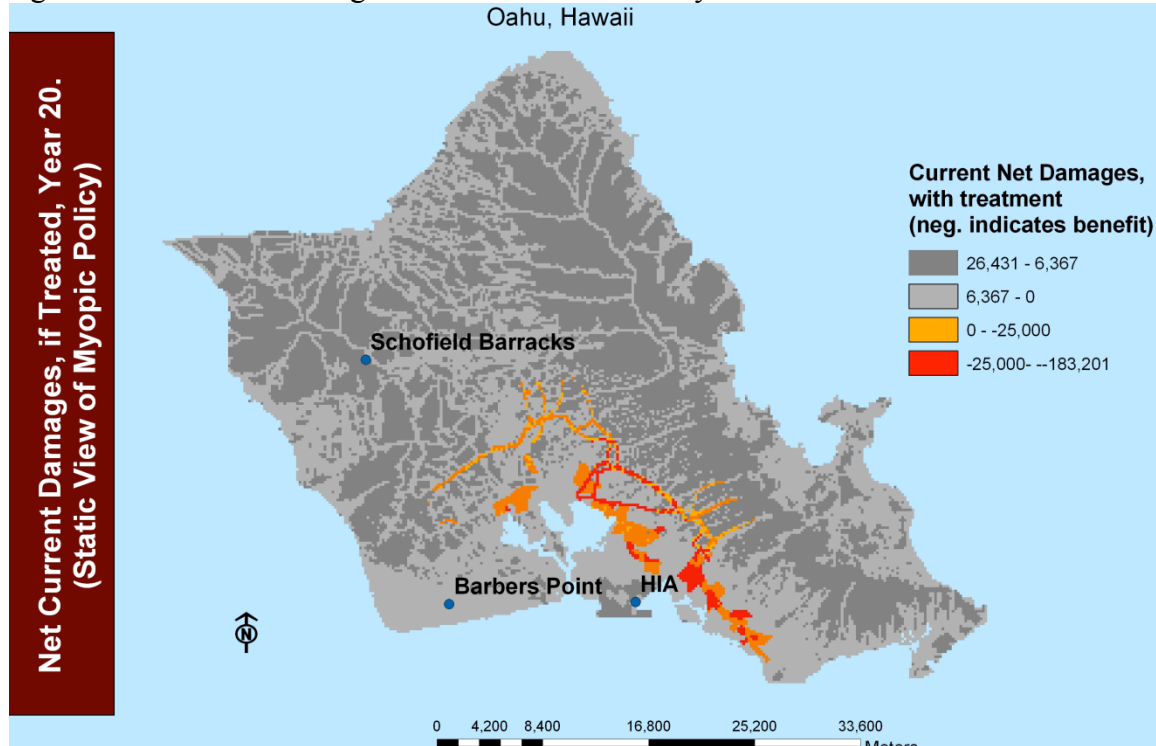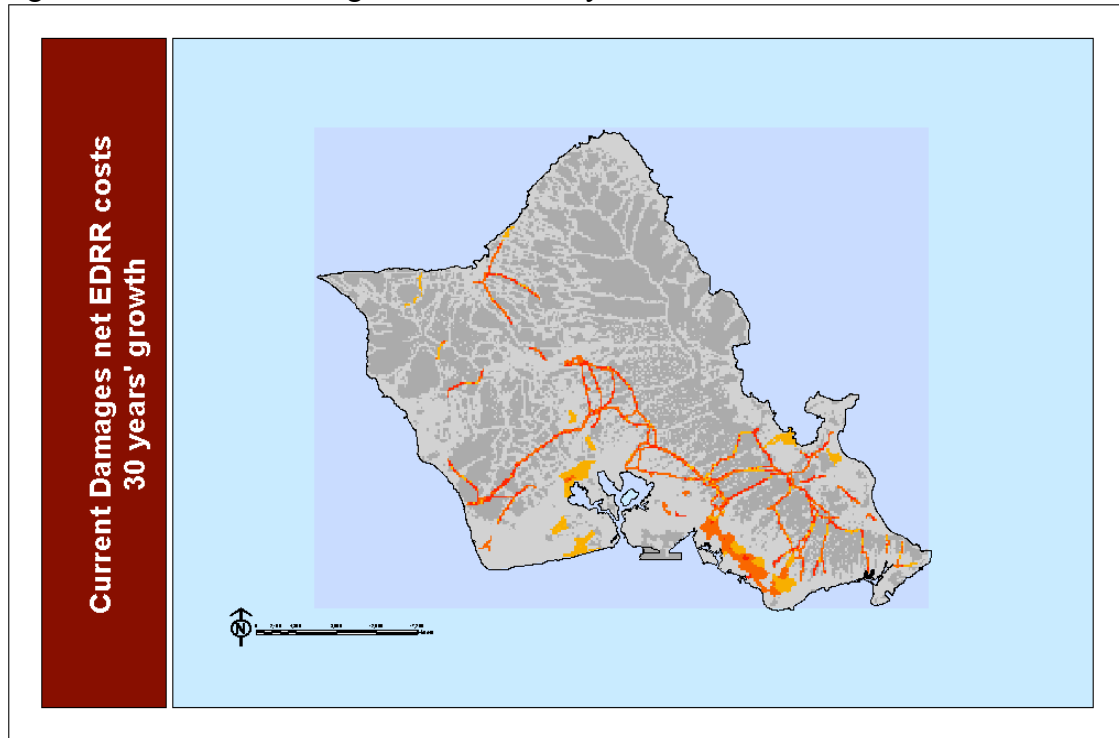0   4,200   8,400   16,800   25,200   33,600
Meters

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

Notes: Orange and red cells have current damages increasingly higher than current costs.
Light gray and dark gray have current costs increasingly higher than current damages.