Economic impacts of non-indigenous species:
Miconia and the Hawaiian economy

Brooks A. Kaiser
Assistant Professor of Economics, Gettysburg College
Dept. of Economics
Gettysburg College Box 391
300 N. Washington St, Gettysburg, PA, 17325
bkaiser@gettysburg.edu.

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Abstract:
Imperfect scientific information regarding potential invasiveness, differences between private and public outcomes for individual decisions regarding planting, and inadequate prevention activity combine to impose costs through a change in native ecosystems susceptible to invasion by hardy, rapidly reproducing non-indigenous species. Concepts and tools from economic theory that may improve policy decisions are explored through the specific example of *Miconia calvescens* in Hawaii. Rapid expansion of *Miconia calvescens*, an ornamental tree introduced to several Pacific Islands over the last century, threatens local watersheds, endangered species, and recreational and aesthetic values in the Hawaiian and Society Islands. Potential welfare losses from the unchecked spread of *Miconia* in Hawaii are illustrated. Policy options investigated include accommodation of these losses, efforts at containment, or eradication. Estimates are determined through an optimal control model describing the potential expansion of the weed and its control costs and damages. Results suggest that cost-effective policies will vary with the level of invasion as well as the expected net benefits from control efforts.

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1. Introduction

The purposeful introduction of species to new habitats for horticultural reasons has historically been viewed as a private decision, based on satisfying the introducer's desires. Communities are learning, however, that these private decisions often have public external consequences that impose costs or benefits on individuals or groups not directly involved in the original decision. When an introduced species escapes cultivation and causes harm to existing ecosystems, and the stream of resources derived from these ecosystems, we deem them invasive species. At that point communities must determine a course of action intended to minimize the damages from the invasion.

As trade routes have become global and introductions have multiplied, experiences like English Ivy (*Hedera helix*), Purple Loosestrife (*Lythrum salicaria*), and Scotch Broom (*Cytisus scoparius*) introductions in the United States have shown that an ounce of prevention would have been, as the old adage says, worth a pound of cure. Planted for aesthetics and erosion control, these species have out-competed native plant species and disrupted native ecosystems, potentially decreasing biodiversity, changing watershed and ecosystem health, and creating a host of ecological changes with accompanying economic effects.

Globally, terrestrial and marine plant invasions are changing ecosystems and the benefits derived from them. Examples abound (see, e.g., Carlton and Geller, 1993; Perrings et al, 2000; Wilcove, 1998). In South Africa alone, there are 161 known invasive introduced plant pests. The country's Working for Water program seeks to remove invasive trees that are reducing the country's water supply, with a current budget of 480 million Rand per annum (~$66 million USD) (Working for Water Programme, 2003). 169,144 hectares were cleared from its inception in 1995 through 2001, mainly of introduced plants from Australia, particularly Black Wattle (*Acacia mearnsii*), *Pinus spp*., and *Eucalyptus spp*. Partial clearing and rehabilitation occurred on almost 200,000 additional hectares. (Haigh, 2001). Hydrilla (*Hydrilla verticillata*) and water hyacinth (*Eichhornia crassipes*) have become expensive nuisance species almost everywhere they have appeared, from native regions of Africa and South America respectively to Europe and the Americas. While local and national governments fret about the increasing costs of preventing and controlling newly arrived pests, a global effort to identify invaders, warn about damages, and disseminate the best practices for prevention and control is underway (The Global Invasive Species Programme (GISP), 2003).

It is impossible to calculate the total sum that has been spent on controlling such invasives by national and local government entities as well as private citizens, but the costs continue to escalate. Though the volume of purposeful and accidental movement of live plant materials that do not change ecosystems far outweigh the numbers of species' introductions that cause damages (Williamson, 2000), a single species may wreak unexpected and substantial damages.

In hindsight, prevention of the invasion seems preferable for society as a whole. The qualities that make the plants successful invaders, particularly hardiness and a high reproduction rate, often make them impossible to eradicate. Hardiness is also a characteristic, along with novelty, that frequently prompts individuals to introduce new plants. We either must acquiesce to the changes and their costs, or fight a continuous, costly battle against the further spread.

Currently, the State of Hawaii is at a critical turning point with respect to one such invasive species, *Miconia calvescens,* a member of the Melastomataceae family. This invader from Central America was purposefully introduced to Hawaii; planted in a handful of back yards and arboretums four decades ago, it has been spreading with increasing rapidity on the islands of Maui and Hawaii. It is also present on Kauai and Oahu, though it has not claimed significant acreage in either location. A model of its potential expansion is available through comparison with Tahiti, where dense, monotypic stands of the tree now cover 65% or more of the main island of Tahiti after a single specimen was introduced to the Papeari Botanical Garden in 1937 (Medeiros, Loope et al. 1997). Miconia has earned itself descriptors like the "green cancer" of Tahiti and the "purple plague" of Hawaii. In its home range, expert botanists have reported that it is "not a particularly common species," and "never…occurs in monospecific formations" (Meyer 1998). Nineteenth and early twentieth century reports of horticultural efforts to grow Miconia, or velvet tree, suggested the plant was difficult to cultivate (Meyer 1998). This inability to predict the effects of an introduction into a foreign ecosystem increases the complexity of the problems facing policy makers and suggests even extensive research before purposeful introductions will not catch every potential invader.

Due to a combination of scientific, political and economic concerns, prevention efforts did not succeed in catching this threat to native ecosystems, biodiversity, and Hawaiian watersheds. Incomplete information and uncertainty, myopic private decisions about trade, and limited public resources for research, education, and monitoring each contribute to such an invasion.

With regard to incomplete information, even the most informed scientists are not able to predict with certainty how a species will interact with a new environment. In the case of *M. calvescens*, the explosive growth in Tahiti was not obvious until aided by two successive hurricanes in the 1980s, at which point the ornamental plant was already established in Hawaii.

Clearly poor information and uncertainty will lead to an inefficient level of political interference in trade. Had a conservative policy, banning, for example, all *Miconia spp.* been put into place before *M. calvescens* was declared a noxious weed in 1992, potential economic benefits might have been missed from one member of the family or another. Even relatively recent advice from mainstream texts (Graf 1974; Graf 1986; McMillan, Barlow et al. 1991)[1] recommends that *M. calvescens* be planted (Meyer 1998). Conservative policies are expected to catch a fraction of invaders while eliminating considerable potential gain. Imagine, for example, the economic differences there would be today if America's European settlers had never introduced wheat. A more liberal policy, on the other hand, that allows species to enter unless they are a known pest elsewhere, will introduce species without adequate assessment of the potentially costly risks for the new environment, leaving one with species along the lines of *Miconia calvescens,*

An additional political concern is that the agencies responsible for intercepting invasive species often have overlapping and even conflicting goals or management mandates. The federal Animal and Plant Health Inspection Service (APHIS) is the

[1] Though out of print, these texts remain listed for sale and highly rated at Amazon.com, for example.

primary line of defense, but the agency must coordinate with state and local governments and private interests, particularly private nurseries. Historically the agency has been poorly funded, and funding has not grown as rapidly as trade and vectors for introduction. Its stature as an agency may be growing; in the summer of 2002 it was placed under Homeland Security, highlighting the serious potential consequences of unwanted introductions. This move, however, may allocate funds more aggressively toward pathogens and other direct biological threats to human health, which will require that the private or state and local governments increase their vigilance regarding less immediately dire introductions like attractive, healthy plants.

Finally, economic benefits of the trade that purposeful introductions create and accidental introductions accompany may be substantial, and reducing the probability of an invasion may significantly reduce these economic benefits. As a typical example, though not the case for *Miconia*, many invasive species in Hawaii and North America in general gained their footholds through plantings at the recommendation of the Natural Resources Conservation Service (NRCS) for quick soil stability or range feed.

With these barriers to comprehensive prevention strategies, we must consider policy aimed at preventing and controlling invasions within the context of the damages that may occur if a species does arrive. The remainder of this paper is concerned with the economic consequences of an invasive species that has established itself as an agent of ecological change in a new habitat. In section 2, we investigate the tools with which economists might measure the damages from an invasive species. Section 3 discusses control policies to combat an invasion from an economic perspective. Section 4 applies the techniques and policies described in sections 2 and 3 to the case of *Miconia* *calvescens*, an aggressive, invasive tree in Hawaii and other Pacific Islands. In section 5 we conclude and investigate the broader lessons we are learning from the efforts to control *Miconia calvescens.*

2. Measuring the Damages from Invasive Species

2.1. Defining Invasive Species in Economic Terms

While economic goods and services increase the well being of society, economic "bads" reduce it. Invasive species reduce the productive capacity of nature for both direct and indirect economic goods. Thus we consider invasive species as an economic bad. Direct goods and services from nature might include water resources, timber resources, or other consumptive uses. Indirect services might include purification of water supplies, or pollination, or other natural processes that enhance nature's provision of social welfare. We consider the stock of environmental and resource assets as natural capital, from which we draw benefits, either as a flow of services provided by the stock or in the form of stock depletion.

In most cases pertaining to biological assets, stocks are renewable and we speak of flows derived from the natural capital available in any given time period. For example, harvesting from fisheries or animal herds can continue indefinitely as long as the remaining stock is sufficiently healthy and large to reproduce at a rate greater than or equal to the harvest rate.[2]

This same biological growth that provides such bounties from nature is the chief complication in controlling invasive species. Since invasive species will continue to reproduce if they are not entirely eradicated, the problem also has a temporal component

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[2] For more on optimal harvesting and use of renewable assets, see Clark, C. (1990). Bioeconomics..

in which today's actions affect tomorrow's welfare. Essentially, in each time period we must consider whether the benefit of removing a marginal unit of the invasive species is greater than the cost of doing so. At the same time, we must consider the present value of the costs and benefits of that action into the future. This requires two steps for economic analysis.

The first step is to measure the marginal benefits and costs of controlling an invasive species. Economists prefer to measure benefits and costs using market interactions, but in the case of natural capital they are rarely available, as for example there is no direct market for scenic beauty. Techniques for valuing such goods fall into one of two broad categories, direct or indirect. The second step, discussed below, merges the economic valuations with models of ecosystem functioning to predict outcomes across time.

2.2. Methods for valuing Economic Damages from Invasive Species

There are two types of valuation techniques that can be used to elicit values for non-market goods such as forest quality. In both cases, the goal is to determine the value that the users gain from the asset. The value for a good is measured by the social welfare generated by its consumption. This is the price a consumer is willing to pay net of the cost to society of producing it, aggregated across consumers of the good. With a market good like syrup, fluctuations in prices and quantities purchased and offered for sale allow economists to trace out demand and supply curves. The demand curve represents what consumers are willing to pay, while the supply curve represents the producers' costs, and aggregation of social welfare is straightforward.

With non-market goods, the lack of price transactions requires one to estimate the consumers' willingness to pay. *Direct* techniques do this by asking consumers "directly", in some sort of survey or referendum format. *Indirect* techniques attempt to break down composite market goods that include non-market amenities in order to value each of the components.

2.2.1. Direct techniques

Direct techniques include the Contingent Valuation Method (CVM) and Conjoint Analysis. These related methodologies survey individuals to identify values for amenity goods like scenic beauty or species diversity.[3]

The techniques are controversial because surveying may not reflect the true actions individuals would take if price tags were attached. Examples of important potential biases include informational mistakes, where respondents are not sufficiently aware of the benefits or costs of the decision they are making, and hypothetical bias, where individuals lie to promote an agenda that they would not support with financial decisions, because there is no commitment. For these reasons, direct techniques have faced great skepticism. However, direct techniques are the only tools economists have to value the benefits that individuals get from a good that is not consumed in any way. For example, an individual might never visit Hawaii, but may be willing to pay for its restoration to a condition containing increased numbers of native species. One's

[3] For further discussion of direct valuation techniques, see Diamond, P. A. and J. A. Hausman (1994). "Contingent Valuation: Is Some Number Better than No Number?" Journal of Economic Perspectives **8**(4): 45-64.
Hanemann, W. M. (Ibid.). "Valuing the Environment through Contingent Valuation.": 19-43.
Portney, P. R. (Ibid.). "The Contingent Valuation Debate: Why Economists Should Care.": 3-17.
Kolstad, C. D. (1999). Environmental Economics. New York, Oxford University Press.

motivation might be ethical, or to leave the asset as a bequest for future generations. Indirect techniques cannot capture such values.

In the case of invasive plant species, direct techniques are likely to be necessary for valuation of any loss in biodiversity. For Miconia, we generate a stream of values for biodiversity loss through contingent valuation estimates of willingness to pay to prevent the loss of endangered species.

2.2.2. Indirect techniques

Indirect techniques tie a market good to a non-market good and "portion out" the value of the market good to its composite characteristics, including the non-market good. As a simplified example, if a plant in a home provides decoration and indoor air quality, a cross-section of data about plants' prices, decorative qualities, and air purification qualities can be used to estimate the part of the price that reflects each attribute.[4]

Indirect techniques are also used frequently to value recreational opportunities. Since recreation often requires expenditure on equipment and travel to the site, data collection on these expenditures then can be used to compare recreational facilities and value them.

Indirect techniques can be used to elicit values for indirect ecosystem services where the final good has direct market value. For example, if an invasive plant species displaces a marketable product like wheat or rangeland, the change in value can be calculated through the change in wheat or rangeland prices that should accompany the greater scarcity and higher costs of production.

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[4] For more on indirect techniques, see Ibid. as well as Freeman, A. Myrick (2003) The Measurement of Environmental and Resource Values, 2nd Ed. Washington DC: Resources for the Future.

The purposeful introduction of a horticultural species may have benefits as well as costs. These benefits must be included in the assessment of the optimal level of control efforts that should be undertaken. Since these species are often purchased, we will have some market prices to guide the valuation of benefits.

Because horticultural species have positive economic value, and their damage levels can only be anticipated with uncertainty in most cases, prevention policies should focus on creating liability rules that create responsibility for any potential damages. If this is done, then the individual decision to introduce a potentially invasive species carries the full expected cost to society, and appropriate resource allocations will be made. If liability is limited or absent, then there is discordance between the incentives of the importer and the incentives to society. Here, we look at the incentives of society, presuming either that there is appropriate liability, or that, in its absence, society will have to make the same choices, but spend public funds on minimizing the costs from an invasion.

2.3. Incorporating biological growth into economic valuation

The second step in analyzing the economic threat from invasives is to combine the valuation information with the biological growth patterns of the species over time in order to integrate the temporal considerations into the problem. For this economists have turned to optimal control theory (Perrings, Williamson et al. 2000; Eiswerth and Johnson 2002). Optimal control theory allows economists to trace a path through time for human intervention control efforts that maximizes the present value of expected net benefits. This technique is also used to determine optimal harvest rates for beneficial species (Clark 1990).

In many cases, an equilibrium level of effort is reached such that the present value of the marginal cost of removing another species today (bad or good) is just equal to the marginal benefit of doing so, and the present value of net benefits are equal across time. Thus social welfare is maximized because there are no resources that could be more effectively used in another period of time.

In the case of an invading species, equilibrium effort levels that maximize social welfare may be less attainable. For an internal solution in which neither is eradication achieved nor is removal entirely abandoned, the equilibrium would require meeting a condition where the social discount rate (society's willingness to trade value today for value delayed until tomorrow) equals the rate of change in the growth rate of the invader plus the ratio of the change in the marginal benefits (net marginal damages) from an increased population to the marginal costs of an increased population. Current and future stochastic changes need to be incorporated as well. Economic valuation techniques generally cannot foretell future costs and damages with certainty, but stochastic dynamic optimization provides an excellent framework for both illustrating the potential outcomes and incorporating what is known about statistical probabilities, and economic and biological factors into a useful policy tool. The better the scientific research, the better the economic estimates.

In the case of invasive species, dynamic optimization illustrates clearly that the most profitable control strategy may neither recommend the entire eradication of the invasive population, nor accept assimilation. A variety of potential interesting permutations are possible. These include both steady state levels of the invasion as well as cases where the optimal strategy is to treat the invasion in "pulses," or pushes toward eradication followed by periods of unchecked re-growth.

Existence of an internal solution, i.e. one where a steady state level of the invasion is achieved through annual control expenditures, will depend on society's rate of time preference, the biological rate of reproduction, the value of benefits and damages and the costs of control. A situation with low discount rates and low biological reproduction, as well as high damages and low control costs, is likely to lead to eradication as the welfare maximizing solution, as eradication is economically viable. In other words, the long-term discounted benefits of the removal of the very last Miconia plant will exceed the long-term costs of its removal. On the other hand, the opposite scenario would likely lead to acceptance and accommodation of the new ecosystem. In this case, control expenditures would never produce more benefits than leaving the invasion to nature. That is, money spent on removals will not reduce damages (increase benefits) by as much as the cost of their removal, for any amount of removals.

For most economic problems, internal solutions are not only nice mathematical outcomes, but also make a great deal of sense. This is because in most cases, the costs of the action grow as the action draws on more resources or resources that are less suitable for that use, while at the same time the benefits produced lessen. This may or may not be true for *Miconia calvescens* in Hawaii, and for other invasive species, due to the biological characteristics of invasions. We investigate the benefits and costs of an invasion of *Miconia* to determine which of these scenarios is most likely.

3. *Miconia calvescens* in Hawaii

3.1. Overview

Miconia calvescens (Melastomataceae), commonly known as purple velvet leaf, is a beautiful ornamental plant from Central America that has been cultivated in Europe and Asia since the mid 19th century (Medeiros, Loope et al. 1997). It is a tree that has a height range of 4-15 m, and large, strongly trinerved leaves (to 80 cm in length) that are dark green on top and deep purple below (Loope 1997). *M. calvescens* reached Hawaii in the late 1950s and early 1960s. In 1961, it was planted at Lyon Arboretum in the Manoa Valley of Oahu. On the Big Island of Hawaii, the first known evidence of Miconia was in North Hilo, at a private estate, and soon after it was also introduced at a botanical garden in Onomea, near Hilo (Medeiros, Loope et al. 1997).

Though there were other early plantings in the state, these two had the best growing conditions, receiving precipitation above the 1800-2000 mm threshold suggested by its growth in Tahiti and Ecuador (Loope 1997). From these two initial plantings, volunteer seedlings began to spread and *Miconia* became established in Hawaii.

On the island of Hawaii, the plant is now present to some degree on over 250,000 acres, ranging from dense, monotypic stands to places where there is a single tree over a 500-acre range (Tavares and Santos 2002). The botanical garden at Onomea and a 3000 acre surrounding region is referred to as the "core" in South Hilo, though several satellite cores, initiated by human, wind and bird dispersal, exist on the island[5]. The estimated potential expansion range, without human intervention, is 500,000 acres.

Maui, which has aggressively targeted the weed since 1991[6], has limited its expansion significantly, though eradication eludes the island. On both Maui and Hawaii, *Miconia* trees have been located in terrain that is inaccessible except by expensive helicopter

[5] Human dispersal has begun an incipient population in West Hawaii.
[6] From 1991 to 1993, over 20,000 *Miconia* plants were removed from West Hawaii Loope, L. L. (1997). HNIS Report for Miconia Calvescens. Honolulu, USGS/BRD**:** 8..

access. On Maui, the determination of net benefits regarding the costs of these last trees will be crucial to determining the ongoing control policies for the island. On Oahu and Kauai, populations remain limited. Figures 1-4, created by the Hawaiian Ecosystems At Risk Project (HEAR) show the most recent mapped information on the locations of *Miconia* on these four islands.

On Kauai, it is possible that there are no remaining trees of reproductive age (Conant and Nagai 1998). If true, this finding confirms the large savings that can occur from early detection and eradication. The plant was not introduced to Kauai until the late 1980s (Medeiros, Loope et al. 1998), and by 1991 was recognized as a state nuisance. If, through this relatively quick identification of the threat for Kauai, the island continues to be spared the extensive control costs mounting on Maui and Hawaii, we will witness directly the cost-effectiveness of immediate action.

3.2. Management Techniques and costs

In the absence of early detection and eradication, control mechanisms include many overlapping activities. The main activities are:

1. Public awareness campaigns for identification and removal, particularly for nurseries and other dispersal mechanisms
2. Public spending on identification of and direct removal of plants; includes spending on research for the most efficient removal techniques
3. Public spending on detection and removal of new growth for the duration of the seed bank

4. Acquisition of permission or compliance from private landowners for removal
5. Development of biological control agents

Each of these should be pursued in a manner that considers the tradeoffs in long-term net economic benefit. In addition, with limited budgets the net benefits of control for other invasive species must also be evaluated; optimal spending will not only equate the expected marginal costs and benefits for removal of an individual species but also across invaders. For example, *Miconia* is just one of several threats to Hawaiian ecosystems from the Melastoma family alone. Other Melastomae already present and spreading rapidly in Hawaii are *Clidemia hirta*, *Tibouchina urvilleana*, *Tibouchina herbacea*, and *Oxyspora paniculata*.

Several of these species are more widespread than *Miconia* at present*; Clidemia hirta,* for one, has become uncontrollable on Oahu (www.hear.org, 2003), but might be stopped from spreading further and out-competing native species on neighbor islands. The benefits of doing so must be weighed not just against the direct costs of control for *Clidemia*, but also against the opportunity cost, or cost of the most profitable alternative use of the funds. If a dollar spent on *Miconia calvescens* reduces damages more than a dollar spent on *Clidemia*, then that dollar should be spent on *Miconia*. As the ratio of *Miconia* infestation to *Clidemia* infestation changes through natural rates of growth and the actions of human intervention, these marginal benefits will also change and must be re-evaluated as necessary.

3.3. Benefits and Costs from *Miconia calvescens*

3.3.1. Creating a Benefit function

Using a combination of the techniques above, the expected net benefits of an introduced horticultural species can be estimated as a function of the species' population in an area. For most invasive species, expected net benefits will be a decreasing, and potentially negative, function of species population, as small levels of invasion are less likely to cause damages like biodiversity loss or change ecosystem functions than large levels of invasion, and benefits of increasing population are unlikely to increase faster than damages:

$$B_t = b(n_t) - \omega \cdot d(n_t)$$

where B_t are the expected net benefits accrued at time *t* from population n_t, comprised of the benefits, $b(n_t)$, net the expected damages, $\omega \cdot d(n_t)$, where, ω is the (constant, species-specific) probability of damages occurring, and reflects the uncertainty of our scientific knowledge, and $d(n_t)$ is the amount of damages, if those damages occur, for a population of n_t. These net benefits will be negative and reduce net social welfare as long as the expected damages outweigh the expected benefits.[7] When this occurs, control costs to reduce the population and increase net benefits may be advisable.

[7] Note that many individuals use GDP as a measure of comparative social well-being. Most economists have a broader definition of social welfare, encompassing returns from and losses to natural capital and spillover effects to individuals and the environment from the production of goods and services, among other non-quantified forms of value. GDP's convenience as a measure of flows in the economy has in the past leant it credibility as a measure of progress. This no longer holds in general. For further discussion of such full-income accounting, see Ellis, G. M. and A. C. Fisher (1987). "Valuing the environment as input." Journal of Environmental Management **25**: 149-56.
Grambsch, A. E., R. G. Michaels, et al. (1993). Taking Stock of Nature: Environmental Accounting for Chesapeake Bay. Toward Improved Accounting for the Environment. E. Lutz. Washington, DC, The World Bank**:** 184-197.
Costanza, R., R. d'Arge, et al. (1997). "The value of the world's ecosystem services and natural capital." Nature **387**(6630): 253-261.

A great deal of uncertainty accompanies the following measures of benefits and damages. These measures should be taken as a first approximation and as an illustration of the technique. The lessons of the techniques are applicable even with large variations in the actual values and specifications of the functional forms that tie the economic values to the plant population.

3.3.2. Benefits

Benefits from *Miconia calvescens* are assumed to be only aesthetic. None of its biological characteristics recommends it for any ecosystem services. Since there are no other known benefits from consumption, these aesthetic benefits for an individual purposefully planting the tree are valued at the price of a tree from a nursery. Since the plant is now no longer sold as an ornamental in North America, no price information is available. We average the price of several subtropical ornamental trees across different stages of development to get an estimated value of $15 per tree. The external benefits accrued to individuals who view the plants but have not purchased them are estimated by assuming that individuals visiting the three botanical gardens in Hawaii that planted the species in the 1960s paid part of the fee to enjoy the Miconia.

The Hawaii Tropical Botanical garden (Hawaii) received 56,000 visitors in 1995, with an adult entrance fee of $15. The garden has over 2000 species, and Miconia dates back to 1959. If on average each visitor received an equal portion of her entrance fee's satisfaction from Miconia as any other plant, each visitor's average value for viewing the tree is $0.0075. The estimated aesthetic value of this Miconia for 1995 is thus $420.

Weitzman, M. L. and K.-G. Lofgren (1997). "On the Welfare Significance of Green Accounting as Taught by Parable." Journal of Environmental Economics and Management **32**: 139-153..

The Hawaii Department of Business and Economic Development and Tourism (DBEDT) reports that there were 31,599 visitors to Wahiawa Botanical Gardens (Oahu) in 1994, with the date of introduction of 1961. The garden's website mentions 28 species for the 27 acre garden, not including Miconia, on their web tour. Admission is free, however, so the value for these visitors will need to be estimated through other means. If the per-species value is the same as that found for the Hawaii Tropical Botanical Garden ($0.0075), then the value of Miconia for the Gardens' 1994 visitors was $237 in 1995 dollars.

Lyon Arboretum (Oahu) received approximately 29,000 visitors in 1994. The arboretum is home to over 7500 species and has no admission fee, but the organization does sell a number of annual memberships at prices ranging from $10 to $100. We again adopt the $0.0075 figure calculated above, and 1994 benefits are thus calculated at $217.50. Miconia was introduced to the Lyon Arboretum in 1964.

Miconia was introduced on Maui in the early 1970's at Helani Gardens in Hana. Tourism data is not available for this site. If as many as 1% of the one million annual tourists to Maui visited the gardens, annual visitors would number 10,000.[8] The estimated annual benefit for 1994 is thus $75.

We estimate that demand for these tourist attractions grew at the same rate as overall Hawaii tourism growth, which averaged 4.5% from 1960-1990, and 2% from 1990-2001, so that the total number of visitors to these gardens since the arrival of Miconia is estimated at 3,344,454. The total value, in 1995 dollars, of the viewing of Miconia in Hawaii at botanical gardens is estimated at approximately $25,000. The total benefits

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[8] Hana, Maui is remote and difficult to access for tourists, and visitor numbers are expected to be significantly lower than at the centrally located Oahu locations.

might increase as populations of Miconia spread spatially and more individuals view the plants, but the marginal benefits, or benefits accrued from another tree, are likely to diminish with ready viewing access and a general consumer dislike for a monotonous landscape cover that would develop with dense monotypic stands. We estimate that the per-person benefit of viewing at least one tree in a botanical garden is \$0.0075, and set this as the maximum per-tree benefit. We assume that marginal benefits are declining and choose a simple functional form for total benefits of:

$$B(n) = 0.0075 \cdot n^{1/2}.$$

3.3.3. Damages

Damages from Miconia are extremely uncertain because they involve indirect ecosystem services as well as non-market goods like biodiversity. They will also differ from island to island as different resources are threatened. The characteristics of the species that have branded it a nuisance species have three major dimensions for potential damages. First, it is an aggressive invader that appears to invade healthy native forest with success (Meyer 1998). Native forest and its biodiversity are replaced with dense, monotypic stands of Miconia that shade out all undergrowth and may change soil chemistry. Second, the seed bank develops quickly once the tree reaches flowering and fruiting size of 4-5 meters (4 cm dbh; at least 4-5 years of age) as a single tree can flower and fruit 2-3 times a year, with a typical fruiting event producing 3 million seeds (Loope 1997). Third, the seed bank has some longevity. It is known to last over 2 years, and may be as long as 6 years (Loope 1997; D. Nelson personal communication).   Canopy openings are quickly taken advantage of by new seedlings. Wind dispersal appears most prevalent, though birds are also dispersal agents.

With sufficient rainfall (greater than 1800 mm / yr) and canopy openings, a single specimen may, in 5-15 years, start a stand that covers several hundred acres. Conditions appear to have been favorable at Onomea, near Hilo, Hawaii, as a five-month removal effort from December 2001 to April 2002 led to the removal of 5980 flowers or fruits, 132,561 seedlings, 118,863 saplings, and 42,813 trees, on only 2788.3 acres. This population is believed to have originated with a private planting (Shipman estate) and a planting at the Hawaii Tropical Botanical Garden in 1959 (Meyer 1998). Figure 5 shows rainfall contours for annual average rainfall greater than 1800 mm, indicating the potential expansion range for *Miconia*.

Hawaii is home to a great percentage of the United States' and the world's endangered species. Changes in forest composition as described may threaten endangered plant species, bird species, and invertebrate species in particular. Hawaii's evolutionary isolation has led to much adaptive radiation of species, where a single ancestor has generated a set of species that each depend on new and different types of habitat; the state is considered to house the most unique and diverse snail population in the world despite the limitation that only 15% of snail families are represented (Asquith 1995). The wet, higher elevations of Maui and Hawaii contain most of the only healthy remaining native forest supporting such diversity in the state, and are now threatened by *Miconia*. For example, the upper Kipahulu Valley on Maui is a conservation district reserve containing stands of Ohia (*Metrosideros polymorphata*) and Koa (*Acacia koa*) that are the primary habitat for rare native Hawaiian birds and insects, and *Miconia* has been discovered in the lower valley (Staff 2001).

In the federal register listing materials for the endangered Elepaio (*Chasiempis sandwichensis*) bird on Oahu, the main justification for protection is based on the bird's reliance on the current forest structure (see Service 2001 for example). Since Miconia poses a significant threat to that structure, the plant is listed directly as one of the concerns for the bird's survival. A set of studies indicates that, on average, each household would be willing to pay $31 (95% confidence interval of $16.66-$48.92) per bird species per year to keep a species from extinction (Loomis and White 1996). This amounts to an annual value for Hawaii state residents of $12.4 million per avian species preserved. From the confidence interval, we assume the damages would lie between $6.7m and $19.6m.

Economic theory and research predicts that households will value invertebrates and plants at lower levels (Loomis and White 1996), and that non-residents will also have lower aggregate values. Thus, as an approximation of the potential damages from *Miconia*, we estimate the full threat of loss in biodiversity on all islands to be equivalent to a loss of half the endangered bird species, or $103-303 million per year (value per bird * 31 birds * 0.5). Note this is expected to be a conservative estimate in that it only includes benefits to the state residents, and that even though virtually all of the 31 species live in the same pristine habitat that the tree is likely to invade, we count at most half of the birds as threatened by the potential invasion. The uncertainty associated with this estimate is particularly high; we create a range of estimates using the 95% confidence interval to underscore a portion of this uncertainty.

Additional damages to watershed functions are also expected from dense stands of *Miconia*. The hydrological properties of Miconia suggest that there may be a significant change in the water balance, with an increase in runoff and a potential reduction in groundwater recharge.[9] Groundwater recharge is of significant consequence for Oahu but less important for Maui, Hawaii, and Kauai, which generate less of their fresh water supply from ground water. Estimates of potential expected losses from an invasion of *Miconia* on Oahu to groundwater recharge may be as high as $137 million per year (Kaiser and Roumasset 2002). Increased sedimentation will incur surface water quality damages on any infested island; costs for Oahu have been estimated to be at least $4.84 million per year (Kaiser and Roumasset 2000). Extrapolating from this figure to Hawaii, Maui, Molokai and Kauai by susceptible land area as a first approximation, damages for the state could increase approximately tenfold, to $48.4 million per year. If the infestation only takes hold in the highly likely cases of Hawaii, Maui, and Oahu, then we estimate these damages at $33.9 million per year.

If all damages occurred, then, the total damages would range from $273.9m to $488.4m, with an estimated average of $377.4 million per year. Assuming that any one tree should be equally responsible for its portion of damages, ceteris paribus, we determine a per-tree damage rate of $3.77. Total expected damages for any given population are described by the function $3.77*n*.[10] Simulations are also run using the

[9] The particular role of Miconia in groundwater recharge is uncertain; on the one hand, increased runoff suggests there is less water available for recharge, but changes in evapotranspiration rates may counteract this loss.

[10] For simplicity, we assume a uniform distribution function where any tree contributes to the loss equally, given the existing population level, and the cumulative distribution as the probability of total losses for any given population, n, is just n/n_{max}. We assume n_{max} is 100,000,000 plants, based on a density of 100 plants per acre and 1,000,000 potential acres of habitat. The marginal damages from loss of biodiversity and watershed quality are thus $3.77 per tree, and the expected damages for any given population is $3.77*$n$.

upper (damages = \$4.88*n*) and lower (damages = \$2.74*n*) bounds created by the confidence interval on the damage estimates.

3.3.4. Control Costs

Expenditures for controlling the species population causing these damages also reduce social welfare, as these are funds that could be spent on other economic goods. Per-unit control costs are expected to decrease as the population increases. Intuitively, removal of the last unit of an invasive species will be significantly more challenging and expensive than removal of the first unit from a large population. We assume that per-unit control costs are $c(n_t)$, and total costs are:

$$C_t = c(n_t) \cdot x_t , \quad c' \leq 0 ,$$

where x_t is the number of units removed from the population.

Table 1 shows historical costs for Miconia control in Hawaii for various removal efforts. From this data we estimate the exponential function that best fits the data showing costs of removal decrease as a function of population:

$$C_t = 3500 * e^{-7*10^{-8} n_t} \cdot x_t .$$

The expected total net benefit (generally negative) of an invasive species, once it has become established, in any time period *t* is $B_t - C_t$. To make the best policy, control activities should be such that they minimize the present value of the sum of these damages plus control costs accrued over time. For simplicity we assume that control

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

efforts reduce the population after it has provided net benefits (damages) in each time period. Net benefits in any given time period will be:

$$B_t - C_t = 0.0075 \cdot n_t^{0.5} - 3.77 \cdot n_t - 3500 \cdot e^{-7*10^{-8} n_t} \cdot x_t \,.$$

We assume that Miconia will reproduce and expand to its 1,000,000 acre potential range following a straightforward logistic growth curve with growth parameter b=0.3. We choose this parameter to accommodate for the minimum 5-6 year lag that will occur between fruiting and trees of reproduction age. The logistic growth function $g(n) = bn(1 - \frac{n}{n_{\max}})$ determines the expected population for any time period after introduction. The change in population between any two time periods will equal the growth rate net of removals from control activities. These choices of growth rate and functional form appear to calibrate well with the existing data for Hawaii and Tahiti, as the maximum population is reached sometime after 70 years and at 40 years there is a high, and rapidly expanding, population.

3.4. Findings

Using a social discount rate of 3%, we find that eradication is economically preferred to accommodation unless it is attempted in the rapidest periods of expansion, and that even failed eradication, where a single tree is missed and allowed to initiate a new population without any subsequent treatments, is preferred to no attempts at control if initiated at any time but the highest periods of growth. Accommodation, with 1,000,000 acres of *Miconia calvescens* causing an annual stream of damages of $377 million, therefore, is eliminated as an economically rational equilibrium outcome. The present values of damages from accommodation over 50, 100 and 200-year horizons are estimated at about $47 million, $1.74 billion, and $2.36 billion dollar losses in economic welfare respectively. Note that at 50 years, the population is still growing rapidly and maximum annual damages have not been reached. Rows 2 to 4 of Table 2 shows the changes in the estimates that occur if the higher or lower damage estimates are used. Rows 5 to 7 demonstrate the importance of the choice of discount rate. Lowering the discount rate from 3% to 1%, which places a greater weight on benefits and costs accrued in the future, more than doubles the damages in just the 50-year horizon, and increases them almost eight-fold in the 200-year horizon. The variation in size of the estimates is much greater from the change in discount rate than it is from using either the higher or lower bound damage estimates, and highlights the fact that most of the damages are expected in the future. For the rest of the discussion, we use the average damages and a 3% discount rate for ease of exposition.

Successful eradication is extremely profitable compared to accommodation if it occurs early, when populations are small. Benefit-cost ratios are above 100 for populations of *Miconia* up to 1000 plants, and social welfare losses are two to four orders of magnitude smaller, ranging from around $100,000 in losses from the earliest detection to $23 million in the 25th year of policy initiation, with growth at about 14,400 plants. Kauai and Oahu should certainly aim for eradication. Even if they were to fail, leaving a single tree that never receives treatment again, the delay in damages is significant and there is a benefit-cost ratio higher than one for this failed eradication (see Figure 6) at low population levels.

As the growth rate increases, however, this benefit-cost ratio falls and is below one for the fastest periods of growth. In this simulation, that occurs between the 44^{th} and 55^{th} years of invasion, as shown in Figure 6. This corresponds to populations between about 2.4 and 3.6 million plants. The islands of Hawaii and Maui appear close to this period of maximum growth and should take care that a failure of activities now does not discourage the state from taking action later, when growth and expansion has slowed and treatment, if it can take place quickly, may be more cost effective.

A policy that removes the base population but does not capture new growth, which may describe the realities involved in a dormant seed bank that cannot be easily controlled, is also economically preferred if initiated at any time but the highest periods of growth, but less beneficial than either eradication or failed eradication, and there is a shorter window for success (see Figure 6).

The possibility of an internal steady state, where removals in each time period are equal to the growth rate, is not found economically preferred to successful eradication, but is preferred to failed eradication with no follow-up for low population levels. At a population growth level under about 14,400 plants (plant population level of about 48000 plants), annual containment of growth will be more cost effective than failed eradication. At higher growth and population levels, failed eradication becomes preferable as the annual costs of containment exceed the damages from failed eradication. Table 3 considers several possible policies targeted at controlling new growth. Clearly, policies targeting 50% or less of growth simply increase the total losses from the invasion by adding the costs of control without reducing or significantly delaying the damages from the invasion, regardless of the initial population at initiation of the policy. At the lowest population level, policies removing 75% of growth or more do reduce expected net damages, and policies with higher removal rates reduce the damages more. At even a relatively low initial population of 48,000 plants, even annual removal of 75% of growth is not sufficient, and at the maximum growth populations of several million plants, just targeting growth alone will add considerably to costs without reducing damages.

Overall, then, policy structure depends on the existing state of invasion. For low levels of invasion, such as exist on Kauai and Oahu, eradication is preferred as long as it can be quickly accomplished, but containment is a more cost effective strategy than failed eradication for growth levels below about 14,400 plants. If the population has surpassed its maximum growth rates, the loss in social welfare can best be reduced by a pulse-like effort at eradication that drives the population below about 48,000 plants, followed by the above advice for low populations.

Unfortunately, the uncertainty regarding the costs of removing the last plant prevents us from fully defining the optimal policy. It is likely, however, that at least for Hawaii and Maui, which have plants in locations that are extremely difficult to reach, containment will be a more economically efficient policy, as eradication efforts are likely to fail.

4. Lessons and Conclusions

The planting of a handful of *Miconia calvescens* trees in Hawaii for ornamental purposes in the 1960s could result in several billion dollars of loss in welfare to the state. This loss stems from a loss in biodiversity and an increase in runoff and sedimentation as well as a reduction in groundwater recharge that would accompany the change in forest structure Miconia would bring about. The forest will shift from a multi-organism, layered canopy to a densely shaded, monotypic cover with no understory and little water interception.

Policy options include accommodation to this new situation, efforts at containment, or attempted eradication. At low population levels, eradication, if successful, has an extremely high payoff, significantly reducing welfare losses. Containment at a low growth rate (perhaps around 14,400 plants) is economically preferred to delayed expansion, and also more cost effective than failed attempts to drive the population below the level that supports this growth rate.

Greater care must be taken to prevent invasions from occurring; the cost of prevention for introduction through horticultural mechanisms is research and the spread of information to nurseries and other distribution channels, which is expected to be significantly lower than eradication. A push for increased research funding should be undertaken, since prevention is most cost effective when it is based on good information. This also prevents undue losses from missed economic opportunities.

The importance of site-specific analysis should not be underestimated. Recall that a hundred year's worth of casual empiricism regarding Miconia's traits, in locations outside of Pacific Islands, caused a belief that it was difficult to grow.

Though not discussed earlier, biocontrol efforts are being pursued for Miconia in Hawaii. The state has released a trial biocontrol agent, the fungus *Colletotrichum gloeosporioides* f.sp. *miconiae* , that is a defoliating agent. This may help in the containment of Miconia, and may hasten the sprouting of the seed bank as openings in the canopy appear. This could increase the cost effectiveness of control by decreasing the lag time between effective treatments and preventing further new growth.

The risk of any biocontrol agent, however, is that it will become its own invasive pest species with a host of new economic and ecological consequences. In Tahiti, biocontrol has less potential as the islands there are home to a few endangered members of the family Melastomaceae, which would be put at risk from most known agents that could attack *M. calvescens*. Again, prudence advises considerable research and caution.

There is hope that *Miconia calvescens* will not attain its full range in Hawaii, and will not cause the severe economic and ecological damages for which it has the potential, especially for locations where the invasion was noticed early. To reach this hope requires quick and determined action; even large expenditures today, in the millions of dollars, will be more profitable than accepting accommodation. Prevention of additional introductions must become an even more important policy. The introduction of additional ecosystem threats not only causes its own set of damages but also reduces the ability to spend now on reducing the damages from existing invasions.

Figure 1: Distribution map for the Island of Hawaii

Source: Hawaiian Ecosytems at Risk Project (HEAR)

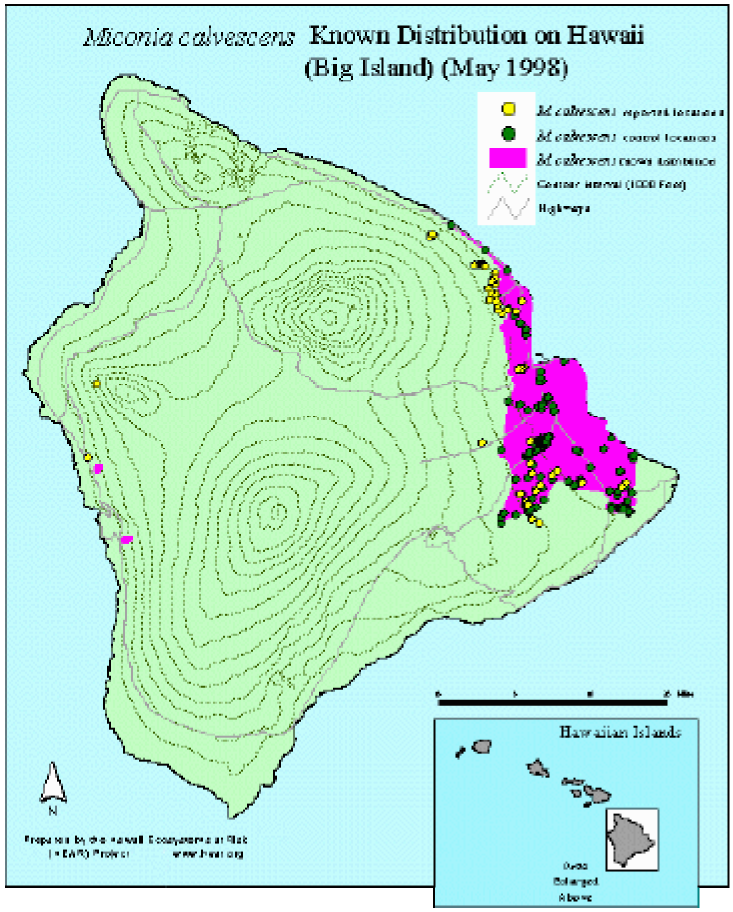


Figure 2: Distribution map for the Island of Maui
Source: Hawaiian Ecosytems at Risk Project (HEAR)

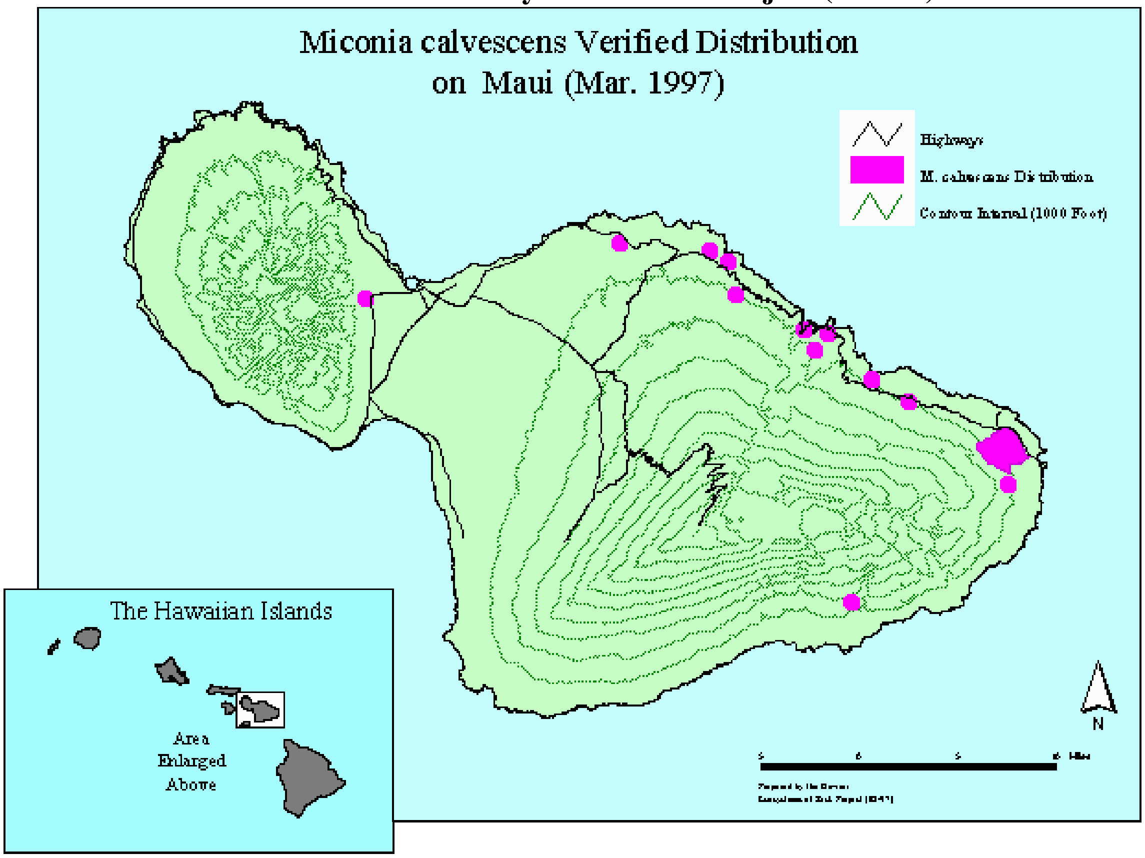


Figure 3: Distribution map for the Island of Oahu
Source: Hawaiian Ecosytems at Risk Project (HEAR)

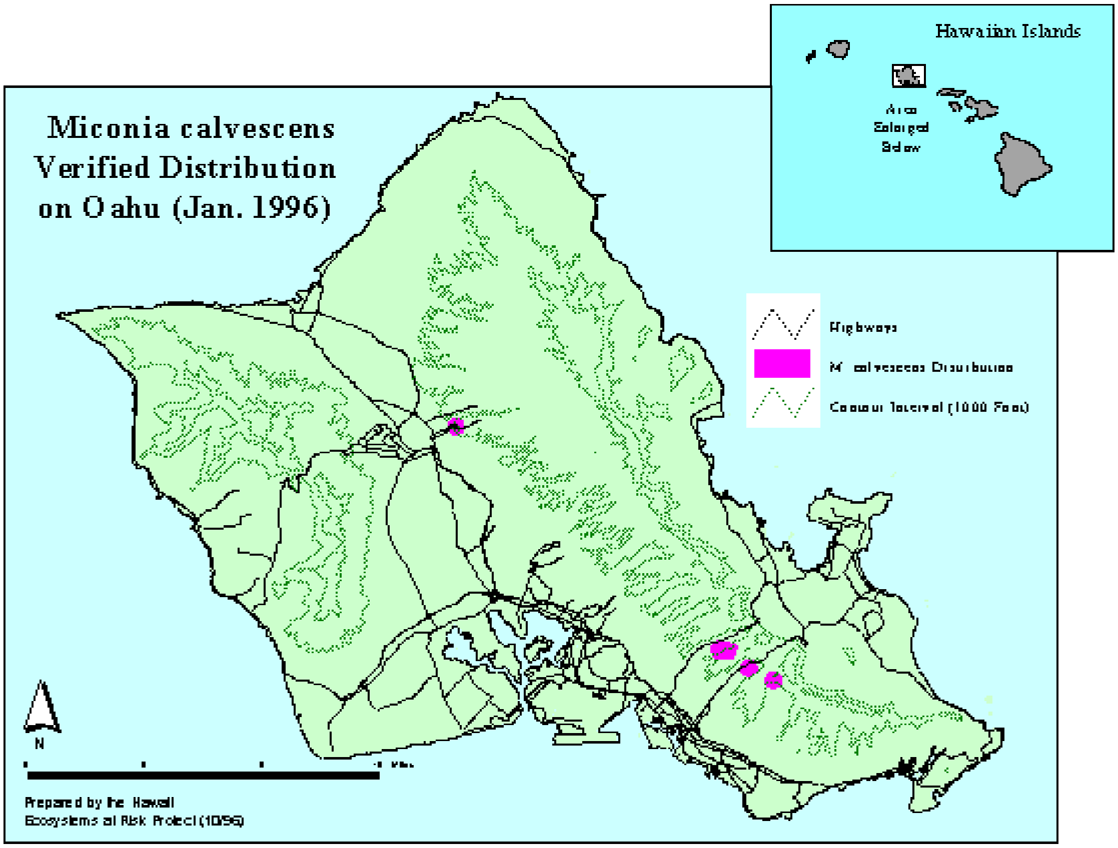


Figure 4: Distribution map for the Island of Kauai
Source: Hawaiian Ecosytems at Risk Project (HEAR)

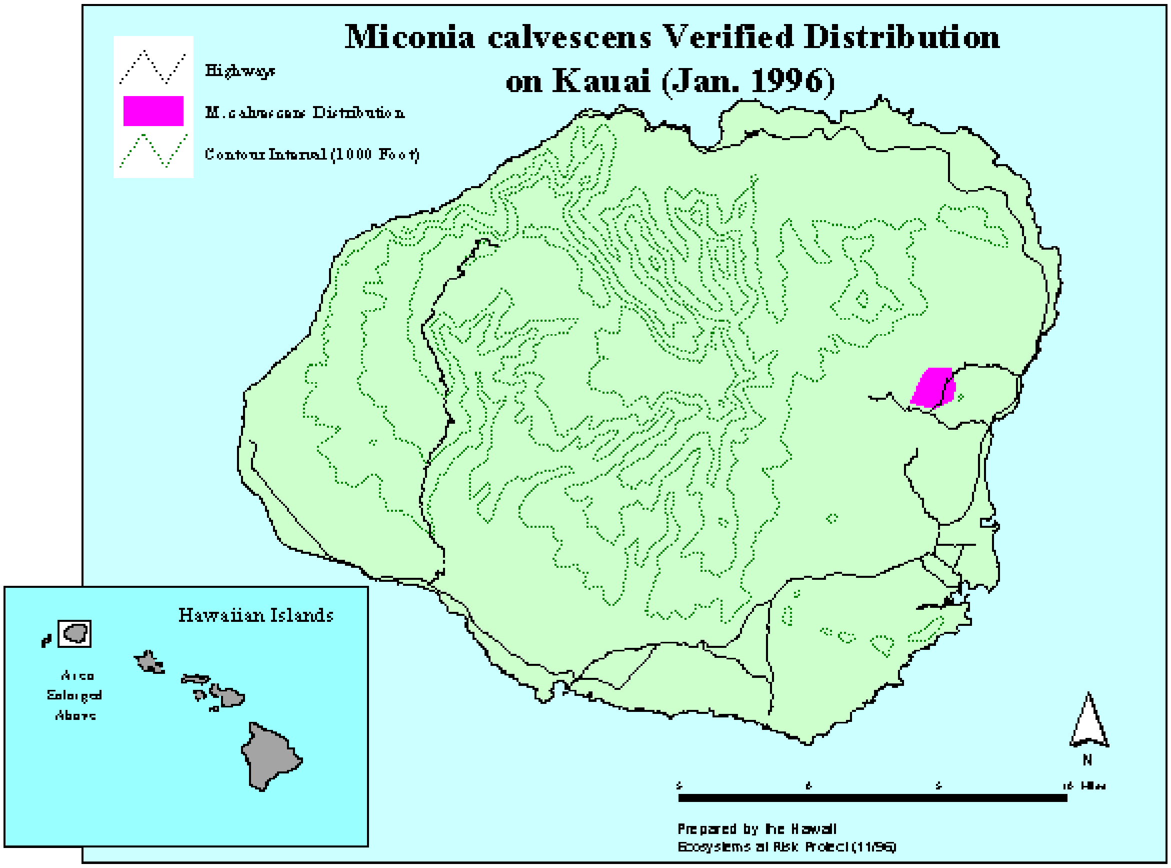


Figure 5: Potential Range for *Miconia Calvescens* in Hawaii

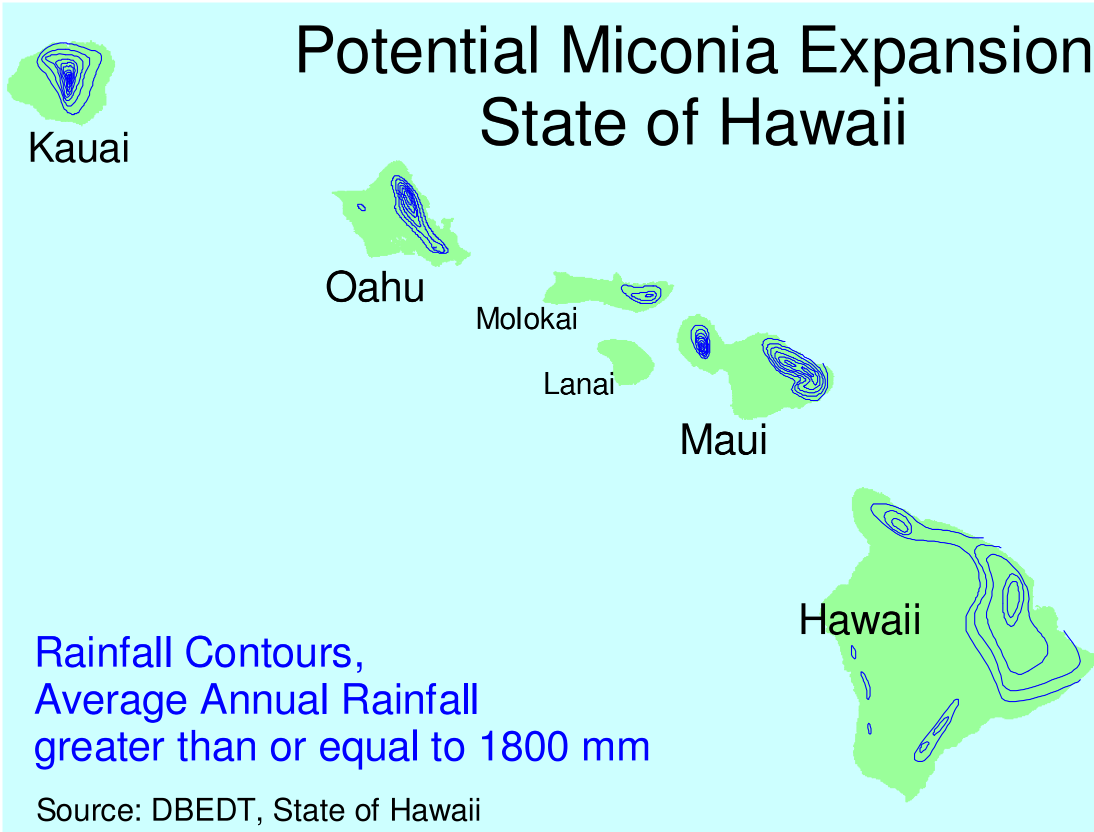


Table 1: Examples of Miconia Control Expenditures

Control Technique	Cost	Level of infestation	Location (Date)
Uprooting and Spraying of individual trees	$9 per plant, average[11]	Heavy (100 plants per acre)	Big Island (2002)
Uprooting and Spraying of individual trees	$12 per plant, average	Moderately heavy (50 plants per acre)	Big Island (2002)
Uprooting and Spraying of individual trees	$100 per plant	Two plants per acre, easy access	Big Island (2002)
Uprooting and Spraying of individual trees	$200 per plant	Single plant per acre, easy access	Big Island (2002)
Helicopter spraying of trees	$850 per plant	Two plants per acre, helicopter access, two treatments	Maui (1997)
Helicopter location and individual removal	Minimum $3500 per plant[12]	Single Plant	Oahu (1996)

[11] A $500,000 make-work project employed 64 persons at $9.96 for 5 months, resulting in destruction of 337,382 plants. Assuming that these areas will need similar treatment at least twice more over the next six years, and adding in costs of equipment and supplies, the resulting average cost per plant is estimated at $9.

[12] 5 hours of helicopter time, labor and supplies. Continued monitoring increases costs further. Repeated helicopter spraying of 110 trees on Maui in 1996 resulting in 72% of trees killed after 1 year and defoliation and reduced fruiting in remaining trees. Defoliation does tend, however, to result in increased seedlings due to canopy openings, at least in the short run Medeiros, A. C., L. L. Loope, et al. (1998). Interagency efforts to combat Miconia calvescens on the island of Maui, Hawai'i. Proceedings of the First Regional Conference on Miconia Control, August 26-29, 1997, Papeete, Tahiti, Gouvernement de Polynésie française/University of Hawaii at Manoa/Centre ORSTROM de Tahiti..

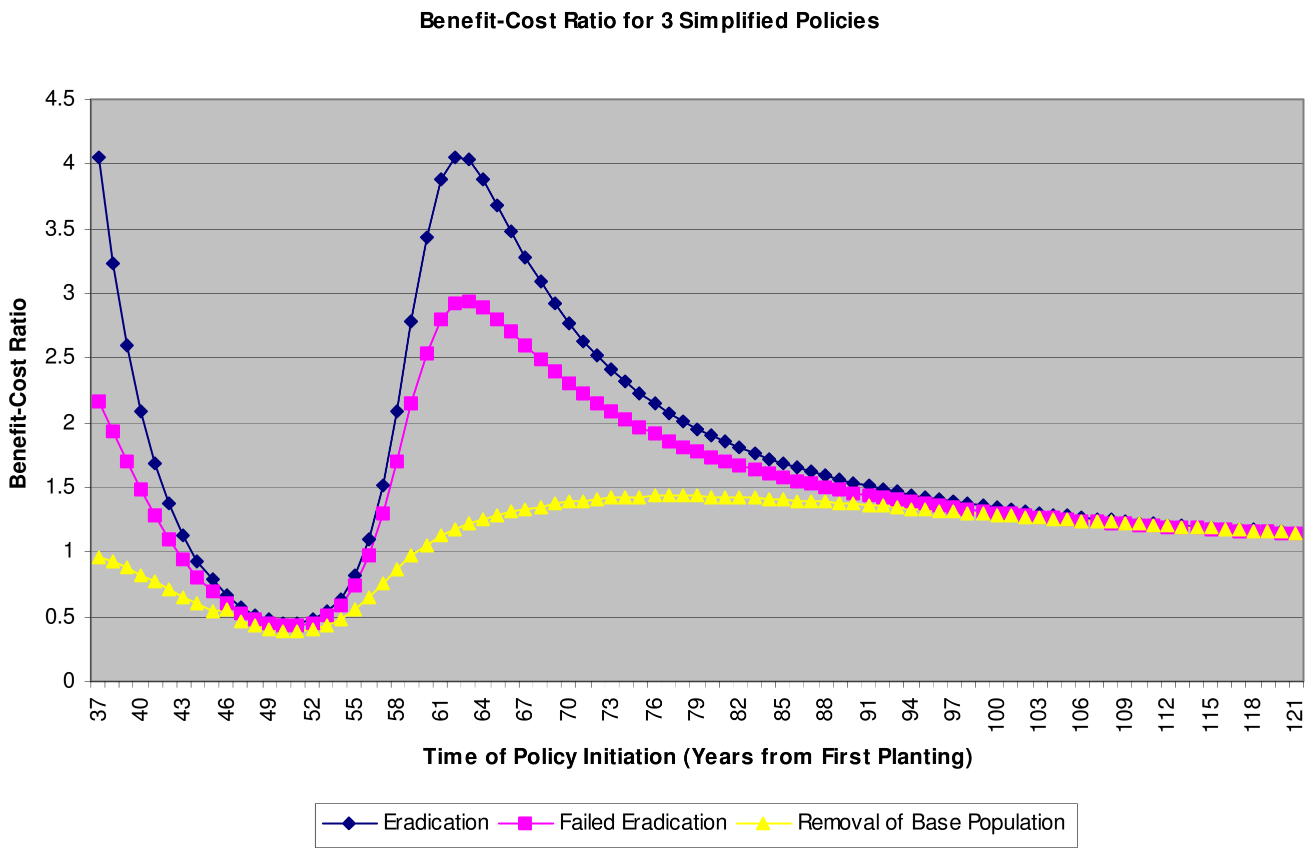
Table 2: Estimated Damages Without Control Expenditures

	Present value of expected damages	Low Estimate	High Estimate
50 years at 3% discounting	$0.05 billion	$0.03 billion	$0.06 billion
100 years at 3% discounting	$1.74 billion	$1.26 billion	$2.25 billion
200 years at 3% discounting	$2.36 billion	$1.72 billion	$3.05 billion
50 years at 1% discounting	$0.12 billion	$0.08 billion	$0.15 billion
100 years at 1% discounting	$7.55 billion	$5.49 billion	$9.77 billion
200 years at 1% discounting	$16.2 billion	$11.8 billion	$21.0 billion

Table 3: Estimated Losses by Population Level and Policy

Removal Policy (% of growth)	Initial Population N=1	Low Population N=48,000	Maximum Growth N=4.6 million
0%	\$2.36 B	\$2.36 B	\$2.36 B
5%	\$2.81 B	\$2.94 B	\$2.37 B
25%	\$4.13 B	\$5.46 B	\$2.44 B
50%	\$3.71 B	\$8.79 B	\$2.59 B
75%	\$0.49 B	\$9.25 B	\$2.97 B
95%	\$257,000	\$1.27 B	\$4.80 B
100%	\$140,000	\$0.73 B	\$7.65 B

Figure 6



Benefit-Cost Ratio for 3 Simplified Policies
Benefit-Cost Ratio
4.5
4
3.5
3
2.5
2
1.5
1
0.5
0
37
40
43
46
49
52
55
58
61
64
67
70
73
76
79
82
85
88
91
94
97
100
103
106
109
112
115
118
121
Time of Policy Initiation (Years from First Planting)
Eradication
Failed Eradication
Removal of Base Population

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