

Some Economic Considerations of the Bloodworm Trade's Potential as an AIS vector

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I. Introduction

A. Tracking a Problem Across Disciplines

Trade in marine bloodworms, part of the live bait industry, generates benefits for recreational fishermen and income for bloodworm gatherers, distributors, retailers and their employees. The trade is made possible in part by the ability to rapidly transport live bloodworms from points of harvest in Maine to retailers and fishermen in the Mid-Atlantic, California, Europe and elsewhere. When the bloodworms are packed in wet seaweed, this rapid transport has the inadvertent effect of conveying with the worms living hitchhikers (Cohen et al., 2001) harbored in the seaweed. Hitchhikers may include any of a wide range of macro-invertebrates (Fowler et al., 2016, Blakeslee et al., 2016), micro-plankton (Haska et al., 2012) and the seaweed packing material, *Ascophyllum nodosum* ecad *scorporides* (Miller et al., 2004).

There is some likelihood that the packing material or the living organisms it harbors will be introduced to a marine coastal habitat similar to that which it left in Maine. There is then a contingent probability that, once introduced, the organism or the packing material will become established in this new place, producing viable offspring who impact, perhaps negatively, the existing ecosystem. Although those probabilities are typically low for any given baitbox, the number of experiments, or, potential introduction events, can be quite large. Fowler and others (2016) estimate that over 1.2 billion macro-invertebrates have been transferred from Maine in bloodworm baitboxes over the past 67 years. Moreover, successful invasions of non-indigenous species can sometimes be catastrophic. Globally, out of 256 vertebrate extinctions with an identifiable cause, 109 are known to be due to biological invaders [Cox, 1993].

In the language of biologists, trade in bloodworms is a vector for the introduction of aquatic invasive species (AIS). In the language of economists, this vector aspect of trade in bloodworms constitutes a negative externality in the live bait market. An externality is an impact imposed (or, in the case of positive externalities, bestowed) on people regardless of whether or not they were involved in the activity that created it. Imagine a farmer who applies nitrogen fertilizer on his fields only to have it wash away in a rainstorm. The runaway nitrogen fuels an algae bloom downstream making fishing and swimming there impractical. The washed away nitrogen is a negative externality. The farmer did not aim for this outcome, but the fishermen and swimmers who cannot fish or swim downstream, among others, bear the burden of the environmental harm done.

The economic cost of the environmental harm imagined for the runaway nitrogen example includes both monetary losses (i.e., increased health care costs or reduced fish harvests) and the value of losses for which there are no markets and, therefore, no readily apparent monetary values. An example of this latter type of loss is the disruption of an ecosystem in someone's favorite natural place. If people were enjoying something which, after the ecosystem disruption, is gone, then they have clearly lost something. Through contingent valuation methods, such as stated preference surveys or voting experiments, it is possible to estimate values for people's "willingness to pay" for keeping some favorite natural phenomenon available (see Carson, 2016b and Bishop and others, 2017).

This report will not develop new non-market valuations but will note where a more complete evaluation of non-market costs could inform decision-making with regard to some of the organisms that were found in bloodworm baitboxes in a recent study supported by Maryland Sea Grant and the Smithsonian

Environmental Research Center's (SERC) Marine Invasions Laboratory. The biological part of that study looked at macro-invertebrates in bloodworm baitboxes in Maine and at delivery points in California and the mid-Atlantic states. Using information from that study, we consider the green crab (*Carcinus maenas*), two species of periwinkles (*Littorina littorea* and *L. saxatilis*), and the packing material itself, *Ascophyllum nodosum* ecad *scorporides* (hereafter, wormweed) as potential invaders imposing costs as described in the following section.

B. The Bio-economic Problem.

The AIS vector problem starts with the possibility that a bloodworm baitbox might carry a living organism (or, propagule) to a place where it is not established and where that organism might then establish. One way to manage that potential outcome is to extend effort to prevent the introduction of the living organism in the first place. It is reasonably expected that there will be costs to such prevention efforts, which raises the question, how much should be invested to prevent the introduction of AIS propagules from bloodworm baitboxes? Given limited resources for competing needs, we do not want to spend more preventing an outcome than the cost that would be incurred if it transpired. To avoid that, we need to know the potential cost of environmental damages if prevention fails.

The sample of species selected for this study spans a variety of potential nuisances. Green crabs have had a significant impact on commercial bivalve fisheries in the northern Atlantic where they have naturalized (Grosholz and Ruiz, 2002), periwinkles have significant and disruptive impacts on ecosystems where they have become established (Carlton and Cohen, 1998) and wormweed could displace indigenous macro-algae or introduce disruptive change by adding new niches to existing ecosystems (Miller, 2004). Damage costs of each of these hitchhiking species depend on their impacts on people's economic welfare. These include commercial effects (market costs), ecosystem function (recreational and existence values) and human health.

AIS costs continue to accrue as long as AIS population impacts persist, so a complete accounting of damages requires that expected future costs be included in any "total" cost. However, these future costs cannot simply be added up without regard to the fact that people have a present time preference for money. That is, in an exchange of money over time, to be willing to part with \$10 today, I want to be compensated with more than \$10 in the future, implying that a dollar is worth more to me today than the promise of one dollar in the future. Consequently, a damage that happens in the distant future has a lower present value than its nominal value in the future. William Nordhaus, 2007, provides a set of good reasons to employ a positive discount rate when considering future costs.

Another factor that affects the costs of environmental damages is the potential for mitigation through control or management efforts. If the negative impacts of established AIS can be mitigated through control efforts in a cost-effective way, then the costs of damages can be reduced. Given that control efforts will carry costs of their own, a complete damage function will include costs of an optimal measure of control (i.e., such that the marginal benefit of more control just equals its cost) plus the costs of whatever damages remain after implementing that optimal level of control. With this third and final cost term, we can characterize the goal of managing the bloodworm trade vector with respect to AIS introductions as: find the least-cost application of resources for prevention, control, and damages costs for the vector. This approach follows Olson and Roy, 2005.

In the following section, impacts are described that either have been imposed by the three species targeted for study here, or which could happen over time. This is constrained by the fact that the future can only be predicted imperfectly and by implication and because what we know about current

circumstances with respect to AIS introduction and establishment is incomplete. It is also worthy of note that our analysis of the hazards of AIS hitchhikers in bloodworm baitboxes is necessarily partial. It is possible and perhaps likely that other hitchhikers identified in the baitboxes research, but not discussed here, could have unforeseen impacts on receiving ecosystems, especially given their small size and current lack of scientific research. It is also possible that hitchhikers which have not yet been found in bloodworm baitboxes but which are found in the ecosystem from which bloodworms and their packing material are drawn could be in play with unknown cost implications. This could include non-native species that become established in the Gulf of Maine.

II. Costs of Some AIS Candidates in the Bloodworm Trade

A. The green crab (*Carcinus maenas*)

Fowler and others, 2016, report finding two dead green crabs in sampled baitboxes packed with wormweed. They found live green crabs in wormweed gathered from areas in Maine where it is sourced as packing material by bloodworm distributors. Finding live green crabs in material sourced from the same place as the wormweed used for commercial bloodworms raises expectations that bloodworm baitboxes will occasionally harbor a live green crab. While these likelihoods are too small to be picked up in the study sample, that sample was very small (N = 15 baitboxes) relative to the universe of bloodworm baitboxes from New England. Cohen and others, 2001 report finding live green crabs in bloodworm baitboxes in the San Francisco Bay area.

The green crab is a nuisance species that came to the American edge of the Atlantic in the 19th century from its native range in Europe (Carlton and Cohen, 2003). It is a successful invader with high fecundity that has established on every continent except Antarctica (Hidalgo et al., 2005). Molecular studies indicate that green crab from the east coast of the US was the source population for its introduction to the west coast (Geller et al., 1997). Although ship ballast is the suspected vector for the earlier global dispersal of green crabs, live specimens were found in retail baitboxes in San Francisco. In San Francisco Bay, green crabs were found living in areas popular with recreational fishermen. It is hypothesized (Cohen et al., 2001) that green crabs were introduced to the west coast by way of baitworms and their wormweed packing material.

Green crabs have an abundance of biological attributes that make them good invaders. They can tolerate salinities between 4 and 52 parts per thousand (ppt) (20 to 35 preferred) and water temperatures from -2 to 35°C (9 to 22.5 preferred). Like most ecto-therms, the colder their habitat, the slower they develop. Females mature in one to three years and, on the east coast of North America, green crabs can live up to six years (Berrill, 1982). They can survive out of water for up to five days (Klassen and Locke, 2007). Sexually mature females typically bear one clutch of eggs per year, each of which can contain 185,000 fertilized eggs (Grosholz and Ruiz, 1996). In their larval stage, green crabs are free-swimming plankton, transported by surface currents for up to 90 days. Once the larvae settle out of the plankton, they can establish in areas with muddy, sandy or rocky substrate (Cohen, 2001).

Green crabs are opportunistic omnivores, voracious eaters and generalist predators of mollusks, polychaetes, and crustaceans with clams, mussels, and other bivalves being the preferred prey (Grosholz and others 2011). Green crabs are credited with undermining the soft clam fisheries in New England and the Canadian Maritimes (Cohen and Carlton, 1995), but they eat a wide range of bivalves (Moulton and others, 1955). As well as eating prey that people like to eat, green crabs also damage new growth in

beds of eelgrass (Neckles, H.A., 2015). These two undesirable traits have received the greatest attention by those seeking to evaluate the harm done by the spread of green crabs.

A.1 Green crab prevention costs

Several different handling, packing, and shipping approaches are considered as possible ways to reduce the introduction of the green crab and other hitchhiker species from Maine through the live bait trade vector. These included using an osmotic shock approach and alternative packing materials. Blakeslee and others, 2016 describe an innovation aimed at reducing the likelihood that hitchhikers will be conveyed with the wormweed packing material used to ship bloodworms from Maine. They found that soaking the wormweed in tap water for 24 hours killed 85% of the hitchhikers and a more thorough treatment of a freshwater bath followed by a hypersaline bath removed 99% (average abundance) and 93% (average species richness) of the hitchhikers in bloodworm baitboxes. However, since soaking the wormweed in two separate baths would add labor costs, change facility requirements and require dependable sources of both fresh and hypersaline water, this suggestion was not well received by entrepreneurs trading in bloodworms.

In the fall of 2016, Robert Wieland and Jeremy Trombley surveyed bloodworm distributors within 200 miles of Boothbay Harbor, Maine. An important goal, among several, was to estimate the costs of employing the freshwater and hypersaline bath treatments described by Blakeslee and others, 2016. During fieldwork, it became apparent that bloodworms were being shipped from Maine to Europe in plastic trays with no packing material other than a few ice packs. This method seems to have arisen at the request of buyers in Europe who found it too time consuming to remove bloodworms when they were packed in wormweed. Some other distributors were using vermiculite as a packing material. Given suppliers' resistance to the osmotic shock preventative and their willingness to employ a shipping method that solved the problem inadvertently, we undertook an evaluation of the plastic tray (and no wormweed) method. We interviewed four of the largest distributors accounting for, perhaps, over two thirds of the market and none of them shipped bloodworms to Europe in wormweed.

Packing the worms in plastic trays without additional packing was found to be very close in price if not cheaper than using wormweed (Table 1). Vermiculite is a more expensive packing material than either wormweed or the naked in trays method, and it was only used at the buyers' request. Importantly, buyers who have been taking bloodworms in plastic trays for several years appear to be sticking with the practice. On the other hand, distributors still use wormweed to pack bloodworms shipped to California and the mid-Atlantic. Given this, and given the likelihood that treating the wormweed in freshwater would add costs for bloodworm distributors, our study shifted focus to evaluate the plastic tray shipping method.

Table 1 shows cost estimates for two comparable bloodworm shipping methods, one using wormweed and the other shipping worms naked in trays. The cost of the worms themselves constitutes by far the largest cost factor for either method. The differences in total costs between either method is miniscule. Assuming that shipping bloodworms without wormweed packing material is an effective preventative measure for the transfer of hitchhiker species, the monetary cost of prevention of the introduction of the green crab is demonstrated here to be negligible. Additional alternatives that reduce the likelihood of conveying AIS include exposing wormweed to osmotic shock through fresh and salt water baths and substituting newsprint for wormweed were not investigated.

Table 1: Comparison of Bloodworm Packing Practice Costs

		Six 125 trays					Six 125 boxes		
		Number	Unit Cost	Total Cost			Number	Unit Cost	Total Cost
Naked in Trays	Worms	750	0.35	262.5	Wormweed in cardboard	Worms	750	0.35	262.5
	Plastic Tray	6	1.25	7.5		Cardboard boxes	6	0.5	3
	Gelpack & ice	2	0.65	1.3		Paper & seaweed	6	0.7	4.7
	Cover box	1	10.1	10.1		Gelpack	2	0.65	1.3
	Total Product & packing			281.4		Cover Box	1	10.1	10.1
	\$/Worm			\$0.375		Total Product & packing			281.6
				\$/worm			\$0.375		

A.2 Green crab damages

The green crab has been credited with destroying value in New England shell fisheries (Athearn, 2008, Beall, 2014, and MacPhail, 1995) and damaging eelgrass beds and marsh grasses in places where they thrive (Holdredge and others, 2009, Garbary and others, 2004 and Belknap and Wilson, 2014). It is noteworthy that the establishment and growth of green crab populations along the northeast coast of America has continued largely unabated for almost 200 years. Green crabs have been present on the west coast for less than 30 years (Cohen et al., 2001). Not unexpectedly, population densities and economic impacts are further along on the east coast than the west (Yamada and others, 2015).

Lovell and others, 2007 evaluate the magnitude and extent of green crab's economic impact on both the east and west coasts. They limit their impact factors to those affecting commercial bivalve fisheries and eelgrass restoration efforts, though they note that green crabs are also thought to have negative impacts on the winter flounder fishery in New England. They estimate green crabs' annual impacts on east coast bivalve fisheries to be about \$22 million per year. This total is calculated by estimating the magnitude of the clam, mussel and scallop harvest reduction, based on a two-stage predation model. Results in this model are determined in large part by crab densities. They then use implied reductions in bivalves marketed to estimate losses in combined consumer and producer surplus, which is a measure of economic welfare.

In their evaluation of commercial shellfish impacts of green crabs on the west coast, Lovell and others (2007) use a deterministic model of dispersal, since current areas of dense populations there are not yet generating large losses. Their predictions for the future spread of green crabs on the west coast use the GARP (genetic algorithm for rule-set prediction) model over the feasible range of the green crab populations from the Baja to the Aleutian Peninsula. Using those predictions, they expect eventual annual welfare losses in the west coast bivalve market of \$840,000 (2006 dollars).

Using crab densities on the west coast and a literature value for restoring an acre of eelgrass (\$35,417), Lovell and others (2007) estimate annual losses of \$6,152 to \$47,018 to eelgrass restoration efforts in 2006 dollars in California only. In comparison, their estimate of east coast eelgrass restoration annual losses due to green crabs ranges from \$60,150 to \$77,433 (2006\$). The mid-point estimate for both coasts sums to a national average annual eelgrass restoration impact of about \$95,000.

A later study by Grosholz and others, 2011 uses data on shellfish harvest losses from green crab predation from six different shellfisheries, employing catch per unit effort (CPUE) using green crab traps as a proxy for green crab population densities. They correlate CPUE with harvest losses and calculate a linear equation to describe shellfish harvest losses as a function of green crab population density. They evaluate expected harvest losses across west coast areas where green crabs are present (prior to 2011) and, for future losses, they develop a set of predictors for the successful spread of green crabs across the west coast and use these to factor potential damages in those areas. Estimated shellfish harvest losses are negligible, even when extrapolated to expected future costs for the entire west coast. These values range from a low of \$900 to \$1,700 at present to \$45,500 to \$88,000 when green crabs spread further along the coast. However, Grosholz and others ignore costs to British Columbia's shellfish industry and place a low probability on green crabs spread above Grays Harbor, Washington. In 2016, this had already happened, as green crabs are now found in Puget Sound¹.

In a yet more recent study, Mach and Chan, 2014 take a different approach, focusing on value at risk instead of risk-weighted potential losses. Risk-weighted potential losses make sense in the world of finance and insurance where expected values are based on large numbers of bets and probabilistic outcomes. For shellfish harvesters in Puget Sound, the question is a little different. They face a given cost function for bringing shellfish to market, whether wild or aquaculture product, in the absence of green crabs, and a different one in the presence of green crabs. Harvesters operating in the absence of green crabs might discount the "with green crab" scenario by the likelihood of their arrival, but once they are found in an area, the uncertainty reduces to whether their population densities will be high, medium, or low. Given that the green crab is expanding its range in both British Columbia (Gillespie and others, 2015) and Puget Sound, considering the value at risk seems justifiable.

To establish value at risk, Mach and Chan consider three different levels of green crab population densities, based on densities in their natural range in Sweden. At middling densities with medium caloric assumptions, 540,000kgs of shellfish landings, valued \$3.72 million per annum are at risk in Puget Sound. Total value of bivalve harvests at risk in their study ranges from a low of \$80,000 to a high of \$23,800,000, depending on population densities and feeding rates. It needs to be noted that these values do not capture the full economic value lost by these diminished harvests, but represent the value of harvest losses if price stayed constant.

A.3 Green crab control costs

On the east coast of the United States, where the green crab appears to be expanding its range (Roman, 2006), control has proven costly and elusive (Maine: Governors Task Force Report, 2014). On the west coast, working with a less well established population, some success has been achieved in limiting local populations through trapping and other management practices (de Rivera and others, 2007). Costs implied by adopting those control efforts are less well-understood. Even so, there is some opportunity to mitigate environmental damages by way of control or management of the AIS, short of extermination.

Pimentel, 2003, without describing methods or sources asserts a value of \$100 million per year in control costs for the green crab in the United States. This national figure covers a range of circumstances from the east coast, where green crabs are long established and creating significant commercial impacts, to the west coast, where impacts are less further along. More importantly, this figure cannot be related to some specific level of control in terms of crab population impacts or limits on environmental damages.

¹ (<https://www.nrdc.org/stories/puget-sound-braces-worst-green-crab-plague>, accessed on 1/30/2017).

Lovell and others, 2007 use California's Green Crab Management Plan (Grosholz and Ruiz, 2002) to estimate green crab management/control costs of roughly \$285,000 per year during 2007 to 2010 in California. Oregon and Washington both have biennial budgets projecting that over the next two years, Oregon will spend \$1.994 million and Washington \$2.414 million on control of AIS, generally. Allocating those budgeted funds to green crabs as a subset of AIS of concerns requires greater detail on budgets of those states' agencies than is readily available.

The work of de Rivera and others (various years) in California provides data that can be used to estimate the effectiveness of different technologies for controlling green crabs, locally. Their work focuses on Bodega Bay but also considers other areas along California's coast where the green crab has established. After an initial survey and population estimate of green crabs in various sub-areas in Bodega Bay, the researchers used a range of trapping and gathering practices and for one year pursued an intensive effort to remove as many adults as possible. They followed this with year-after population assessments both within Bodega Bay and at points where no control efforts were being employed. They found a significant decrease in estimated green crab populations in Bodega Bay, which was not coincident with population declines in other areas. Since up-current larval sources were not a part of this control effort, their gains were relatively short-lived when new crabs recruited to Bodega Bay's green crab population.

Evaluating the costs of control efforts was not a part of the de Rivera and others research. However, since they do describe their effort in terms of traps employed over time, an estimate for cost per unit of effort allows one to impute a cost range. A cost estimate for green crab trapping is taken from the literature, following St-Hilaire and others, 2016. They propose a cost-calculating tool for fishing green crabs in the waters around Prince Edward Island (PEI), in the Northwest Atlantic. Their goal is to estimate the price which at some level of catch per unit effort would provide a motivating benefit for fisherman. They use partial budgeting and cost information gathered from PEI fishermen to create cost scenarios for situations such as when there is no other harvest targeted (dedicated fishery) and when green crabs are fished in connection with some other fishery (auxiliary fishery). They consider several different gear types.

St-Hilaire and others' cost calculating tool can be used as a first approximation for estimating what it might cost to replicate a level of green crab population reduction similar to that achieved through research efforts at Bodega Bay. Our interest is to obtain a value for what it might cost to have fishermen fishing at a level of effort comparable to that employed by researchers in their first year of control effort in Bodega Bay (July 2006 - October 2007). To do this, we run the cost side of St-Hilaire and others' green crab cost calculator and factor the results by the level of effort employed by de Rivera and others.

A first adaptation to the cost calculator is to convert from Canadian to US dollars. Increasing traps per day to 70², and trapping days to 50 and imagining a level of effort 22 percent greater than the number of traps utilized in the Bodega Bay study, but otherwise using the cost estimates in St. Hilaire's cost calculator³, delivers a cost of between \$21,000 and \$27,000 to achieve population reduction similar to that achieved under de Rivera and others' experiment. The increase in minnow and collapsible trap effort balances the other harvest methods investigated by de Rivera and others (and, contributing to stock decrease) but not costed in St-Hilaire and others. A matrix of cost categories and cost estimates under both the dedicated and auxiliary fishing scenarios is appended as Appendix 1. To employ this estimate for a California-wide measure, an estimate of green crab habitat is needed.

² Personal conversation with Thomas Therriault, Fisheries and Oceans Canada.

³ We also replaced St-Hilaire and others' trap and fyke net costs with Fukui (50%) and minnow trap (50%) prices.

The green crab control costs enumerated above are for one year in one small embayment along a habitat range that extends from Baja, Mexico to Alaska. Moreover, recruitment continues when up-current larval sources persist (de Rivera, 2010). An effective effort to reduce adult green crabs along the southern origins of their Pacific coast range might have recruitment effects in Washington, Oregon, British Columbia and Alaska (Yamada and others, 2015). But no characterization of the costs of trying to beat back the green crab on the Pacific coast is available in the economic literature and creating a credible estimate is beyond the scope of this study.

A.4 Green crab economic impacts

Using the \$840,000 estimate of annual welfare losses from bivalve harvests on the west coast from Lovell and others, 2007, the present value of that loss calculated at a social discount rate of 3 percent over one hundred years is \$26.54 million. Using the lower estimate from Grosholz and others 2011 of \$88,000 per year, the present value of this loss over a 100 years at 3 percent is \$2.78 million. The middling estimate from Mach and Chan of assets at risk of \$3.72 million per year, when considered as a constant stream of losses, has a present value of \$117.55 million and their estimate of \$23.8 million in annual losses generates a present value loss of \$752 million. Any of these numbers is significantly larger than the near zero predicted costs of prevention. Given uncertainty about costs and effectiveness of controlling green crabs before they become more firmly established on the west coast, we cannot quantify how these damages might be mitigated by management, except to say that they might be reduced.

B. Periwinkles

Three species of periwinkles were found in bloodworm bait boxes in Fowler and others, 2016. One, *Littorina obtusata* is not a known invader anywhere and is not treated here. Among the other two, *Littorina littorea*, is larger and is a favored food for humans, while *Littorina saxatilis*, is smaller and generally only eaten by crabs and shore birds. *L. saxatilis* is native to the east coast of North America, and *L. littorea* was brought to the new world from Europe, and both were found frequently in sampled bloodworm baitboxes in the study reported by Fowler and others, 2016. Field data from that study shows that 611 live specimens of *L. saxatilis* were found in 118 baitbox samples and 132 live *L. littorea* were found in that same (total) number of samples. Both species were found more frequently in field samples of wormweed.

Carlton and Cohen, 1998 report data suggesting that the two populations of *L. saxatilis* found in San Francisco Bay were introduced via the trade in Atlantic coast baitworms. In 14 years, the two populations reported in 1998 had become nine (Cohen, 2012). All of these sites were near popular fishing spots or boat ramps. *L. saxatilis* broods its larvae instead of releasing them into the water column as other periwinkle species do. Although this slows their spread over a wide area, this characteristic also increases prospects for creating a breeding population from a small number of invaders. In addition to potentially competing with native, west coast periwinkles, *L. saxatilis* is a vector for the parasitic trematode, *Microphallus sp.* (Cohen, 2012).

L. littorea might also have arrived via the live bait trade, but, because it is a favored food source, it is also possible that introductions are made purposefully by people interested in growing their own snails for eating (Cohen, 2012).

B.1 Prevention costs for *Littorina sp*

Since both periwinkle species harbor in the wormweed in which bloodworms are packed, and not among the worms themselves, it is highly likely that shipping bloodworms without wormweed, as

discussed in the context of the green crab, would significantly limit their prospects for introduction via the livebait trade. As also previously discussed, direct money costs of this practice are very near to zero.

B.2 Periwinkle Damages

The money value of environmental damages owing to an invasion by periwinkles is not readily apparent. In the case of *Littorina littorea*, which people eat, there will be an economic welfare gain from introducing it, if it provides food for those who want to eat it at a lower cost than periwinkles brought from further away. On the other hand, field experiments suggest that *L. littorea* has modified or reduced mudflat and saltmarsh habitat in its Atlantic coast range (Cohen and others, 2001). Lubchenco, 1978 showed that the introduction of *L. littorea* shifted the composition of algae on the shore in New England. At high densities, *L. littorea* can graze the intertidal shore down to bare rock. The consequences of grazing by *L. littorea* are much less dramatic on European shores than in North America because of the presence of large, herbivorous limpets in Europe that are absent in North America (CABI).

Littorea littorea can physically alter salt marsh habitats and cobbled beaches by affecting sediment accretion (Bertness, 1984). As *L. littorea* browse cobble surfaces, they remove (“bulldoze”) sediment from these surfaces. Grazing by *L. littorea* also reduces the amount of physically benign habitat at low tide for small organisms by removing leafy and filamentous green, red, and brown algae, which are otherwise often found in mats or turfs on the shore. This grazing also reduces recruitment of many benthic intertidal organisms and larger sessile organisms (e.g., rockweeds). *L. littorea* on the east coast of North America partly displaced the mud snail *Ilyanassa obsoleta* from mudflats, which may have had effects on the composition of infauna of mudflats (Brenchley and Carlton, 1983).

As noted above, *L. saxatilis* is a vector for the parasitic trematode, *Microphallus sp.*, and it affects algal bloom dynamics (Lotze and Worm, 2000). These effects can change ecosystem function.

B.3 Periwinkle control costs

So far, it has not been difficult to control introductions of exotic *Littorina sp.* on the west coast. Cohen (2012) reports costs of \$24,600 for removing *L. littorea* from three different sites in San Francisco Bay. Chang and others 2011 attribute *L. littorea*'s failure to establish on the west coast to its reproduction and recruitment strategy. Specifically, “populations of *L. littorea* larvae released into the water column may end up settling in locations far from congregated populations of the snail—thus impeding the creation of self-sustaining, established populations in this region.” *L. saxatilis*, however, does not send its larvae into the water column to be carried where currents may take them, but rather carries them until they crawl away. Given this strategy, *L. saxatilis* appears to be both more invasive and more easily eradicated from specific sites than *L. littorina*.

B.4 Periwinkle economic impacts

While the introduction of *L. littorea* and *L. saxatilis* to a novel location might generate significant ecological disruptions, in order to place a value on those disruptions some measure of the value of the native, intact ecosystem is needed. This value does not need to be an impact with a monetary component that is visible in the national accounts. National accounts do not capture value derived by people from the existence of natural phenomena, except when those phenomena engender spending decisions (e.g., I not only like it, I am going to go camp, or fish, or sight-see there). The fundamental value of an existing ecosystem is not included in national accounts, hence, neither is its loss when it is disrupted. Evaluating the loss of important ecosystem characteristics resulting from the successful introduction of an AIS such as *Littorina sp.* requires targeted study. There is an extensive economic literature on such losses (Arrow and others, 1993, Cameron and Englin, 1997, Whitehead, 1993).

If there were health impacts from the trematodes that *L. saxatilis* can harbor, a valuation of that loss would include the money costs of treating the illness plus the value of consequent lost productivity. But, if the trematode only carries health impacts for the birds or fish that eat the snails harboring it, then the question requires an assessment of people's willingness to pay to protect those birds from this introduced threat, or their willingness to accept monetary payment as compensation for the (unwanted) impacts. While some economists hold that it is possible to get useful and policy-relevant information by surveying people's stated preferences and contingent valuations (Carson, 2012a), others are less sure (Hauseman, 2012). Economists have developed elaborate experiments for ascertaining preferences which they then use to estimate non-market values (Carson, 2012b), but best practices have not always been employed. If government policy provides a satisfactory arbiter for this question, the incorporation of the NOAA Contingent Valuation Expert Panel's report into the Federal Register (Arrow and others, 1993) supports the pursuit of nonmarket passive use values when researchers adhere to rigorous technical guidelines to impute those values.

***C. Ascophyllum nodosum ecad scoporides*(Wormweed)**

Miller (K.A.) and others, 2011 point out that "Seaweeds are the engineers of near shore environments, providing primary production and habitat structure for intertidal and sub-tidal communities. Humans use more than 200 species as the basis of an international industry in food and phycocolloid products that has doubled since 1984 and was valued at U.S. \$6.2 billion for 1994-5." Given that seaweeds are important to both natural systems and the human economy, it would be useful to be able to predict the effects that invasive species of seaweeds will have in new environments. Unfortunately, making such predictions is not so straight forward.

Williams and Smith, 2007, review 407 global seaweed introduction events and although ecological effects were studied across only 6 percent of introduced species, those studies show mostly negative effects or changes to the native biota where introductions occurred. Wormweed was not included among their list of invasive seaweeds, but a closely related species was. *Fucus evanescens* reduced species abundance and diversity as an invader.

C.1 Wormweed prevention costs

Clearly, if wormweed were not used to pack bloodworms, the baitbox vector would have a reduced likelihood of introducing wormweed to new ecosystems, or to ecosystems with ongoing efforts to eradicate the macroalgae. As for the many hitchhikers harboring in wormweed when it is used to pack bloodworms and other live bait, the direct cost of avoiding introduction of this potential invasive species is very near zero - except to the extent that retailers or final users believe that wormweed is integral to the product.

C.2 Wormweed damages

In its natural range across much of the northern Atlantic coasts, wormweed is a valuable resource, harvested for fertilizer, feed, and supplements, in addition to its perceived utility as packaging material for live shipments of marine invertebrates. When it is attached to rocks, it sways in slow wave action and provides habitat for a wide range of animals. If this intertidal ecosystem function was not already filled along the west coast of America, it is possible that the introduction of wormweed to San Francisco Bay would simply increase species richness there. However, California has other seaweeds occupying that niche and it is likely that these species would compete (See: <http://sanctuaries.noaa.gov/science/sentinel-site-program/channel-islands/invasive-species.html>; Accessed, 3/13/2017).

C.3 Wormweed control costs

Miller and others, 2004, report the occurrence and eradication of wormweed in San Francisco Bay in 2002. A mat of free floating wormweed believed to have grown from a smaller plant, was found during a survey of the intertidal zone. The decision to eradicate the algae was taken rapidly and efforts were made to remove it. At less than \$5,000, removal did not impose a high cost. Unfortunately, while that effort seemed to be successful, substantial mats of wormweed were again found at Bay Farm Island in 2008 (Miller (A.K.) and others 2011). This mat was also removed, but apparently this species continues to be reintroduced to San Francisco Bay. Since the type of wormweed found is the same type used for packing bloodworms, the live bait trade seems an obvious suspect for facilitating these continuing introductions.

C.4 Wormweed economic impacts

When the wormweed used as packing for bloodworms is discarded in marine environments that are not a part of this macroalgae's natural range, there is some likelihood that it will be successfully introduced. Cohen and others, 2001 estimate that over 9 metric tons of wormweed are imported into San Francisco bait shops each year as packing for Atlantic baitworms and that three tons of that material is dumped into San Francisco Bay. While it appears that early in its introduction, it is a simple matter to remove the introduced material and to prevent its becoming established, continued introductions increase the likelihood that eventually wormweed will be successfully introduced on the west coast. Certainly, limiting its use as a packing material would reduce those likelihoods.

The economic impacts of a successful introduction of wormweed are uncertain. If it establishes successfully and out-competes a native intertidal macroalgae, it could change the ecosystem in ways that are difficult to predict. If it provides habitat for other invasive species it could facilitate the establishment of AIS. It could become a nuisance or a hazard to navigation. The salient point is that the successful introduction of wormweed holds uncertain consequences for both the coastal ecosystem and humans who use or who value the existence of that ecosystem. While there is some prospect that these consequences will be positive (i.e., perhaps it will provide a new source of habitat for some native creature), there is also a likelihood that they will be negative and, given the likely irreversibility of a successful establishment, it is reasonable to want to avoid such an introduction, if practical.

III. Policy Implications of Invasive Species Spread Through the Bloodworm Trade

A. Preventing successful introductions

Benefit-cost analysis helps to answer the question, how do we end up with the most benefit? But, benefit-cost analyses depend on the context in which benefits and costs happen. If consideration is limited to the exchange of live bait, which provides jobs for some and recreational fishing pleasure for others, benefits of the trade in bloodworms are thought to be highest when buyers and sellers are free to discover them, unimpeded by taxes or regulatory constraints. However, the evaluation of costs changes if we expand the context to include consideration of the industry as a vector for the introduction of aquatic invasive species (AIS). As argued in the introduction, the AIS vector aspect of the bloodworm trade generates probabilistic external costs which we would like to minimize.

Since probability plays such a large role in the problem of AIS introductions, minimizing AIS costs requires expectations for likelihoods governing three different aspects of the problem. Likelihoods of preventing an introduction tend to rise with investments in prevention but as more prevention is bought

additional prevention benefits become more expensive. If prevention fails, then the costs of damages caused by AIS will be shown, after the fact. To estimate those costs before they occur, probabilities must be taken into account. These costs may include commercial effects as in reduced landings of commercial species, increased health care costs, or non-market costs as in reduced recreational or passive use values. And, thirdly, if damages can be reduced by investments in control, then it is necessary to account control costs alongside the (reduced) costs of damages. As with the cost of damages, before fact these costs can only be estimated.

This paper has considered only three of the over 150 distinct taxa found in wormweed packing material in bait boxes (Blakeslee and others, 2016), but this sample provides examples which can be weighed individually against the benefit generated by the live bait trade. Ultimately, a responsible decision-maker would sum the expected costs of prevention, damages and control over all of the potential invaders enabled by the vector. But since prevention costs are so low for the bloodworm trade, even taking each invader individually it takes very little expected damage or control cost to show prevention to be the best choice for minimizing the cost of this externality.

Because it is easier to provide peer-reviewed references for hitchhikers that have been studied, and being an invader increases the likelihood that a hitchhiker has been studied, our study sample of hitchhikers is biased. But, it makes sense to start with the most costly invaders when trying to account costs of the externality. The potentially larger problem is limited information with respect to the set of potential hitchhikers. The sample taken from Fowler and others, 2016 in this paper does not include microalgae, which could be another costly class of invaders. Haska and others, 2012 found a number of microalgae in wormweed baitworm packing material, several of which are potentially toxic. Especially in association with nutrient over-enrichment, such algae can generate health impacts for both aquatic species and humans. This limited information aspect of the problem argues for including uncertainty in the benefit cost analysis of AIS vectors.

Even though the cost of preventing AIS introductions from the bloodworm trade is very low (if we consider retailers and fishermen indifferent to how they are packed), live bloodworms continue to be shipped in wormweed to both mid-Atlantic states and to the west coast. Bloodworm distributors have shown themselves willing to accommodate demand for their product packed without wormweed if customers ask for it that way. If policymakers in areas that still receive bloodworms packed in wormweed had a mandate to minimize AIS costs, it is reasonable to expect that they would ask retailers to find a new medium for this purpose.

B. When prevention fails

Although prevention appears to be the most cost-effective investment for minimizing AIS costs from the bloodworm trade, this loses relevance for specific invaders after they are successfully introduced and established. Once an invader becomes established, choices reduce to living with its effects or trying to reduce those effects through some type of control practice, or, some combination of the two. Among our sample of hitchhikers in bloodworm bait boxes, the green crab presents the clearest example of both commercial and non-monetary damages and control costs.

The green crab probably came to North America from Europe in the bilges of wooden ships, but it is likely that its introduction to the west coast was via the live bait vector (Cohen and others, 2001). The fact that it has been established on the east coast for a long time but has only recently been moved to the west coast is material. Two hundred years after its introduction, the green crab has established itself across a wide range of niches along the Atlantic coast of North America, from Delaware to Newfoundland (NEANS). Control efforts have only been attempted during the last quarter of that period

and these have met with mixed but, mostly, limited success. In New England, the green crab is dominating coastal marine and estuarine ecosystems (Beal, 2014). In Canada, where the green crab has expanded northward recently, efforts are being made to create markets for it as bait for lobster and, when recently molted, as a food delicacy for humans. St-Hilaire (2016) reports costs and break-even prices for various green crab products, but those markets are not yet taking off.

On the west coast, where the green crab is less thoroughly well established, control efforts start from a much lower population than that faced on the east coast. Catch rates there are generally a small fraction of east coast catch rates. This is a double-edged sword with regard to control costs. If we consider a baited trap in green crab habitat in New England that rapidly fills with green crabs which harvesters are reimbursed to deliver to a composting site, it will not take a high price per pound to generate a sufficient return to motivate fishing effort. On the west coast, where catch rates are much lower, it would take a much higher price per pound to motivate fishing. Depending on how we calculate control costs, these will be higher, the less widely established the invader. Yet, the alternatives to paying that control cost includes a scenario wherein green crabs become sufficiently numerous on the west coast to justify lower per unit control costs. This is not a desirable outcome.

Approaches for reducing the direct costs of managing green crabs have consequently included utilizing people's willingness to solve shared problems through volunteerism. De Rivera and others, 2010 describe getting help from renters and owners of water-facing condominiums, working crab traps and delivering their catch to collection points. Similar efforts were tried in Newfoundland (McKenzie and others, 2013) where local fishermen along with community and school groups were enrolled to support a green crab composting program. In both cases, activities were motivated by paid staff who may not be available across time and coastal polities. These efforts notwithstanding, there does not appear to be a large scale effort to control the green crab invader on the west coast. Instead, monitoring is ongoing of the green crab's progress in stock and extent.

The other two hitchhikers discussed in this paper do not threaten the type of commercial impacts that green crabs do, though they will likely generate negative impacts with respect to economic welfare. If exotic periwinkles out-compete native snails on the west coast, if they change ecosystem relationships that lead to dramatic changes in ecosystem function, or if they bring new vectors for harmful pathogens, these changes will reduce the utility of those who enjoyed the ecosystem the way that it was before they were successfully introduced. These effects can be ascertained by research into people's valuation of existing ecosystem characteristics, and the loss implied by hypothetical changes in those characteristics. To the extent that these ecosystem characteristics motivate recreation and sightseeing, it is possible to estimate demand for such visits denominated in dollars. Human health impacts can also be valued in dollar terms.

C. Managing the vector cost effectively.

Following Olson and Roy, 2005, an optimal solution to the question, *how best to manage potential AIS introductions from the live bait trade* requires an estimate for the likelihood of introduction and establishment success, an estimate of the cost effectiveness of spending for prevention and control, and an estimate of the cost of environmental damages. For much of this problem, one needs to know what is going to happen before it happens. One then must associate a cost or benefit with those outcomes. Economists use available data and probability theory to establish likelihoods in making these estimates but those probabilities often are not easily explained. This is not the biggest problem, however.

The way in which we have addressed the question of managing potential AIS introductions from the live bait trade presumes a "responsible party" empowered to act to achieve whatever optimal solution is

discerned, given the cost minimization problem. While this often is not the case, it is required as an intellectual construct to motivate the calculus. Numerous entities exist to monitor the progress of invasive species, but taxing or regulating the markets that cause introductions and invasions is outside the job description of most those. Researchers and resource managers are sometimes consulted by policy-makers seeking a way to deal with current, visible problems, but problems that depend on probabilistic predictions tend to garner less attention. Consequently, it is not clear how one makes use of the observation that, with respect to the AIS vector aspect of the bloodworm trade, prevention in the form of wormweed-free packing material would easily pay for itself in damages foregone.

If the money cost of readying bloodworms for shipment (F.O.B.) were the sole determinant for how bloodworms are shipped, distributors in Maine might be indifferent to a rule preventing the use of untreated wormweed in bloodworm shipments. If the "naked in trays" method were adopted, no additional cost would be incurred, distributors would not have to gather or buy wormweed and an environmental concern could be mitigated. The "naked in trays" method might even improve efficiency in the industry by helping to standardize processes. However, packing costs are not the sole determinant for how bloodworms are shipped.

Decisions about packing bloodworms for shipment are driven by demand for the product. Shipments of bloodworms to the mid-Atlantic states and to California continue to be packed in wormweed because that is the way fishermen expect to see them. On the other hand, social science research supported by SERC and Maryland Sea Grant considered both producers and users of bloodworms found that a high percentage of fishermen do care about the potential hazards of AIS (Citation for Paoliso and Trombley?). Eighty percent of their sample was very concerned about the quality of the fishing environment and between 43 and 51 percent were concerned about AIS. These fundamental concerns might be leveraged to motivate changes in demand for bloodworms and how they are packed.

Banning the use of wormweed at the source would solve the problem of AIS hitchhikers that harbor in that material. However, to the extent that worms in wormweed are what buyers want, banning the use of wormweed would likely restrict demand for bloodworms, lowering producer prices and reducing quantities traded. Requiring bloodworm distributors to use wormweed that had been exposed to two different water baths would increase distributor costs, shifting the supply curve up and resulting in higher market prices and smaller quantities of worms traded. On a normative basis, it can be argued that policy-makers in Maine should protect the rest of the world from potential AIS being conveyed through the bloodworm trade. If those policy-makers were liable for the costs that are being imposed by AIS conveyed through the bloodworm trade, it is likely that they would implement the most cost-effective prevention policies available.

There is no mechanism for imposing the costs of AIS introductions on the source of those introductions. Given that, and given that the losses discussed in this paper reside most immediately with citizens in the receiving areas, it is up to policy-makers in receiving areas to find a way to protect their aquatic environment from AIS introduced through the bloodworm trade. Distributors in Maine have shown themselves willing and able to ship bloodworms without wormweed. If policymakers in California or the mid-Atlantic states decided that preventing further introductions from the bloodworm trade had value, they might ask or require consumers and retailers to use wormweed-free shipping.

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Appendix 1 - Adapting a Prince Edward Island Green Crab Harvest Calculator for Bodega Bay, California

St-Hilaire and others, 2016 use cost information gathered from fishermen and lobstermen working on Prince Edward Island to develop a cost calculator which associates harvest costs with varying levels of effort. They use several scenarios to develop costs, relative to effort. The most cost effective way to harvest green crab under their analysis is as an auxiliary fishery, next to their eel fishery. Under the assumption that no one would buy a boat to harvest green crabs, vessel capital costs are not a part of their analysis. Trap costs were factored by their useful life in a straight line depreciation and fuel, bait and labor costs were based on reports from operators.

In transferring this cost calculator to California, we are implicitly assuming green crab harvesters there would be working at least one other fishery and that alternative harvest effort would leave sufficient slack for operators to take up a green crab fishery. This application of the tool to California makes several adaptations, however. Whereas the scenarios in St-Hilaire and others, 2016 assume the use of 60 baited traps and fyke nets which are set for eel but which also catch green crabs, in adapting the tool to California, we assume harvesters would use fukui traps and minnow traps and that 70 of these would be worked each day. The values showing in Table 1 are reported in Canadian dollars.

Table 1: Harvest costs adapted from St. Hilaire and others, 2016

	auxiliary <i>Scenario 2</i>	dedicated <i>Scenario 2</i>
i) Days in Season	50	50
ii) Bait cost per lb of bait (CAD)	0.50	0.50
iii) Average bait per trap (lb)	0.50	0.50
iv) Cost per Fukui/minnow trap (CAD)	64	64
v) No. of traps per day per boat	70	70
vi) Average crabs caught per trap per day (no.)		
vii) Labour per day (man hrs)	6	6
viii) Labour cost per man hr (CAD)	20	20
ix) Other charges per trip (i.e gas, other...)	25	120
x) Average weight per crab (g)	40	40
xi) Cost of other gear used in fishing	300	300
xii) Cost of boat modification	1000	1000
xiii) License fee	0	0
xiv) Interest rate on bank deposits (%)	3	3

The difference between the auxiliary scenario and the dedicated scenario derives from the other charges per trip, which rise in the case of the dedicated fishery, due to trip costs being allocated to the green crab fishery solely. The values in **Table 1** are gathered and factored in **Table 2**.

Table 2: Scenario costs for Bodega Bay - level effort.

Costs	<i>Economic life (yrs)</i>	auxiliary	dedicated
Fixed capital			
Fukui/Minnow cost (CAD)	10	\$4,495	\$4,495
Other gear (CAD)	5	300	300
Boat modifications (CAD)	10	1000	1000
Total fixed capital cost		\$5,795	\$5,795
i) Depreciation on fixed capital (CAD)		\$609	\$609
ii) Interest on fixed capital (CAD)		\$174	\$174
iii) License fee (CAD)		0	0
Total fixed cost per season (i+ii+iii)		\$783.30	\$783.30
Variable Costs			
i) Bait cost (CAD)		875	875
ii) Labor cost (CAD)		6000	6000
iii) Other costs (CAD)		1250	6000
Total variable costs per season (i+ii+iii)		\$8,125	\$12,875
Total Variable cost per Season @12,000 trap days		\$27,857	\$36,182
US Dollars (At \$1US = 1.32385 CAD)		\$21,042.52	\$27,331.21

Total variable costs per season are based on 50 days of effort at 70 traps per day, or 3,500 trap days. However, the harvest effort at Bodega Bay that achieved a significant short term decrease in green crab populations required approximately 12,000 trap days. The break-even tool was developed to inform an individual operator what price they would need to receive at various catch rates and still pay their costs. It is being employed here to ascertain the cost of a certain level of fishing; that is, the level of fishing that generated a decline in green crab populations in Bodega Bay as reported by de Rivera and others, 2010. To ramp the cost estimate up to 12,000 trap days we factor \$8,125 by 3.43, the ratio of 3,500 to 12,000, generating a cost of 36,182 Canadian dollars, or 27,331 US dollars.

This exercise is second-best to gaining market information from fisheries in California. For working 12,000 trap days in the Chesapeake Bay, these values seem low.