



Invasion of Japanese eelgrass,
Zostera japonica in the Pacific
Northwest:
A Preliminary Analysis of
Recognized Impacts, Ecological
Functions, and Risks

Prepared for:

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1 Introduction

Japanese (dwarf) eelgrass, *Zostera japonica*, is thought to have been introduced to the West coast of Canada and the U.S. from oyster shipments in the 1930's. It has since colonized and spread into many embayments along the West coast. In Washington State, it's role as an ecosystem engineer has not historically been differentiated from that of the native species, *Zostera marina* by state regulatory agencies, and as such was afforded regulatory protection under existing 'no net loss' provisions for native eelgrass, despite its non-native status and propensity in many locations to radically alter the intertidal environment and the biological communities inherent within those environments. The Washington State Department of Fish and Wildlife, however, has recently acknowledged this discrepancy and has instructed the state legislature to remove protections afforded this species, primarily in recognition of its impacts to shellfish growing regions of the state where the species has colonized and effectively displaced formally productive shellfish growing grounds (P. Anderson, WDFW Director, Personal Communication to Brian Blake, House Chair, Agricultural and Natural Resources Committee, Feb. 2011).

While ecological functions and benefits can be prescribed to virtually all species, including non-native invasive species, the habitat and biological community changes that result from the establishment and spread of invasive species can adversely impact native species, and the socioeconomic fabric of local communities. Indeed, invasive species are broadly recognized as the second leading cause of threatened and endangered species loss after habitat destruction (Pimental et al. 2000). The broad recognition of the impacts of invasive species—estimated by Pimental et al. (2000) at greater than \$137 billion per year to the U.S. economy, exclusive of lost ecosystem services costs, led President Clinton to sign Executive Order 13112 which outlined requirements for the federal government to develop a national invasive species management plan and establish a national invasive species council, and instructed federal agencies to cooperate in coordinated responses to address invasive species risks.

This white paper reviews published and unpublished information on Japanese eelgrass, and examines its risk potential using an invasive species risk assessment paradigm initially developed by the Aquatic Nuisance Species Task Force, and applied to address risks from other introduced aquatic species (Orr and Fisher, 2009). This paradigm essentially evaluates invasive species risks in two distinct analyses: (1) an assessment of the probability of organism establishment, and (2) an assessment of the consequences of establishment. The former analysis considers pathways for introduction, the probabilities for entry through the pathways, and the potential for colonization and spread. The latter 'consequence' analysis considers the economic, environmental and social (cultural) impacts possible from the colonization and spread of the species. While this paper clearly focuses more on the latter—the consequences of establishment (i.e., as the species is already well established in numerous locations), assessing the probability of further establishment at refined spatial scales is still necessary as numerous embayments exist where it has not become established yet suitable conditions are available.

In some states such as California active eradication efforts are ongoing, consistent with that state's Strategy 4a of their Aquatic Invasive Species Management Plan (CRA & CDFG 2008). Clearly, from a management perspective, it is important to understand the mechanisms that have led to its establishment so that further spread can be addressed with appropriate

management measures where appropriate. It is equally important to consider how the species should be managed in those areas where it has become firmly established and spread, particularly where economic and environmental impacts have been identified. This strategy is in keeping with the broad recognition of the harm invasive species are causing, and is wholly consistent with the provisions of the National Invasive Species Management Plan (NISC 2008), as mandated by Executive Order 13112, which expressly directs federal efforts to prevent, control and minimize invasive species and their impacts.

2 Probability of *Zostera japonica* Establishment: Pathway of Entry, Ecology, Habitat

Zostera japonica is presently distributed from Vancouver Island, British Columbia to Humboldt Bay, California (McBride 2002). It generally occurs higher in the intertidal (0.1 to 1.5m mean lower low water [MLLW]) than native *Z. marina* (generally 0.6 m MLLW and below), colonizing open tidal mudflats and sandflats within sheltered bays and inlets of the Pacific Northwest (Ruesink et al. 2010). Aside from the development of monocultures in the upper intertidal zone, there are also reports of mixed beds in the transition zones between the two eelgrass species (0.3 to 0.6 m MLLW), and even range extension of *Z. japonica* into *Z. marina* beds (Harrison 1982, Thom 1987, Bulthuis et al. 2005, Ruesink et al. 2010). Since its introduction to Washington State, *Z. japonica* has spread along Washington's outer coast and throughout northern and central Puget Sound (Table 1).

Zostera japonica is thought to have been introduced to Washington State with shipments of Japanese oyster (*Crassostrea gigas*) spat in northern Puget Sound in the 1930s (Bulthuis et al. 2005, Mumford 2007), and subsequently observed on the Washington State coast in 1957 (Posey 1988; Harrison 1982). *Z. japonica* may have been used as packing material for the Japanese oyster stock, with the eelgrass being dumped into bays after removal of the stock (Harrison and Bigley 1982 as cited by Merrill 1995); however, this cannot be confirmed definitively. It is also possible that seed which hitchhiked on the oyster shipments may also have been responsible for the introduction.

The Willapa Bay and Grays Harbor ecosystems share many similarities with the other smaller areas *Z. japonica* has colonized in Washington State, although due to their size, the ecosystems support a greater diversity of species. These habitats, as well as other coastal habitats of Washington where smaller colonies of *Z. japonica* have established, support several priority species, some of which are listed as threatened or endangered (T&E) under the Endangered Species Act (ESA).

Region	Location	County	Source
Canada-USA border	Bellingham Bay	Whatcom	Harrison and Bigley 1982
	Chuckanut Bay	Whatcom	Harrison and Bigley 1982
	East of Ferndale	Whatcom	Gaeckle et al. 2009
	Birch Bay	Whatcom	Harrison and Bigley 1982
	Semiahmoo Spit	Whatcom	Gaeckle et al. 2009
	Drayton Harbor	Whatcom	Gaeckle et al. 2009
	SE of Cherry Point	Whatcom	Gaeckle et al. 2009
San Juan-Strait of Juan de Fuca	Eastsound County Park (Orcas Island)	San Juan	Gaeckle et al. 2009
	North Side of Crane Island	San Juan	Gaeckle et al. 2009
	Picnic Cove	San Juan	Gaeckle et al. 2009
North Puget Sound	Padilla Bay	Skagit	BMNHC 2006, Gaeckle et al. 2009
	Samish Bay	Skagit	Gaeckle et al. 2009
	Similk Bay	Skagit	Gaeckle et al. 2009
	North Possession	Island	Gaeckle et al. 2009
	Useless Bay (Whidbey Island)	Island	Gaeckle et al. 2009
	Ebey's Slough	Snohomish	BMNHC 2006
	Hat Slough	Snohomish	BMNHC 2006
	Jetty Island	Snohomish	BMNHC 2006
	Tulalip Bay	Snohomish	Gaeckle et al. 2009
	Snohomish Delta N	Snohomish	Gaeckle et al. 2009
	Edgewater, Possession Sound	Snohomish	Gaeckle et al. 2009
Kilisut Harbor	Jefferson	ENVIRON 2009	
Hood Canal	Oak Bay	Jefferson	Gaeckle et al. 2009
	S. of Tala Point	Jefferson	Gaeckle et al. 2009
	E. of Squamish Harbor	Jefferson	Gaeckle et al. 2009
	N. of Thorndyke Bay	Jefferson	ENVIRON 2009, Gaeckle et al. 2009
	Dabob Bay	Jefferson	Gaeckle et al. 2009
	S. of Long Spit	Jefferson	Gaeckle et al. 2009
	Quilcene Bay	Jefferson	Gaeckle et al. 2009, USFWS 2009
	Toanados Peninsula	Jefferson	Gaeckle et al. 2009
	Dosewallips	Jefferson	Gaeckle et al. 2009
	N of Port Gamble	Kitsap	Gaeckle et al. 2009
	Warrenville	Kitsap	Gaeckle et al. 2009
	Anderson Cove	Kitsap	Gaeckle et al. 2009
	Stimson Creek	Mason	Gaeckle et al. 2009
	Annas Bay	Mason	USFWS 2009
	Lynch Cove	Mason	Gaeckle et al. 2009
Forest Beach	Mason	Gaeckle et al. 2009	
Central Puget Sound	Sinclair Inlet	Kitsap	USFWS 2009
	Agate Pass Bridge SE (Bainbridge Island)	Kitsap	Gaeckle et al. 2009
	Murden Cove (Bainbridge Island)	Kitsap	Gaeckle et al. 2009
	Quartermaster Harbor	King	BMNHC 2006
	Tramp Harbor (Vashon Island)	King	Gaeckle et al. 2009
	Paradise Cove (Vashon Island)	King	Gaeckle et al. 2009
	Poverty Bay	King	Gaeckle et al. 2009
	Dumas Bay	King	Gaeckle et al. 2009
	Piner Point (Maury Island)	King	Gaeckle et al. 2009

Region	Location	County	Source
South Puget Sound	North Bay, Case Inlet	Mason	USFWS 2009a
	Taylor Bay, Case Inlet	Mason	ENVIRON 2009
	Harstine Island, Case Inlet	Mason	ENVIRON 2010
	Totten Inlet	Thurston	ENVIRON 2009
	Burley Spit, Carr Inlet	Pierce	Gaeckle et al. 2009
Washington Coast	Willapa Bay	Pacific	Harrison and Bigley 1982, BMNHC 2006
	Grays Harbor	Grays Harbor	Harrison and Bigley 1982, BMNHC 2006
Canada-USA border	Bellingham Bay	Whatcom	Harrison and Bigley 1982
	Chuckanut Bay	Whatcom	Harrison and Bigley 1982
	East of Ferndale	Whatcom	Gaeckle et al. 2009
	Birch Bay	Whatcom	Harrison and Bigley 1982
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	Annas Bay	Mason	USFWS 2009
	Lynch Cove	Mason	Gaeckle et al. 2009
Forest Beach	Mason	Gaeckle et al. 2009	

*The above table simply references locations where Japanese eelgrass has been found, and does not reflect its rapid spread in some of the embayments cited, in particular Grays Harbor and Willapa Bay over the past 5 years.

2.1 Pathways showing potential for further introduction

While invasive species prevention mechanisms in place in the shellfish industry should prevent the future introduction of *Z. japonica* into uninfested state waters from abroad (PCSGA 2011), pathways remain open for its continued introduction into embayments of the Puget Sound (in particular) from local sources where it is not currently found. These pathways include hitchhiking through recreational and commercial vessel traffic, dispersal of seeds passed through the alimentary tract of waterfowl, and natural seed and reproductive shoot dispersal mechanisms through drift.

2.2 Spread potential

An important reproductive feature of *Z. japonica* is that it reproduces vegetatively via rhizomatous (root) growth and also produces seeds. These mechanisms increase ‘propogule pressure’ for uninvaded areas, and allow for both short and long distance dispersal. However, there appears to be a limit to how far along the intertidal gradient *Z. japonica* populations may extend due to the narrow band of suitable habitat available along tidal elevations (Almasi and Eldridge 2008). Dumbauld and Wyllie-Echeverria (2003) demonstrated that *Z. japonica* is also found in less saline portions of estuaries possibly due to decreased abundance of ghost shrimp (*N. californiensis*), which can reduce *Z. japonica* survival through sediment bioturbation.

An example of facilitation between invading species comes from a series of manipulation experiments conducted by Wonham et al. (2005) in Padilla Bay, WA. In their study, percentage cover of *Z. japonica* increased significantly in the presence of another introduced non-native species, the Asian hornsnail *Batillaria attramentaria*. The authors posited the mechanism of enhancement to be habitat provision or ecosystem engineering through bioturbation by these snails. However, it was noted that since the exact arrival dates of the two invaders are unknown, the importance of positive interactions in their early establishment as opposed to later spread cannot be determined.

Competition with native eelgrass appears to retard the spread of Japanese eelgrass into deeper waters where the two species are collocated (Ruesink 2010). However, Japanese eelgrass can also assert competitive suppression of native eelgrass in some instances. Merrill (1995) reported that *Z. japonica* inhibited leaf growth and shoot recruitment of *Z. marina* in August within Padilla Bay. In a separate study, Hourdequin (1994, as cited in Riggs 2002) found that *Z. marina* grew significantly faster in areas isolated from *Z. japonica*. Harrison (1982b) reported that under simulated spring conditions (9°C, 12 hr light: 12 hr dark, low irradiance), *Z. japonica* could compete successfully with *Z. marina* when both were submerged continuously. In contrast, under simulated summer conditions (18°C, 14 hr light: 10 hr dark, higher irradiance), the vegetative growth of *Z. marina* was more than twice that of *Z. japonica*. While Ruesink et al. (2010) reported that the two eelgrass species had similar patterns of productivity (growth) in Willapa Bay, *Z. japonica* significantly outperformed *Z. marina* in flowering and seed germination, mechanisms that would likely favor its spread. Conversely, *Z. marina* was shown to negatively affect the distribution of Japanese eelgrass in the lower intertidal range where it is found. Increased desiccation and wave energy regimes typically retard native eelgrass establishment in tidal ranges above around +1 to +2 MLLW, however, below this zone, the native grass appears to be suppressing, at least to a limited degree, the density of *japonica* on those portions of the bed where the two species co-occur. In summary, the ability of *Z. japonica* to spread

rapidly reflects its broad environmental tolerances to seasonal and tidal variation, though the mechanisms through which it competes with native species are not fully resolved.

Almasi and Eldridge (2008) developed a spatially explicit model in attempt to understand patterns of *Z. japonica* growth and spread. The model incorporated field observations from Yaquina Bay, OR, to test the relative importance of stochastic (random) abiotic disturbance, interspecific competition, and vegetative and seedling survival. This model predicted that, at least in Yaquina Bay, vegetative shoot and seedling survival were limited by competition with native eelgrass and this competition was the most important factor limiting *japonica* growth, though stochastic disturbance was also limiting. The model's prediction appears to be borne out by anecdotal reports: Willapa Bay researchers and oyster growers have observed that the establishment of *Z. japonica* in the middle intertidal range has caused changes in sediment composition and water retention, facilitating the spread of native eelgrass into shallower waters than it would normally be found.

Many invasive species do not exhibit pronounced spreading for many years after their initial introduction. This lag period, or Allee effect, is often the result of stochastic disturbance events that periodically eliminate potential donor populations (patches). The Allee effect is likely responsible for the absence of Japanese eelgrass in embayments in southern Puget Sound where habitat is suitable, but where the species is mostly observed ephemerally and has not been observed to develop large meadows. However, if the frequency or magnitude of such random disturbance events does not impact isolated patches, vegetative growth (in particular) can allow isolated patches to grow into established clonal beds. Under such circumstances, future elimination becomes less and less likely as the bed area expands and a threshold population is established from which future expansion occurs much more rapidly than it had initially. In the Almasi and Eldridge (2008) study, they observed that the strongest relationship to spread potential was associated with vegetative propagule survival. That is, when colonization habitat is not limited, vegetative propagation appeared to contribute more to the spread of established populations than propagation and spread through seed dispersal and germination.

In Willapa Bay, Japanese eelgrass has been documented since the mid 1950s (Harrison and Bigley 1982) and until about 1998 remained relatively confined in plant density and location. Since that time it has exploded and aggressively carpeted many areas of Willapa Bay (Figure 1). This anecdotal lag period observed with Japanese eelgrass in Willapa Bay may be a reflection of a critical threshold size being achieved, following which, interannual variation is reduced as the population expands beyond the threshold that was initially required for establishment and growth (Almasi and Eldridge 2008). In Willapa, Japanese eelgrass appears to colonize intertidal hillocks that are at an elevation that would not initially support native eelgrass.

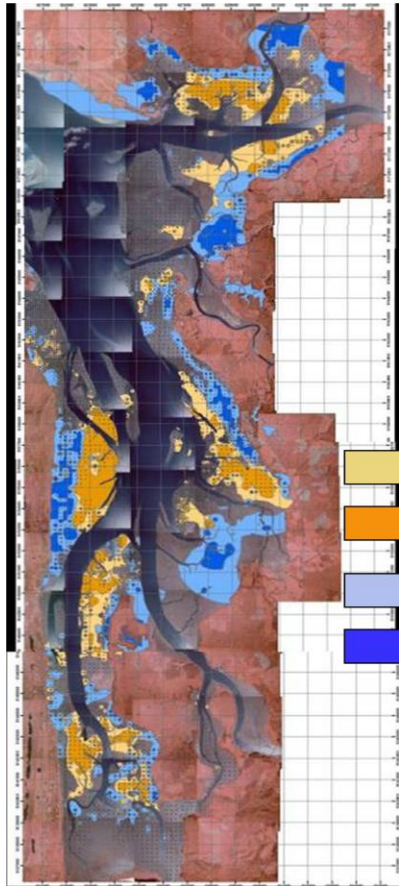


Figure 1. Current Distribution of Japanese eelgrass in Willapa Bay (Source: B. Dumbauld)

From its initial establishment, Japanese eelgrass has then been observed to spread into deeper waters, competing with and sometimes replacing native eelgrass at depths where native eelgrass growth is marginal. Where 20 years ago it inhabited areas between approximately between 4' and 7' MLLW, it now can be observed at the approximate MLLW (0') tidal elevation occupying vast monotypic beds (Figure 2). Spread is also observed farther out into the bay, far from the shoreline, where the elevation is still suitable for growth (B. Sheldon, Northern Oyster, personal communication), and further inland into existing salt marsh habitat (Figure 3). Further establishment at lower tidal elevations can therefore conceivably increase propagule pressure for continued spread into other lower tidal elevation areas where light and competition factors do not otherwise limit its spread.

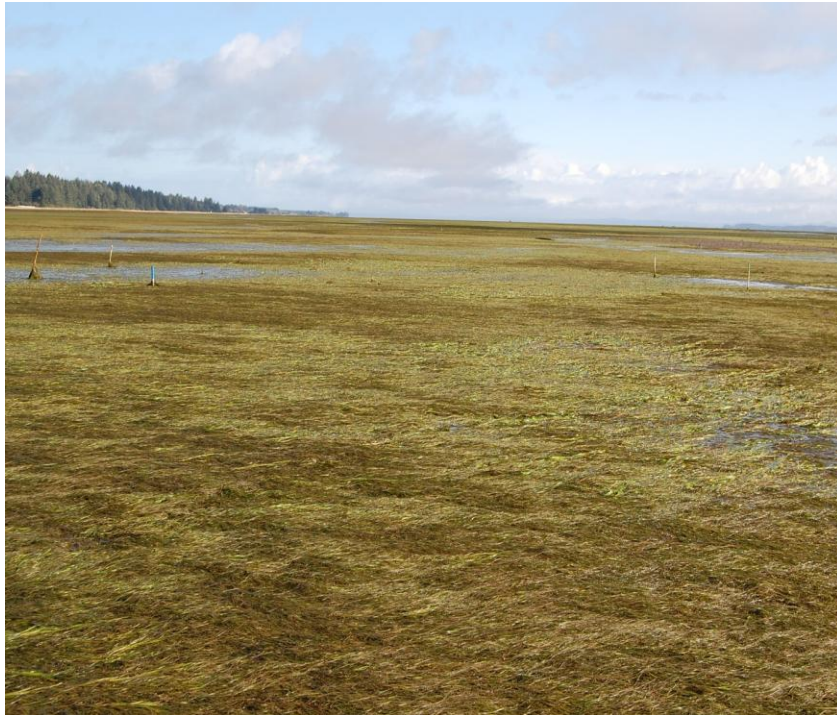


Figure 2. *Z. japonica* currently dominates the majority of the upper intertidal mudflats in many areas of Willapa Bay. These sites were essentially *Z. japonica* free ten years ago (Source: K. Patten)



Figure 3. Salt marsh encroachment of Japanese eelgrass into existing salt marsh habitat in Willapa Bay (Source: K. Patten)

Predictions of sea level rise under climate change would also appear to favor additional spread of the species higher in the intertidal range. In addition, *Z. japonica* has been shown to tolerate higher temperatures, as predicted with climate change, than *Z. marina* (Shafer et al. 2007).

3 Consequences of Establishment

3.1 Socioeconomic Impact Potential

Economic impacts from Japanese eelgrass are primarily associated with the interference it creates with shellfish growing areas that were formerly bare mud and sand flats used for shellfish culture. In particular, the invasive eelgrass has colonized intertidal zones formerly used for Manila clam culture to such a degree and density in some locations as to prevent effective shellfish planting and harvesting. In a study conducted by Tsai et al. (2010), Manila clam meat was reduced within plots of *Z. japonica* at the highest clam density tested, though there were no effects on clam size or recruitment. WSU researchers also evaluated production indices in Manila and soft shell clams (*Mya arenaria*) among five sites in Willapa Bay to consider potential effects on commercial clam production, by treating portions of the beds infested with Japanese eelgrass with herbicide to remove the grass, and comparing production indices from areas on the same bed that were not treated. Site characteristics among these locations are summarized in Table 1. As a result of the herbicide treatments for the studies, the time which the plots remained free of Japanese eelgrass varied (Table 1).

Site	History	Site quality and tidal elevation for clam production	Months free of <i>Z. japonica</i>
Oysterville -1	Well graveled clam farm, abandoned due to thick infestations of <i>Z. japonica</i>	Excellent, 2-3'	9
Oysterville -2	Well graveled clam farm, abandoned due to thick infestations of <i>Z. japonica</i>	Excellent, 2-3'	9
Leadbetter	Moderately graveled site, never farmed, medium <i>Z. japonica</i> infestation	Good, 3-4'	18
Stackpole	Moderately graveled site, never farmed, medium <i>Z. japonica</i> infestation	Fair, 3-4'	18
S. Nahcotta	Moderately graveled site, never farmed, thin to medium <i>Z. japonica</i> infestation	Fair 3-4'	9

As demonstrated in Figure 4, the number and weight of market size Manila clams on herbicide treated beds (i.e., with no Japanese eelgrass) was higher at four of the five locations sampled, and significantly higher at three of the five sites studied.

Commercial Manila Clams

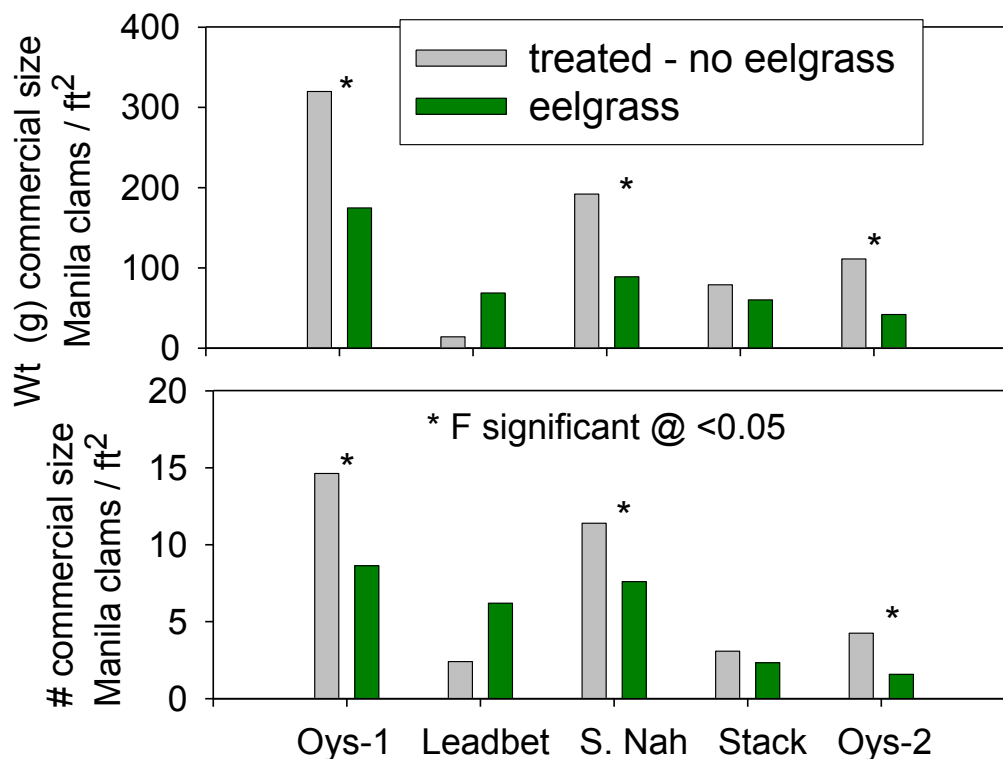


Figure 4. Effects of *Zostera japonica* infestation (green bars) on commercial Manila clam weight (top) and abundance (bottom) in paired plots treated with the herbicide imazamox to remove the invasive eelgrass (grey bars).

The number of Manila clams per square foot of sediment surface of all size classes (combined) was also found to be higher among all five sites sampled, and significantly so at three of the five sites (Figure 5, bottom panel). Results with soft shell clams when all size classes were considered were more variable, with three of the five sites showing higher abundance in the absence of Japanese eelgrass (herbicide treated), but two of the sites showing significantly higher total abundance in beds with Japanese eelgrass (Figure 5, top panel).

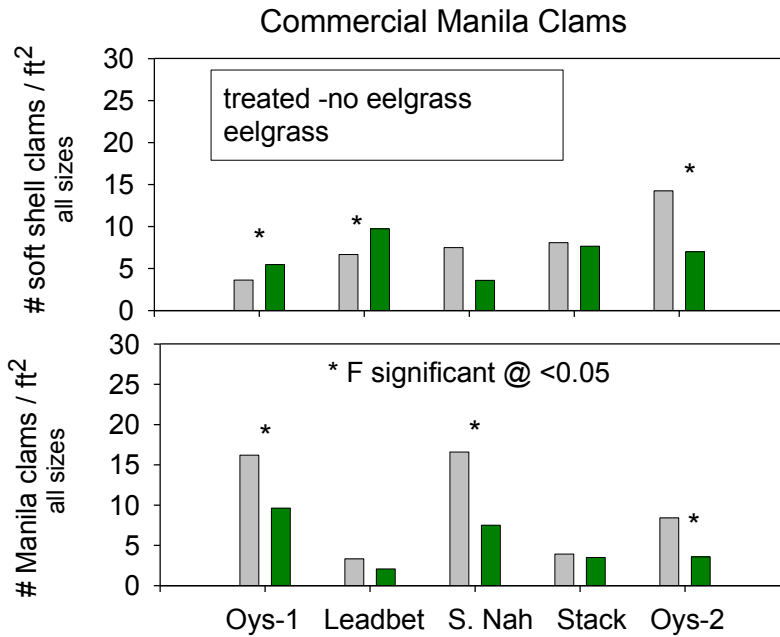


Figure 5. Clam production, all size classes, on five production beds with (green bars) and without (grey bars) Japanese eelgrass. Beds without eelgrass had been previously treated with herbicide at approved label application rates.

Other studies of shellfish bed economic viability have documented significant impacts or revealing trends from Japanese eelgrass invasion. These measures included increased summer length gain and clam weight (K. Patten, unpublished), and clam quality (meat wt/shell wt.) –in nearly all cases showing higher and often significantly higher values on beds where Japanese eelgrass had been removed from herbicide treatment relative to infested beds. Data relating results from ongoing WSU research on clam quality are depicted in Figure 6.

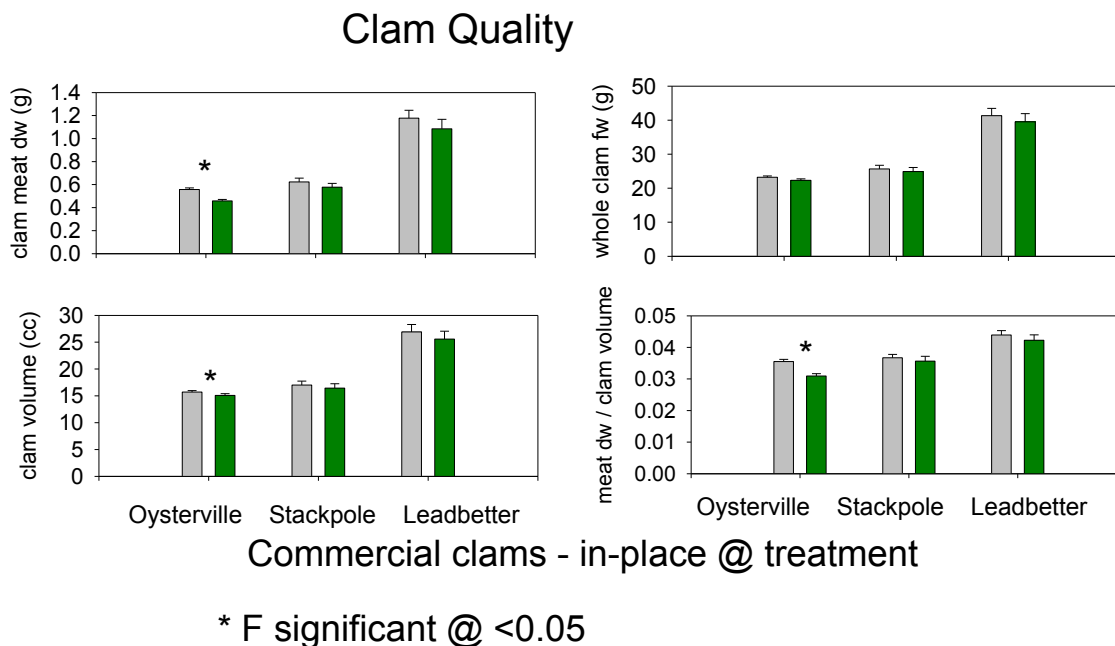


Figure 6. Clam quality metrics, as recorded among three sites sampled in Willapa Bay with (green bars) and without (grey bars) Japanese eelgrass.

Clam sets of your the three sites examined *arenaria* clams in recruitment are sc

Clam set (6/15/2010 to 11/1//2010)
Clams <2 mm/ trough

variable results, with two of Manila and soft shell *Mya* more variable results on from Tsai et al. 2010.

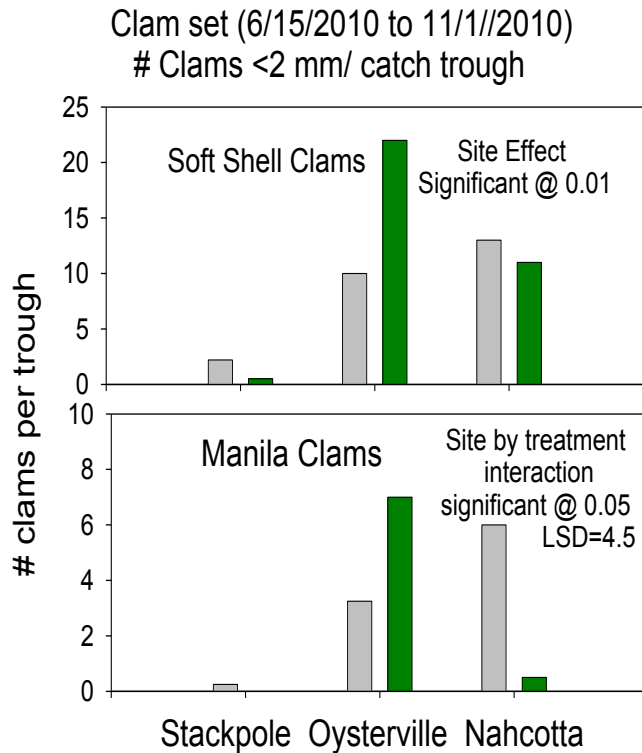


Figure 7. Young of the year clam sets onto beds vegetated with Japanese eelgrass (green bars) versus unvegetated, herbicide treated beds (grey bars).

To quantify these biological effects on the local economy, Patten et al. (2011, unpublished) projected net income based on mean yield of clams calculated at five sites in Willapa Bay, with and without Japanese eelgrass infestations (Table 2). Over all sites examined, they estimated: (1) an average decrease in net return of 44% due to *Z. japonica* infestation, assuming a yield of 1 lb./ft², an estimated profit of \$1/lb., and a 4-year crop cycle; (2) an estimated loss of \$4,000/acre/year; and (3) a \$4 million impact for every 1,000 acres affected by Japanese eelgrass.

Table 3. Projected net income (\$) of Manila clam harvests at Willapa Bay, WA sites with and without Japanese eelgrass*						
Treatment	Oysterville -1	Oysterville. -2	Leadbetter Point	Stackpole	South Nahcotta	All Sites
No japonica	32,303	11,230	10,036	7,970	19,385	17,040
Z. Japonica	15,959	3,821	6,271	5,487	8,130	8,888
Difference	16,344	7,409	3,765	2,483	11,255	8,152

*assumes: (a) \$2.35/lb. wholesale; (b) \$0.65/lb. digging cost, \$0.65/lb. to produce, clean, package, and market product; (c) an extra \$0.05/lb. cleaning cost for *Z. japonica*-infested sites; and (d) 5 percent crop loss for *Z. japonica* sites due to clams left unharvested, or damaged.
Source: K. Patten, WSU, unpublished.

The economic impact data cited in Table 1 really only reflect one season in the bay not two to three growing seasons, where the most significant differences may be realized. Additional economic impacts that have yet to be calculated may also accrue from reduced oyster seed recruitment, ground already lost to infestation, and lost investments in cultivating beds.

Japanese eelgrass can also impact public tidelands in the same manner as private farms and can act to reduce or eliminate shellfish harvest there. Businesses that rely on recreational harvesters for a source of their revenue can be impacted through this mechanism, though precise figures as to this impact are as yet lacking.

3.2 Environmental Impact Potential

Potential Adverse Impacts

The valuable ecological functions of normally unvegetated mudflats can be adversely affected by the introduction of *Z. japonica* (Harrison 1987, Posey 1988, Nomme and Harrison 1991). According to the Alaska Natural Heritage Program (ANHP 2005), “colonization of sparsely vegetated or bare intertidal flats by Japanese eelgrass represents a drastic modification of habitat.” As previously mentioned, *Z. japonica* grows mainly in the mid-intertidal regions, overgrowing previously unvegetated mudflats and growing in mixed beds with *Z. marina* lower in the intertidal zone (Harrison 1982, Harrison 1987, Posey 1988, Nomme and Harrison 1991, Ruesink 2010). Nomme & Harrison (1991a) found *Z. japonica* density was lower in the mixed zone, but *Z. marina* density was not significantly affected by tidal zone. Tidal zone, however, affected morphology of each species. In cool temperatures and high light, *Z. japonica* leaf elongation was the same rate as *Z. marina* (Harrison 1982). Nomme & Harrison (1991b) transplanted *Z. japonica* and *Z. marina* into the higher *Z. japonica* zone, the mixed species zone, and the lower *Z. marina* zone, and found that shoot density (counts) showed no consistent significant difference in density of either species that could be explained by depth, but found that *Z. japonica* transplants had lower shoot growth in all but one transplant in the lower *Z. marina* zone.

Although it is possible that *Z. japonica* and *Z. marina* negatively impact the growth of one another, it is unclear whether *Z. japonica* will ultimately expand farther into lower intertidal zones dominated by native *Z. marina* in all areas. A study in Washington State by Bando (2006) found that disturbance substantially enhanced *Z. japonica* productivity and fitness while at the same time decreasing *Z. marina*'s performance. Bando (2006) cleared plots in the *Z. japonica* and *Z. marina* mixed tidal zones and measured plant growth back into the plots. *Z. japonica* recruited in first, and in the two years the study was run, *Z. marina* never recolonized. The authors suggested that *Z. japonica*'s success as an invasive species stems dually from its ability to persist in competition with *Z. marina* coupled with its positive response to disturbance. Notably, climate change predicts increased frequency of extreme climatic disturbance events that could facilitate further spread with associated impacts.

Z. japonica colonization can also substantially decrease water flow to mudflats, reportedly by up to 40% (Tsai et al. 2010). This action recruits fine sediment and detritus into the mid-intertidal zone, which displaces some benthic invertebrates, increases the diversity and abundance of

others, and alters the use of the habitat by shorebirds, and other species dependant on bare mudflat habitat (Harrison 1987, Posey 1988, Lee et al. 2001). The general trend noted by Grosholz and Ruiz (2009) was a reduction in larger, surface feeding taxa and concurrent increase in smaller, subsurface detritivores. The authors hypothesized that such global shifts in the benthic community would have potentially negative impacts for higher trophic level consumers including crabs, fishes, and birds. For example, in a mesocosm study by Semmens (2008) undertaken in a net enclosure in Willapa Bay, WA, Chinook salmon (*Oncorhynchus tshawytscha*) smolts preferred to remain in native *Z. marina* beds whereas no preference existed for habitats composed of *Z. japonica* or a non-native cordgrass (*Spartina alterniflora*).

As has been observed with many invasive plant species (Ehrenfeld 2003; Hawkes et al. 2005), *Z. japonica* has also been found to alter nitrogen cycling in ecosystems it invades. In a Yaquina Bay, Oregon study, unvegetated sediments colonized by *Z. japonica* switched from functioning as net sources to net sinks of inorganic nutrients; further, NO_3^- and NH_4^+ fluxes in *Z. marina* beds were twice that of *Z. japonica* beds (Larned 2003). Bulthuis and Margerum (2005) showed nitrogen reduction of up to 90% during the summer as water passed through an eelgrass community in Padilla Bay, WA that included both *Z. marina* and *Z. japonica*. In contrast, Tsai et al. (2010) showed that there were no differences in ammonium levels in sediment porewater from *Z. japonica* plots and mudflat areas. Clearly, nutrient fluxes may change based on plant nitrogen requirements or changes in associated nitrogen-cycling microbial composition and activity (Silver 2009). It is conceivable, therefore, that in estuaries that are nitrogen limited, Japanese eelgrass could further limit nitrogen sources used by other resources (phytoplankton, native eelgrass, etc.). This switch in nutrient use can ultimately affect energy flows and resultant biological community structure in the estuary. However, the variable evidence about *Z. japonica*'s use of nitrogen and how this use may explain or contribute to its environmental impacts on community structure indicates that more research is necessary to better explain variability.

In Padilla Bay, WA, Hahn (2003) showed that *Z. japonica* can also accelerate rates of decomposition and alter the decomposer community, which could lead to higher carbon and nutrient turnover when compared to *Z. marina* beds. *Z. japonica*'s decomposition rate in this study was 1.65% of total mass lost per day relative to *Z. marina* -- 1.35% of total mass lost per day (Hahn 2003). Possible explanations for its rapid decomposition include that it is simply smaller than *Z. marina*, which gives it a higher surface area to volume ratio, or that its chemical or structural composition makes it easier to break down than *Z. marina* (Hahn 2003). Hahn (2003) also found that microbial assemblage is strongly influenced by the type of associated vegetation present and differs among *Z. marina* and *Z. japonica* communities. These differences could lead to differing rates of decomposition and nutrient retention, alter the interaction between the microbes and the vegetation, and affect higher levels of productivity. Ultimately, the additional dissolved organic matter (DOM) generated from decomposition can be assumed to be available for consumption by organisms such as zooplankton and filter feeders, however the effect of this additional material in the food chain has not been studied or modeled.

Changes in water flow resultant from Japanese eelgrass establishment can affect how organisms use mudflats. Migratory behavior of mobile epifauna and fish that generally move in and out with the tide on unvegetated and unobstructed tide flats may be disrupted. At its peak

in summer, *Z. japonica* stems may be as much as 18' long in areas that were previously bare sand and mud. Tideflats in many sections of Willapa Bay drain regularly in the winter and go dry on a minus tide, but in the summer will have 0.25 to .5' of water on them due to entrapment by *japonica* (K. Patten, WSU, personal communication, 3/27/11). With *Z. japonica* stem infesting the flats, it is possible that some biota are either blocked or delayed from returning to deeper water as the tide recedes- (perhaps getting tangled in the dense stems?), or they may become 'navigationally' disoriented and become essentially stranded on the exposed tide flat. To this end, growers report it is becoming more common to see crab tangled in stems, or larger fish trapped at low tide (B. Sheldon, Northern Oyster Company, personal communication), though we are aware of no study that has expressly quantified this impact. Similarly, species that recruit to *Z. marina* beds may accidentally recruit into the upper intertidal zone where *Z. japonica* grows and thereby become susceptible to stranding. Exposure to temperature and desiccation stress in the higher intertidal could also increase stress and mortality in juveniles and/or egg masses that are deposited or stranded higher in the intertidal than would be typical. In support of these anecdotal observations, Ruesink et al. (2010) sampled 14 transects in Willapa Bay on two time points four years apart and found *Z. marina* moved upshore into *Z. japonica* zones. Ruesink et al. suggested that this is caused by *Z. japonica* retaining water, thus physically altering the upper intertidal zone to mimic a lower tidal elevation (i.e., thereby making the habitat more suitable for native eelgrass migration into higher tidal elevations).

Excess water trapped or otherwise 'ponded' on the upper tide flats from *Z. japonica* infestation is exposed during low tide, heats more readily, and is thus able to hold less oxygen. At the same time, these elevated temperatures can increase decomposition rates and sediment-associated biological oxygen demand, further reducing dissolved oxygen concentrations required for respiration. Species requiring high water flow for respiration or filter feeding could therefore be negatively impacted by *Z. japonica*. These anoxic conditions may result in what growers have described as a 'putrid soup', with salmonid smolts and other fish observed gaping on the surface on the incoming tide (B. Sheldon, Northern Oyster Co., personal communication); however, quantitative examination of these observations are needed to formally document this type of impact.

The reduced water flow in the middle and upper intertidal zone caused by established *Z. japonica* can result in a higher proportion of fine sediments accreting on intertidal beds (Posey 1988; Figure 8). In turn, the accumulation of silts in these beds can create a thicker muck layer and alter bed elevation, as has been examined and documented in Willapa Bay at two sites previously treated with herbicide to remove the Japanese eelgrass (Figure 9). These fine grained sediments now entrained amongst the grass can be more susceptible to local resuspension, resulting in higher background turbidity from even minor wave action. The increase in fines may also exacerbate spreading, by providing a more suitable habitat for seed germination, root growth and vegetative propagation.

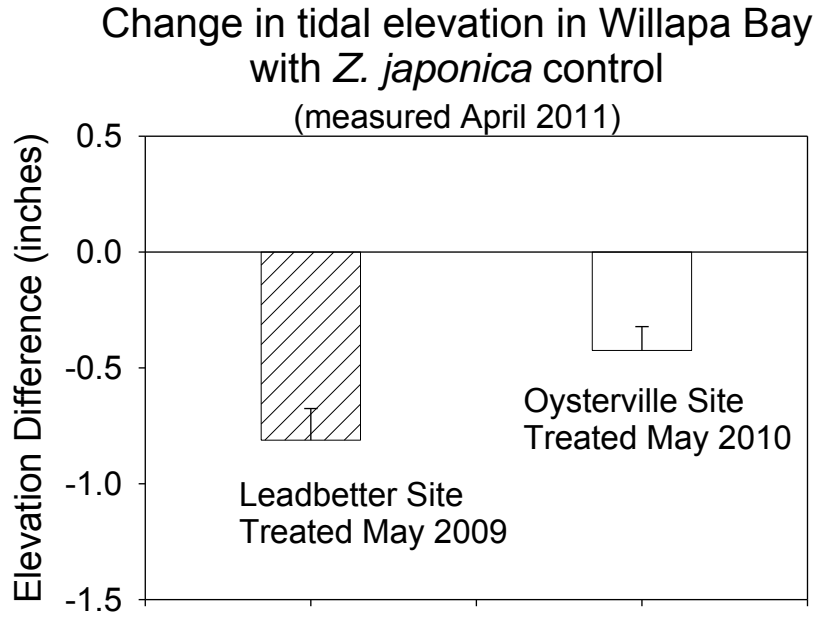


Figure 8. |
treatment c

; removed relative to pre-

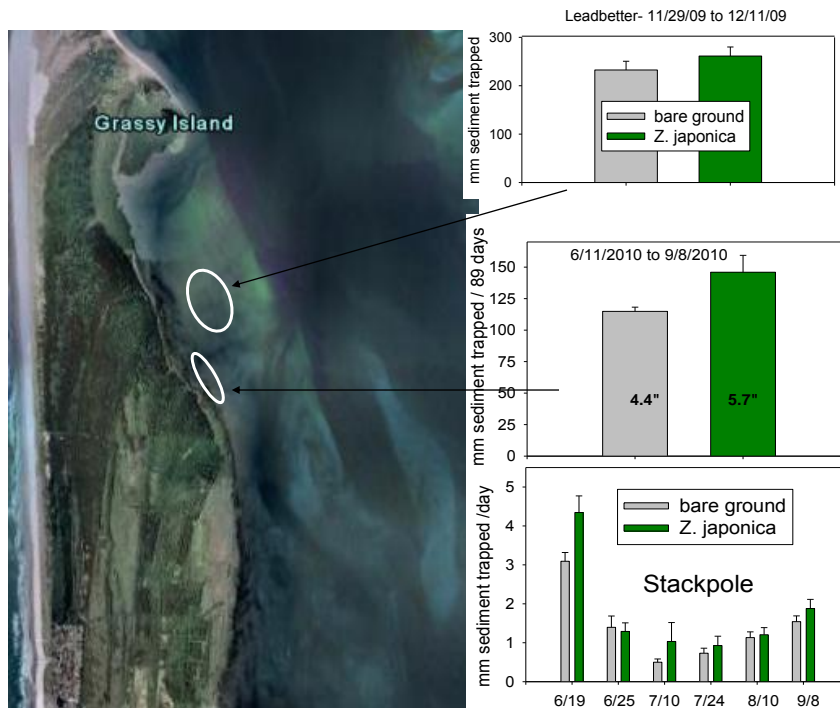


Figure 9. Sediment accumulation on sites where eelgrass was removed with herbicide treatment (grey bars) relative to pre-treatment conditions when Japanese eelgrass was present (green bars). Top panel: Leadbetter Pt., lower panels, Stackpole. (Source: Patten, unpublished).

The benthic community changes documented in Japanese eelgrass beds are likely a result of both changes to nutrient flux and sediment composition alterations that result from the establishment of the beds and changes in water flow. Berkenbusch et al. (2007) found that Japanese eelgrass did not naturally recruit into ghost shrimp control plots in the presence of *Z. japonica* in Netarts Bay, yet manual transplants into ghost shrimp beds were successful. Dumbauld and Wyllie-Echiverria (2003) found that spaying mudflats with carbaryl to reduce ghost shrimp densities increased *Z. japonica* density, illustrating the competition between these species.

Bioturbators like burrowing shrimp are likely limited by the rhizomatous growth of *Zostera* spp. In support of this hypothesis, Harrison (1987) removed all shoots in a test plot, which allowed adult shrimp to colonize the sediment, while addition of transplanted *Z. japonica* shoots caused a temporary decrease in shrimp abundance. Within a few weeks after the transplant, however, the shrimp destroyed the plant transplants by disturbing their root structures. These data suggest that where burrowing shrimp are established and not controlled, successful colonization and/or expansion of Japanese eelgrass could potentially be inhibited. Observations on the ground, however, suggest that seedlings establish in new sites very rapidly, and will persist if there are ample seed sources available (i.e., high propagule pressure), irrespective of shrimp presence. Thus, while burrowing shrimp populations conceivably may suppress seedling survival via bioturbation, and controlling the shrimp population could therefore conceivably facilitate japonica expansion, shrimp bioturbation appears unlikely to suppress seedling survival in practice if seed propagule pressure is too high. In Willapa Bay, for example, shrimp have been controlled since 1963 on select shellfish beds, but japonica infestation on these (and other) beds did not appear to become a significant problem until the late 1990's.

The findings of adverse effects of Japanese eelgrass on burrowing shrimp brings to light an additional potential adverse effect from Japanese eelgrass, in that burrowing shrimp are a key food item of green and white sturgeon, the former species of which is listed under the Endangered Species Act. Indeed, foraging pits left by sturgeon in Japanese eelgrass beds in Willapa Bay (Figure 10) have been anecdotally observed to be less abundant than on beds treated with herbicide to remove the grass by one of the authors. Such observations, if confirmed, could be a reflection of these fishes' preferences for foraging where food resources are greatest, or where the energy expended to obtain food resources is lessened because of less interference with Japanese eelgrass.



Figure 10—Typical sturgeon pit, as observed in a moderately dense Japanese eelgrass bed. Source: Patten 2011.

Potential Positive Impacts

Fish, birds and a broad range of infaunal and epifaunal invertebrates utilize *Z. japonica* for food and habitat. In Boundary Bay, British Columbia, Baldwin and Lovvorn (1994) documented that migratory waterfowl grazed preferentially on *Z. japonica* versus *Z. marina*, and made up a significant portion of the esophagus contents of brant (57.2%), American wigeon (84.8%), and mallard (72.3%). The leaves of *Z. japonica* had higher caloric value than leaves of *Z. marina* though there was no significant difference in reported caloric values of the rhizomes. The preferential grazing found in this study may be a reflection of the greater accessibility of Japanese eelgrass higher in the intertidal zone, thus potentially requiring less energy expenditure to feed on a diet of similar caloric value, an example of optimal foraging theory. In contrast, in ongoing work conducted in Willapa Bay, Patten et al. (unpublished) have analyzed about 150 gullet contents of waterfowl and found no appreciable amount of Japanese eelgrass. It is also possible that observed consumption of the eelgrass is inadvertent to the birds consumption of benthic infauna that are simply associated with the root mass of the plant.

Japanese eelgrass provides habitat functions similar to native eelgrass that are likely beneficial to some organisms, as it creates three dimensional structural complexity not present on bare mudflats. This structural complexity is thought to be the primary factor that regulates fauna associations in native eelgrass, rather than interactions tied to the plants autotrophic processes. The structural complexity and epiphytic food resources attached to the grass are attractive to fish and other organisms (Simenstad et al. 1994), and the structure provides predator protection and potential spawning substrate. The degree to which such trophic linkages primarily studied in native eelgrass are also provided by Japanese eelgrass has not been examined thoroughly, but several studies have focused on species relationships. For example, Posey (1988) measured abundance and richness of infaunal species in *Z. japonica* beds and in bare mudflats from core samples and found that small infaunal species densities were higher in *Z. japonica* cores than in those from the mudflats without rooted vegetation. Javier (1987) found significantly higher densities of the four most common spionid polychaetes in beds of *Z. japonica* relative to unvegetated plots and plots caged to exclude predators. Results from this caged exclusion test supported the hypothesis that the Japanese eelgrass was providing prey refuge, facilitating the higher density of spionids found in both of these tested habitats relative to unvegetated mudflats. In its native waters of Hong Kong, where it is not considered invasive, macrofaunal species richness was significantly higher in the seagrass bed (n =118) than in unvegetated beds (n = 70). Infaunal species richness exhibited high similarity between the two habitat types and was not significantly different though the opposite was true for epifauna, which was significantly more species diverse in Japanese eelgrass habitat. In contrast, abundance of both infauna and epifauna was higher in the eelgrass bed than the unvegetated bed (Lee et al. 2001).

In invaded habitats, however, the effect of Japanese eelgrass on infauna and epifauna abundance and diversity important to priority commercial and recreational species such as ESA-listed salmonids and waterfowl remains to be fully examined. Presumptions that the non-native eelgrass provides functions for ESA-listed salmonids that are superior to unvegetated mudflat, gravel or shellfish beds—simply because it is a species of eelgrass—cannot be supported or refuted with either the published or unpublished information currently available.

4 Summary of Risks

The environmental and socioeconomic impacts of established Japanese eelgrass beds appear variable, based on the location, prevalence, and effects of the infestation on ecological, recreational and commercial resources. Results from numerous studies of its effects following introduction suggest that its spread potential remains high for low energy intertidal areas with suitable fine grained substrate, tidal elevation, and water quality. Stochastic disturbance events and sea level, both predicted to increase under climate change scenarios, will likely facilitate the continued spread of the species further up the beach and into areas not presently colonized. Natural dispersal mechanisms for the species and its broad distribution among estuaries along the West coast indicate broad scale eradication is not possible.

Adverse socioeconomic impacts currently identified from the species affect the shellfish growing community in locations within Willapa Bay, Grays Harbor and other coastal embayments on the West coast by displacing existing cultivated beds, increasing costs associated with bed management where it invades, and reducing yield from shellfish beds. Preliminary economic impact assessments to shellfish growers specifically are significant, and indirect economic impacts to supporting industry would likely increase the estimate of impact.

Infestation of tidal beds with Japanese eelgrass generally results in significant changes in biological communities. Some species benefit, while others are impacted. Ecological impacts identified from Japanese eelgrass establishment largely relate to the physical changes to habitat that result from its introduction. Establishment slows water movement through the middle and upper intertidal zone, leading to sediment fines and water retention on the beds, and commensurate changes in biological community structure relative to unvegetated sand/mudflats that are the habitat type typically invaded. Though not formally researched to our knowledge, heavily invaded beaches may affect reproduction and survival, and/or migratory behavior of species that atypically and perhaps inadvertently utilize the habitat created, without consideration for the increased desiccation, heat stress and mortality that accompany middle and upper intertidal conditions.

The relative ecological functions supported by Japanese eelgrass compared to native eelgrass communities may be similar, with functional producer and consumer groups partitioning between the two types of beds based more on depth preference, wave tolerance, and desiccation and heat tolerance. Where they co-occur, the species appear to compete for space in a lower intertidal transitional zone, below which the native species dominates, and above which the non-native dominates. Both species produce oxygen which is obviously helpful for basin water quality in general, though the effects of decaying Japanese eelgrass during decomposition may reduce oxygen concentrations in waters unnaturally retained on beds during low tide. Japanese eelgrass also appears to affect nutrient cycling at an accelerated rate relative to native eelgrass, serving as a nitrogen sink during the growing season and a source of DOM during seasonal senescence with a more rapid rate of decomposition. The long term effects of altered nutrient cycling have not been fully researched and site-specific differences in impact could occur, for example, where nitrogen is limiting phytoplankton growth. An economic analysis of the relative ecosystem services provided and lost as a result of the introduction of the Japanese eelgrass is needed to better gauge the impacts from invasion over the range of areas where the species has been found.

Management of risks from the spread of Japanese eelgrass has been complicated by regulatory provisions in several branches of state, local and federal government that essentially stipulated no-net-loss requirements for actions potentially affecting eelgrass, without differentiating among species. The WDFW has recently rescinded the protections formerly afforded the species with policy changes submitted through the Washington State legislature. Full implementation of this policy change will undoubtedly require interagency recognition, as multiple agencies continue to play roles in aquatic resource management. Regulated in-water activities overseen by multiple agency review processes must be exercised with consistency with regards to the recent WDFW policy change or interpretations regarding the status of the species will continue to be muddled in permit review decisions. Any lack of clarity leaves open to interpretation whether state and federal policy against non-native invasive species should be disregarded in lieu of interpretations that could be made from inconsistent interagency policy when such regulations result in the protection of a non-native invasive species—even when significant socioeconomic and ecological impacts from infestation are documented.

The risk of the species on ecosystem and socioeconomic functions appears density dependent. In Willapa Bay, for example, the infestation may reflect a case of too much of a good thing', wherein eradication appears to represent a supreme challenge. The broad infestation there may be a reflection of elevated carbon dioxide in coastal waters that have resulted from ongoing climate change. Whatever the cause(s), the broad scale and dense distribution of the species in this embayment makes eradication a supreme challenge and emphasizes the need for assertive management controls to minimize economic impacts. The broad distribution in Willapa Bay, however, may be a bell weather for potential impacts in other embayments where the grass is less established but continues to spread (e.g., Coos Bay, OR), or occurs only sporadically (South Puget Sound) The changes wrought by this non-native eelgrass on ecosystem function relate primarily to its effects on physical habitat structure, and when heavily infested, the species clearly represents an ecosystem engineer' These physical habitat changes are visibly dramatic and obvious. As with other invasive species, early detection and rapid response should be fully considered to eradicate new infestations where they are observed before radical changes in ecosystem structure result.

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