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A bioeconomic model for estimating potential economic damages from a hypothetical Asian beetle introduced via future trade with Cuba

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Abstract
Although present United States (U.S.) policy restricts very nearly all Cuban commercial exports to the U.S., there is potential for the restrictions to be relaxed or perhaps even lifted at some point in the future. In light of the potential increased trade with Cuba, the potential arrival of invasive species such as bark beetles and ambrosia beetles—particularly from Asia via Cuban imports—could represent a serious threat for the southern U.S. forestlands. We develop a bioeconomic model that estimates potential economic damages to southern pine forests caused by the hypothetical introduction of an unknown Asian bark or ambrosia beetle via future trade with Cuba for the study period 2018–2050. We examine individual policies and combinations of them that could be considered in response to this hypothetical situation. Using the pre-revolution Cuban level of imports, the economic damages in absence of "any policy or management action" could reach $2.44 million ($76,250/year). These damages could be reduced to between $469,000 ($14,656/year) and $1.02 million ($31,875/year) if a risk mitigation policy (forest thinning) were implemented. When prevention polices are considered as the baseline, the risk mitigation policy (combined with prevention) is again observed to be the dominant policy. The differences between the various combinations with respect to the prevention policy are relatively minor ranging between $856,000 ($26,750/year) and $1.30 million ($40,635/year). These findings should be interpreted with caution given the limited amount of empirical data and policy assumptions, but they nonetheless can inform policy choices related to trade, pests, and Cuba.

Keywords Bioeconomic modelling · Asian bark/ambrosia beetle · Forests · Damages · Trade · Cuba

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1 Introduction

Before the Cuban revolution (pre-1958), the United States (U.S.) and Cuba were main agricultural partners, with U.S. agricultural exports to Cuba amounting $600 million and U.S. imports from Cuba totaling around $2.2 billion annually (2014 dollars, Zahniser et al. 2015). Trade restrictions imposed on Cuba after 1958 by the U.S. have resulted in limited U.S. agricultural exports without any allowance for imports from Cuba (until the last few years when Cuba has been permitted to export a few small shipments of charcoal into the U.S. market). Since 2000, when the Trade Sanctions and Exports Enhancement Act allowing the sale of agricultural (and selected medicine) to Cuba was passed, U.S. firms have shipped over $5.7 billion worth of agricultural and food products to Cuba. In 2017, the U.S. agricultural exports to Cuba totaled over $215 million (USDA 2019). As diplomatic relations between the US and Cuba became more normalized during the presidency of Barack Obama, there had been an expectation that a normalization of trade between the countries would follow at some point in the future (Penca et al. 2016). Although the current U.S. administration has aimed to roll back some aspects of the Obama administration strategy, trends indicate a gradual progress between both countries' commercial relationships in the medium to long term (Piccone 2017).

A negative implication of international trade is that it provides opportunity for the introduction of new invasive species compromising a country's biosecurity (Levine and D'Antonio 2003; Paini and Yemshanov 2012). An example of a recently arrived lethal invasive species to the U.S. associated with international trade (through wooden packing materials), is the redbay ambrosia beetle (*Xyleborus glabratus*). Native to Asia and detected in 2002 near Savannah, Georgia, it has widely proliferated in several states in the Southern U.S. (MFC 2016); it has led to 90% mortality (300 million trees) of red bays (*Persea borbonia*), resulting from vectoring the fungus that causes laurel wilt disease, and it has caused damage to the avocado (*Persea americana*) industry of around $4.2 million (Hughes et al. 2017). More recently, the invasive shot hole borers *Euwallacea fornicatus*, *Euwallacea whitfordiodendrus*, and *Euwallacea kuroshio*, have been causing severe damage to the avocado industry and native environments in Florida and California after being introduced from Asia (Eskalen et al. 2012; Boland 2016; Carrillo et al. 2016).

In view of the potential for increased trade with Cuba, it is reasonable to consider the introduction of invasive species from Cuba to the U.S. due to several related-factors:

1. Cuba is the largest country in the Caribbean and the second closest country to Florida after the Bahamas. The gradual expansion of agricultural imports from Cuba over time is expected to increase the likelihood of alien species introduction. Tropical forests outside of the U.S. have become a reservoir of potential invasive bark beetles, which are a major concern but about which our knowledge is limited (Susaeta et al. 2016).
2. Recent occurrences of financially relevant quarantine pests in the Caribbean region suggests continuous changes in the agriculture-pest situation in those

countries with which U.S. has trading relationships (Penca et el. 2016). A recent assessment of Cuban bark and ambrosia beetle diversity recorded several new genera and species for the island (Bright 2019; Gomez et al. 2019).

3. The proximity of Cuba to the Southern U.S. requires a prudent amount of caution as an invasive pest pathway (Penca et al. 2016). Notably, the port of Miami in Florida receives over 85% of the live nonnative plant shipments that arrive each year in the U.S. (ISWG 2002).
4. Florida is prone to nonindigenous species invasions with landscape features and environmental conditions that can facilitate the easy spread of invasive species (Simberloff 1997). Furthermore, pests can be carried to Florida not only by traded commodities and by travelers, but also by hurricane winds.
5. Although China is the country that contributes the most invasive alien species to other countries (Turbelin et al. 2017), Florida directly imports very little from Asia. Over 70% of the U.S. imports from Asia are received in West coast ports, Gulf ports and Northeast ports (New York-New Jersey-Norfolk) (JOC 2017).
6. Finally, Upgrades to the Panama Canal and expected re-routing of trade flows through the Caribbean in general and Cuba in particular are a cause for concern, which this paper addresses. Cuba may serve as midway vector for invasive species arriving to the U.S. from third countries. For example, China is Cuba's top trading partner, supplying approximately 30% of Cuban imports in 2016 (The Observatory of Economic Complexity 2019).

Cuba, like China, is a member of the International Plant Protection Convention (IPPC) with a robust system of phytosanitary regulations and enforcement, and has adopted the International Standards for Phytosanitary Measures No. 15 (ISPM 15). While ISPM-15 would decrease the likelihood pest invasion, changes in projected trade through Cuba to the Southeastern US should increase the expected risk to southern pines (Leung et al. 2014). As such, the potential arrival of invasive wood-boring pests—such as bark beetles and ambrosia beetles[1]—particularly from Asia via future imports from Cuba, could represent a serious threat to the southern U.S. forest sector. Covering around 100 million hectares, southern forests support an industry that supplies 16% of the global industrial wood and contributes 5.5% of the jobs and 7.5% of the industrial economic activity of the region (Southern Regional Extension Forestry 2019). They also can sequester around 23% of the regional carbon emissions (Han et al. 2007), provide high levels of biodiversity and wildlife habitats (Kirkman and Jack 2018), and supply 34% of the regional water yield (Lockaby et al. 2013).

[1] From the 500 bark and ambrosia beetle species that occur in the U.S. (Atkinson 2017), 60 are established non-natives (Haack and Rabaglia 2013), and half of those are ambrosia beetles, which are one the most successful group of invaders (Brockerhoff and Liebhold 2017). Nevertheless, pine trees (*Pinus* spp.), the most valuable American tree commodity, has not yet been substantially impacted by an invasive tree-killing borer in the Southeast. The situation might vary in other parts of the U.S. For example, *Sirex noctilio,* which has not yet caused excessive mortality in eastern pines, has become the most abundant woodwasp colonizing pine in northeastern North America (Dodds et al. 2010; Foelker et al. 2016; Gomez et al. 2016).

The cause for concern about invasive bark and ambrosia beetles is that their encounter with a naïve host in the U.S. could amplify their damage potential and trigger an epidemic. It is important to point out that ambrosia beetles are most likely to arrive to the region due to lower requirements for establishment, compared to bark beetles which need large numbers to overcome tree defenses, and find mates through pheromones (Brockerhoff and Liebhold 2017). Once an aggressive wood-borer gets established on undefended host, there are limited management options to prevent its spread and damage (Maner et al. 2013; Hanula and Mayfield 2014). Such a scenario has already been manifested by many recent invaders, including the redbay ambrosia beetle, the emerald ash borer (*Agrilus planipennis*) (Herms and McCullough 2014), and the Asian long-horned beetle (Haack et al. 1997; Lingafelter and Hoebeke 2002). In addition, recent fieldwork in Asia indicates that some species of bark beetles possess characteristics that suggest they are candidates for becoming potential damaging invasive pests in the pine forests of the southeastern U.S by colonizing live tissues of pine trees, e.g. the Old World species *Tomicus yunnanensis, Tomicus destruens, Dendroctonus armandi,* and several *Cryphalus* species (Mendel 1987; Ye 1998; Yang and Youqian 2000; Lieutier et al. 2003; Chen and Tang 2007).

The purpose of this study is to simulate economic damages to public and private loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine forests in Alabama, Florida, and Georgia given the introduction of an unknown Asian bark or ambrosia beetle (hereafter AB) to the US as a result of future imports from Cuba by using a stochastic bioeconomic model. We select loblolly and shortleaf pine because the former is the main commercial species, planted on more than 10 million hectares in the region (Schultz 1997), while the latter has the most extensive geographic range of the southern pines and is a critical species for silvicultural and ecological reasons, particularly in publicly managed lands (Stewart et al. 2016).

Based upon a Markov-chain structure, the probabilistic bioeconomic model evaluates the efficiency of AB management policy options in terms of net costs associated with each policy. International trade data is linked with empirical invasive species interception (monitoring and detection) data to estimate an establishment rate for exotic ambrosia and bark beetles originating from Asia (Koch et al. 2011). This rate is then adjusted downwards to model the probability that the AB is introduced from Cuba, which is assumed to be the intermediate transfer pathway to arrival in the U.S. A stand-level, event-based risk model for loblolly pine[2] (Susaeta et al. 2016) and empirical data from national forests (Chabreck et al. 2013; Meeker 2017) provide the underlying per hectare forest and timber values. We use research on the enormous destruction caused by the southern pine beetle (SPB; *Dendroctonus frontalis*) and mountain pine beetle (MPB; *Dendroctonus ponderosae*), past and present, as a guide for the development of a plausible

[2] We note the distinction between our model and the one presented by Susaeta et al. (2016), whose model relies on an extended version of the Faustmann framework developed by Reed (1984), and only determines the net present value of perpetual forest rotations and the optimal harvest age under the risk of stochastic natural disturbances. That model is, by nature, static over time. Notably, our bioeconomic model presented here uses optimal control theory offering a more flexible mathematical framework, and describes a dynamic biological process.

model of an AB invasion—in particular, the likely epidemiology and appropriate prevention and control responses. SPB is considered as one of the most destructive insect species that inhabit pine forests throughout the South, responsible for over \$1 billion in damage between 1991 and 2004 (Waldron 2011). MPBs have killed pines trees over a vast area since the mid-1990s in the Western U.S. (Hart et al. 2015). These input data are used to simulate the potential impacts of an AB infestation for the 2018 to 2050 time horizon and provides upper and lower bound estimates of the hypothetical economic impacts of a Cuba-originated AB.

The remainder of this paper is as follows. A mathematical model of the AB control is presented in Sect. 2. Empirical simulations with different policy control scenarios, associated costs, and arrival rates of AB are described in Sect. 3. We present and discuss the results in Sect. 4. Finally, we offer concluding remarks of our study.

2 The bioeconomic model

Our bioeconomic model is in line with those proposed by Lee et al. (2009) who assessed the efficiency of control programs for invasive upland plants on public conservation forestlands in Florida; and Adams and Lee (2012) and Adams and Lee (2007) who evaluated the efficiency of control methods for invasive mollusks and aquatic plants in public lakes in Florida, respectively. Our stochastic dynamic programming approach was employed to evaluate the net present value of benefits Z under the risk of AB infestation over a planning horizon T is:

$$Z = \sum_{t=0}^{T} (1+r)^{-1}\left(\left(c' + b'\right)XS_t + d(X)' S_t + e' S_t\right)$$

where X is the management strategy that maximizes Z, S_t is the state of the AB, r is the discount rate, and c, b, d, and e, are vectors of management costs, benefits, economic damages and environmental damages. S_t depends on the number of states s_i of AB in relation to the loblolly/shortleaf pine forests for the area of study. The probability S_t that AB is in each of the n states is as follows:

$$S_t = \begin{bmatrix} s_1 \\ \cdot \\ \cdot \\ s_n \end{bmatrix}_t \text{with } 0 \le s_{it} \le 1;\ \sum_{i=1}^{n} s_{it} = 1$$

Unlike Adams and Lee (2012) and Adams and Lee (2007), we specify five states of nature and assume a biennial transition for AB between states once it has arrived. We assume a 2 year-period due to our assumptions regarding the likely biology and life-cycle of the AB data, and the resultant outbreak and spread dynamics over a massive spatial expanse (i.e., Alabama, Florida, Georgia). The states ($i = 1, \ldots, 5$) for AB are as follows: s_1 = not arrived, s_2 = established in port of entry (POE) city, s_3 = incipient epidemic phase (beyond POE locale), s_4 = epidemic outbreak phase, and s_5 = widespread regional proliferation.

At time $t = 0$, there is no invasion of AB, such that $S_0 = \begin{bmatrix} 1 \\ 0 \\ 0 \\ 0 \\ 0 \end{bmatrix}_{t=0}$; after one period, $S_{t+1} = A_0 S_t$ where A_0 represents the initial transition probability matrix. Thus:

$$A_0 = \begin{bmatrix} a_{11} & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21} & a_{22} & a_{23} & a_{24} & a_{25} \\ a_{31} & a_{32} & a_{33} & a_{34} & a_{35} \\ a_{41} & a_{42} & a_{43} & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & a_{54} & a_{55} \end{bmatrix}$$

where an element a_{ij} implies the probability of transitioning from state j to state i in one period of time (e.g. a_{43} indicates a transition from incipient epidemic state to epidemic outbreak state). Each element of the matrix diagonal ($i = j$) represents the probability for a given state to remain in that particular state. Only consecutive "jumps" between states are allowed, i.e., the transition of AB between states is sequential as it advances from state 1 to 5 (e.g., $a_{35} = 0$).

In this initial transition probability matrix $a_{11} = 1 - a_{21}$, with a_{21} as the probability of establishment. Furthermore, $a_{32} = a_{43} = a_{44} = a_{54} = 1$, which reflects that AB progresses to a state of widespread regional proliferation without any policy or management control, and the probability of all other $a_{ij} = 0$.

The transition between states depends on the management actions or preventive control policies to combat the risk of AB and its associated damages. We include the following policy responses to AB infestation: (1) prevention efforts, including eradication, at the POE and its urban surroundings; (2) mitigation of forest stand-level establishment risk; (3) remedial action taken to prevent or limit the spread of the AB, and; (4) a containment response based on extensive clear-cuts. Thus, the transition probability matrix associated with prevention policy A_p:

$$A_p = \begin{bmatrix} a_{11} - a_{12} f_p & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21}(1 - f_p) & a_{22} & a_{23} & a_{24} & a_{25} \\ a_{31} & a_{32} & a_{33} & a_{34} & a_{35} \\ a_{41} & a_{42} & a_{43} & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & a_{54} & a_{55} \end{bmatrix}$$

Here, the original probability of establishment a_{21} is multiplied by $\left(1 - f_p\right)$, where f_p is the effectiveness of prevention control. Thus, the state of *not arrived* $= a_{11} - a_{21} f_p$. Furthermore, the probability that the AB moves from establishment to the incipient epidemic state $a_{32} = 0.7$; thus $a_{22} = 0.3$. The rationale for these estimate values is discussed in the next section. Furthermore, $a_{43} = a_{54} = a_{55} = 1$.

The second policy considered is risk mitigation policy through proactive forest stand thinning. This policy (A_m) is modeled as follows:

$$A_m = \begin{bmatrix} a_{11} - a_{21}f_p & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21}(1 - f_p) & a_{22} - a_{32}f_m & a_{23} & a_{24} & a_{25} \\ a_{31} & a_{32}(1 - f_m) & a_{33} & a_{34} & a_{35} \\ a_{41} & a_{42} & a_{43} & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & a_{54} & a_{55} \end{bmatrix}$$

where f_m is the risk mitigation effectiveness. In A_m, $a_{43} = a_{54} = a_{55} = 1$.

The third policy considered is remedial treatment, using both cut-and-leave and cut-and- remove, to suppress incipient spot infestations in an attempt to prevent the emergence of a widespread AB infestation. The transition matrix associated with this policy (A_r) with its effectiveness rate f_r is as follows:

$$A_r = \begin{bmatrix} a_{11} - a_{21}f_p & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21}(1 - f_p) & a_{22} & a_{23} & a_{24} & a_{25} \\ a_{31} & a_{32} & a_{33} - a_{43}f_m & a_{34} & a_{35} \\ a_{41} & a_{42} & a_{43}(1 - f_r) & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & a_{54} & a_{55} \end{bmatrix}$$

with $a_{22}=0.3$ and $a_{32} = 0.7$; $a_{54} = a_{55} = 1$.

The fourth policy response is containment which reflects extensive clear cutting of forestlands infested by AB. Thus, the transition matrix associated with this policy $\left(A_c\right)$, with its effectiveness rate $f_c = 0.95$, is as follows:

$$A_c = \begin{bmatrix} a_{11} - a_{21}f_p & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21}(1 - f_p) & a_{22} & a_{23} & a_{24} & a_{25} \\ a_{31} & a_{32} & a_{33} & a_{34} & a_{35} \\ a_{41} & a_{42} & a_{43} & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & a_{54} & a_{55} \end{bmatrix}$$

where with $a_{22}=0.3$ and $a_{32} = 0.7$; $a_{23}=1$.

3 Empirical simulation

3.1 Establishment rate for AB

We estimated a nationwide establishment rate for the hypothetical AB,[3] apportioned it to individual domestic regions and metropolitan statistical areas (MSA) in the U.S., and treated the results as the probability of arrival/establishment of a new exotic ambrosia or bark beetle at a given region or MSA (UF/SFRC 2017). Summing over the relevant regions and MSAs of these states and then subtracting the joint probabilities, an aggregated annual establishment probability was estimated for Alabama, Florida, and Georgia (UF/SFRC 2017). Accounting for a 2-year modeling time-period (discussed below), and the so-called "Rule of Tens" (approximately 10% of newly established, non-indigenous species become invasive pests that cause significant ecological and/or economic damages; see Williamson and Fitter 1996), a probability of 0.0063 was estimated for the establishment of the AB in Alabama, Florida, and Georgia (UF/SFRC 2017).

In the present study, we adjust this value to assume that the AB arrives at the study area via Cuban imports. Two separate adjustment factors are calculated: one based upon current imports from the Dominican Republic (used as a proxy for future Cuban imports), and the other based upon Cuban exports to the U.S. for 1958 (i.e. pre-revolution). Since both the size and population of the Dominican Republic are similar to Cuba, it is considered to be a good proxy for future Cuban agricultural trade flows with the US (Valdez 2016). The proportion of the value of 2016 imports from the Dominican Republic to the US total value was 0.002134% (Table 6, USCB 2017); thus, the adjusted AB establishment rate a_{21} is 0.000000134 (Table 1). Since Cuban exports to the U.S. in 1958 accounted for 0.04055% of the total value of US imports for that year (USDA 1959), we also calculate an alternative AB establishment rate of 0.0000025 to serve as an upper bound.

3.2 Policy scenarios

The specific assumptions about the different policy responses and their estimates of efficacy rates associated to each option are described below. In our bioeconomic model, we consider two options to assess the performance of policy responses. In the first option, we test each policy separately to determine the feasibility of each individual policy. In the second option, we test a combination of different polices to assess viable actions to reduce the AB damages in southern forests. The summary of the model parameters is shown in Table 1.

Prevention scenario (*P*) it represents actions taken at the location of the hypothetical introduction of the AB, and consists of three levels of effort: detection,

[3] We assume that AB kills trees through mass accumulation vectoring a mild pathogen, although the most feared approach of AB-related tree mortality is vectoring a virulent pathogen (Hulcr and Stelinski 2017). We choose this type of AB damage, which can last for months, since it is the most common mode of AB-related tree death (Hulcr and Stelinski 2017).

Table 1 Asian Beetle/Cuban pathway bio-economic model parameter and unit cost data

Parameter	Definition	Input data
a_{11}	Probability (Pr.) of AB not being introduced	0.999999866
a_{21}	Pr. of accidental establishment of hazardous invasive AB from Cuba	0.000000134
a_{32}	Pr. AB moves from establishment to incipient-epidemic state	1
a_{43}	Pr. AB moves from incipient-epidemic to epidemic state	1
a_{54}	Pr. AB moves from epidemic state to widespread percolation	1
f_p	Effectiveness of detection & spread prevention	0.2, 0.4, 0.8
f_m	Effectiveness of risk mitigation	0.8, 0.6, 0.9
f_r	Effectiveness of remediation policy	0.8, 0.9
f_c	Effectiveness of containment policy	1.0, 0.95
r	Discount rate of 3% (2-year period = 5.873%)	0.0587334
t	Time (2-year periods) 1, …, 32 years (to 2050) ≡ t = 1, …, 16	1, …, 16
T	Terminal period of simulated scenarios	16
b_{tc}	Benefits of risk mitigation commercial thinning ($/ha)	$1000
c_{cl}	Remedial direct control costs—cut & leave ($/ha)	$3930
c_{cr}	Remedial direct control costs—cut & remove ($/ha)	$544
d_{n1}	Private economic damages—plantations, no AB management ($/ha)	$3140
d_{n3}	Public/PNCF economic damages—no AB management ($/ha)	$2258
d_{b1}	Private economic damages—plantations, base management level ($/ha)	$2437
d_{b2}	Private economic damages—plantations, mitigated risk level ($/ha)	$1837
d_{b3}	Public/PNCF economic damages—base management level ($/ha)	$1752
d_{b4}	Public/PNCF economic damages—mitigated risk level ($/ha)	$1321
e	Value of ecosystem reduced services ($/ha)	$1276

quarantine, and monitoring *(Pd)*; detection, etc. plus suppression *(Ps)*; and detection, etc. plus POE city-wide early eradication *(Pe)* efforts. The *Pd* scenario comprises the monitoring and detection of exotic species that have evaded (or non-complied) pre-shipment International Standards for Phytosanitary Measures No. 15 (ISPM 15), and thus become introduced into the US at a given POE city, and is based largely on the Cooperative Agricultural Pest Surveys (CAPS), of the Plant Protection and Quarantine (PPQ) unit of the United States Department of Agriculture (USDA), and the Early Detection and Rapid Response (EDRR) program of the USDA Forest Service.

Furthermore, the *Pd* scenario incorporates on-site visual surveys of AB by taxonomic experts. The *Pd* scenario represents detection, quarantine, and monitoring efforts; any engagement in terms of suppression is assumed to be limited and incidental (i.e. not a result of proactive efforts). It is implicitly assumed that *Ps* and *Pe* policies are cumulative, e.g. $Ps = Pd +$ suppression; $Pe = Pd +$ eradication in management costs and efficacy.

We employ data on the portion of the annual CAPS budget expenditures related to surveys of wood-boring exotic insects at the national level (John Bowers, US Department of Agriculture, APHIS, personal communication) and for

Florida (Leroy Whilby, Florida Department of Agriculture and Consumer Services, personal communication), and EDRR program budget information, to calculate an annual cost of \$763,290 for the detection and prevention policy applied to the study area; this value is multiplied by 2 (time-period conversion due to our assumption of biennial transition for AB between states) to arrive at a management cost of \$1.527 million per period. This value corresponds to an assumed prevention effectiveness rate (f_p) of 0.20. Furthermore, it is assumed $a_{32}=0.7$ in the transition matrix probability A_p –i.e., the probability that AB moves from establishment to the incipient epidemic state. The probability of all other $a_{ij}=0$. Since the CAPS and EDRR programs exist and are contemporary, the *Pd* simulation serves as the actual baseline scenario of the model.

Two alternative prevention-oriented scenarios, *Ps* and *Pe,* and their associated efficacy rates ($f_p=0.40$ and $f_p=0.80$, respectively), represent management efforts to suppress and eradicate the AB in the urban environment of the POE, in a manner reflecting urban responses to the Asian long-horned beetle (Haack et al. 1997; Nowak et al. 2001). Based on PPQ data (Mary Palm, U.S. Department of Agriculture, APHIS, personal communication), and considering a state contribution, we assume *Ps* with an annual budget expenditure of approximately \$4.9 million that targets the vicinity of the POE (i.e. the areas immediately surrounding the maritime port, or airport), as well as limited locations throughout the city, to prevent the AB from translocating to forested areas outside of the urban POE city. The average annual costs to eradicate the Asian long-horned beetle (\$31.2 million; 2019 values) in the states of Illinois, New Jersey and New York (Haack et al. 2010)—are used as reference to determine the eradications costs of AB. We assume 80% of the eradication costs of the Asian long-horned beetle (\$25 million per year). This lower value is based on the way AB attacks the tree, first the main trunk and then the branches (the reverse of how the Asian long-horned beetle attacks) thus eradication costs are reduced.

Risk mitigation scenario (M) this policy is based on proactive stand thinning which is an important management tool for beetle epidemiology that simultaneously disrupts beetle pheromonal communication and improves tree health (Schmitz et al. 1989; Thistle et al. 2004; Fettig et al. 2007; Coops et al. 2008). We assume that thinning is the most appropriate tool for mitigation for a particular group of bark beetles, including our AB. However, the potential of tree mortality reduction due to thinnings also depends on the type of bark beetles and forest species. For example Egan et al. (2018) found that thinnings had no effect on reducing the mortality of mixed conifer stands attacked by Fir engraver beetles *(Scolytus ventralis)*. Other bark or ambrosia beetle pests may be best addressed with different policies.

We assume that, upon the confirmed introduction of an AB within period $t=1$, federal and state governments arrange contracts for commercial thinning operations that are applied to 5% of all moderate- and high-hazard public loblolly/shortleaf pine forestland (approximately 15,000 ha) each year, on a rolling basis, beginning in the year of AB establishment ($t=1$). This thinning regime represents additional area to be thinned beyond that which has already been planned for irrespective of an AB threat. It is further assumed that thinning will be undertaken annually on 2% (37,640 ha) of the moderate- and high-hazard loblolly/shortleaf pine forestland held by those private non-corporate forestland (PNCF) owners who do not engage in rotational timber harvests and have overstocked stands due to a lack of management. The areas targeted in the initial years of the thinning program will concentrate in the region around the city in which the beetle was introduced.

The base efficacy rate of risk mitigation is assumed to be $f_m = 0.3$, which considers thinning that has already been conducted on public and private forests as part of silvicultural management operations and/or SPB risk reduction efforts. For example, it is assumed that 40% of all private loblolly/shortleaf pine land is private corporate forestland (PCF) and already treated, as is the land owned by PNCF owners that manage their land primarily to grow and harvest timber.

We assume a risk mitigation rate of $f_m = 0.8$ for the thinning regime described previously; this value is based upon the findings of research trials designed to evaluate thinning impacts on MPB outbreaks (Cole et al. 1983; McGregor et al. 1987; Schmitz et al. 1989; Whitehead and Russo 2005), and a statistical evaluation of the effectiveness of thinning on an SPB outbreak (Nowak et al. 2015). Nevertheless, we also simulate alternative scenarios ($f_m = 0.6$, $f_m = 0.9$) to examine other efficacy rates. We assume that the commercial thinning of overstocked stands anywhere from 10 to 50 years of age will yield an average return, net of cost, of \$1000 per hectare (ha). This low value for commercial thinning considers that pulpwood and sawtimber prices will be depressed given anticipated copious local supply (Pye et al. 2011; Chabreck et al. 2013), without a price rebound typically observed following an isolated salvage-inducing event such as a hurricane (Kinnucan 2016). We examined thinning literature (Folegatti et al. 2007; Dickens et al. 2014), consulted forestry experts (Dale Greene, Warnell School of Forestry and Natural Resources, University of Georgia; Mathew Smidt, School of Forestry and Wildlife Sciences, Auburn University, personal communication), and considered empirical salvage supply impacts on prices (Brissett 2002) to arrive at a reasonable "ballpark" value for commercial thinning Time constraints and the complexity of the issue (multiple product types, multiple sub-regional markets in the study area, and the expected supply impact on stumpage and mill prices) precluded detailed modeling of the supply/price relationship as a component of this study.

Remedial treatment scenario (R) Prompt responses to spot outbreaks in the form of small clear cuts are the customary remedial action taken to help prevent the spread of outbreak areas of bark beetles deemed to be a threat. Such direct control responses include cut-and-leave and cut-and- remove (Billings 2011), widely employed to control SPB, which if conducted in a timely and thorough manner, can be very effective in checking the spread of an epidemic (Meeker 2017). For ambrosia beetles, sanitation is the most effective action, managing the infested wood through burning or chipping (Jones and Paine 2015). Although cut-and-leave and cut-and- remove can be less feasible for AB control (Self et al. 2015), we include both alternatives to suppress incipient spot infestations in an attempt to prevent the emergence of a widespread AB infestation. For this policy scenario, economic benefits from cut-and-remove are not included in our model.

We assume 200 spot outbreaks per year will require remedial treatment once they begin to develop (t = 2) in forests beyond the POE city: 15 outbreak areas/year on public forestland will be treated with cut-and-leave, and 185 outbreak areas/year on private (mainly PNCF) land will be treated with cut-and-remove. Average spot size (1.79/ha cut & leave, 5.84/ha cut & remove) is based on historical spot outbreak data, on national forests in the study area from 2000 to 2017, obtained from the SPB Information System Status Reports (USDA 2017).

The operational unit cost of cut-and-leave is estimated from empirical values of $1430/ha (Chabreck et al. 2013) and $1915/ha (Meeker 2017); we therefore use $1673/ha as a conservative mid-point value and add $2258 as the value of the timber lost (discussed later), to arrive at a total cost of $3930/ha. The total cost of cut-and-remove is assumed to be $544/ha, which is the operational cost of cut-and-leave minus half the value of the timber extracted. The efficacy rate of remediation ($f_r = 0.8$) is approximately what Chabreck et al. (2013) indicate from empirical observations following the SPB outbreak on the Homochitto National Forest (Mississippi) in 2012. We also examine a slightly higher efficacy rate ($f_r = 0.9$).

Containment scenario (C) this scenario includes extensive clear-cutting, and is also simulated based upon the same 200 spot outbreaks per year, but with the amount of forest area that is clear-cut scaled up by a factor of 10 and 100% effectiveness assumed. Due to the enormous quantities of timber cut under this scenario (11,073 ha), the cost basis is almost exclusively derived from cut-and-leave unit costs; cut-and-remove unit costs are only applied to the area that would have been remediated given the remediation policy. Similar to the remedial treatment scenario (R), we rule out the possibility of economic benefits due to cut-and-remove.

The containment policy is implemented in the first period when spot outbreaks are detected (t = 2), and the discounted cost of this policy is also attributed to this period. Since, in reality, a containment policy is not likely to be 100% effective, we employ an alternative efficacy rate ($fc = 0.95$) to examine such a scenario.

We also simulate several combinations of the above policy responses. This implies that the adjustments of the transition probabilities given these "new policies" are made concurrently in the matrix. For example, a policy combining risk mitigation and remediation *(MR)* is represented as:

$$A_{mr} = \begin{bmatrix} a_{11} - a_{21}f_p & a_{12} & a_{13} & a_{14} & a_{15} \\ a_{21}(1 - f_p) & 0.8 & a_{23} & a_{24} & a_{25} \\ a_{31} & 0.2 & 0.8 & a_{34} & a_{35} \\ a_{41} & a_{42} & 0.2 & a_{44} & a_{45} \\ a_{51} & a_{52} & a_{53} & 1 & 1 \end{bmatrix}$$

Under this particular scenario, we assume a synergistic effect such that thinning will reduce the number, and average size, of outbreak areas that eventually develop and require remedial treatment. We therefore assume 150 outbreak areas per year (a 25% decrease), at 0.9/ha for cut-and-leave and 2.92/ha for cut-and-remove (50% decrease for each).

Finally, it is relevant to mention that our bioeconomic model and the polices defined do not account for the possible total loss of southern forests and related ecosystem services, and the consequent impacts on the aggregated behavior of the forest industry.

3.3 Economic parameters

The management costs (c) comprise the CAPS/EDRR outlays, and the unit costs of pre-commercial thinning and remediation applied to the appropriate forest areas, while the benefits (b) are derived from commercially thinned areas as part of risk mitigation. The model comprises $t=16$ time periods that, given a specification of 2 years per period, cover an actual time horizon of 32 years—which encompasses one rotation on a typical commercial loblolly plantation (i.e. 22–28 years). We specify $r=0.0587334$, which is equivalent to an annual discount rate of 3%.

Damages (d) are specified in the form of timber value lost or forgone due to an AB infestation, applied to two categories of forest landowners. Using values obtained from a model of disturbance risk for loblolly plantation production (Susaeta et al. 2016), we estimate damages $d_{n1}=\$3140$/ha that represent forgone timber values, in terms of the difference in land expectation value (LEV) between the LEV for no policy *(N)* scenario and LEV given 65% tree mortality due to unimpeded AB proliferation. The mortality figure is a mid-range estimate based upon data from Cole et al. (1983), McGregor et al. (1987), and Schmitz et al. (1989) for the MPB, and Billings (2011) for the SPB, and is conservative in the sense that it is much less than the 90% mortality observed for the redbay ambrosia beetle. The d_{n1} estimate is applied to all loblolly/shortleaf pine forestland held by private corporate forestland (PCF) owners and those private non-corporate forestland (PNCF) owners that produce timber as their primary use for their forestland.

The area of loblolly/shortleaf pine was obtained for Alabama, Florida, and Georgia from the USDA Forest Inventory Analysis (USDA-FIA 2017); the data consists of four categories: national forest, other federal, state and local, and private (Table 7). In order to subdivide the private category, we used PCF data from Butler and Wear (2013, p. 113) to estimate the ratio of PCF land to total forestland for Alabama, Florida, and Georgia; this ratio was applied to the loblolly/shortleaf pine area in the same three states, allowing us to estimate PNCF area as well.

We then assume that 40% of all PNCF loblolly/shortleaf pine area was under rotational harvest and applied d_{n1} to this area. Damages $dn_2=\$2258$/ha represent lost timber value on multiple-use public loblolly/shortleaf forestland and 60% of the loblolly/shortleaf PNCF whose owners do not engage in rotational harvests of timber. This unit value is based upon the loss of timber to the SPB in 2016 on national forests in Mississippi (Meeker 2017).

Since the LEV model relies on the AB establishment rate, we also estimate damages considering some level of policy control for AB. In the case of private landowners, we estimate damage $d_{b1}=\$2437$/ha, based on the difference in LEV under the baseline policy *(P_d)* scenario and LEV with 65% tree mortality due to the AB, for the combined PCF + 0.4 PNCF area under timber production. An additional estimate $d_{b2}=\$1837$/ha is also applied to this area for the policy scenarios in which thinning is applied to mitigate the risk of AB establishment and spread (M); it is calculated in the same manner as d_{b1} but with a mortality rate of 15% (Cole et al. 1983; McGregor et al. 1987; Schmitz et al. 1989; Billings 2011). In the case of public ownership, damages d_{b3} = \$1752/ha represent lost timber value on the non-timber production lands defined previously (i.e. multiple-use public forestland and 60% of the PNCF land) and is calculated by applying the ratio of d_{b1}/d_{n1} (0.776) to d_{n2}. To obtain d_{b4} = \$1321/ha, the damage estimate applied to the multiple-use forest areas that are thinned under the risk mitigation policy simulation, we multiply d_{b3} by 0.754 (i.e. d_{b2}/d_{b1}). In a similar manner, separate forest values are derived for the *Ps* and *Pe* policy scenarios based on the effects of these policies on the AB establishment rate.

Our ecosystem service damages parameter, e = \$1276/ha, is 15% of the estimate (\$8505/ha) reported by Escobedo and Timilsina (2012) as the value that forests in the Florida Forest Stewardship Program provide in terms of water purification services, as compared to developed land. Given that the impact of an AB infestation on water quality would likely be similar to that of clear-cuts, and thus minimal (Grace 2005), we nevertheless assume a negative impact (ecosystem service loss) of 15%. We posit a slight loss of ecosystem service value (2–5%) due to changes in water quality, with the balance of 15% assumed to be lost amenity values in terms of decreased aesthetic value, which is likely to be the real impact on non-market forest values of an AB infestation. The latter assumption is due to our inability to find in the literature any recent forest amenity valuation studies that were applicable to this study. The damage parameter e is applied to 65% of all loblolly/shortleaf forestland (public and private), which reflects the base mortality rate discussed previously; for simulations in which the risk mitigation policy is applied, e applies to 15% of the forest area under consideration.

4 Results and discussion

4.1 Individual policies and the MR combination

Given a hypothetical introduction of an AB from Cuba, Table 2 lists the change in net present value (ΔNPV) of forest damages plus policy costs between the NPV of the theoretical baseline of "no management action taken" (*N*) and each of several potential management policies. These policies consist of government-instituted prevention policies (*Pd, Ps, Pe*) proactively applied within the POE city, three policy responses (*Mn, Rn, Cn*) applied to peri-urban and rural forestlands, a combination policy (*MRn* = thinning + remedial cuts), and some alternate sensitivity analyses (e.g. *Mnl, Rna*) based on varied efficacy rates. The results clearly indicate that all policies except for risk mitigation (*Mn, Mnl, Mnh*) are inferior to the "no policy" (*N*) simulation. In the absence of any policy, an AB infestation via Cuba is estimated to result in NPV −\$128,300 of potential economic damages to forestland of the study area (sum of columns 3, 4, and 5), which is two orders of magnitude less than the NPV of the policy cost (−\$16.1 million) for the prevention policies.

Depending upon the efficacy rate of thinning assumed, the risk mitigation policies would reduce economic damages to a range of NPV −\$24,000 to NPV

Table 2 Simulation results of individual policies and risk mitigation + remediation ($ million)

Policy	Public policy cost	Private management cost	Private economic loss	Public economic loss	Ecosystem service loss	NPV (costs + losses)	ΔNPV[b]
N[a]	–	–	−0.0919	−0.0060	−0.0304	−0.1283	–
Pd[c]	−16.15	–	−0.0534	−0.0035	−0.0175	−16.22	−16.09
Ps	−16.15	–	−0.0380	−0.0025	−0.0131	−16.20	−16.07
Pe	−16.15	–	−0.0119	−0.0008	−0.0044	−16.16	−16.03
Mn	0.000136	0.000340	−0.0328	−0.0021	−0.0040	−0.0385	0.0897
Mnl	0.000085	0.000212	−0.0453	−0.0030	−0.0056	−0.0535	0.0748
Mnh	0.000189	0.000471	−0.0207	−0.0014	−0.0026	−0.0240	0.1043
Rn	−0.000001	−0.000004	−0.0445	−0.0029	−2.84	−2.89	−2.76
Rna	−0.000001	−0.000005	−0.0262	−0.0017	−2.83	−2.86	−2.73
Cn	−0.000009	−0.000340	–	–	−28.3	−28.3	−28.1
Cna	−0.000008	−0.000308	−0.0115	−0.0008	−28.3	−28.3	−28.1
MRn	0.000189	0.000470	−0.0117	−0.0008	−1.06	−1.07	−0.9461
MRna	0.000237	0.000591	−0.0036	−0.0002	−1.06	−1.06	−0.9363

T = 16 periods (32 years), r = 5.873% per period (3% per year)

[a]The *No Policy or Management (N)* scenario is the theoretical baseline simulation

[b]With NPV of the *No Policy or Management* scenario serving as the basis

[c]Base prevention, *Pd*, is a stand-alone policy; as are the forest-based policies (except the *MR* combination*s*) which are estimated using the theoretical baseline values (i.e. no prevention policy assumed, thus no interaction) as indicated by the “n” subscript

– $54,500 that translates into a 58% to 81% reduction of damages over the base "no policy" simulation. Reduced damages to private forestland comprise the majority of the impact of risk mitigation (thinning benefits). In contrast, the remediation, containment, and combined mitigation-remediation policies are all impractical policies because each of their simulations result in continuous damages + policy costs (time horizon of 32 years) that exceed the theoretical maximum economic damages (NPV – $128,300) under the "no policy" simulation– no economic benefits from silvicultural interventions. The NPV values of these policies are almost exclusively due to impacts on ecosystem services during the time horizon (column 5, Table 2), resulting from clear-cutting that is the basis of the containment policy (widespread clear-cuts) and the remediation policy (limited scale *cut-and-leave* and *cut-and-remove* treatments).

Table 3 presents ΔNPV of forest damages plus policy costs for simulations that utilize the alternative AB establishment rate, which is based upon the level of pre-revolution Cuban exports to the US. An AB infestation via Cuba under this assumption would result in maximum economic damages to forestland in the study area of NPV – $2.44 million (– $76,250/year), in the absence of any policy. This represents an upper-bound damage value. All policies except for risk mitigation (*Mn, Mnl, Mnh*) and the mitigation/remediation combination (*MRn, MRna*) are inferior to the "no policy" (*N*) scenario. Note that the implementation of these policies would actually return revenue in the form of timber sales from thinning, and from *cut-and-remove* treatments when remediation is paired with mitigation. Nevertheless, the mitigation policy is the preferred policy due to the greater value of ΔNPV.

4.2 Potential policy combinations

Although the prevention policies are not viable in the above simulations, we cannot exclude base prevention from consideration since, in reality, prevention-detection (*Pd*) represents the *status quo* in terms of current invasive species policy in the U.S. We therefore examine various policy combinations with *Pd* serving as the baseline by hypothetically assuming the federal government would not commit any additional funds beyond what is currently spent on monitoring and detection. Given such a scenario, what action would policy makers in the study area (Alabama, Florida, Georgia) take in terms of policy response to an AB invasion originating from Cuba?

Table 4 indicates that the only viable policy combinations would be risk mitigation paired with any of the prevention policies; the gain in terms of ΔNPV would be modest, however, ranging from NPV $45,000 (*PdM*) to NPV $68,000 (*PeM*). It should be noted that the prevention-detection-suppression (*Ps*) and prevention-detection-eradication (*Pe*) policies would also result in ΔNPV of $21,000 and $57,000 (data not shown), respectively.

Table 5 displays simulation results for policy combinations using the AB establishment rate based on the pre-revolution Cuban trade scenario. Prevention policies paired with either risk mitigation (e.g. *PdM*) or the mitigation + remediation combination (e.g. *PeMRn*) are feasible in terms of positive ΔNPV, although the gains over the base policy (*Pd*) for combinations featuring mitigation + remediation are

Table 3 Simulation results for the alternative establishment rate simulations ($ million)

Policy	Public policy cost	Private management cost	Private economic loss	Public economic loss	Ecosystem service loss	NPV (costs + losses)	ΔNPV[b]
N[a]	–	–	−1.75	−0.1144	−0.5776	−2.44	–
Pd	−16.15	–	−1.02	−0.06641	−0.3323	−17.56	−15.12
Ps	−16.15	–	−0.7236	−0.04733	−0.2492	−17.17	−14.73
Pe	−16.15	–	−0.2273	−0.01487	−0.08308	−16.47	−14.03
Mn	0.00260	0.00647	−0.6245	−0.04086	−0.07699	−0.7333	1.71
Mnl	0.00162	0.00403	−0.8613	−0.05635	−0.1062	−1.02	1.42
Mnh	0.00360	0.00897	−0.3941	−0.02579	−0.04859	−0.4690	1.98
Rn	−0.00001	−0.00008	−0.8465	−0.05538	−3.11	−4.01	−1.57
Rna	−0.00002	−0.00010	−0.4993	−0.03266	−2.99	−3.52	−1.08
Cn	−0.00018	−0.00647	–	–	−28.3	−28.3	−25.8
Cna	−0.00016	−0.00587	−0.2184	−0.01429	−28.3	−28.6	−26.1
MRn	0.00359	0.00895	−0.2229	−0.01458	−1.09	−1.31	1.13
MRna	0.00451	0.01124	−0.06940	−0.00454	−1.07	−1.13	1.31

T = 16 periods (32 years), r = 5.873% per period (3% per year)

[a]The *No Policy or Management (N)* scenario is the theoretical baseline simulation

[b]With NPV of the *No Policy or Management* scenario serving as the basis

Table 4 Simulation results with prevention-detection (*Pd*) as the baseline ($ million)

Policy	Public policy cost	Private management cost	Private economic loss	Public economic loss	Ecosystem service loss	NPV (costs + losses)	ΔNPV[b]
Pd[a]	−16.15	0	−0.0534	−0.0035	−0.0175	−16.22	–
PdM	−16.15	0.00027	−0.02485	−0.002	−0.00324	−16.17	0.045
PdR	−16.15	0.00000	−0.02425	−0.002	−2.83	−19.0	−2.79
PdC	−16.15	−0.00022	–	–	−28.3	−44.4	−28.2
PdCa	−16.15	−0.00020	−0.00583	0.000	−28.3	−44.4	−28.2
PdMR	−16.15	0.00038	−0.00887	−0.001	−1.06	−17.2	−0.997
PdMRa	−16.15	0.00047	−0.00276	0.000	−1.06	−17.2	−0.989
PsM	−16.15	0.00020	−0.01678	−0.001	−0.00243	−16.2	0.054
PsR	−16.15	0.00000	−0.01728	−0.001	−2.83	−19.0	−2.78
PsC	−16.15	−0.00017	–	–	−28.3	−44.4	−28.2
PsCa	−16.15	−0.00015	−0.00437	−0.00029	−28.3	−44.4	−28.2
PsMR	−16.15	0.00028	−0.00599	−0.00039	−1.06	−17.2	−0.994
PsMRa	−16.15	0.00035	−0.00186	−0.00012	−1.06	−17.2	−0.989
PeM	−16.15	0.00007	−0.00499	−0.00033	−0.00081	−16.2	0.068
PeR	−16.15	0.00000	−0.00543	−0.00036	−2.83	−19.0	−2.76
PeC	−16.15	−0.00006	–	–	−28.3	−44.4	−28.2
PeCa	−16.15	−0.00005	−0.00146	−0.00010	−28.3	−44.4	−28.2
PeMR	−16.15	0.00009	−0.00178	−0.00012	−1.06	−17.2	−0.989
PeMRa	−16.15	0.00012	−0.00055	−0.00004	−1.06	−17.2	−0.987
PeMal	−16.15	0.00004	−0.00688	−0.00045	−0.00112	−16.2	0.066
PeRa	−16.15	0.00000	−0.00314	−0.00021	−2.83	−19.0	−2.76

T = 16 periods (32 years), r = 5.873% per period (3% per year)

[a]The *Prevention-detection* (*Pd*) scenario is the baseline simulation

[b]With NPV of the *Prevention-detection* simulation serving as the basis

Table 5 Results for *Pd* as the baseline, plus the alternative establishment rate ($ million)

Policy	Public policy cost	Private management cost	Private economic loss	Public economic loss	Ecosystem service loss	NPV (costs + losses)	ΔNPV[b]
Pd[a]	−16.15	0	−1.02	−0.0664	−0.332	−17.56	–
PdM	−16.14	0.00518	−0.473	−0.03093	−0.062	−16.7	0.856
PdR	−16.15	−0.00006	−0.461	−0.03018	−2.98	−19.6	−2.05
PdC	−16.15	−0.00420	0	0	−28.3	−44.4	−26.8
PdCa	−16.15	−0.00389	−0.111	−0.0073	−28.3	−44.6	−27.0
PdMR	−16.14	0.00716	−0.169	−0.0110	−1.08	−17.4	0.161
PdMRa	−16.14	0.00899	−0.0525	−0.0034	−1.07	−17.3	0.303
PsM	−16.14	0.00388	−0.319	−0.0209	−0.0462	−16.5	1.03
PsR	−16.15	−0.00004	−0.329	−0.0215	−2.94	−19.4	−1.88
PsC	−16.15	−0.00315	0	0	−28.3	−44.4	−26.8
PsCa	−16.15	−0.00291	−0.0832	−0.0054	−28.3	−44.5	−27.0
PsMR	−16.14	0.00537	−0.114	−0.0075	−1.08	−17.3	0.222
PsMRa	−16.14	0.00674	−0.0355	−0.0023	−1.07	−17.2	0.319
PeM	−16.15	0.00129	−0.0949	−0.0062	−0.0154	−16.3	1.30
PeR	−16.15	−0.00001	−0.103	−0.0068	−2.86	−19.1	−1.56
PeC	−16.15	−0.00105	0	0	−28.3	−44.4	−26.8
PeCa	−16.15	−0.00097	−0.0277	−0.0018	−28.3	−44.4	−26.9
PeMR	−16.15	0.00179	−0.0339	−0.0022	−1.07	−17.2	0.313
PeMRa	−16.15	0.00225	−0.0105	−0.0007	−1.06	−17.2	0.342
PeMal	−16.15	0.00081	−0.131	−0.0086	−0.0212	−16.3	1.25
PeRa	−16.15	−0.00002	−0.0598	−0.0039	−2.85	−19.1	−1.50

T = 16 periods (32 years), r = 5.873% per period (3% per year)

[a]The *Prevention-detection* (*Pd*) scenario is the baseline simulation

[b]With NPV of the *Prevention-detection* simulation serving as the basis

significantly less than for the prevention mitigation pairs. Thus, the risk mitigation policy is again observed to be the dominant policy, in this case in combination with prevention. The differences between the various combinations with respect to ΔNPV are relatively minor, however: $856,000 (*PdM)* ($26,750/year), $1.03 million (*PsM)* ($32,187/year), and $1.30 million (*PeM)* (– $40,635/year).

5 Conclusion

This study estimates potential economic damages to southern pine forests in the event that an unknown invasive bark or ambrosia beetle (AB) from Asia becomes introduced to the U.S. via Cuba, given the assumption of future trade normalization between the countries. This is a salient policy issue, particularly in the state of Florida, which has a contentious history with its Caribbean neighbor. We also examine policy/management actions that would likely be considered in response to this hypothetical situation. We employed a bio-economic AB model, adjusting expected AB establishment rates to account for an introduction from Cuba (as opposed to the rest of the world). Two adjustments are made to define lower and upper bounds for potential economic damages, and we also examine policies individually and in combination.

The theoretical "no policy or management action" scenario indicates that the lower bound of potential damages to forest resources and associated ecosystem services from the hypothetical AB is estimated to be – $128,300 in net present value (NPV) for the study period (2018–2050). This value is based upon the current level of imports to the U.S. from the Dominican Republic, which serves as a proxy for future Cuban imports to the U.S. These damages could be reduced to between NPV – $24,000 and NPV – $54,500 if a risk mitigation policy, based upon thinning of moderate- and high-hazard forestland, were employed at the onset of AB detection in the U.S. When analyzing the impacts of combined policies, risk mitigation paired with prevention policies including suppression and eradication result in small monetary improvements—total costs are reduced by $45,000 to $68,000.

In the absence of any policy, an upper bound of potential damages of NPV – $2.44 million (– $76,250/year) is obtained when basing the AB establishment rate on Cuban imports from 1958, the last year prior to the Cuban revolution. Implementing a risk mitigation policy could potentially lower these damages to between NPV – $469,000 (– $14,656/year) and NPV – $1.02 million (– $31,875/year), depending upon the assumed efficacy rate of the commercial thinning that comprises this policy.

It is important to highlight the hypothetical foundation of this research; findings of the study rely upon many assumptions due to the inherently speculative nature of a pest that is not yet known, and a limited amount of (or lack of, in some cases) solid empirical data to inform model parameters. For example, we have assumed that only the risk mitigation strategy can generate economic benefits due to thinnings, while remedial and containment policies only result in damages and costs. However, this kind of modeling is often given the critical need for information to inform policy choices and interventions before invasion, when interventions are most likely to achieve success and be most cost-effective (Samset and Christensen 2017). Bioeconomic modelling in this context is particularly useful, as it can help policymakers evaluate the outcomes of policies before they are implemented, and to understand the impacts over time. This is the typical case of land-use problems in agriculture, forestry and fisheries (Castro et al. 2018).

We have also assumed that our AB covers both bark and ambrosia beetles. Although they are similar, they have different ecology and invasion patterns (Rassati et al. 2016). Future avenues of research could include the incorporation of potential economic benefits from clear-cutting based policies, and different infestation paths of tree beetles. The economic analysis of potential pest infestation with other countries as intermediate vectors and due to increases in trade volume could be also included. Finally, given the discrete nature our data and limitations inherent in our bioeconomic model, we were not able to incorporate the ecology of the AB damage at the moment it attacks the tree in a granular way, i.e., the rate of spread of the pest and how quickly a tree dies after being infested. Such an approach would be helpful, as it could provide further insights about the interplay between the timing of invasive species arrival and adoption of early mitigation strategies, and subsequent flows of costs affecting the estimates of economic damages. Future work is needed to develop a more flexible model framework that would support that kind of detailed analysis.

Finally, we caution that in the unfortunate event that an AB were to become established in a port of entry city in the southern U.S., it would be prudent for policy makers to view the output values of this study in a qualitative fashion—that is, in relation to each other for the purposes of comparison and ranking. This is typically how we present them throughout this report; however, for the exceptions to this rule (e.g. presenting the maximum theoretical damages for a scenario), we suggest that they be considered in a general sense that downplays their absolute value in favor of being a "ballpark" value.

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Appendix

See Tables 6 and 7.

Table 6 Data for the establishment rate adjustment factors

Country/year	Exports to U.S. ($ Billion)	Total U.S. imports ($ Billion)	% of U.S. total
Dominican Republic/2016	4.68	2188	0.002134
Cuba/1958	0.517	12.75	0.04055

Source USCB (2017)

Table 7 Loblolly/shortleaf pine forest area, by category (acres)

State	National forest	Other federal	State and local	Private
Alabama	132,907	43,523	154,575	8,704,940
Florida	207,585	65,390	204,220	1,353,939
Georgia	123,062	175,640	226,265	6,928,925
Total	463,554	284,553	585,060	16,987,804

Source USDA FIA 2017

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