AN ABSTRACT OF THE THESIS OF

Daniel M. Sund for the degree Master of Science in Marine Resource Management presented on May 22, 2015.

Title: Utilization of the Non-Native Seagrass, *Zostera japonica*, by Crab and Fish in Pacific Northwest Estuaries

Abstract Approved:

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The introduction of non-native species often results in fundamental changes in the structure and function of disturbed environments. In the Pacific Northwest (PNW), the introduced seagrass *Zostera japonica* is rapidly expanding in distribution, impacting stakeholders and public use of the intertidal. *Z. japonica*'s expansion has prompted a number of different management strategies and many research studies that examine its ecology in the PNW. A number of studies have compared the benthic and epifaunal communities in *Z. japonica* to those of the native *Z. marina*, but to date, none have contrasted the nekton communities using the two seagrasses. The goals of this project were to 1) examine the community composition of a variety of available estuarine habitats in Willapa Bay, Washington, and Yaquina Bay, Oregon, via paired deployment of cameras and small fish traps; and 2) to explore the different management strategies used in the PNW and identify strengths and weaknesses associated with invasive species management, as indicated by short interviews with professionals working on *Z. japonica*.

In Willapa Bay, *Z. japonica*, *Z. marina*, clam aquaculture, and on-ground oyster

aquaculture were examined. In Yaquina Bay, *Z. japonica*, *Z. marina*, and bare substrate were examined. A total of 11 species, with 10 occurring in Willapa Bay and 9 in Yaquina Bay, were observed in video footage. Habitat was a significant predictor of catch per unit effort (CPUE) for the most abundant species in Yaquina Bay but not those in Willapa Bay. Community composition was significantly different between habitats in each bay but not between the bays. Explicit comparisons of seagrass habitat in each bay indicate some evidence that community composition of the two seagrasses differs in Yaquina Bay, but not in Willapa Bay. We conclude that community composition varies little between seagrass structure in Yaquina Bay and Willapa Bay and that local variation is highly dependent on the availability of structured habitats. Additionally, the distribution of *Z. japonica* relative to *Z. marina* may drive these differences in community composition between seagrass habitats in these estuaries.

In short, unstructured interviews with professionals working on *Z. japonica* in the PNW, ecological characteristics that prompted management consideration; historical and potential management approaches; and suggestions to improve invasive species management at the local, regional, and national levels were discussed. Interview participants highlighted *Z. japonica*'s expansion into historically unstructured regions of the intertidal, its role as an ecosystem engineer, and the intrinsic value of the local, native ecology as reasons for management. The need for collaboration across all levels (local, state, regional, and federal) of invasive species management, public outreach and education, professional development, and explicit statement of management position were all stressed as potential improvements to invasive species management.

Utilization of the Non-Native Seagrass, *Zostera japonica*, by Crab and Fish in Pacific Northwest Estuaries

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

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Chapter 1: Introduction

1.1 Seagrass Ecology

Seagrasses are highly productive, flowering vascular plants that grow in brackish or marine conditions. In comparison to the diversity of freshwater vascular plants, there are very few marine species of seagrasses in the marine environment (Larkum et al. 2006; Water Quality Program 2014). Globally, seagrasses grow in both temperate and tropical conditions, typically within shallow and protected estuarine waters. Seagrasses are considered to be ecosystem engineers within estuaries, both structuring intertidal habitats and providing a host of ecosystem services (Larkum et al. 2006; Orth et al. 2006). Among the most widely recognized services provided by seagrasses are their role as: primary producers; sources of organic matter; a source of oxygen; sediment traps for suspended material; important foraging ground for multiple life stages of estuarine organisms; and, most prominently, as a source of structured habitat refuges and valuable nursery habitat for many species (Short and Wyllie-Echeverria 1996; Heck Jr. et al. 2003; Larkum et al. 2006).

One of the most widely reported functions of seagrass habitat is its role as a nursery for myriad oceanic species. Several high-value commercial fisheries utilize seagrass nurseries. In Australia, seagrass beds were found to support the harvest of shrimp and crab worth up to $1,436 per hectare (Barbier et al. 2011). In the Pacific Northwest (PNW), Pacific herring are dependent on seagrass beds as spawning grounds, so much so that when seagrasses declined significantly at a number of sites in the Puget Sound local extinctions were observed (Wyllie-Echeverria et al. 2009).

In the PNW, the Dungeness crab fishery is one of the largest commercial shellfish

fisheries, with an estimated total ex-vessel value of $42 million dollar in Oregon in 2012 (Oregon Department of Fish and Wildlife 2013; The Research Group LLC 2014). Holsman et al. (2003) found that adult and sub-adult subtidal populations of Dungeness crab heavily subsidize their diet by foraging within intertidal environments. Juvenile Dungeness crab have been shown to utilize structured estuarine habitats preferentially over unstructured ones (Pauley et al. 1989). They use seagrass structure as both a foraging ground and refuge (Fernandez et al. 1993; Eggleston and Armstrong 1995).

In addition to providing nursery grounds for a number of species, commercially fished and otherwise, seagrass beds help maintain and improve water quality in nearshore areas (Larkum et al. 2006; Barbier et al. 2011). Costanza et al. (1997) valued nutrient cycling by seagrasses at approximately $19,000 per hectare per year. Multiple seagrass species have been shown to be sinks for dissolved nitrogen, providing not only a natural buffer against eutrophication and hypoxia, but also improving water quality (Larkum et al. 2006; Mach et al. 2010). In the PNW, shellfish hatcheries have started timing the intake of seawater to correspond with daily peak carbon dioxide uptake by seagrass beds to mitigate acidic water (Barton et al. 2012).

Physical characteristics of seagrass beds have also been found to be important to the function of coastal ecosystems and human communities. Seagrass beds slow the movement of water flowing into them, mitigating coastal erosion and the transport of sediment (Larkum et al. 2006; Barbier et al. 2011).

Despite all of the benefits afforded by healthy seagrass ecosystems globally, many are on the decline due to a suite of problems. Seagrasses have been unable to adapt to stresses associated with water quality degradation, increased turbidity, and higher

volumes of marine traffic that have rapidly changed aquatic conditions. These issues have caused large declines in the distribution of seagrasses on a global scale (Short and Wyllie-Echeverria 1996; Orth et al. 2006). Of the 50 to 60 known species of seagrass, over 20 have had reported declines in recent years (Short and Wyllie-Echeverria 1996). Concern is mounting about further degradation of seagrasses on a global scale because of the looming impacts of sea level rise, increased sea surface temperatures, and changes in sea water chemistry associated with climate change (Short and Wyllie-Echeverria 1996; Orth et al. 2006; Waycott et al. 2009).

Yet, contrary to the declines observed in seagrass distribution in other regions of the globe, the PNW is actually experiencing expanding seagrass populations (Young et al. 2008; Ruesink et al. 2010). However, the long term direction of this trend is unknown, and a portion of the expansion is driven by the dispersal of the introduced *Zostera japonica* (Young et al. 2008; Ruesink et al. 2010).

1.2 Zostera japonica in the Pacific Northwest

Introduction of *Zostera japonica*, or dwarf eelgrass, to the PNW occurred between 50 and 100 years ago (Young et al. 2008; Mach et al. 2010; Ruesink et al. 2010). The species, native to the coasts of the Western (Asian) Pacific, is believed to have been introduced into the PNW by the oyster aquaculture industry, where it was used as a wrap for exporting oysters to the region from Japan (Mach et al. 2010; Shafer et al. 2014). Although believed to be present since the early 20th century, *Z. japonica* was not recognized as an introduced species until the mid-1980s (Bigley and Barreca 1982).

Scientists and researchers initially identified *Z. japonica* as a novel species,

Zostera americana, until further genetic and morphological information indicated it was, in fact, *Z. japonica* (Bigley and Barreca 1982). Even after the discovery that *Z. japonica* was present along the shore of the PNW, management response was slow to determine a course of action. The initial response to *Z. japonica* was to afford it statutory protection because of a perception that it would provide the same ecosystem services as its native congener, *Zostera marina*. The long gap between introduction and identification, compounded by the delayed management response at state and federal levels, provided ample opportunity for *Z. japonica* to become established within the region. Recent comparisons of the historic range of *Z. japonica* to the current distribution show a nearly 400% increase over the last nine years within Yaquina Bay, Oregon (Young et al. 2008).

The rapid spread of *Z. japonica* in the PNW has raised concerns within the shellfish aquaculture industry because of reported declines in harvest and increased work effort associated with the higher *Z. japonica* densities, particularly in the harvest of Manila clams (Mach et al. 2010; Patten 2014). Preliminary evidence suggests that *Z. marina* may be competitively excluding *Z. japonica* in the lower intertidal (Ruesink et al. 2010). However, natural co-occurrence of the two species in *Z. japonica*'s native range suggests that competitive exclusion observed in the PNW is an expression of differing community dynamics in PNW waters (Ruesink et al. 2010). The twofold impact of changing ecosystem structure driven by climate change and further perturbation from an introduced ecosystem engineer calls into question the resilience of PNW seagrass habitats and the communities directly and indirectly dependent on the many services that they provide.

In its native range, *Z. japonica* spans from temperate to sub-tropical environments

(Young et al. 2008; Shafer et al. 2014). Geographically, this range extends from northern Japan down through southern Vietnam. The large native range of *Z. japonica* has raised concerns regarding its potential distribution in North America, because it demonstrates the species' ability to survive across a wide range of environmental conditions. The ability of an introduced species to invade a new habitat is often determined by the range of conditions that species inhabits in its native range. *Z. japonica*'s potential latitudinal range on the west coast of North America extends from British Columbia in the north well into Central America in the south (Shafer et al. 2014).

1.3 Invasive Species Ecology and Management

Invasive species are species whose introduction to an area causes, or is likely to cause, economic or environmental harm, or is detrimental to human health (Executive Order 13112, p. 1683). Invasive species are a specific subset of species introduced to areas outside of their native range. Listings for invasive species are made on a case-by-case basis, with the same species potentially having differing status in between states and at the federal level. For a listing determination to be considered by a federal or state agency, claimants are required to support the claim that there is actual or potential for economic or environmental harm, or considerable detriment to human health (NISMP 2008). In some instances, the "potential invasibility" of a species is a primary consideration in listing a species, especially if it has a history of invasion in other regions of the globe (Byers et al. 2002; Williams and Grosholz 2008).

Invasive species are managed through a number of federal statutes. Among the most direct at addressing invasive species are the National Invasive Species Act (NISA)

and the National Aquatic Nonindigenous Prevention and Control Act (NANPCA) (National Invasive Species Council 2008; Shafer et al. 2014). However, the efficacies of NISA and NANPCA at addressing invasive species are called into question because of their failure to address multiple mechanisms of introduction.

The executive mechanism for denoting and managing invasive species under NISA and NANPCA is the National Invasive Species Management Plan (NISMP). NISMP is managed collaboratively by a network of 35 federal and numerous state agencies all working towards five strategic goals:

1) **Prevention** of the introduction and establishment of invasive species in order to reduce their impact on the environment, economy, and health of constituents of the United States.

2) Development and enhancement of **early detection capabilities & rapid response** efficacy.

3) **Management & control** of populations of established invasive species to reduce their impact and further spread.

4) **Restoration** of native species and high-value environments that have been deleteriously effected by invasive species presence.

5) Maximize the effectiveness of **organizational collaboration** between international, federal, state, tribal, private, and institutional entities on invasive species issues.

NISMP's structure is intended to be synergistic by creating a framework and implementing tiers of management goals that depend on the initial conditions invasive species managers are starting from, relative to the introduction of invasive species. However, if the goals of prevention and early detection and rapid response are not met, then the importance of management response and restoration efforts increases

significantly while the achievability of the entire suite of goals is called into question. Failure to meet early goals of the NISMP has led to contrasting management strategies between states and the federal government. It can be said that these contrasting management approaches can be a direct result of poor implementation of the plan itself, which calls for organizational collaboration. Yet organizational collaboration does not always work within or between states, and the network of 35 management agencies at the federal level often leaves managers wondering whose place it is to step in regarding each individual case.

The listing process is designed to incorporate the arguments of all interested parties through collaboration and stakeholder engagement, but the science of proper stakeholder engagement is still being implemented by managers. The requirement of demonstrated or potential economic harm is weighed against the utility of a given species in a given region. Different social focuses and economic forces have resulted in contrasting listings of the same species across state or regional borders. The action of listing often has both detractors and proponents. Those interested in addressing economic damage or operational costs related to a proposed species advocate for listing and management. Lobbyists for chemical manufacturers and other management technologies also contribute support to listing. Listings are most often opposed because the species is utilized for some economic value, has socially acknowledged aesthetics, provides ecosystem services, or has historical significance.

Management of invasive species in estuaries and the near shore environment presents a unique situation for marine resource managers. In terrestrial systems, the introducer of a non-native species is usually the party experiencing harm if the species

ends up becoming invasive. However, in the coastal environment, it is often not the party transmitting the introduced species that is harmed by its introduction but rather a diverse array of stakeholders. The harm experienced by a stakeholder in the near shore environment is hard to pinpoint and is often overlooked for long periods of time. The nature of the marine environment does not lend itself to discerning changes in function. It's an inherently dynamic environment that requires considerably more effort and understanding to know when the system is acting within normal tolerances or has been perturbed by an outside influence.

In the coastal and marine environment, considerable diversity in the uses of the system as well as its broad scale are factors that often reinforce the establishment of invasive species. Coastal stakeholders damaged by invasive species are especially diverse and disconnected from one another (Williams 2007). With stakeholders that often purposely avoid or are unaware of each other's uses, establishing the actual or potential damage caused by invasive species, yet alone determining the presence and range of any given invasive species, is a remarkable challenge. The issue of aquatic invasive species is further compounded by complex connectivity that arises in the aquatic environment and a diverse array of acting management agencies with occasionally overlapping and unclear administrative duties (Stocker 2004; Williams 2007; NISMP 2008; Brown et al. 2009; Shafer, Kaldy III, and Gaeckle 2014).

1.3.1 Management of Z. japonica in the PNW

1.3.1.1 California

Attempted eradication of *Z. japonica* in California started in 2003 upon the

discovery of a population in Humboldt Bay in 2002 (Schlosser et al. 2011; Schlosser and Eicher 2012). Currently, *Z. japonica* is listed as both a noxious weed and an invasive species in California (Dean et al. 2008; Shafer et al. 2014). The goal of the eradication efforts were congruent with NISMP and the managers' intent to restore the invaded areas back to pre-invasion conditions (Schlosser et al. 2011).

The discovery of a *Z. japonica* population in the Eel River estuary south of Humboldt Bay eventually led to a suspension of eradication efforts in 2011 due to a lack of funding. Throughout eradication efforts, California managers attempted to utilize a number of different control tactics. Among some of the attempted control techniques were excavation, covering, flame heat treatments, and heater cartridges. The most effective technique tried to date in California is manual excavation of *Z. japonica* beds from the intertidal. However, preliminary results of the use of heater cartridges to heat the sediment to temperatures approaching 100 degrees centigrade suggest it may be a cheaper and less labor-intensive method of killing *Z. japonica* beds (Schlosser et al. 2011). Use of aquatic herbicides to control *Z. japonica* were not pursued in California, and due to a lack of interest in chemical control regimes, this status will likely remain unchanged.

1.3.1.2 Oregon

Oregon has not taken an official management position for or against *Z. japonica* in its coastal zone (Dudoit 2006; Shafer et al. 2014). To date, the species is afforded de facto protection alongside the native species, *Z. marina*. Responding to changes in attitude in Washington and California, a number of researchers have conducted small-scale experimental studies examining the ecology of *Z. japonica* as well as viable

removal methods. In Coos Bay, manual removal was found to be the most effective at preventing the return of *Z. japonica* in previously colonized areas (Dudoit 2006). Yamada & Rumrill (pers. communication) conducted a small scale eradication experiment using manual removal on the tideflats of the Coquille estuary in 2005. Follow-up monitoring in 2006 indicated that the removed colonies had fragmented and reestablished.

1.3.1.3 Washington

Washington State's management of *Z. japonica* has transitioned from a historical explicit requirement of zero net loss of *Zostera* spp. to listing *Z. japonica* as a noxious weed, allowing for limited control on privately owned intertidal beds (Huppert et al. 2003; Mach et al. 2010; Shafer et al. 2014). Historical protection of all seagrass species in Washington stemmed from an acknowledgement of positive ecosystem services provided by both the native and exotic species and no discernible negative impacts related to expanding *Z. japonica* populations, at least at the time the legislation was written (Shafer et al. 2014).

In 2011, Washington State decided to revisit the issue of *Z. japonica* in its coastal zone because of complaints associated with declines in aquaculture production, an important industry in the state (Fisher et al. 2011; Patten 2014; Shafer et al. 2014). Many aquaculture operators expressed concern with the colonization of a large portion of the upper intertidal by *Z. japonica*, where large, monospecific stands existed where there had previously been expanses of bare habitat (Patten 2014). After a review of the claims and lobbying by interested parties, Washington State decided to list *Z. japonica* as a noxious weed. Managers are allowing control of the introduced seagrass on private beds by

application of an aquatic herbicide (Mach et al. 2010; Fisher et al. 2011). This will be the first attempt at controlling *Z. japonica* with the herbicide Imazamox (Hamel 2012; Water Quality Program 2014).

In Washington, application of Imazamox is preferred over mechanical control, because it allows aquaculture operations to remain in production continuously rather than having to be cleared for removal operations. Additionally, chemical treatment is much less labor intensive. The decision to utilize chemical control for *Z. japonica* raises a number of questions that need to be resolved before full commitment to the program moves forward (Water Quality Program 2014).

Concerns regarding the impact of Imazamox are discussed in detail within the final environmental impact statement released by the Washington State Department of Ecology (Water Quality Program 2014). Imazamox is a highly soluble compound that breaks down into inert chemicals within a pH range of 5 to 7. Solubility in water and its low adhesion to all soil types make it unlikely that sediment chemical composition will be directly impacted (Hamel 2012; Water Quality Program 2014). Mitigation of the impact of Imazamox on *Z. marina* is achieved by limiting application to the peak growing period of *Z. japonica* (April 15th through June 30th) and not permitting the application to swales leading to *Z. marina* beds. Monitoring to determine an appropriate buffer between spray treatments of *Z. japonica* and adjacent *Z. marina* beds is to be carried out(Water Quality Program 2014). The ability to mitigate harm to *Z. marina* varies given the different relative distribution patterns associated with the two species (Shafer et al. 2014). However, as distributions of the two species move closer together, it becomes more difficult to mitigate damage to the native species. In the case where the

two species overlap, mitigating damage to *Z. marina* becomes impossible (Water Quality Program 2014).

1.3.1.4 Federal

The federal government has not taken an official position regarding the invasive status of *Z. japonica* in the United States. Management agencies have been involved with the listing status at the state level, research into best management practices and the ecology of the species, funding state management efforts, and regulating importation of potential vectors (Dean et al. 2008; Mach et al. 2010; Schlosser et al. 2011; Shafer et al. 2014). Management of aquatic invasive species has proven to be a weakness in the US national invasive species policy that arises from difficulty in controlling non-traditional vectors (Williams 2007). The current status of *Z. japonica* in the federal government is reflective of their preference for the use of state level management to control aquatic nuisance species that fall between the cracks of the NISA and NANPCA (Stocker 2004; Williams 2007; NISMP 2008). However, this strategy has resulted in the varied and often contrasting management approaches observed in states in the PNW.

1.4 Regional Professional Perspectives

Any cursory examination of management practices for *Z. japonica* within its introduced range in the US reveals the starkly contrasting management practices discussed previously. To better understand this, a series of short, semi-structured interviews – a fundamental tool utilized in social science to understand and document perceptions of the participant (Dicicco-Bloom and Crabtree 2006; Berg and Lune 2012) –

were carried out with marine resource managers and scientists from the region.

Considering the distinct roles that scientists and managers play in examining and responding to topics of environmental concern, the goal of the interviews was to identify regional differences in issues emphasized by marine resource managers and scientists studying *Z. japonica*. Interviewing individuals involved with management and science addressed the question of *why* differing approaches to management arose in the face of access to the same suite of scientific information.

1.4.1 Participant Selection

Interview participants were selected under advice from key informants in the field; sampling was not intended to be random or representative of the entire field (Berg and Lune 2012). Participants from California, Oregon, Washington, and the federal government were selected on the basis that they were (or would be) involved with research or management of *Z. japonica*.

From each state, two participants were initially selected: one from the natural resource management community and one from the research community. Only one representative was selected from the federal government because of its current deferment to state management of the species. In the case of California, only one participant was selected because of the breadth of their involvement in both management and research of *Z. japonica*. This resulted in six total interviews: three researchers, two resource managers, and one working in both capacities.

1.4.2 Data Collection

Prior to each interview, a set of open-ended questions following the protocols set forth in semi-structured interviews (Berg and Lune 2012) was provided for the participants to consider (Appendix 1). Interviews were conducted in-person whenever possible; in the event that in-person meetings were not possible, telephone interviews followed. Interviews lasted between 30 and 60 minutes; the natural pace set by each participant determined interview length and overall direction. Interview participants were allowed to spend as much time as they wanted on any given topic; "probing" questions were asked as topics arose in the discussion. Participants were not compensated for their involvement. All interviews were recorded, facilitating a more conversational atmosphere and natural flow of dialog and enabling direct transcription.

1.4.3 Professional Themes: The Ecology and Management of Z. japonica

Natural resource professionals working with *Z. japonica* listed a number of ecological characteristics pertinent to management approaches (Table 1.1). Among these are *Z. japonica*'s colonization of previously bare habitat, role as an ecosystem engineer, high reproductive and dispersal capacity (particularly when disturbed), and degradation of the intrinsic diversity of PNW ecosystems. Although each of these factors was brought up by multiple participants, the final interpretation and implementation of management across the region differed greatly depending on what aspects the ecosystem managers perceived were most important to their stakeholders.

1.4.4 Professional Themes: Invasive Species Management

Marine resource managers and researchers that discussed aspects of their

experience working with *Z. japonica* listed a number of themes (Table 1.2) that would enhance invasive species management, at both regional and national levels, in the United States. At the regional level, these were: more collaboration between all parties involved; standardizing policy across all levels; more outreach and education to the general public; and more opportunities for professional development. At the national level participants suggested expediting the process for listing invasive species, political consistency across all agencies/parties involved, and movement towards a system similar to New Zealand's.

1.5 Organization and Motivation

Understanding the ecological role that invasive species play within their introduced range is important for two reasons. The first is that any attempted management of an introduced species requires a knowledge of the niches and interactions occurring within the new range: *novel*, a new role previously unseen; *recovered*, a role previously observed but historically lost; or *superseded*, a role overtaken from other organisms currently occupying a similar station. Management action may be attempting to return to a state that is no longer obtainable without a sound understanding of the current and initial condition of an invaded environment. The second is that such introductions – while undesired and potentially detrimental – provide a venue to examine and refine ecological theory under real world conditions, thereby providing insights on how to better respond to and prevent future introductions. Without a reasonable grasp of the ecology of an introduced species, management action may:

1) Fail to identify characteristics that are ultimately detrimental to management efforts;

2) Fail to identify characteristics of an introduced species that enhance the health or function of an ecosystem and need to be accounted for in determination of management action; and/or

3) Fail in identifying characteristics of the invader or invaded environment that may better inform prevention of the spread of invasive species.

In the instance of *Z. japonica* in the PNW, the research available describes the plant's physiology, reproduction, growth, and direct interaction with *Z. marina*. Research on both the native species, *Z. marina*, and seagrasses in other coastal ecosystems has demonstrated that seagrass habitat enhances the survival of juvenile fishes and crabs when compared to unstructured habitat (Heck Jr. et al. 2003). Outside of work done by Baldwin and Lovvorn (1994) on preferential use of *Z. japonica* by waterfowl and Posey (1988) and Ferraro and Cole (2007, 2011, 2012) regarding the role of *Z. japonica* in structuring benthic infaunal communities, only Rose et al. (2010) examined how the species is used by the endemic community of fish and crab. This work attempts to elucidate these patterns of use in two estuaries within the introduced range of *Z. japonica* in the PNW.

The goal of my research is to better understand the composition of the fish and crab community that utilizes intertidal and aquaculture habitats within the PNW. The fundamental questions that my research attempts to answer are:

1) How is the introduced seagrass, *Z. japonica*, used by the intertidal community of fishes and crabs within the PNW?

2) How does the use of *Z. japonica* by this community differ in comparison to that of *Z. marina*?

3) Is the use of *Z. japonica* by this community the same across two bays in the PNW?

4) Finally, given the complex political arena for invasive species

> management, are there any suggestions that can be offered to inform more robust management of *Z. japonica* in the PNW?

While this research will not provide all-encompassing answers to these questions, my investigation does provide a glimpse into the murky waters of PNW estuaries for both researchers and managers to build upon. It also adds to the base of knowledge on *Z. japonica* within the PNW, thus informing further research and management oriented around both environmental function and social value.

This thesis is comprised of three chapters. Chapter 1 provided an introduction to seagrass ecology, an overview of the introduction and management of *Z. japonica* in the PNW, a discussion of invasive species management within the US, and an introduction to interview methods. Chapter 2 examines the community composition of Willapa Bay and Yaquina Bay using univariate and multivariate analysis of observations recorded from video footage and deployment of small breder traps. Finally, Chapter 3 summarizes the findings and offers a series of "lessons learned" from interviews of researchers and managers of *Z. japonica* across the PNW.

1.6 Tables and Figures

Table 1.1 Ecological characteristics of Z. japonica and associated concerns that prompted/would prompt management in the PNW.

	Washington	Oregon	California
Colonization of historically bare substrate	i. Negative interaction with the shellfish aquaculture industry	i. Change from historically bare to vegetated substrate	i. Concern that the migratory shorebird community would change as a result of decreased un-vegetated substrate.
Role as an ecosystem engineer in its native range	i. Potential competition with *Z. marina* ii. Facilitation of the establishment of other non-natives	i. Potential competition with *Z. marina* ii. Facilitation of the establishment of other non-natives	i. Potential competition with *Z. marina* ii. Facilitation of the establishment of other non-natives
Intrinsic value of native ecology	i. Facilitation of the establishment of other non-natives ii. Homogenizes native diversity iii. Degradation of ecosystem health	i. Facilitation of the establishment of other non-natives ii. Homogenizes native diversity iii. Degradation of ecosystem health	i. Facilitation of the establishment of other non-natives ii. Homogenizes native diversity iii. Degradation of ecosystem health
Climate change	i. Expansion under warmer conditions ii. Competition with *Z. marina*	i. Expansion under warmer conditions ii. Competition with *Z. marina*	i. Expansion under warmer conditions ii. Competition with *Z. marina*

Table 1.2: Suggested improvements to invasive species management in the US by natural resource professionals in the PNW.

Suggestion	Details	Desired Outcome
Ecological cost of management	i. Assess ecological impact of management decision ii. Identify social values associated with the end state after management	i. Prevent undue environmental damage ii. Provide clear objectives for management iii. Weighs fiscal, environmental, and social cost of action
Explicit statement of management position	i. Consistency in official position across all areas and parties. ii. Explain necessity of multiple positions	i. Incorporates EBM principles ii. Designates clear and obtainable goals iii. Clarity leads to better public compliance
Regional Collaboration	i. Discussion across all levels and parties ii. Centralized management (i.e., New Zealand)	i. Clarification of management goals ii. Quicker responses to introductions iii. Standardization of prevention and monitoring
Outreach, education, and professional development	i. Separate engagement of public and stakeholders ii. Regular professional training opportunities	i. Clarifies *why* invasive species are a problem ii. Creates personal investment in outcome iii. Utilization of stakeholder knowledge iv. Standardization of monitoring and management goals
Standardization and expedition of invasive species listing	i. Standardize listing process across all levels ii. Shorter process in official designation of invasive status	i. Easier compliance and communication of objectives. ii. Lower fiscal cost associated with process iii. Less confusion with process
Incorporate and respond to social pressures	i. Active, internal reexamination of practices and policies	i. More fluid and acceptable policies ii. Reflective of social pressures

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Chapter 2: Contrasting Fish and Crab Associations in the Introduced Seagrass *Zostera japonica* and Its Native Congener, *Zostera marina*

2.1 Abstract

Introduction of non-native species often results in fundamental changes in the structure and function of disturbed environments. In the PNW, the introduced seagrass *Zostera japonica* is rapidly expanding. A number of studies have compared the benthic and epifaunal communities in *Z. japonica* to those of the native seagrass, *Z. marina*. However, none have examined the communities of fish and crab using the two seagrasses. The goal of this project was to examine the community composition of a variety of available estuarine habitats in Willapa Bay, Washington, and Yaquina Bay, Oregon, via paired deployment of cameras and small fish traps. In Willapa Bay, *Z. japonica*, *Z. marina*, clam aquaculture, and on-ground oyster aquaculture were examined. In Yaquina Bay, *Z. japonica*, *Z. marina*, and bare substrate were examined. A total of 11 species, with 10 occurring in Willapa Bay and 9 occurring in Yaquina Bay, were observed in video footage. Habitat was a significant predictor of Catch Per Unit Effort (CPUE) for the most abundant species in Yaquina Bay but not for those in Willapa Bay. Community composition was significantly different between habitats in each estuary but not between estuaries. Explicit comparisons of seagrass habitat in each estuary indicate that community composition of the two seagrasses differs in Yaquina Bay but not in Willapa Bay. We conclude that community composition of seagrass structure varies little between the two bays and that local variation is highly dependent on the availability of structured habitats. Additionally, the distribution of *Z. japonica* relative to *Z. marina* may drive differences in community composition between seagrass habitats in these estuaries.

2.2 Introduction

Seagrasses are principal components of highly productive and diverse coastal ecosystems (Larkum et al. 2006). Serving as an ecosystem engineer (Wright and Jones 2006), seagrasses influence the physical, chemical, and biological conditions of the coastal environment, providing many services that influence pelagic and benthic systems as well as coastal communities (Costanza et al. 1997; Huppert et al. 2003; Larkum et al. 2006). Providing a primary food source for large marine herbivores and promoting the growth of secondary food such as epiphytes, seagrasses support robust and diverse ecological communities (Heck Jr. et al. 2003; Orth et al. 2006; Hughes et al. 2009). Seagrass habitats stabilize coastal shorelines by dampening wave energy and slowing water flow, accumulating sediments in the nearshore zone and preventing offshore transport (Larkum et al. 2006; Barbier et al. 2011). They also serve as a sink for organic carbon and dissolved nitrates (Larned 2003; Orth et al. 2006).

Z. japonica is a small intertidal seagrass native to the shores of the western Pacific that has been present in the PNW since before 1957 (Bigley and Barreca 1982; Harrison and Bigley 1982). Believed to have been introduced to Willapa Bay, Washington, via the aquaculture industry where it was used as a packing material to transport Pacific oysters from Japan, *Z. japonica*'s current range extends from British Columbia, Canada to Humboldt Bay, California (Harrison and Bigley 1982; Young et al. 2008; Mach et al. 2010). Expansion of *Z. japonica* has been attributed to both the dispersal of seeds via transport with migratory waterfowl and delivery via wrack on trailers and propellers (Figuerola and Green 2002; Shafer et al. 2014).

In the Pacific Northwest (PNW), the importance of the native seagrass, *Z. marina*,

in supporting biodiversity (Hosack et al. 2006) as a nursery (Hughes et al. 2014) is well recognized. Due to their importance, seagrasses are afforded considerable protections throughout the entire region (Shafer et al. 2014). However, the impact of this rapidly expanding population of introduced seagrass on the coastal ecology of the PNW is poorly understood (Mach et al. 2010).

Within its introduced range, a number of studies have tried to clarify the role of *Z. japonica* in the context of its introduced environment. Colonizing the upper intertidal, a region previously comprised of unstructured tidal flats, *Z. japonica* fundamentally alters the structure of the benthic macrofaunal community (Posey 1988; Ferraro and Cole 2012). Baldwin and Lovvorn (1994) found that *Z. japonica* is used extensively by migratory waterfowl and is often preferred over its native counterpart. Additionally, *Z. japonica* has been found to be a sink for dissolved nitrates in the PNW, compared to the unstructured habitats it colonizes (Larned 2003). In Washington, there is evidence that *Z. japonica* negatively impacts the culture of Manila clams, a multimillion dollar industry. Patten (2014) observed both higher clam quality and growth in removal plots than in adjacent *Z. japonica* beds, but the observed patterns may have been artifacts of the small scale of the removal plots. The effects of *Z. japonica* on recruitment of Manila clams remain unclear, with multiple studies reporting contrasting results (Tsai et al. 2010; Patten 2014; Ruesink et al. 2014). While a number of studies to date have examined community composition of *Z. marina* or *Z. japonica* (Posey 1988; Murphy et al. 2000; Hosack et al. 2006; Ferraro and Cole 2010), few have explicitly contrasted the communities using these species. Knight et al. (2015) contrasted the epifaunal communities, and Ferraro and Cole (2011, 2012) compared benthic macrofaunal

communities, but to date no studies have examined use by larger nekton.

The purpose of this study was to examine the composition of nekton that utilize several intertidal habitats in two bays in the PNW using underwater video cameras. Habitats studied in Willapa Bay, Washington, were *Z. japonica* and *Z. marina* beds, graveled manila clam aquaculture beds, and on-ground oyster aquaculture. In Yaquina Bay, *Z. japonica* and *Z. marina* beds were examined in addition to unstructured bare habitat. We hypothesized that the overall community structure would not differ between the two estuaries, but community composition within the different habitats would differ in both Willapa Bay and Yaquina Bay.

2.3 Methods

2.3.1 Study Locations

This study was conducted in two PNW estuaries: Willapa Bay on the southwest coast of Washington State (Figure 2.1) and Yaquina Bay on the central Oregon coast (Figure 2.2). Willapa Bay is a drowned river estuary draining an area of roughly 2,900 km^2 and has a mean tidal range at its mouth of 1.9 m, with approximately 55% of the intertidal tide flats exposed during extreme low tides (Hickey and Banas 2003). Willapa Bay is characterized by large scale, high-production bivalve aquaculture. Of the available 21,502 ha of intertidal, 4,888 ha have aquaculture, 1,764 ha of which is active oyster aquaculture (Dumbauld and McCoy, in press; Dumbauld et al. 2011). *Z. japonica* primarily occupies a tidal elevation of approximately 0 ft. mean lower low water (MLLW), while *Z. marina* occurs at tidal heights less than 0 ft. MLLW. Zonation patterns of *Z. japonica* and *Z. marina* in Willapa Bay differ from those observed in

Yaquina Bay (Figure 2.3), with the two species showing both “mosaic” (patches of *Z. marina* growing in depressions within *Z. japonica* beds in the mid and upper intertidal) and “overlap” (large beds of *Z. japonica* and *Z. marina* converging and intermixing in the mid intertidal) (Shafer et al., 2014).

The fourth largest estuary in the Oregon coastal zone, Yaquina Bay is a drowned river estuary with a drainage area of roughly 660 km^2 and mean tidal range at itsmouth of 1.8 m (Percy et al. 1974; Hickey and Banas 2003; Northwest Area Committee 2005). Relatively large seagrass meadows of both *Z. marina* and *Z. japonica* occur primarily in the lower and middle bay relative to the bay mouth (Specht et al. 2000; Northwest Area Committee 2005; Young et al. 2008). In Yaquina Bay, *Z. japonica* occurs at approximately 4 ft. above MLLW, while *Z. marina* occurs at a tidal height below 0 ft. MLLW (Clinton et al. 2007). The distribution of *Z. japonica* in Yaquina Bay relative to its native congener is described as “disjunct” by Shafer et al., 2014 (continuous bands of the non-native species along the upper intertidal separated from *Z. marina* in the lower intertidal by a distinct band of bare substrate (Figure 2.3)).

The predominant intertidal habitat types examined in Willapa Bay (Figure 2.1) were: (1) on-ground clam aquaculture beds (approximately 4 ft. MLLW); (2) on-ground oyster aquaculture beds (approximately 2 ft. MLLW); (3) *Z. japonica* beds (approximately 4 ft. MLLW); and (4) *Z. marina* beds (approximately 1 ft. MLLW). The primary study site was located adjacent to Oysterville (Figure 2.1) because of the availability of overlapping beds of *Z. japonica* and *Z. marina* as well as graveled clam aquaculture beds in the upper intertidal. A secondary site offshore Nahcotta, Washington (Figure 2.1b), was chosen because of the availability of an on-ground oyster bed. The two

sites chosen were relatively close in proximity to one another. The site of Nahcotta has a much higher degree of hard structured habitat than those at Oysterville due to the presence of oyster culture and relatively large oyster hummocks.

In Yaquina Bay, this study was conducted at Sally's Bend (Figure 2.2), located in the lower estuary on the north shore of the bay, because of the availability of all three predominant intertidal habitats, which were: (1) bare substrate (approximately 3 ft. MLLW)*;* (2) *Z. japonica* beds (approximately 4 ft. MLLW); and (3) *Z. marina* beds (approximately -0.5 ft. MLLW).

2.3.2 Breder Traps

To assess the accuracy of identifying organisms in the often turbid estuarine environment, small Plexiglas breder traps (Figure 2.4) were deployed in tandem with video cameras as an independent measure of the efficacy of the video system.

2.3.2.1 Breder Trap Design & Deployment

Breder traps used for this study were constructed from three pieces of molded Plexiglas epoxied together, creating a small box with a depth of 31 cm, width of 15 cm, and height of 15 cm (Figure 2.4). In the field, we attached two 31 cm Plexiglas wings to the front of the box with wing nuts, creating a 31 cm wide v-shaped funnel leading into a 1 cm opening to the box of the trap. Traps were not baited and were secured in each habitat using two 0.5 m stakes driven through the box of the trap and roughly 20 cm into the sediment.

Breder traps were deployed during the low tide prior to video camera deployment and retrieved on the first low tide of the next day, after approximately 24 hours. All traps were deployed within 20 m of cameras, in sets of three, within each of the intertidal habitats. Within each set, two traps were deployed with wings and openings parallel to the shoreline, and one faced towards the tidal channel. The total length of fish caught in the trap was recorded to the nearest millimeter. Carapace width for any crustaceans was measured using calipers and recorded to the nearest millimeter.

2.3.3 Camera Deployment

2.3.3.1 Video Camera Mount Design

Each video camera was mounted on a 1 ½ inch diameter PVC stand with a 1 inch diameter PVC base pole (Figure 2.5a). A high definition (HD) GoPro camera was held 50 cm above the substrate and 50 cm away from the base pole by two 50 cm lengths of PVC connected to a tee. One of the 50 cm lengths was set in the bottom of the tee and placed against the sediment surface, providing the 50 cm offset. The second length was placed in the normal, perpendicular position of the tee to hold the camera out of view of the base pole. Stands were deployed and retrieved by means of a third length of PVC tied into the top of the tee. The length of the top PVC piece was either 2.5 m, for deployment into *Z. marina* and oyster aquaculture beds, or 1.5 m, for deployment into *Z. japonica* beds, clam aquaculture beds, and bare substrate. The total length of the stands was either 2 m or 3 m depending on whether they were to be used in the upper or lower intertidal, respectively. The extra length of the top piece of PVC was set to extend roughly 1 m out of the water approximately 2 hours before the high tide in order to allow for deployment and retrieval

(Figure 2.5b).

All of the stands were secured onto a base pole during deployment. The base pole was a length of 1 inch diameter PVC that was 1 m longer than the total length of the stands, resulting in a 3 m base pole that was paired to the 2 m stands and a 4 m base pole that was paired to the 3 m stands. The extra 1 m of length of the base poles was driven into the sediment to secure the base pole and stand. The base poles were also cut to stand approximately 1 m out of the water 2 hours before a high tide in order to allow for deployment of the stands onto the base poles. Stands were secured to the base poles using an eye bolt and wing nut threaded through both the stand and base pole (Figure 2.5b). The cameras, oriented straight down, were secured to the arm of the stand with adhesive mounts provided by the manufacturer and reinforced with zip-ties.

2.3.3.2 Video Deployment

On the low tide prior to camera stand deployment, the base poles were driven roughly 1 m into the substrate. In *Z. marina* habitat, the seagrass within the viewing area of the camera was trimmed to a height of roughly 30 cm to prevent seagrass from wrapping around the arm of the camera or reducing the ability to identify observed organisms. The date and habitat type were recorded on each camera prior to deployment. Stands and cameras were slid over the base poles and secured approximately 2 hours prior to an incoming high tide and retrieved roughly 2 hours after high tide.

2.3.3.3 Camera Settings

Prior to deployment, each camera was fitted with an additional BacPac battery,

fully charged and placed into a manufacturer-provided, polycarbonate waterproof case. Every camera was set to record video at 720p resolution, 30 frames per second (fps), and a wide angle. Use of the BacPac provided approximately 4 hours of video at the above settings. Video was recorded continuously once deployed. Each 4 hour block of video was saved in approximately 30 minute segments.

2.3.3.4 Data Collection

In Willapa Bay, video was recorded from July 20 - 24, 2013, in *Z. japonica*, clam, oyster, and *Z. marina* habitats. This resulted in nearly 104 hours of continuous video. In Yaquina Bay, video was recorded from August 5 - 9, 2013, in bare, *Z. japonica*, and *Z. marina* habitats. This resulted in approximately 140 hours of continuous video.

An observation was recorded for each unique individual that entirely entered the frame. In instances where an individual entered, left, and then re-entered the frame, a new observation was recorded unless there was absolute certainty that the individual was the same. This was only performed when an organism with distinct markings or of relative rarity was seen in close succession. To account for bias associated with the presence of the boat or researchers near the camera, no observations were recorded until the boat motor was no longer audible in the film and it was apparent the set up was no longer being handled. For every observation, the time the individual completely entered the frame, the direction it entered from, and its lowest taxonomic level were recorded.

For every video segment, the start time (T_s), usable time (T_u), and an obstruction value (P_{obst}) were recorded. T_s was the time within the deployment at which the video segment began. T_u was the amount of time within that video where observations could be

recorded as per the limitations set above. P_{obst} was a proportion denoting an estimate of how much of each video segment was unable to be used for recording observations. An obstruction value of 0 corresponded to a video void of obstructions, while a value of 1 denoted a video where no observations could be recorded. Conditions that contributed to high obstruction values were suspended sediment, algae or seagrass wrapping around the camera lens, or the incomplete submersion of the camera for all or part of the video.

Using the amount of T_u in each video segment (v) and P_{obst}, the amount of time adjusted for the proportion of obstruction within each video where observations could be made ($T_{effort\ adj.}$) was calculated for each video segment:

$$T_{effort\ adj.} = \sum_{i=1}^{v} \mathrm{T_u}(1 - P_{obst}) \quad (2.1)$$

Subsequently, the total number of observations (T_{obs}) per hour of $T_{effort\ adj.}$, or CPUE for each video segment, was calculated for each species (s_i):

$$CPUE\ s_i = \sum_{i=1}^{v} (T_{obs}/T_{effort\ adj.}) \quad (2.2)$$

2.3.4 Statistical Methods

For breder trap deployments, each day and each orientation were used as a sampling unit. Three breder traps were deployed alongside the cameras within each habitat ($n_{\mathrm{Habitat\ WB}} = 4$, $n_{\mathrm{Habitat\ YB}} = 3$) for four 24 hour deployments. This resulted in a sample size of 48 traps in Willapa Bay and 36 traps in Yaquina Bay (Tables 2.1, 2.2). For video camera deployments (Tables 2.3, 2.4), we used each sampling day as the individual

sampling unit. Five samples were collected for each habitat type resulting in a total sample size of 20 in Willapa Bay and 15 in Yaquina Bay.

Preliminary univariate parametric analyses were carried out in the '*stats*' package of the R statistical computing software (Chambers et al. 1992; R Development Core Team 2014). For each species captured in the breder traps, a two-way ANOVA was performed examining the mean catch as predicted by habitat type and orientation direction. For each species observed in the video footage, a one-way ANOVA of mean CPUE (Table 2.5) as predicted by habitat type was performed on individual subsets for Willapa Bay and Yaquina Bay. In instances where mean catch or mean CPUE for a given species was significant, a Tukey's Honest Significant Difference (TukeyHSD) test with Bonferroni adjusted p-values was performed (Table 2.6).

Prior to carrying out the one and two-way ANOVA's and TukeyHSD contrasts, we examined how breder trap and video data fit the assumptions of each respective test. For both the ANOVA models and the TukeyHSD contrasts, the assumptions are the same: 1) independence of samples, 2) normality, and 3) homogeneity of variances.

Independence between both breder trap deployments and samples of the video data were thought to be reasonable. Since breder traps are a passive collection technique similar to the video camera, we do not believe that the presence of one trap influences the catch experience in another, especially between habitat types. Unpublished video data in Willapa Bay suggests that distributions of intertidal organisms do not significantly change across 24-hour time scales (Dumbauld 2014, pers. communication). All of the samples in a given bay were recorded in the same tidal series ± 2 hours relative to the high tide, making each video sample independent. Distributions of each species were

relatively normally distributed. Moderate deviations from normality did not preclude use of the one-way ANOVA because of the technique's robustness against non-normal data (Schmider et al. 2010). Homogeneity of variances were examined and only found to deviate significantly between habitat types for three-spine stickleback (*Gasterosteus aculeatus*) observed in the video footage and for shiner perch (*Cymatogaster aggregata*) catch in breder traps, as predicted by habitat. Since sample sizes were equal and ANOVA has been found to be most robust against the violation of homoscedasticity in groups of equal population size, we decided to continue to use the technique (Rogan and Keselman 1977).

In addition to parametric statistics, multivariate analyses on community composition were employed, including Non-metric Multidimensional Scaling (NMDS) within the '*ecodist*' package of R (Clarke 1993; Clarke and Ainsworth 1993; McCune and Grace 2002; Goslee and Urban 2007). NMDS, MRPP and ISA analyses were performed to determine significant habitat associations and examine the community composition as it differed between:

1) Seagrass structure (*Z. marina* and *Z. japonica* combined) in Willapa Bay and Yaquina Bay;
2) Oyster, clam, *Z. japonica*, and *Z. marina* habitats in Willapa Bay;
3) Unstructured tide flat, *Z. japonica*, and *Z. marina* beds in Yaquina Bay;
4) *Z. japonica* and *Z. marina* beds in Yaquina Bay; and
5) *Z. japonica* and *Z. marina* beds in Willapa Bay.

NMDS analyses were paired with Multi-Response Permutation Procedure (MRPP) analyses utilizing the '*vegan*' package of R (Oksanen et al. 2015) to provide a means of

testing the hypotheses of no effect of bay, habitat type, or seagrass species on overall species composition (Mielke et al. 1981; Biondini et al. 1988; McCune and Grace 2002). Additionally, an Indicator Species Analysis (ISA) using the R package '*labdsv*' (Roberts 2013) was used to examine associations of observed species with bay, habitat type, and seagrass species (Dufrêne and Legendre 1997; McCune and Grace 2002).

Adjusted CPUE values were converted to proportions for use in multivariate analyses (NMDS, MRPP, and ISA). The coefficient of variation and skew of the data were checked and minimized using relativization to each species' respective maximum (Biondini et al. 1988; Clarke 1993; Dufrêne and Legendre 1997; McCune and Grace 2002). The threshold used to denote acceptable variation in the data was a coefficient of variation < 200. A relatively normal distribution was insured by minimizing the skew of each dataset to between ±2. No further transformation was necessary after relativization.

NMDS analyses were conducted using the Sorenson distance measure (McCune and Grace 2002) in the '*ecodist*' package of the R computing software and were iterated 1000 times with random starting positions. One runthrough with real data was completed for each NMDS analysis. To determine the final configuration and dimensionality, the stress of all 1000 iterations was plotted by the number of dimensions of the solution, ranging from 1 to 3 dimensions. Once stress was minimized below 0.2, the dimensionality was chosen. Each NMDS was then iterated 1000 more times with the chosen number of dimensions, providing a pool of 1000 possible configurations of the desired dimension. The configuration with the lowest stress coefficient was chosen as the final configuration for all analyses. For every species examined, a weighted average score was calculated and plotted on the ordination diagrams using the `*vegan*` package in R

(Oksanen et al. 2015).

For all MRPP analyses, the Sorenson distance measure was also used (McCune and Grace 2002). Sorenson distance was chosen because of its common use with ecological datasets and its propensity to give less weight to outliers than other non-proportional and Euclidean distance measures (McCune and Grace 2002). For MRPP, the p-value and the chance with group agreement (A-value) are reported (Table 2.7). The p-value denotes the probability that selected groups differ significantly within ordination space from groups randomly chosen from the population. The A-value is a measure of the effect size occurring in observed groups. For groups where all items are exactly the same, A = 1, while for groups where items are as similar as to be expected by chance, A = 0. Within the field of community ecology, often A < 0.10 (McCune and Grace 2002). Both the A-value and the p-value should be used when interpreting the results of an MRPP analysis.

ISA analyses were conducted in the '*labdsv*' package of the R statistical computing software (Roberts 2013). Each ISA analysis for the comparisons described above was conducted using 10,000 randomizations in the Monte Carlo test. For ISA, indicator values rating the strength of a species' association with a given habitat are reported in addition to a p-value examining the significance of the indicator values (Table 2.8). Indicator values range from 0 to 1 for each species. A species with an indicator value of 1 for a given habitat means the presence of that species is associated entirely with the indicated habitat type without error, while an indicator value of 0 means there is no indication that species is associated with that habitat (McCune and Grace 2002).

2.4 Results

2.4.1 Breder Trap Community

Seven species of fish and crab were found in breder traps across both bays (Tables 2.1, 2.2; Figure 2.6). All of these species were caught in Willapa Bay, with the three most common being three-spine stickleback (*Gasterosteus aculeatus* (n = 78)), shiner perch (*Cymatogaster aggregata* (n = 64)), and Dungeness crab (*Metacarcinus magister* (n = 16)). Trap orientation was not found to be significant for any species in Willapa Bay.

In Willapa Bay, habitat was found to be the only significant predictor of mean catches of shiner perch in breder traps (two-way ANOVA, $F_{2, 50} = 3.088$, $p = 0.035$). More shiner perch were caught in *Z. marina* than in any other habitat type. On average, approximately 1.58 more shiner perch (TukeyHSD, $p = 0.03$) were caught in *Z. marina* than in clam aquaculture beds. Additionally, 1.45 more shiner perch (TukeyHSD, $p = 0.06$) were caught in *Z. marina* than in oyster beds. This effect, however, was only marginally non-significant.

Catches of three-spine stickleback in Willapa Bay were found to be significantly predicted by habitat (two-way ANOVA, $F_{Hab.\ 2,50} = 6.387$, $p_{Hab} < 0.001$). Approximately 1.5 times more three-spine stickleback were caught in *Z. marina* than in any other habitat type. On average, approximately 3.85 more three-spine stickleback (TukeyHSD, $p < 0.001$) were caught in *Z. marina* than in clam aquaculture beds. Approximately 2.45 more three-spine stickleback (TukeyHSD, $p = 0.041$) were caught in *Z. marina* than in oyster aquaculture beds. Approximately 2.71 more three-spine stickleback (TukeyHSD, $p = 0.019$) were caught in *Z. marina* than in *Z. japonica*.

In Yaquina Bay, only 3 of the 7 taxa were represented (Tables 2.1, 2.2): shiner

perch (n = 19), Dungeness crab (n = 8), and staghorn sculpin (n = 3). Mean catch of shiner perch, Dungeness crab, and staghorn sculpin were not significantly predicted by trap orientation or habitat type (two-way ANOVA, $p >> 0.05$).

2.4.2 Video: Univariate Tests

Between Willapa Bay and Yaquina Bay, a total of 13 distinct taxa were identified from video footage. Ten occurred in Willapa Bay (Figure 2.7), and 9 occurred in Yaquina Bay (Figure 2.8). Total CPUE for all species varied greatly between habitats and between bays. In Willapa Bay, *Z. marina* and *Z. japonica* had the highest total mean CPUE for all species, with 123.05 individuals per hour and 96.09 individuals per hour, respectively (Table 2.4). In Yaquina Bay, the highest CPUE was in bare substrate and *Z .marina*, with 336.59 individuals per hour and 212.63 individuals per hour, respectively (Table 2.4).

The three most common species observed in Willapa Bay video were surfperch (n = 1422) (Figure 2.10), staghorn sculpin (n = 1173) (Figure 2.11), and three-spine stickleback (n = 242) (Tables 2.3, 2.4). Adjusted CPUE differed significantly between habitats only for three-spine stickleback (one-way ANOVA, $p < 0.001$) (Table 2.5). However, *post hoc* Tukey contrasts did not have the necessary power to determine where these differences originated (Table 2.6a).

In Yaquina Bay video, the most commonly observed taxa were staghorn sculpin (n = 2541) (Figure 2.11), surfperch (n = 2148) (Figure 2.10), and Dungeness crab (n = 100) (Figure 2.9). For all species except Dungeness (Figure 2.9) and staghorn sculpin (Figure 2.11), habitat was not found to be a significant predictor of adjusted CPUE (one-way ANOVA, $p > 0.05$) (Table 2.5). CPUE of Dungeness crab was found to significantly

differ across habitat types (one-way ANOVA, $F_{2,12} = 4.71$, $p = 0.031$). Approximately 12.68 (TukeyHSD, $p < 0.05$) more Dungeness crabs were observed per hour of video footage in bare substrate than in *Z. japonica* (Table 2.6b). There was no significant difference in the CPUE of Dungeness crab between *Z. japonica* and *Z. marina* (TukeyHSD, $p > 0.05$) (Table 2.6b).

The adjusted CPUE of staghorn sculpin was found to differ significantly (one-way ANOVA, $F_{2,12} = 8.80$, $p = 0.004$) between habitat types. Approximately 213.6 (TukeyHSD, $p < 0.01$) more sculpin were observed per hour of video in bare substrate than in *Z. japonica* (Table 2.6b). Adjusted CPUE did not differ significantly (TukeyHSD, $p > 0.05$) between either the two seagrass species or between bare substrate and *Z. marina* (Table 2.6b).

2.4.*3 Video: Multivariate Tests*

2.4.3.1 Community Composition Between Bays

The final configuration of the NMDS analysis examining the overarching community composition between Willapa Bay and Yaquina Bay using only habitats available in both bays (*Z. marina* and *Z. japonica* beds) was composed of three dimensions, with a stress coefficient of 0.1175 explaining 92.96% of the total variation observed in the data. The NMDS plots examining community by bay show slight overlap in the first and second dimensions (Figure 2.12a) and some separation in the first and third dimensions (Figure 2.12b).

Results of the MRPP (Table 2.7) for the groups of Willapa Bay (n = 20) and Yaquina Bay (n = 15) indicated that patterns observed in fish CPUE are significantly

different than what would be expected from random. There is some evidence that communities in the two bays are distinct from one another, but the effect strength is weak (MRPP, p = 0.016, A = 0.07023). The low A-value also indicates that the point clouds overlap each other considerably within ordination space.

The observed differences in communities between bays can be parsed out by ISA (Table 2.8). Three-spine stickleback were found to be associated with Willapa Bay (ISA, indicator value = 0.90, p = 0.001) since none were observed in Yaquina Bay. Locations of the plotted species values relative to the bay centroids in Figures 2.4a and 2.4b show three-spine stickleback closely associated with Willapa Bay, having little interaction with the Yaquina Bay centroid. Gunnels (family *Pholidae*) were significantly associated with Yaquina Bay (ISA, I-value = 0.70, p = 0.004).

2.4.3.2 Community Composition Between Habitats in Willapa Bay

The final configuration of the NMDS analysis examining the community composition among habitats in Willapa Bay was composed of three dimensions, with a stress coefficient of 0.1320 explaining 92.85% of the total variation in the data. Examination of the ordination plots (Figure 2.13) indicates that there is overlap between the four habitat centroids in Willapa Bay across all dimensions. The centroids of the clam and oyster habitats are particularly large, indicating that there was a large amount of variance between samples. The least amount of overlap of the habitat centroids appears to occur between the two seagrass species. The close overlap of the clam aquaculture centroid and the *Z. japonica* centroid is likely because the graveled clam aquaculture habitat included patches of clams within a larger *Z. japonica* meadow.

Results of the MRPP (Table 2.7) for clam (n = 5), oyster (n = 5), *Z. japonica* (n = 5), and *Z. marina* (n = 5) indicate the examined habitats are significantly different than groups randomly selected from the population (MRPP, p = 0.012). There is some evidence indicating that community structure differs between habitats in Willapa Bay, but the effect size is relatively small (MRPP, A = 0.0994). An effect strength bordering on the margin of 0.10 indicates community structure is relatively similar, but the observed delta values for *Z. japonica* and oyster habitat appear to differ the most from one another. Upon examination, however, the strength of even this greatest difference appears weak, with considerable overlap of the points and centroids for not only oyster aquaculture beds and *Z. japonica*, but all other examined habitats as well.

ISA results show that both Dungeness crab and three-spine stickleback are significantly associated with individual habitats in Willapa Bay (Table 2.8). Dungeness crab were significantly associated with oyster aquaculture habitat (ISA, I-value = 0.755, p = 0.004), and weighted average species scores on the ordination plot show Dungeness crab clearly within the oyster aquaculture centroid. Three-spine stickleback were found to be associated with clam aquaculture habitat (ISA, I- value = 0.464, p = 0.03).

2.4.3.3 Community Composition Between Seagrasses in Willapa Bay

The final configuration of the NMDS analysis examining the community composition between seagrasses in Willapa Bay was composed of two dimensions, with a stress of 0.1108 explaining 95.34% of the total variability in the data. The ordination plot (Figure 2.15) shows the centroids of *Z. japonica* and *Z. marina* interact some, and that the individual samples cover a large range of values.

MRPP results (Table 2.7) for the groups of *Z. japonica* (n = 5) and *Z. marina* (n = 5) show that the two seagrass species are not significantly different than groups selected from the population at random (MRPP, p = 0.328). The wide range within the individual samples for each of the groups are just as likely to come from a random grouping within ordination space as they are to comprise *Z. japonica* and *Z. marina* beds. This suggests no significant difference exists between the community structure of *Z. japonica* beds compared to *Z. marina* beds in Willapa Bay.

ISA results (Table 2.8) indicate that under an explicit comparison of *Z. japonica* to *Z. marina* within Willapa Bay, none of the species observed in video footage are more likely to select one seagrass species over another (ISA, p > 0.5) (Table 2.8). ISA results are similar to those of the MRPP.

2.4.3.4 Community Composition Between Habitats in Yaquina Bay

The final configuration of the NMDS analysis examining the community composition between habitats in Yaquina Bay was composed of three dimensions, with a stress of 0.0998 explaining 96.23% of the total variation in the data. The NMDS plots (Figure 2.14) show the centroids of the three habitats as being relatively distinct from one another across all of the dimensions examined, with the bare and *Z. marina* centroids overlapping.

MRPP results (Table 2.7) for the groups of bare substrate (n = 5), *Z. japonica* (n = 5), and *Z. marina* (n = 5) show that the habitats are significantly different from groups chosen at random from the population (MRPP, p = 0.023, A = 0.1396), and that habitats in Yaquina Bay are moderately distinct from one another. Differences in deltas indicate

that communities are most different in *Z. japonica* and bare substrate and most similar in bare substrate and *Z .marina*.

ISA results (Table 2.8) indicate that there is some evidence that Dungeness crab and surfperch show habitat associations in Yaquina Bay. Dungeness crab are significantly associated with bare substrate in Yaquina Bay (ISA, interaction value = 0.563, p = 0.048), which supports the TukeyHSD contrasts reported in Table 2.6. There is some evidence to suggest that surfperch are relatively closely associated with *Z. japonica* beds. However, the results are only marginally significant (ISA, interaction value = 0.591, p = 0.066).

2.4.3.5 Community Composition Between Seagrasses in Yaquina Bay

The final configuration of the NMDS analysis examining the community composition between seagrasses in Yaquina Bay was composed of two dimensions, with a stress of 0.1002 explaining 96.34% of the total variability in the data. The ordination plot (Figure 2.16) shows that the centroids of *Z. japonica* and *Z. marina* in Yaquina Bay are relatively separated. Individual sample points within ordination space appear to have a much more constrained range than that observed of seagrass beds in Willapa Bay.

MRPP results (Table 2.7) for the groups of *Z. japonica* (n = 5) and *Z. marina* (n = 5) show some evidence that the two seagrasses are different from groups chosen randomly from the population, and that the community composition of the two species are moderately different from one another (MRPP, p-value = 0.059, A = 0.1552). Visual examination of the ordination plots shows that community composition between the two species is markedly separate from one another, and both have relatively narrow ranges in

ordination space.

ISA results (Table 2.8) suggest some evidence that surfperch and staghorn sculpin are associated with individual seagrass species in Yaquina Bay. Surfperch are more associated with *Z. japonica* (ISA, indicator value = 0.765, p = 0.084), and staghorn sculpin are more associated with *Z. marina* (ISA, indicator value = 0.680, p = 0.084).These results are only marginally statistically non-significant.

2.5 Discussion

The goal of this study was to contrast the community composition of *Z. japonica* and *Z. marina* in two PNW bays. Results both demonstrate the efficacy of video cameras in assessing the composition of fish and crab in soft bottom intertidal estuarine habitats and provide evidence of different community composition in similar habitats in both Willapa Bay and Yaquina Bay.

2.5.1 Efficacy of Video in the PNW Intertidal

Use of video cameras to examine freshwater or marine communities has increased markedly in recent years as video equipment and data storage have become more affordable. Other studies using video cameras in aquatic environments have used a variety of different metrics to record behavior, community composition, foraging activity, and abundance (Babcock et al. 1999; Guidetti 2006; Wilson et al. 2014). Among those metrics used are presence-absence data, the total number of fish or taxon observed over an entire video (maximum counts), maximum number of individuals from observed taxa present on screen at any given time, and the mean number of individuals present over the

course of the video (Gillanders et al. 2003; Chidami et al. 2007).

In contrast to previous work conducted with underwater video cameras, this study is one of the first to use all recorded data rather than a subset. To overcome significant biases afforded to faster, more gregarious species, we used a metric (CPUE) that represents observations relative to unit time and adjusts for the amount of video frame obstructed across each video segment. CPUE calculated in this study is not the same as that used in fisheries science, because it is not necessary to incorporate the catchability of each species for the specific gear type into the metric. For the sake of video footage, catchability (the portion of each species caught (observed) in the field of view of the camera) can be assumed to be 100%, because (in theory) all individuals that enter the frame can be recorded. This addresses concerns in using CPUE to assess community composition due to differences in catchability between species occurring from the occupation of different niches (Maunder et al. 2006).

In contrast to results from studies that used active collection techniques (trawls, seines, etc.) (Murphy et al. 2000; Hosack et al. 2006), the community observed in this study has lower diversity and is comprised of fewer species. These differences may be explained by the small temporal window during which cameras were deployed, deployment in close association with spring tides, or the small area sampled. Short, four hour deployments were performed due to limitations in battery length and logistics. The symmetry about the high tide and assurance that habitats of interest would be covered by water during this time window were also important. Studies conducted by Holsman et al. (2003, 2006) indicated that intertidal habitats are heavily used during flood tides, so we anticipated seeing the greatest numbers of individuals during this time. However, it may

be that the diversity of the community is reduced during these times, despite increased overall use, because only select community members utilize intertidal habitats during the temporal window that we sampled. The small area sampled may also limit the number of individuals and taxa observed, particularly if a taxon is rare (i.e. salmonids). Fyke nets in seagrass habitats, which sample a much greater area, observed rarer taxa than were observed in video sampling (Hosack et al. 2006; Sund & Dumbauld unpublished data). Future studies would benefit from increasing the temporal window of camera deployment, deploying across a greater range of tidal conditions rather than just spring tides, and deploying across twilight and night time conditions.

Use of breder traps as a baseline comparison of the community composition in video footage was moderately effective. The observation of gunnels within breder traps but not in video footage in Willapa Bay, as well as the smaller observed community and markedly lower catches in comparison to video in Yaquina Bay, raise questions about underrepresentation, sampling bias, and gear selectivity that may limit comparisons between the two techniques.

Under-representation of the biodiversity in breder trap catches can in part be attributed to the low density of target species, passive nature of the traps, and small size of traps resulting in limited sampling effort. Within other studies in the PNW, both gunnel and English sole were found to compose between 2 % and 7 % of the intertidal community, respectively (Murphy et al. 2000; Hosack et al. 2006). Partial feeding by Dungeness crab on both surfperch and staghorn sculpin was observed within breder traps. This predation within traps may have introduced bias by underestimating prey species and overestimating predator density. Additionally, small fish (i.e. gunnels) are

considered to be among available prey items consumed by staghorn sculpin (Lane et al. 1975). Despite these issues, the community of fish in breder traps and those seen in video were similar to those observed in other studies of intertidal habitats in the region (Bayer 1981; Armstrong et al. 1995; Murphy et al. 2000; Hosack et al. 2006).

2.5.2 Intertidal Estuarine Community

Several studies have examined the littoral nekton community of estuaries in the PNW. Multivariate contrasts of Willapa Bay and Yaquina Bay seagrass habitats indicate that the communities of fish and crab observed in both bays were similar in composition. Community composition was also similar to that reported from the Gulf of Alaska (Murphy et al. 2000) and Humboldt Bay, California (Pinnix et al. 2005; Garwood et al. 2013). The most common species observed in this study were also the most abundant as observed in other studies in Yaquina Bay and Willapa Bay (Bayer 1981; De Ben et al. 1990; Hosack et al. 2006; Ferraro and Cole 2010; Lewis 2014). Multivariate contrasts of Willapa Bay and Yaquina Bay seagrass habitats also indicate that the nekton communities in both bays were similar in composition between habitats of the same structure type. In this study, only 11 total taxa were observed in Yaquina Bay and Willapa Bay, which is markedly fewer than that observed in other studies that examined the region (De Ben et al. 1990; Ferraro and Cole 2010; Lewis 2014). Because of the small timeframe and small sampling areas (2 m^2), it is possible that some species were missed through a mistiming in daily use of intertidal habitats, seasonal occurrence, or inadequate sampling effort.

2.5.3 Willapa Bay

Within Willapa Bay, both univariate and multivariate analyses indicate that nekton community composition was relatively similar across all intertidal habitats. Individual univariate tests identify habitat as significant for only three-spine stickleback. However, there was not enough power in subsequent analyses to determine which habitat saw higher CPUE. ISA indicated that both three-spine stickleback (I = 0.85) and Dungeness crab (I = 0.755) were associated most with clam and oyster aquaculture, respectively, in Willapa Bay. Explicit comparisons of *Z. marina* and *Z. japonica* detected no significant difference in community composition.

Large indicator values for stickleback (I = 0.85) suggest that a higher proportion of stickleback were observed in clam aquaculture relative to all other taxa. Movement of three-spine stickleback into the shallows has been known to occur during breeding to allow males of the species to construct nests across a range of subtidal and intertidal habitats, with some populations utilizing rocky substrates (Iersel 1953; Macdonald et al. 1995). Some populations have been observed to return to areas near nesting sites following tidal exchanges (Macdonald et al. 1995). The clam aquaculture habitat examined in Willapa Bay was comprised of a base layer of large, rounded gravel (2-4 cm grain size) covered in biogenous and *Z. japonica* shoots. The strong association of stickleback to clam beds may be due to collection of filamentous algae or seagrass shoots from clam beds that are used in nest construction or as a substrate on which to build nests (Mori 1994; Macdonald et al. 1995).

The strong association of Dungeness crab to oyster aquaculture is supported by previous research including the use of oyster shell as an effective means of enhancing

intertidal habitat available to juvenile Dungeness following dredging activity (Dumbauld et al. 1993). Fernandez et al. (1993) observed enhanced survival of juvenile Dungeness crab tethered in hard structured habitat over both unstructured substrates and the soft structure of seagrass beds. Sub-adult Dungeness crab are also known to rely heavily on foraging in intertidal environments to supplement the relatively scarce resources available in the lower intertidal (Holsman et al. 2003; Holsman et al. 2006).

Due to logistical constraints, the oyster habitat that was examined in Willapa Bay was at a different site than the other habitats investigated. While the site at Nahcotta was chosen because of its proximity and similar tide height to the *Z. marina* beds at Oysterville, it was farther from the mouth and located on a much narrower intertidal bench. Ferraro and Cole (2011) indicate that the benthic macrofaunal communities within *Z. marina* and oyster habitats are relatively similar, which suggests that the habitat associations observed in this study are not due entirely to a site effect. However, without contrasts between analogous habitats of the two sites, it is difficult to discern whether the observed differences in community composition in oyster aquaculture beds are due to site or habitat alone.

Z. japonica and *Z. marina* inhabit the entire range of tidal elevations available at the Oysterville site in Willapa Bay and overlap in the middle of the intertidal bench (Shafer et al. 2014). This results in one homogeneously structured habitat in place of the unstructured habitat that previously occupied this location. With this conversion from bare substrate to one homogenous seagrass structure, the ability of the community to move throughout the landscape without reaching an edge or transition is greatly increased. Without the presence of such natural barriers, members of the community

previously associated with seagrass structured habitat are able to move further across the intertidal landscape, creating a community that is more generalized than that occurring in a heterogeneously structured habitat.

One examination of habitat fragmentation in seagrass habitats showed declining abundances of fish and crab as the distance between patches increased (Johnson and Heck Jr. 2006) However, Johnson and Heck (2006) believed fragmentation to have little impact because the majority of organisms that they studied were generalists, having no preference of the edge of a patch versus the center. Additionally, Horinouchi et al. (2009) observed lower diversity in continuous seagrass landscapes than in fragmented ones, which is analogous to the results of this study.

Some potential explanations for the considerable overlap between the community occupying clam aquaculture and *Z. japonica* beds are the relatively small size of the areas we examined as clam habitat and its shared tidal height with the upper limits of *Z. japonica*. The similar tidal height of where cameras and breder traps were deployed for clam and *Z. japonica* habitats may have also led to a very similar community using these substrates. This potential for a shared community is increased further when one considers that the clam habitat examined consisted of small patches of un-colonized gravel distributed throughout a nearly continuous *Z. japonica* bed in the upper intertidal.

2.5.4 Yaquina Bay

In contrast to Willapa Bay, nekton community composition was significantly different between habitat types in Yaquina Bay. Parametric tests indicated that staghorn sculpin and Dungeness crab were both significantly higher in bare substrate than *Z.*

japonica. ISA results also indicated that Dungeness crab have a significant association with bare habitat. Multivariate analyses indicate that the two seagrasses were the most different out of all the habitats in Yaquina Bay, and explicit comparisons of *Z. marina* and *Z. japonica* beds revealed marginal evidence that community composition was significantly different between the two seagrasses.

This difference in community composition between the two seagrasses may be driven by the large difference in tidal height, with tidal elevation as the primary architect of community structure instead of habitat type. If so, we would expect to see similar communities between *Z. japonica* and bare substrate. However, NMDS plots indicate much more similarity between unstructured habitat and *Z. marina*, even though the places we sampled these habitats were at much different tidal elevations than between the two seagrasses or bare substrate and *Z. japonica*. Different distributions of *Z. japonica* relative to *Z. marina* have been described by Shafer et al. 2014. In Yaquina Bay, the two species are separated by a large band of unstructured bare substrate, creating two distinct bands.

We hypothesize that without connectivity between the habitats, some species may not be moving out of the structure afforded by *Z. marina* in the lower intertidal. This lack of connectivity between the two seagrasses was likely compounded by the large tides present when we deployed cameras, because individuals that invest energy to move out of structured habitats risk becoming exposed with the ebbing tide. Many more Dungeness crab (a relatively slow moving macro-fauna (Holsman et al. 2003; Holsman et al. 2006)), staghorn sculpin (a well described saltatory predator (Armstrong et al. 1995)), and gunnels (known to be less active during the day (McGlory and Gotthardt 2005)) were

observed in *Z. marina* than *Z. japonica*. However, surfperch (a highly mobile intertidal fish) were observed to be an order of magnitude more common in *Z. japonica* than in *Z. marina*. The association of more stationary, sedentary, and slow-moving organisms with *Z. marina* and more mobile surfperch with *Z. japonica* lend some evidence to this hypothesis.

Life history stage has been previously shown to be important in Dungeness crab use of estuarine intertidal habitats. Eggleston and Armstrong (1995) found young of the year Dungeness crab to preferentially settle in structured substrates, with oyster shell being preferred over seagrass, which was, in turn, preferred over bare substrate. In other work performed in conjunction with this study, we observed higher settlement of young of the year Dungeness in seagrass habitats than unstructured mud flats but saw higher densities of Dungeness juveniles (< 1 year old) in bare substrate than in adjacent *Z. japonica* (Patten et al. 2013). However, in areas where shell habitat is not available, densities of Dungeness crab have been found to be much higher in seagrass beds than unstructured habitat (Iribarne et al. 1995; Blackmon et al. 2006).

Dungeness crab observed in video of unstructured substrate were smaller than those observed at the lower tidal elevations (personal observation). Unfortunately, quantitative analysis of measurements recorded was not possible because of a lack of a standard size reference. Without a standard size reference, the distortion created by the lens of the GoPro cameras was too large to be used for quantitative comparisons across replicates or days. Dungeness crab are known to utilize structured habitats in the upper intertidal until they are more able to compete with large conspecifics (Fernandez et al. 1993; Iribarne et al. 1995; Eggleston and Armstrong 1995). However, at sites in Yaquina

Bay, the only structured habitat available higher in the upper intertidal is the small narrow bed of *Z. japonica*, which is within 10 m of the riprap placed along the side of the bay. In 2013 at Sally's Bend, we observed a large amount of *Hemigrapsus* spp. moving into hard substrate that was placed into the area to assess settlement of Dungeness crab. It is possible that Dungeness crab emigrated out of the structured seagrass habitat because of interactions with *Hemigrapsus* spp. Visser et al. (2004) documented such competitive exclusion of early benthic phase Dungeness by *Hemigrapsus* spp. in hard structured substrates in Gray's Harbor, Washington.

The limited time frame of the video deployment may also be a factor explaining the observed association of Dungeness crab to open substrate. The greatest amount of activity on seagrass beds by Dungeness crabs has been observed to be during large tidal exchanges (Holsman et al. 2006). Holsman et al. (2006) also noted less use of structured oyster and seagrass by larger individuals despite increased availability of prey items, which they attributed to an aversion to the risk of stranding. The high tidal elevation of the open substrate and *Z. japonica* beds that were examined may have only been immersed for short periods of time, exposing individuals to increased mortality or stress via predation, desiccation, or thermal stress. The higher number of crab seen in open substrate may have been due to immigration with the high water and a very quick emigration with the ebbing tide, resulting in individuals present in open substrate to be overestimated. It is also possible there was a bias against detecting smaller Dungeness in the structured *Z. japonica* habitat simply because they were harder to see. Without an assessment of how individuals were using these habitats and the associated timing ascertaining whether this observed pattern is an actual preference for some characteristic

of open substrate habitat or a secondary association related to movement through it is impossible.

Surfperch were also found to be marginally associated with *Z. japonica* habitat in Yaquina Bay. Higher numbers of surfperch observed in *Z. japonica* may be explained by the availability of higher epifaunal biomass per unit area in *Z. japonica* than in *Z. marina* (Knight et al. 2015), which may facilitate more efficient foraging in *Z. japonica* than *Z. marina*. While this pattern of invertebrate biomass association has not yet been explored in Yaquina Bay, invertebrate biomass has been found to be higher in *Z. japonica* than in *Z. marina* within other Oregon estuaries (Ferraro and Cole 2012). Characterizing both the diet of surfperch in Yaquina Bay and how surfperch were using these habitats would be useful in elucidating whether more feeding was occurring in *Z. japonica* than in the other examined habitats. Another possible explanation for the marginally significant association of surfperch is that there may have been detection bias due to the high density of *Z. japonica*, which may have interfered with detection. However, the order of magnitude difference of surfperch observed in *Z. japonica* relative to other habitats suggests that this pattern was real.

In contrast to Willapa Bay, no three-spine stickleback were observed in Yaquina Bay despite their range being known to span the entire eastern Pacific. Three-spine stickleback may have been missed in Yaquina Bay due to either a mismatch in spatial location or temporal patterns. Previous researchers have observed three-spine stickleback (Bayer 1981; De Ben et al. 1990) in Yaquina Bay, but they were more commonly found further upriver than the site that was examined. Lewis (2014) also did not find three-spine stickleback in the lower portion of the bay in monthly monitoring of *Z. marina* beds that

were closer to the mouth than study sites examined in this study. Stickleback may have also moved out of the intertidal by the time (mid-August) that video recording was performed in Yaquina Bay.

Gunnels were only observed in video recorded in Yaquina Bay. The lack of detection of gunnels in Willapa Bay video may be due to temporal patterns of habitat use that occurred outside of the timeframe of recording. In experimental conditions, some species of gunnel have been observed to be highly active near dawn and dusk and relatively sessile the remainder of the day (McGlory and Gotthardt 2005).

Examination of all habitats in Yaquina Bay reveals that seagrass structured habitats were more similar to each other than the unstructured bare habitat, despite the large difference in tidal elevation that exists between them. Habitat complexity is often associated with increased survival (Fernandez et al. 1993; Selgrath et al. 2007; Hovel and Lipcius 2009), higher diversity (Murphy et al. 2000; Blackmon et al. 2006; Hosack et al. 2006; Horinouchi et al. 2009) and increased habitat use (Whitlow and Grabowski 2012) of fish and invertebrates.

2.6 Conclusion & Recommendations

This study demonstrates that cameras are an effective means of assessing the community composition of underwater habitats by providing novel data on the nekton community composition of intertidal habitats in the PNW, including that of an invasive seagrass. Improvements to the deployment design could increase the length of deployment, allowing for examination of diurnal patterns and examination of more extensive spatiotemporal patterns. Sampling a larger area or towing the camera along

transects may resolve issues associated with detection of rare taxa. Collection of additional variables such as associated behaviors and time within frame would provide more detail for little additional investment in processing. Nevertheless, the amount of detail gleaned from video deployment as a passive sampling method make it a very powerful tool for understanding how intertidal habitats are used, who is using them, and how this varies spatially and temporally.

Comparisons of the communities observed in video footage taken in seagrass structured habitats in Willapa Bay and Yaquina Bay reveal that community composition is similar between the two bays across a high tide, with only gunnels and stickleback showing significant associations to any one estuary. This suggests that type of structural habitat is a predominant determinant of community composition at a regional scale.

Examinations of communities associated with habitats in individual bays revealed contrasting patterns. Willapa Bay habitats had similar community compositions. However, in Yaquina Bay habitat associations were distinctly different, with *Z. japonica* and *Z. marina* habitats being the least similar. The relative distribution of *Z. marina* and *Z. japonica* were different between bays, with those in Yaquina Bay separated by a large band of unstructured mud and those in Willapa Bay growing together into one uniformly structured habitat type. This suggests that the presence of a continuous structured habitat serves to homogenize community composition across the entire range of tide heights. The sharp separation between the two seagrass habitats in Yaquina Bay appears to structure communities more than tide height alone.

2.7 Tables & Figures

Table 2.1: Mean daily catch from breder trap deployment. Values reported are the mean number of individuals caught within each bay and trap orientation (Channel, North, South) for all habitat types in both Yaquina Bay (n_{total} = 36) and Willapa Bay (n_{total} = 48).

		Willapa Bay					**Yaquina Bay**				
Orientation	Species (common name)	Clam	Oyster	*Zj*	*Zm*	Bay Total	Bare	*Zj*	*Zm*	Bay Total	Total
Channel	*Crangon* spp. (bay shrimp)	-	-	-	-	-	-	-	-	-	0.00
	Metacarcinus magister (Dungeness)	0.00	0.20	0.00	0.67	0.87	0.00	0.25	0.25	0.50	1.37
	Embiotocidae aggregata (shiner perch)	0.40	0.40	0.20	2.33	3.33	0.25	0.00	0.75	1.00	4.33
	Pholidae (gunnel)	-	-	-	-	-	-	-	-	-	0.00
	Leptocottus armatus (staghorn sculpin)	0.00	0.40	0.00	0.33	0.73	0.00	0.25	0.00	0.25	0.98
	Gasterosteus aculeatus (three-spine stickleback)	0.00	1.20	0.20	9.00	10.4	-	-	-	-	10.40
	Syngnathus leptorhynchus (pipefish)	-	-	-	-	-	-	-	-	-	0.00
North	*Crangon* spp. (bay shrimp)	0.20	0.00	0.00	0.00	0.20	-	-	-	-	0.20
	Metacarcinus magister (Dungeness)	-	-	-	-	-	0.50	0.75	0.00	1.25	1.25
	Embiotocidae aggregata (shiner perch)	0.40	0.20	1.00	2.00	3.60	0.50	0.75	0.75	2.00	5.60
	Pholidae (gunnel)	-	-	-	-	-	-	-	-	-	0.00
	Leptocottus armatus (staghorn sculpin)	0.20	0.00	0.00	0.00	0.20	-	-	-	-	0.20
	Gasterosteus aculeatus (three-spine stickleback)	0.20	1.40	3.00	0.25	4.85	-	-	-	-	4.85
	Syngnathus leptorhynchus (pipefish)	-	-	-	-	-	-	-	-	-	0.00

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Table 2.1: Continued

		Willapa Bay					**Yaquina Bay**				
Orientation	Species (common name)	Clam	Oyster	*Zj*	*Zm*	Bay Total	Bare	*Zj*	*Zm*	Bay Total	Total
South	*Crangon* spp. (bay shrimp)	0.00	0.00	0.00	0.25	0.25	-	-	-	-	0.25
	Metacarcinus magister (Dungeness)	0.00	0.00	0.80	0.00	0.80	0.25	0.00	0.00	0.25	1.05
	Embiotocidae aggregata (shiner perch)	0.20	0.80	1.00	1.75	3.75	0.75	0.75	0.25	1.75	5.50
	Pholidae (gunnel)	0.00	0.20	0.00	0.00	0.20	-	-	-	-	0.20
	Leptocottus armatus (staghorn sculpin)	-	-	-	-	-	0.00	0.00	0.50	0.50	0.50
	Gasterosteus aculeatus (three-spine stickleback)	0.00	1.80	0.40	2.25	4.45	-	-	-	-	4.45
	Syngnathus leptorhynchus (pipefish)	0.00	0.00	0.00	0.25	0.25	-	-	-	-	0.25
Total	*Crangon* spp. (bay shrimp)	0.20	0.00	0.00	0.25	0.45	-	-	-	-	0.45
	Metacarcinus magister (Dungeness)	0.00	0.20	0.80	0.67	1.67	0.75	1.00	0.25	2.00	3.67
	Embiotocidae aggregata (shiner perch)	1.00	1.40	2.20	6.08	10.68	1.50	1.50	1.75	4.75	15.43
	Pholidae (gunnel)	0.00	0.20	0.00	0.00	0.20	-	-	-	-	0.20
	Leptocottus armatus (staghorn sculpin)	0.20	0.40	0.00	0.33	0.93	0.00	0.25	0.50	0.75	1.68
	Gasterosteus aculeatus (three-spine stickleback)	0.20	4.40	3.60	11.50	19.70	-	-	-	-	19.70
	Syngnathus leptorhynchus (pipefish)	0.00	0.00	0.00	0.25	0.25	-	-	-	-	0.25

Table 2.2: Total catch from breder trap deployment. The number of individuals for all species caught across all trap deployments for each orientation direction (Channel, North, South) for each available habitat in both Yaquina Bay ($n_{total} = 36$) and Willapa Bay ($n_{total} = 48$).

		Willapa Bay					**Yaquina Bay**				
Orientation	Species (common name)	Clam	Oyster	*Zj*	*Zm*	Bay Total	Bare	*Zj*	*Zm*	Bay Total	Total
Channel	*Crangon* spp. (bay shrimp)	0	0	0	0	0	0	0	0	0	0
	Metacarcinus magister (Dungeness)	0	1	0	2	3	0	1	1	2	5
	Embiotocidae aggregata (shiner perch)	2	2	1	7	12	1	0	3	4	16
	Pholidae (gunnel)	0	0	0	0	0	0	0	0	0	0
	Leptocottus armatus (staghorn sculpin)	0	2	0	1	3	0	1	0	1	4
	Gasterosteus aculeatus (three-spine stickleback)	0	6	1	27	34	0	0	0	0	34
	Syngnathus leptorhynchus (pipefish)	0	0	0	0	0	0	0	0	0	0
North	*Crangon* spp. (bay shrimp)	1	0	0	0	1	0	0	0	0	1
	Metacarcinus magister (Dungeness)	0	0	0	0	0	2	3	0	5	5
	Embiotocidae aggregata (shiner perch)	2	1	5	8	16	2	3	3	8	24
	Pholidae (gunnel)	0	0	0	0	0	0	0	0	0	0
	Leptocottus armatus (staghorn sculpin)	1	0	0	0	1	0	0	0	0	1
	Gasterosteus aculeatus (three-spine stickleback)	1	7	15	1	24	0	0	0	0	24
	Syngnathus leptorhynchus (pipefish)	0	0	0	0	0	0	0	0	0	0

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Table 2.2: Continued

		Willapa Bay					Yaquina Bay				
Orientation	Species (common name)	Clam	Oyster	*Zj*	*Zm*	Bay Total	Bare	*Zj*	*Zm*	Bay Total	Total
South	*Crangon* spp. (bay shrimp)	0	0	0	1	1	0	0	0	0	1
	Metacarcinus magister (Dungeness)	0	0	4	0	4	1	0	0	1	5
	Embiotocidae aggregata (shiner perch)	1	4	5	7	17	3	3	1	7	24
	Pholidae (gunnel)	0	1	0	0	1	0	0	0	0	1
	Leptocottus armatus (staghorn sculpin)	0	0	0	0	0	0	0	2	2	2
	Gasterosteus aculeatus (three-spine stickleback)	0	9	2	9	20	0	0	0	0	20
	Syngnathus leptorhynchus (pipefish)	0	0	0	1	1	0	0	0	0	1
Total	*Crangon* spp. (bay shrimp)	1	0	0	1	2	0	0	0	0	2
	Metacarcinus magister (Dungeness)	0	1	4	2	7	3	4	1	8	15
	Embiotocidae aggregata (shiner perch)	5	7	11	22	45	6	6	7	19	64
	Pholidae (gunnel)	0	1	0	0	1	0	0	0	0	1
	Leptocottus armatus (staghorn sculpin)	1	2	0	1	4	0	1	2	3	7
	Gasterosteus aculeatus (three-spine stickleback)	1	22	18	37	78	0	0	0	0	78
	Syngnathus leptorhynchus (pipefish)	0	0	0	1	1	0	0	0	0	1

Table 2.3: Total counts of video observations. The total number of counts for the lowest taxonomic group within each habitat type in Willapa and Yaquina bays. The top three species in each bay are in bold italics and denoted as species 1[a], species 2[b], and species 3[c], in decreasing order. The total number of observations for each bay and the total between bays are in **bold**.

	Willapa Bay					**Yaquina Bay**				
Species (common name)	Clam	Oyster	*Zj*	*Zm*	Bay Total	*Zj*	*Zm*	Bare	Bay Total	*Species Total*
Crangon spp. (bay shrimp)	0	0	0	1	1		–			1
Metacarcinus magister (Dungeness crab)	21	27	22	52	122	3	27	70	***100***[c]	222
Neotrypaea calforniensis (ghost shrimp)			–			0	1	0	1	1
Hexagrammidae (greenling)	2	0	1	15	18	0	2	0	2	20
Pholidae (gunnel)	0	1	0	0	1	2	42	18	62	63
Hemigrapsus spp. (shore crab)	1	0	0	0	1	0	1	2	3	4
*Paguriadae (*hermit crab)	7	1	1	0	9		–			9
Embiotocidae (surfperch)	197	78	452	695	***1422***[a]	1393	325	430	***2148***[b]	3570
Syngnathus leptorhynchus (pipefish)	1	0	0	0	1		–			1
Cancer productus (red rock crab)			–			0	1	0	1	1
*Leptocottus armatus (*staghorn sculpin)	341	137	338	357	***1173***[b]	71	1253	1217	***2541***[a]	3714
Pleuronectidae (sole)	2	7	1	3	13	1	1	9	11	24
Platichthys stellatus (starry flounder)			–			0	3	0	3	3
Gasterosteus aculeatus (three-spine stickleback)	108	10	71	53	***242***[c]		–			242
Unknown	1	2	4	1	8	12	9	11	32	40
Habitat Total	681	263	890	1177	**3011**	1482	1665	1757	**4904**	**7915**

Table 2.4: Mean and standard deviation of adjusted video CPUE. Values are reported as the mean number of individuals observed per hour of video in each habitat in Willapa Bay and Yaquina Bay for all species recorded.

Species (common name)	Willapa Bay			Yaquina Bay		
	Habitat	Mean CPUE	Standard Deviation	Habitat	Mean CPUE	Standard Deviation
Crangon spp. (bay shrimp)	Clam	0.00	0.00			
	Oyster	0.00	0.00	Bare		
	Z. japonica	0.00	0.00	*Z. japonica*	–	
	Z. marina	0.07	0.16	*Z marina*		
	Species Total	0.07	0.16	*Species Total*		
Metacarcinus magister (Dungeness crab)	Clam	1.44	3.12			
	Oyster	12.13	15.04	Bare	12.83	10.80
	Z. japonica	3.04	6.62	*Z. japonica*	0.15	0.33
	Z. marina	4.60	4.38	*Z marina*	3.80	4.35
	Species Total	21.21	29.16	*Species Total*	16.78	15.48
Neotrypaea calforniensis (ghost shrimp)	Clam					
	Oyster			Bare	0.00	0.00
	Z. japonica	–		*Z. japonica*	0.00	0.00
	Z. marina			*Z marina*	0.11	0.25
	Species Total			*Species Total*	0.11	0.25
Hexagrammidae (greenling)	Clam	0.07	0.16			
	Oyster	0.00	0.00	Bare	0.00	0.00
	Z. japonica	0.09	0.19	*Z. japonica*	0.00	0.00
	Z. marina	2.10	3.52	*Z marina*	0.20	0.29
	Species Total	2.26	3.88	*Species Total*	0.20	0.29
Pholidae (gunnel)	Clam	0.00	0.00			
	Oyster	0.08	0.18	Bare	3.92	2.68
	Z. japonica	0.00	0.00	*Z. japonica*	0.31	0.53
	Z. marina	0.00	0.00	*Z marina*	6.64	7.23
	Species Total	0.08	0.18	*Species Total*	10.87	10.44
Hemigrapsus spp. (shore crab)	Clam	0.31	0.69			
	Oyster	0.00	0.00	Bare	0.45	0.66
	Z. japonica	0.00	0.00	*Z. japonica*	0.00	0.00
	Z. marina	0.00	0.00	*Z marina*	0.08	0.18
	Species Total	0.31	0.69	*Species Total*	0.53	0.84
Paguriadae (hermit crab)	Clam	0.46	0.92			
	Oyster	0.07	0.15	Bare		
	Z. japonica	0.07	0.15	*Z. japonica*	–	
	Z. marina	0.00	0.00	*Z marina*		
	Species Total	0.6	1.22	*Species Total*		

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Table 2.4: Continued

Species (common name)	Willapa Bay Habitat	Willapa Bay Mean CPUE	Willapa Bay Standard Deviation	Yaquina Bay Habitat	Yaquina Bay Mean CPUE	Yaquina Bay Standard Deviation
Embiotocidae (surfperch)	Clam	12.67	13.47			
	Oyster	17.47	24.10	Bare	96.75	107.37
	Z. japonica	49.75	42.36	*Z. japonica*	84.63	179.54
	Z. marina	76.77	91.65	*Z marina*	64.53	97.05
	Species Total	156.66	171.58	*Species Total*	245.91	383.96
Syngnathus leptorhynchus (pipefish)	Clam	0.07	0.16			
	Oyster	0.00	0.00	Bare		
	Z. japonica	0.00	0.00	*Z. japonica*	–	
	Z. marina	0.00	0.00	*Z marina*		
	Species Total	0.07	0.16	*Species Total*		
Cancer productus (red rock crab)	Clam					
	Oyster			Bare	0.00	0.00
	Z. japonica	–		*Z. japonica*	0.00	0.00
	Z. marina			*Z marina*	0.08	0.18
	Species Total			*Species Total*	0.08	0.18
Leptocottus armatus (staghorn sculpin)	Clam	31.39	18.57			
	Oyster	14.19	11.77	Bare	218.98	104.78
	Z. japonica	34.98	17.29	*Z. japonica*	5.35	4.61
	Z. marina	34.55	59.28	*Z marina*	135.96	93.68
	Species Total	115.11	82.89	*Species Total*	360.28	203.07
Pleuronectidae (sole)	Clam	0.11	0.16			
	Oyster	0.57	0.68	Bare	1.86	2.34
	Z. japonica	0.07	0.15	*Z. japonica*	0.05	0.11
	Z. marina	0.36	0.39	*Z marina*	0.11	0.25
	Species Total	1.11	1.38	*Species Total*	2.02	2.70
Platichthys stellatus (starry flounder)	Clam					
	Oyster			Bare	0.00	0.00
	Z. japonica	–		*Z. japonica*	0.00	0.00
	Z. marina			*Z marina*	0.19	0.42
	Species Total			*Species Total*	0.19	0.42
Gasterosteus aculeatus (three-spine stickleback)	Clam	7.96	1.83			
	Oyster	0.86	1.17	Bare		
	Z. japonica	7.81	1.05	*Z. japonica*	–	
	Z. marina	4.41	4.18	*Z marina*		
	Species Total	20.25	8.23	*Species Total*		
Unknown	Clam	0.07	0.16			
	Oyster	0.16	0.35	Bare	1.80	1.10
	Z. japonica	0.28	0.43	*Z. japonica*	1.07	1.50
	Z. marina	0.19	0.42	*Z marina*	0.92	0.87
	Species Total	0.69	1.36	*Species Total*	3.79	3.47
Habitat Totals	Clam	54.54	39.23			
	Oyster	45.53	53.44	Bare	336.59	229.72
	Z. japonica	96.09	68.24	*Z. japonica*	91.55	186.62
	Z. marina	123.05	116.35	*Z marina*	212.63	204.76
	Total	319.21	277.26	*Total*	640.77	621.10

Table 2.5: One-way ANOVA results for adjusted video CPUE predicted by habitat type. Results of one-way ANOVA for each species within Willapa Bay and Yaquina Bay using the model of mean adjusted CPUE predicted by habitat type. Significant p-values ($p < 0.05$) are in bold denoted by an '*'. For every statistical significant one-way ANOVA result, a Tukey contrast was performed.

	Willapa Bay			**Yaquina Bay**		
Species (common name)	DF (Habitat, Residuals)	F- value	p-value	DF (Habitat, Residuals)	F- value	p-value
Crangon spp. (bay shrimp)	3, 16	1.00	0.42	2, 12	-	-
Metacarcinus magister (Dungeness crab)	3, 16	1.50	0.25	2, 12	4.71	**0.03***
Neotrypaea calforniensis (ghost shrimp)	3, 16	-	-	2, 12	1.00	0.40
Hexagrammidae (greenling)	3, 16	1.68	0.21	2, 12	2.51	0.12
Pholidae (gunnel)	3, 16	1.00	0.42	2, 12	2.54	0.12
Hemigrapsus spp. (shore crab)	3, 16	1.00	0.42	2, 12	1.87	0.20
Paguriadae (hermit crab)	3, 16	0.97	0.43	2, 12	-	-
Embiotocidae (surfperch)	3, 16	1.64	0.22	2, 12	0.08	0.93
Syngnathus leptorhynchus (pipefish)	3, 16	1.00	0.42	2, 12	-	-
Cancer productus (red rock crab)	3, 16	-	-	2, 12	1.00	0.40
*Leptocottus armatus (*staghorn sculpin)	3, 16	2.12	0.14	2, 12	8.80	**0.004***
Pleuronectidae (sole)	3, 16	1.67	0.21	2, 12	2.86	0.10
Platichthys stellatus (starry flounder)	3, 16	-	-	2, 12	1.00	0.40
Gasterosteus aculeatus (three-spine stickleback)	3, 16	9.69	**< 0.001***	2, 12	-	-
Unknown	3, 16	0.30	0.83	2, 12	0.79	0.48

Table 2.6(a): Significant TukeyHSD contrasts for CPUE of species by habitat in Willapa Bay. Cells denote the difference (rows - columns) between hourly catch per unit effort of two habitat types. A value of -0.088 in the row denoting clam habitat and the column denoting oyster is read as 0.088 less stickleback were observed per hour of video recorded in oyster than in clam.

Species (Common Name)		**Willapa Bay**			
		Clam	Oyster	*Z. japonica*	*Z. marina*
Gasterosteus aculeatus (three-spine stickleback)	Clam	0	-0.088 (*ns*)	-0.212 (*ns*)	-0.116 (*ns*)
	Oyster		0	-0.124 (*ns*)	-0.028 (*ns*)
	Z. japonica			0	-0.096 (*ns*)
	Z. marina				0
		(ns)p>0.05	****p<0.05***	*****p<0.01***	

Table 2.6(b): Significant TukeyHSD contrasts for CPUE of species by habitat in Yaquina Bay. Cells denote the difference (rows - columns) between hourly catch per unit effort between two habitat types. A value of 12.6 in the row denoting bare habitat and the column denoting *Z. japonica* is read as 12.6 more Dungeness crab were observed per hour of video recorded in bare than in *Z. japonica*.

Species (Common Name)		**Yaquina Bay**		
		Bare	*Z. japonica*	*Z. marina*
Metacarcinus magister (Dungeness crab)	Bare	0	***12.682 ****	9.026 (*ns*)
	Z. japonica		0	-3.356 (*ns*)
	Z. marina			0
		Bare	*Z. japonica*	*Z. marina*
Leptocottus armatus (staghorn sculpin)	Bare	0	***213.632*****	83.020 (*ns*)
	Z. japonica		0	-130.612 (*ns*)
	Z. marina			0
		(ns)p>0.05	****p<0.05***	*****p<0.01***

Table 2.7: MRPP results for all groupings. Results of Multi-Response Permutation Procedure (MRPP) analyses that examined the similarity of the communities between 1) Willapa and Yaquina Bays, 2) habitat types within Willapa Bay, 3) habitat types in Yaquina Bay, 4) seagrass species in Willapa Bay, and 5) seagrass species in Yaquina Bay.

Environmental Parameter	Group	Average Distance	Observed Delta	Expected Delta	Interaction Strength (A-value)	Within Group Significance (p-value)
Bay	Willapa	0.5099	0.5458	0.587	0.07023	**0.016***
	Yaquina	0.5816				
Habitat (Willapa Bay)	Clam	0.4591	0.4803	0.5332	0.09932	**0.011***
	Oyster	0.5763				
	Z. japonica	0.3781				
	Z. marina	0.5077				
Habitat (Yaquina Bay)	Bare	0.4192	0.4831	0.5615	0.1396	**0.023***
	Z. japonica	0.6046				
	Z. marina	0.4254				
Seagrass Type (Willapa Bay)	*Z. japonica*	0.492	0.5048	0.5149	0.01271	0.338
	Z. marina	0.5247				
Seagrass Type (Yaquina Bay)	*Z. japonica*	0.497	0.3848	0.4555	0.1552	0.059
	Z. marina	0.2726				

p<0.05

Table 2.8: ISA results for all groupings. Results of Indicator Species Analyses (ISA) examining the association of species that occurred in > 5% of video to 1) seagrass habitats in Willapa and Yaquina bays, 2) habitat types within Willapa Bay (clam, oyster, *Z. japonica*, and *Z. marina*), 3) habitat types in Yaquina Bay (bare, *Z. japonica*, and *Z. marina*), 4) seagrass species in Willapa Bay (*Z. japonica* and *Z. marina*), and 5) seagrass species in Yaquina Bay (*Z. japonica* and *Z. marina*). Only species with some evidence ($p \leq 0.10$) of association are reported.

Species (common name)	Group	Association	Indicator Value	p-value
Metacarcinus magister (Dungeness crab)	Bay[1]	Willapa	0.553	0.396
	Habitat[2] (Willapa Bay)	Oyster	0.755	**0.004***
	Habitat[3] (Yaquina Bay)	Bare	0.563	**0.048***
	Seagrass Type[4] (Willapa Bay)	*Z. marina*	0.553	0.396
	Seagrass Type[5] (Yaquina Bay)	*Z. marina*	0.601	0.157
Pholidae (gunnel)	Bay[1]	Yaquina	0.700	**0.004***
	Habitat[2] (Willapa Bay)	Oyster	0.200	1.000
	Habitat[3] (Yaquina Bay)	*Z. marina*	0.510	0.182
	Seagrass Type[4] (Willapa Bay)	-	-	-
	Seagrass Type[5] (Yaquina Bay)	*Z. marina*	0.651	0.193
Embiotocidae (surfperch)	Bay[1]	Willapa	0.532	0.705
	Habitat[2] (Willapa Bay)	*Z. marina*	0.341	0.328
	Habitat[3] (Yaquina Bay)	*Z. japonica*	0.591	0.066
	Seagrass Type[4] (Willapa Bay)	*Z. marina*	0.539	0.689
	Seagrass Type[5] (Yaquina Bay)	*Z. japonica*	0.765	0.084
Leptocottus armatus (staghorn sculpin)	Bay[1]	Yaquina	0.573	0.288
	Habitat[2] (Willapa Bay)	Clam	0.311	0.384
	Habitat[3] (Yaquina Bay)	*Z. marina*	0.416	0.307
	Seagrass Type[4] (Willapa Bay)	*Z. japonica*	0.516	0.850
	Seagrass Type[5] (Yaquina Bay)	*Z. marina*	0.680	0.084
Gasterosteus aculeatus (three-spine stickleback)	Bay[1]	Willapa	0.900	**0.001***
	Habitat[2] (Willapa Bay)	Clam	0.464	**0.029***
	Habitat[3] (Yaquina Bay)	-	-	-
	Seagrass Type[4] (Willapa Bay)	*Z. japonica*	0.668	0.210
	Seagrass Type[5] (Yaquina Bay)	-	-	-

***p<0.05** ***p<0.01**

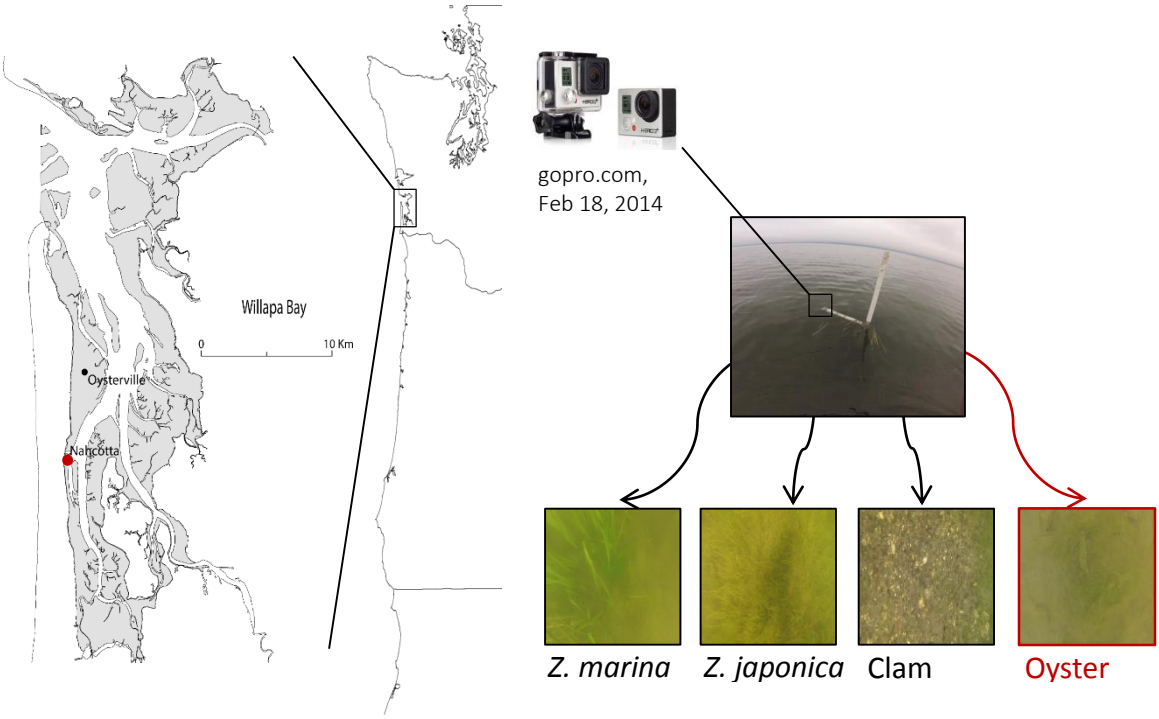

Figure 2.1: Field and study locations in Willapa Bay. Habitats sampled at Nahcotta are denoted in red. Habitats sampled at Oysterville are denoted in black. Images are examples taken from downward-facing video cameras in each habitat.

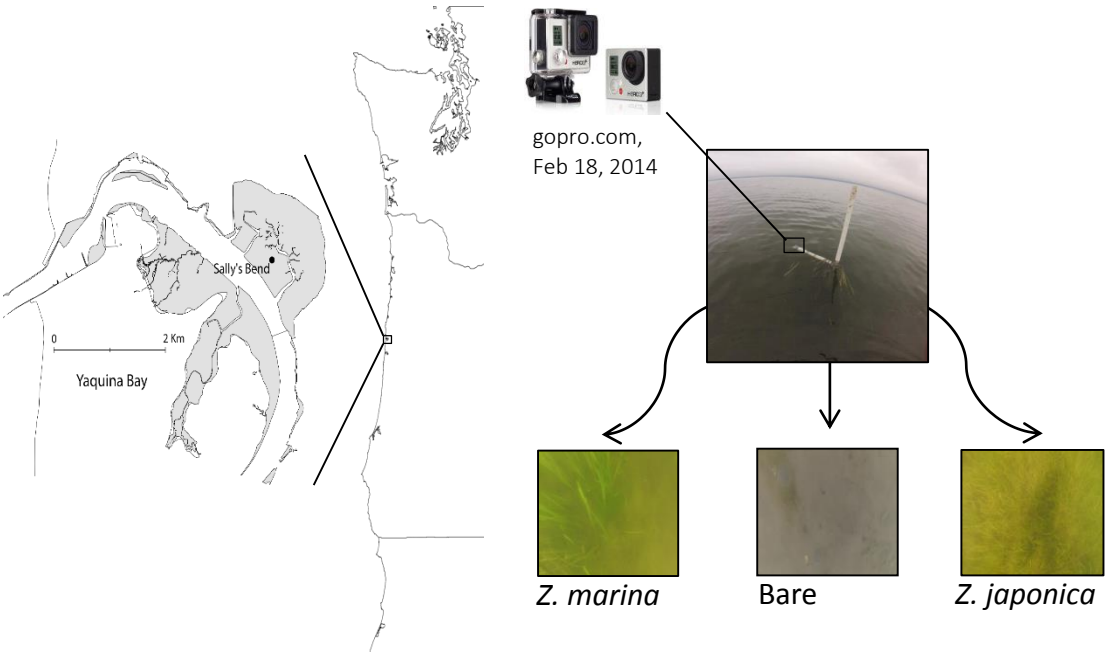

Figure 2.2: Field and study locations in Yaquina Bay. All habitats were sampled at Sally's Bend, denoted in black. Images are examples taken from downward-facing video cameras in each habitat.

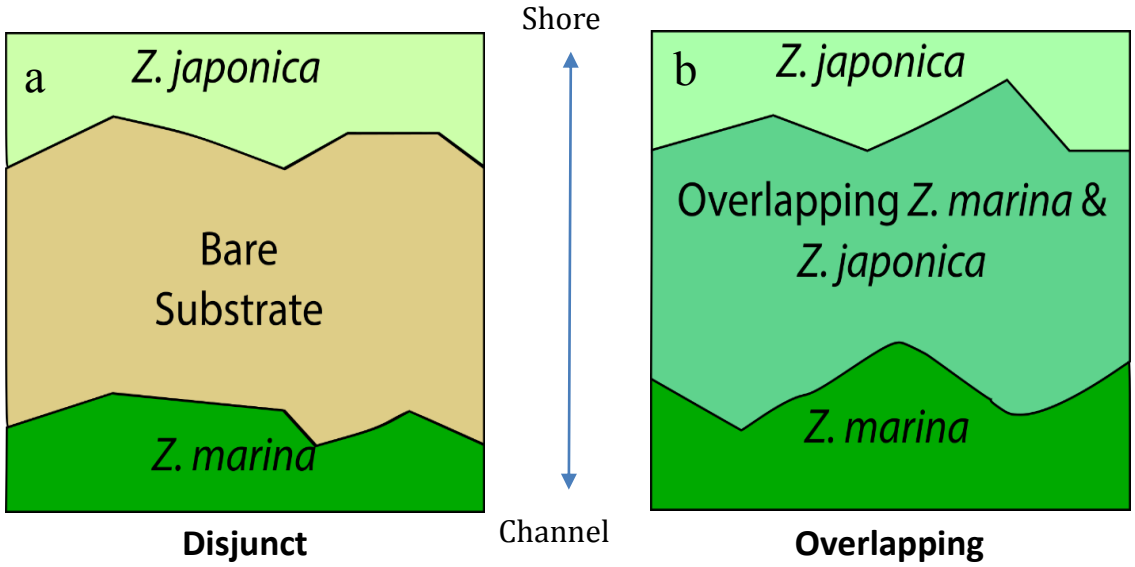

Figure 2.3: (a) Disjunct zonation of *Z. japonica* relative to *Z. marina* observed in Yaquina Bay. (b) Overlapping zonation of *Z. japonica* relative to *Z. marina* observed in Willapa Bay. Zonation patterns are described in Shafer et al. (2014).

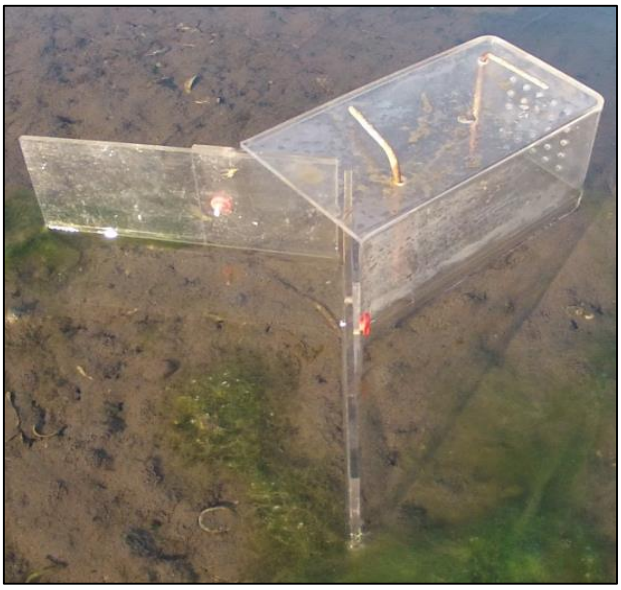

Figure 2.4: 31 cm x 15cm x 15 cm Plexiglas breder traps with 31 cm x 15 cm wings that were deployed in unison with the cameras.

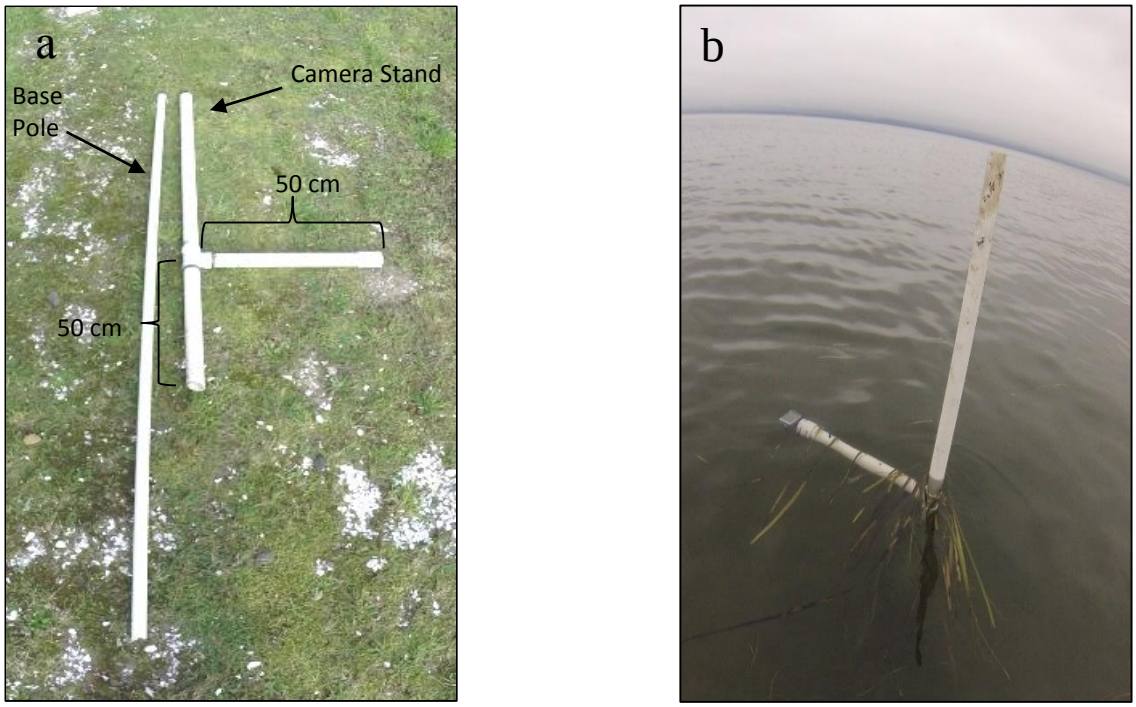

Figure 2.5: (a) Camera mount and base pole example: The base pole (left) and camera stand (right) that comprised the camera mounts used to deploy cameras. (b) Deployed camera mount and camera. Camera is oriented straight down towards the substrate.

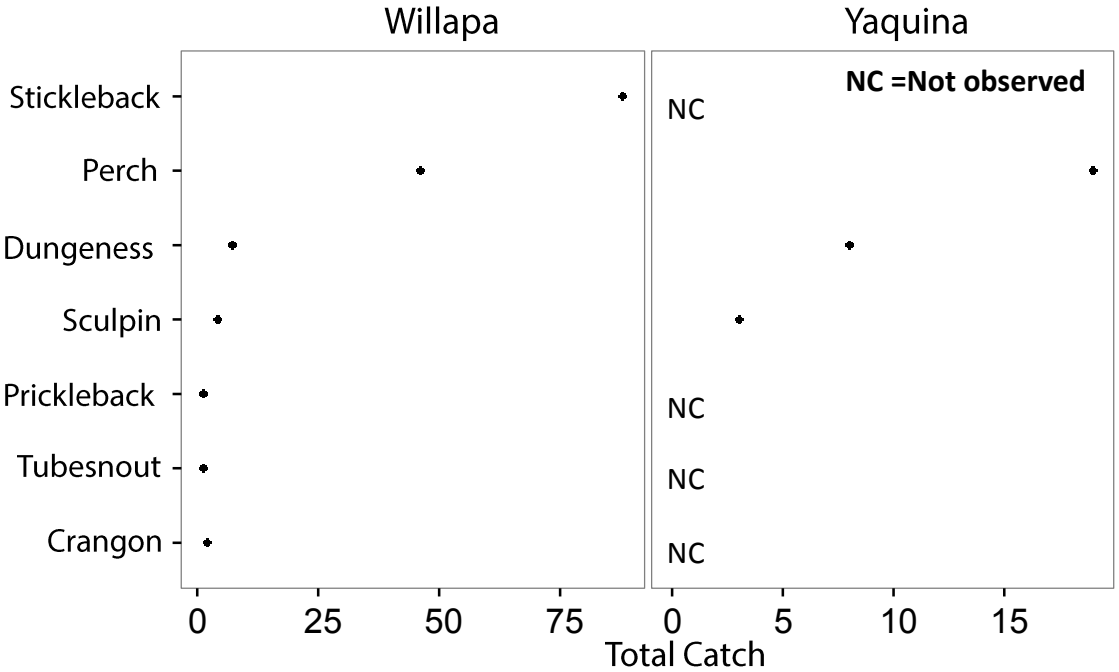

Figure 2.6: Total of all species caught in breder traps in Willapa Bay and Yaquina Bay. NC indicates the species was not observed in breder traps Yaquina Bay.

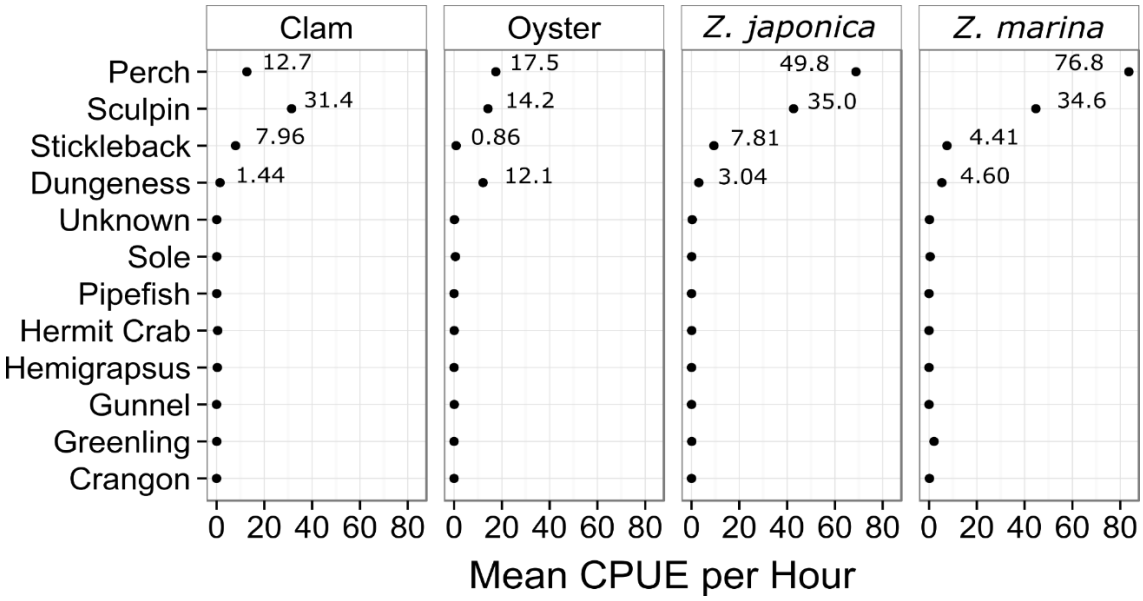

Figure 2.7: Mean hourly CPUE for each species observed in video in Willapa Bay. The mean hourly CPUE for the top four species is labeled.

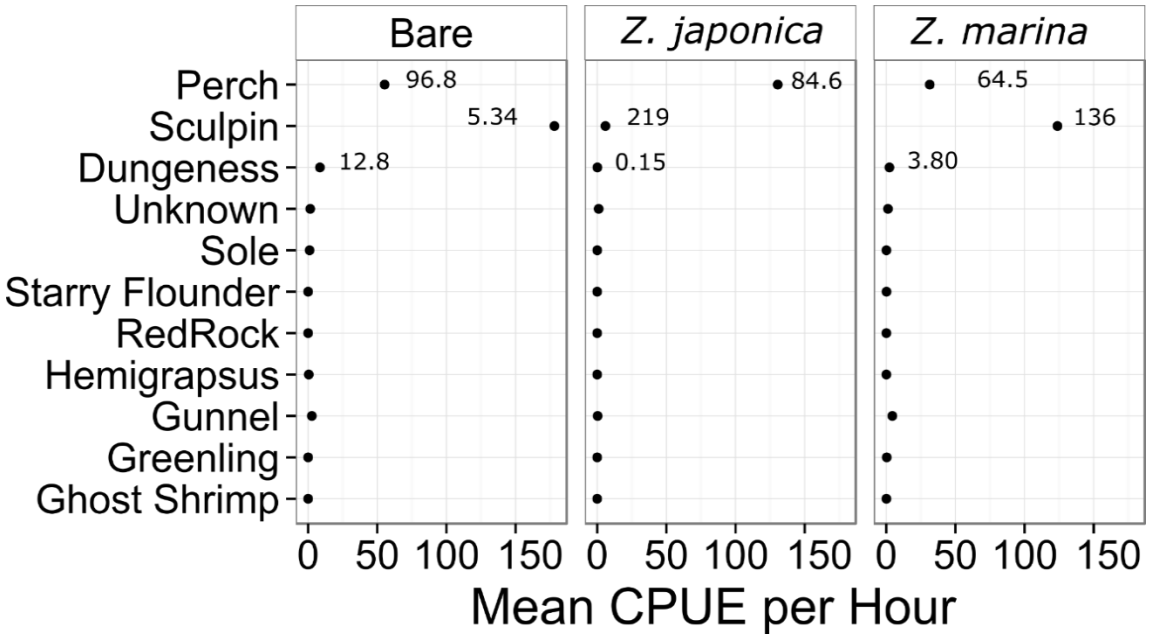

Figure 2.8: Mean hourly CPUE for each species observed in video in Yaquina Bay. The mean hourly CPUE for the top three species is labeled.

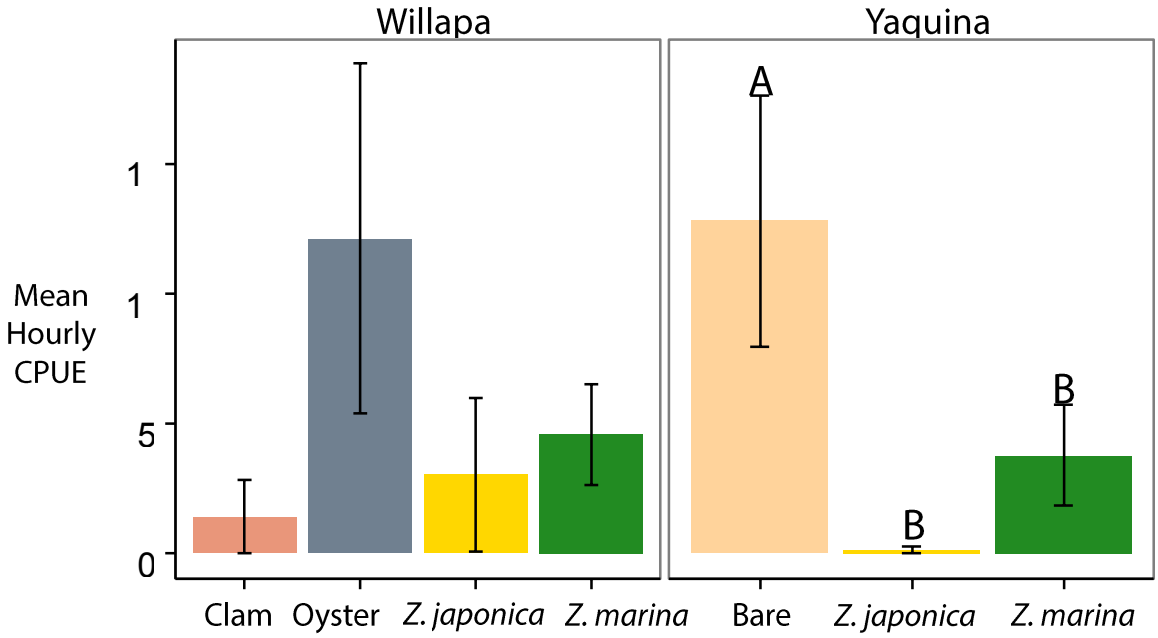

Figure 2.9: Mean CPUE of Dungeness crab in Willapa and Yaquina Bay. Error bars denote Standard Error. “A” and “B” denote significant difference indicated by Tukey contrasts.

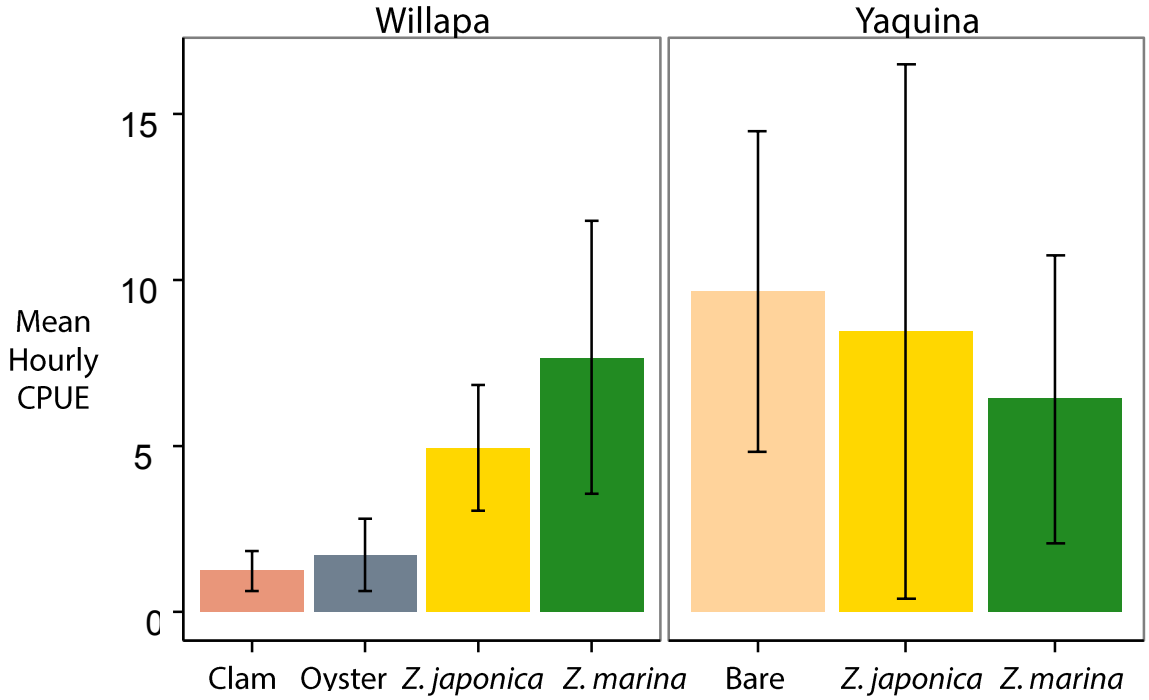

Figure 2.10: Mean CPUE of surfperch in Willapa Bay and Yaquina Bay. Error bars denote Standard Error.

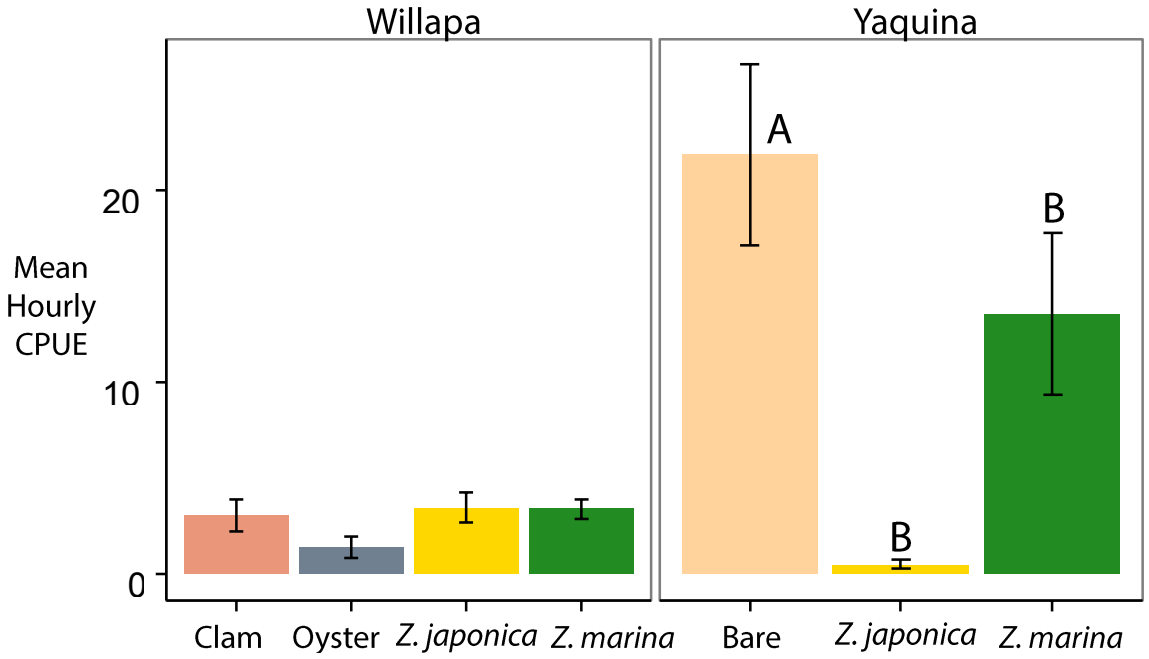

Figure 2.11: Mean CPUE of staghorn sculpin in Willapa Bay and Yaquina Bay. Error bars denote Standard Error. “A” and “B” denote significant difference indicated by Tukey contrasts.

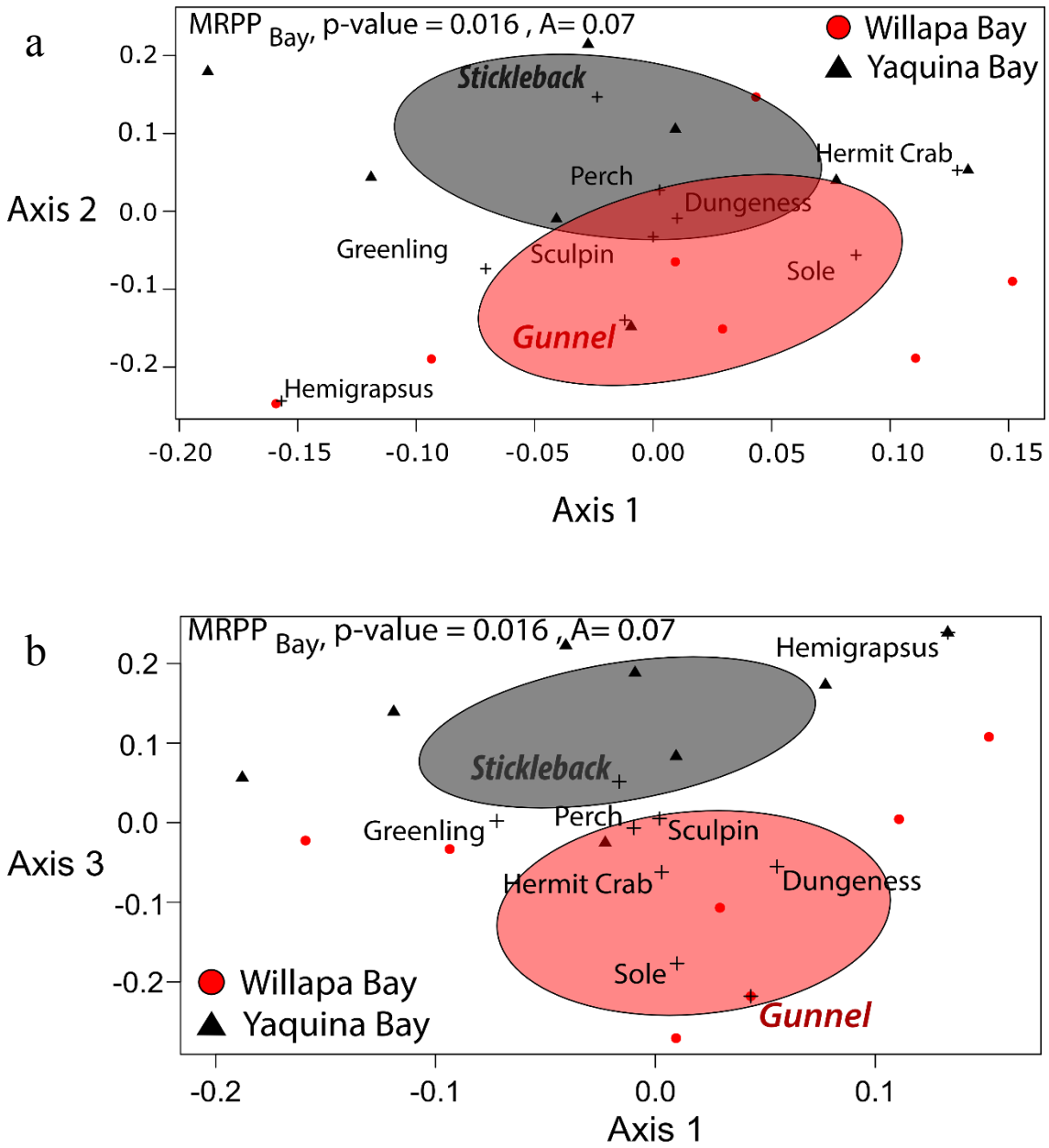

Figure 2.12: (a) Axes 1 & 2 and (b) Axes 1 & 3 of an NMDS plot of seagrass structure (*Z. japonica* and *Z. marina* habitats combined) visualizing the community composition of Willapa Bay and Yaquina Bay. Weighted mean species scores for the lowest identified taxa are denoted by "+" and labeled. Significant ISA associations are denoted in bold, with the text color indicating the association.

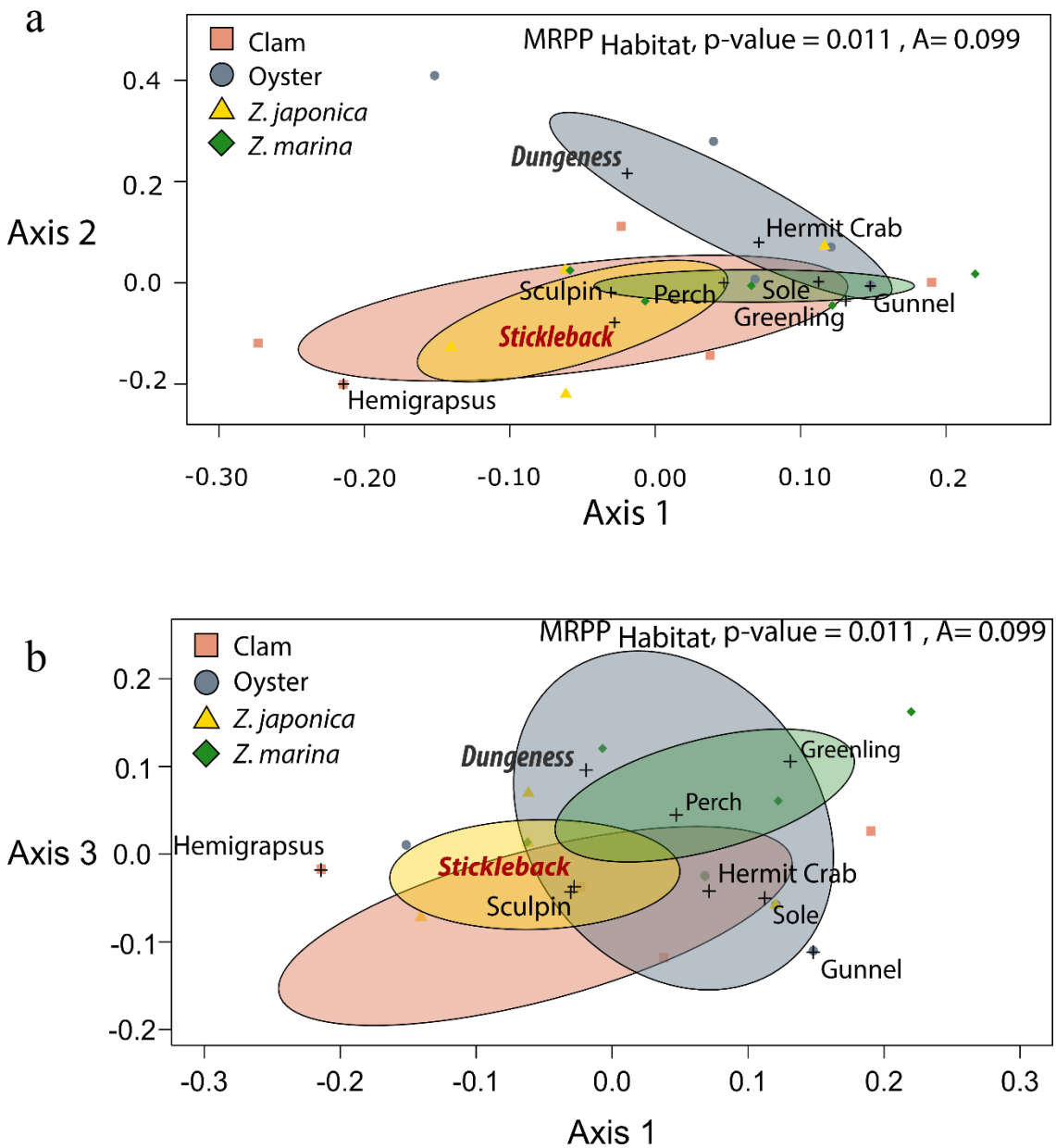

Figure 2.13: (a) Axes 1 & 2 and (b) Axes 1 & 3 of an NMDS plot visualizing the community composition of habitats (clam, oyster, *Z. japonica*, and *Z. marina*) in Willapa Bay. Weighted mean species scores for the lowest identified taxa are denoted by "+" and labeled. Significant ISA associations are denoted in bold, with the text color indicating the association.

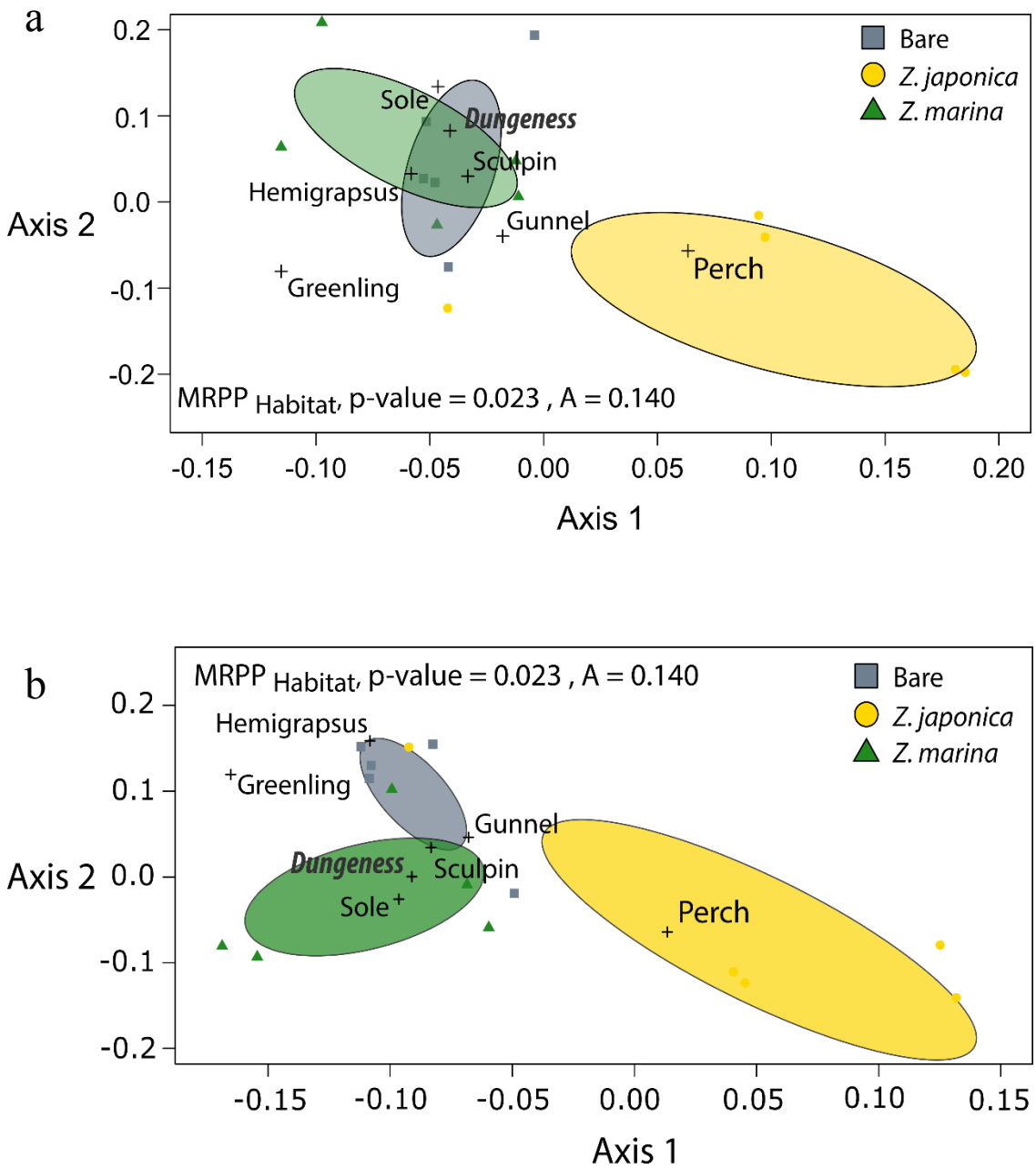

Figure 2.14: (a) Axes 1 & 2 and (b) Axes 1 & 3 of an NMDS plot visualizing the community composition of habitats (bare substrate, *Z. japonica*, and *Z. marina*) in Yaquina Bay. Weighted mean species scores for the lowest identified taxa are denoted by "+" and labeled. Significant ISA associations are denoted in bold, with the text color indicating the association.

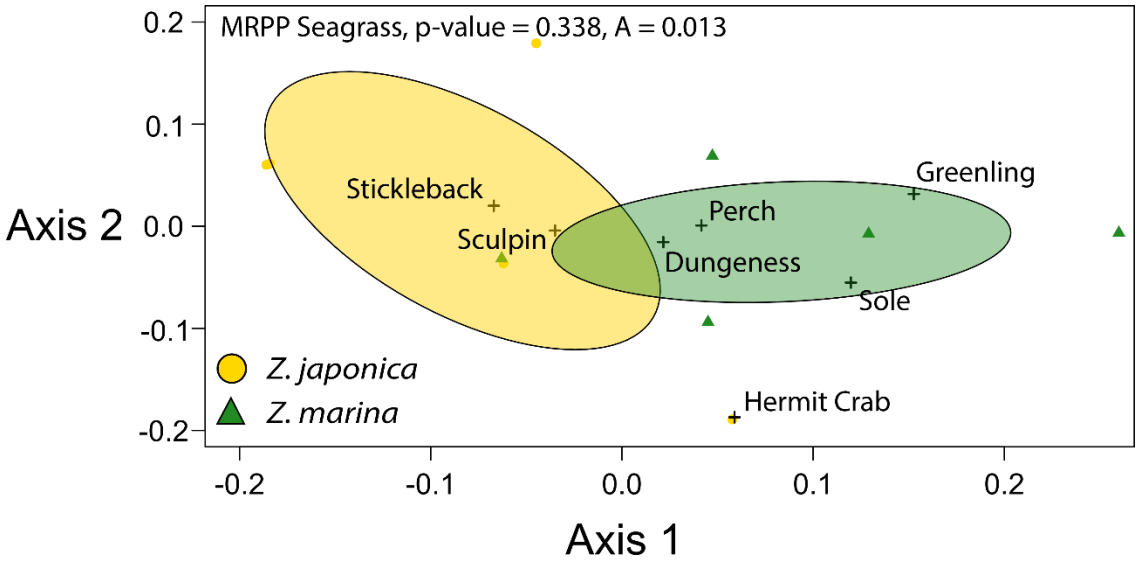

Figure 2.15: NMDS plot providing a visual representation of the community composition of seagrass habitat (*Z. japonica* and *Z. marina*) in Willapa Bay. Weighted mean species scores for the lowest identified taxa are denoted by "+" and labeled. Significant ISA associations are denoted in bold, with the text color indicating the association.

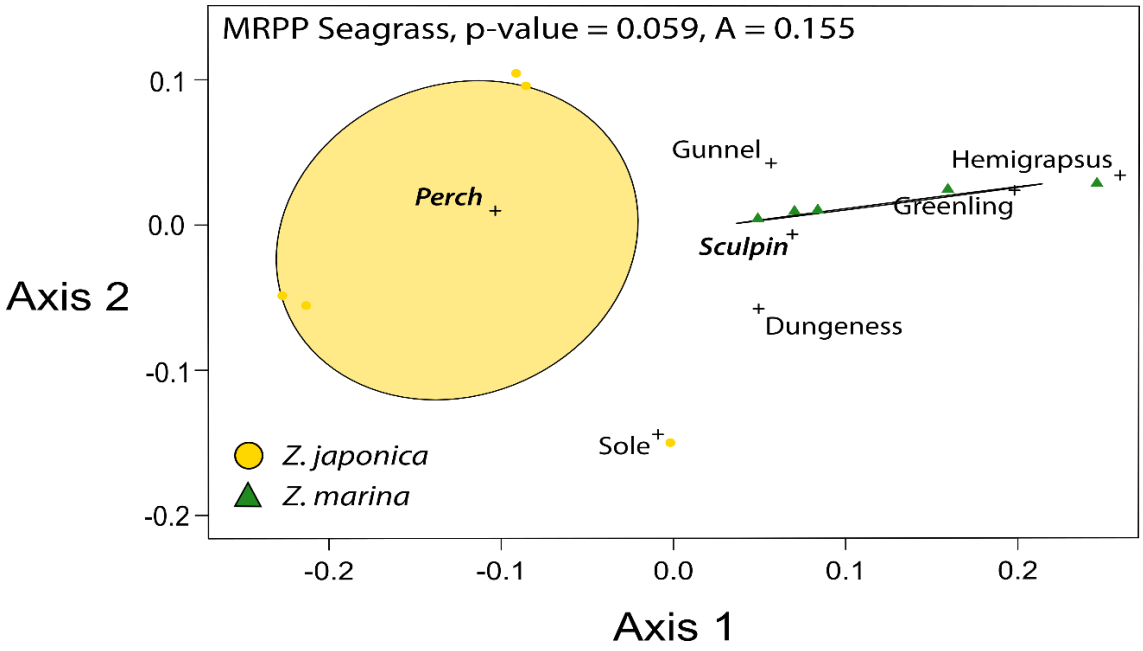

Figure 2.16: NMDS plot providing a visual representation of the community composition of seagrass habitat (*Z. japonica* and *Z. marina*) in Yaquina Bay. Weighted mean species scores for the lowest identified taxa are denoted by "+" and labeled. Significant ISA associations are denoted in bold, with the text color indicating the association.

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Chapter 3: Discussion

Sound and effective natural resource management can be described as a synthesis of social pressures and scientific perspective. Synthesizing new scientific observations with social pressures can become especially complex when results are complicated and contextually dependent, as they often are in ecology. Interpreting and integrating the results of baseline nekton community contrasts between *Z. japonica* and *Z. marina* in Willapa Bay and Yaquina Bay are no different.

The purpose of this chapter is to discuss what the social and ecological patterns observed in the previous chapters mean at a broad scale, as well as how they potentially play into management of *Z. japonica*. To do so, I will discuss the patterns observed and how they relate to current and historic management of the species, as well as takeaways and suggestions from interviews of professional researchers and natural resource managers working with *Z. japonica* in the PNW.

3.1 Pattern, Process, & Management

Results of this study indicate that the similarity of the nekton community between *Z. japonica* and *Z. marina* is contextually dependent on the relative distribution of the two species. Understanding that the overall structure type (structured versus unstructured) of intertidal habitats dictates community composition is important in determining management approaches.

Currently, management across all of the PNW states is limited; only Washington actively controls *Z. japonica*. If current management in Washington State remains

constrained to aquaculture beds, then utilization of Imazamox is an effective way to redress stakeholder concerns over *Z. japonica* interaction with aquaculture. This approach appears to control *Z. japonica* where its expansion has had the greatest impacts, while simultaneously limiting the adverse effects of Imazamox on *Z. marina* through a limited scale of action by allowing removal only on aquaculture beds. However, if management expands beyond aquaculture beds, considerable consideration should be placed on whether chemical control methods are appropriate because of accidental removal of *Z. marina* and social stigma of large scale use of herbicides/insecticides in the intertidal.

Results of this research indicate that site conditions result in significantly different community compositions depending on how the two species of seagrass are distributed relative to one another. If chemical control is scaled up, managers must account for higher incidental impacts on *Z. marina*, particularly on intertidal beds where the two species grow together. In discussion with managers and researchers in Washington State, it is highly unlikely that any eradication will be successful in the state without untold perceived impacts to *Z. marina* populations.

Considering that management of natural resources is under the same social, political, and economic pressures experienced by all governmental services, it is unlikely that management positions of states will remain static on the issue of *Z. japonica*. In the face of inevitable change, managers must weigh the social, economic, and political pressures when drafting potential management changes. If California and Oregon decide to undertake further control of *Z. japonica*, the characteristics of the environment most impacted by management will need to be identified. Considering that managers in both

states felt that use of chemical control methods would be unsuitable for stakeholders, it is highly likely that the characteristics most pertinent in structuring management in Oregon and California will be different from those in Washington. Other control regimes open to Oregon and California are excavation, heating, or covering of intertidal *Z. japonica* beds. Whatever the direction these states decide to pursue, the environmental cost of control as well as identification of who will 'foot the bill' must be identified.

3.2 Interview Takeaways

3.2.1 Lessons Learned: Ecological Characteristics of an Invasive Seagrass

3.2.1.1 Colonization of Bare Substrate

In Washington, colonization of the upper intertidal by *Z. japonica*, a region that was previously unoccupied by vegetation in the PNW, and its perceived conflict with clam aquaculture has been the primary driver for control in the state. Managers and researchers alike reported that *Z. japonica*'s nature as an r-selected (high reproductive capacity) species combined with the commercial utilization of the upper intertidal for clam aquaculture ultimately led to allowing selective removal of the species on aquaculture beds. If *Z. japonica* had invaded a different region of the intertidal in Washington State, participants are unsure if there would have been as much push for control. Any management – current, future or hypothetical – would ideally be decided upon after weighing the ecological services provided by *Z. japonica* against the economic and ecological costs of managing it.

The importance of bare habitat in supporting migratory shorebirds was heavily considered by managers in California. While aware of studies that have demonstrated

preferential feeding of *Z. japonica* by some waterfowl (Baldwin and Lovvorn 1994), participants stated that managers in Humboldt Bay were concerned that the large change in habitat structure seen elsewhere would be detrimental to the shorebird community and that there was enough habitat available to support waterfowl. Concern for intertidal habitats as they were used as a migratory flyway combined with the knowledge of *Z. japonica*'s highly reproductive nature ultimately led to the decision to attempt removal. Even though initial removal efforts failed as of 2011, natural resource managers in California are still interested in pursuing eradication of the species in their estuaries given appropriate levels of funding.

In Oregon, participants reported that management of *Z. japonica* would be undertaken if there was public interest in pursuing it and that, currently, managers feel there is both little knowledge of the species among the general public and relatively little negative perception between various stakeholders about *Z. japonica*. The much smaller size of the aquaculture industry (in comparison to Washington State) and the emphasis on oyster aquaculture over clam aquaculture have resulted in no known lobbying for management of the species by aquaculture stakeholders. Additionally, no other stakeholder groups have shown interest or concern over its presence in Oregon. Participants suggested that if management of the species were to occur in Oregon, a control regime similar to Washington's application of herbicides was unlikely to be pursued because of local negative sentiment towards the addition of chemicals to aquatic habitats.

3.2.1.2 An Invading Ecosystem Engineer

Most participants identified *Z. japonica*'s role as an ecosystem engineer in its native range as a point of particular concern on a broad scale, because there is a general understanding that invasive species that are also ecosystem engineers have a much broader impact than those that are not ecosystem engineers. Researchers also suggested that interaction between *Z. japonica*'s role as an engineer and its ability to colonize areas previously unused by vegetation make its potential for persisting in the PNW even more of a concern. The fact that *Z. japonica* is an ecosystem engineer seemed to be a feature that demonstrated the need for research and potential management of the species in the PNW.

3.2.1.3 Climate Change

A number of participants also highlighted concerns that climate change may facilitate further expansion of *Z. japonica* through both increased fitness and the ability to outcompete *Z. marina* under the new climate regime. One participant noted that *Z. japonica* expansion appears to be temperature limited in the Puget Sound when compared to areas like Willapa Bay, which is much warmer. There is a concern by participants that *Z. japonica*'s introduced range may stop being temperature limited if climate change sufficiently warms surface waters. There is also concern that interaction between rising sea level, sea surface temperature, and changes to water chemistry may synergistically alter the range, community, and health of intertidal seagrasses.

3.2.1.4 Intrinsic Value of Native Ecology

A number of participants – particularly researchers – raised the notion that non-native species (like *Z. japonica*) degrade the local diversity of the native community.

> *"There is an intrinsic ecological value in the native community and shared evolutionary history of any given ecosystem. Introduction of non-native species homogenizes these communities, harming the local diversity and diluting the intrinsic value of a given locale."*

The question of whether a non-native species should be controlled even if it provides beneficial services to the community was a point of contention for both managers and scientists. The notion that a Non-natives species should be removed simply because of its alien origin is a widely held view in the field of ecology. However, one participant noted that,

> *"If that stance is used, then all introduced species should be removed regardless of their role in the environment …" including the clams that Z. japonica is displacing.*

It was also noted that all species provide services, both positive and negative, to the environment. Manager participants were quick to note that while science informs management decisions, management from either perspective must ultimately be a reflection of society's view on the matter.

In contrast to the view that there is intrinsic value in maintaining the native community associated with an ecosystem, some participants noted that if you look at an ecosystem as the sum of all of the parts currently incorporated, including non-native species, then it is unclear that eradication/removal is entirely necessary. Participants reported that in the case of *Z. japonica*, the services that it provides have yet to be determined in the PNW. The final choice to attempt large scale control measures, at an

estuary level rather than a piecemeal selection of shellfish growing sites like in Washington, will need to be the result of a cost-benefit analysis. If the cost-benefit analysis determines that the services provided by *Z. japonica* are less than those provided by the previous condition, then removal would be considered. Multiple participants noted that control measures, large or small, are going to have both ecological and monetary costs that soar as scale increases.

3.2.2 Lessons Learned: Invasive Species Management

3.2.2.1 Understanding the Cost of Action

Close scrutiny of the ecological cost is also an important feature that participants felt should be carefully considered. If the cost is too high (i.e. removal causes more damage to the environment than taking no action), then management must account for this. Researcher participants pointed out that the ecological cost is highly contextually dependent, with the potential damage caused by removal in one location not being the same as that in another. Both managers and scientists agreed that management action may need to be different even in very similar ecosystems. However, some participants stressed that while the need may be there for different logistical approaches, it can be very difficult to manage natural resources this way. It was also noted that having different management strategies for the same species might lead to confusion and a perception of inconsistency in regulation among the general public.

Participants generally agreed that eradication of *Z. japonica* isn’t feasible in areas where it has become well established. They also noted that if eradication were to be

attempted, a number of conditions must be met before proceeding. One participant stated,

> *"Above everything, there must be absolute certainty that Z. japonica is a non-native species. If found to be non-native, then the ecosystem services provided by Z. japonica must be quantified before any action is taken."*

If all conditions are met, then participants suggested that eradication move forward only if it can be done without damage to *Z. marina*.

All participants concluded that eradication only be undertaken if the cost to the ecosystem is understood beforehand. If the impacts of eradication are not known, participants agreed that alternative management strategies should follow. Some of the suggested alternatives were limited removal (discussed above), mechanical removal (which has been studied in Oregon and California), and simply removing the regulatory protections currently afforded to *Z. japonica* in Oregon.

Discussion with a California participant indicated that the state felt that it had a firm understanding of what the impact of eradicating *Z. japonica* would be and ultimately moved forward with an eradication attempt. Since detection of *Z. japonica* in Humboldt Bay was believed to have occurred shortly after introduction to the area, scientists and managers believed eradication to be feasible. Yet, the decision to eradicate *Z. japonica* in California did not come without significant consideration. Managers and stakeholders from the region contacted their peers in Oregon, Washington, and British Columbia in an effort to gain a better understanding of how *Z. japonica* was impacting other ecosystems.

After considerable discussion, eradication via mechanical removal was attempted in Humboldt Bay. Despite the ultimate failure of this eradication effort, it was stated that there is still significant interest in continuing eradication efforts if the fiscal means to do

so become available. If eradication were to be attempted again, more research on the role of *Z. japonica* in Humboldt Bay would be desired, as well as broader monitoring efforts than were previously required. Interviews with participants from Washington and Oregon shared that they believe that eradication would have been more successful if California had used chemical control measures because of the small population they were initially dealing with. However, application of chemical controls to the intertidal environment was unacceptable to the general public.

Other participants noted that while eradication of detrimental invasive species may be desired, it is not a particularly fruitful exercise when the species has already become established. For instance, in Washington State, it is generally conceded that it is likely "too late" for eradication of *Z. japonica* without causing excessive damage to native eelgrass. Some also believe that there may be more negative impacts in trying to remove an invasive species like *Z. japonica* than by selectively controlling it. In comparison to the eradication of *Spartina* in Washington State, eradication of *Z. japonica* is believed to have the potential for much greater impacts on native species because it overlaps with *Z. marina* (and the large, diverse communities associated with it).

Researcher and manager participants alike had a number of suggestions for managing *Z. japonica* in areas where eradication was not feasible:

1) Selective removal and management of the species in areas of particular concern.

2) Removal of the protections afforded *Z. japonica.*

3) Creating education and outreach programs on *Z. japonica* in an effort to get the public involved.

4) Instituting road checks or boat launch inspections to determine if seeds or reproductive shoots are being transferred.

Several participants noted that in terms of invasion biology, rapid response to the discovery of an introduced species is the best possible response in terms of controlling the spread and mitigating damage that may arise. However, one participant considered the "appropriateness" of eradication to be "the only option when one is working from the vantage that keeping invasive species out is 'good'" and that "determination of the 'goodness' of such a perspective should be done by managers, not scientists, and be a reflection of society's views."

3.2.2.2 Regional Collaboration

Interview participants identified a number of areas where invasive species management could be improved on both a regional and national scale. One topic discussed by participants was an increase in collaboration and discussion between local, state, regional, and federal levels. Some participants felt that local bays and estuaries were part of regional level research but were not aware of the scope, purpose, or findings because of little to no communication between research agencies and local level managers. Effective management of invasive species necessitates regional collaboration between all parties presently or potentially impacted (Williams 2007; NISMP 2008). Australia and New Zealand have demonstrated that centralized management of invasive species allows for quicker responses, more coordination, and standardization of management and detection efforts (Stocker 2004; Williams 2007; Boonstra 2011).

Some participants explicitly suggested a system more akin to that used in New Zealand, which operates on three fronts: (1) prevention and exclusion of unwanted organisms; (2) surveillance and response to detect unwanted organisms; and (3) management of established pests (Boonstra 2011). Such a plan would require considerably more investment in vector control and large scale environmental monitoring for invasive species, which were also independently suggested by participants. However, it was also noted that there are considerable political hurdles in creating any new national management plan, requiring a significant increase to monetary costs.

With respect to management plans already in place for aquatic invasive species, it was suggested that a mandate to follow federal level regulations for ballast water treatment would alleviate confusion in what regulations apply to US waters. Additionally, by mandating international compliance with ballast water treatment regulations, further confusion could be reduced as well as mitigation of risk associated with transfer of aquatic organisms in ballast water. The need for collaboration, however, extends beyond management decisions (Stocker 2004); educational outreach, extension, stakeholder engagement, public volunteer efforts, political involvement, and resource allocation must all be integrated at the ecosystem, or regional, scale.

Recent legislative calls requiring the incorporation of Ecosystem Based Management (EBM) incorporate many aspects of collaboration between parties (National Ocean Policy Implementation Plan 2010). However, EBM efforts are still in their infancy, in that what truly defines “ecosystem-based” is often poorly defined or misconstrued (McLeod and Leslie 2009). Within the PNW, some manager participants

expressed that while some discussion occurred, federal agency research was not conveyed effectively to local level managers in a watershed.

3.2.2.3 Explicit Statement of Management Position

Another issue brought up by multiple interview participants was the belief that an official management position should be explicitly stated and remain consistent across all political levels. Participants believed that management objectives were more obtainable and compliance greater when consistent rather than when multiple sets of regulations exist. This follows closely with the core concepts of EBM, which integrates the concept of explicit management goals and tradeoffs (McLeod and Leslie 2009).

Explicitly stating goals and the effects that reaching them will have upon the range of potential stakeholders inherently requires managers to look at effects across the entire spectrum of spatial and stakeholder scale. State and federal managers' initial efforts to manage *Z. japonica* failed to explicitly consider all stakeholders. Furthermore, these efforts, at times, failed to identify whether there was even a management plan in place (Shafer et al. 2014).

Despite a call to have a consistent management position across all scales, interviewees stated that consistency does not preclude different levels of management. They felt this to be particularly true under environmental or spatial conditions where one set of rules is impractical. If multiple management regimes were necessary for a species, it is absolutely necessary to convey to the public why such an approach is taken. Without a clear understanding of why control regimes are different, the public could perceive

management of the issue as inconsistent or ill-conceived.

In order for future management to be effective, managers should inform the public about the issue, engage them regarding its effects, and communicate the expectations and reasoning for management strategies. Utilizing the principles of EBM, future management of *Z. japonica* should integrate management efforts of all entities involved – from local to federal. Management action taken without explicit inclusion of all management agencies may result in contrasting goals and programs that ultimately compete with one another. Without full public cooperation in the process, natural resource management becomes a task akin to that of Sisyphus'.

3.2.2.4 Expediting the Invasive Species Listing Process

One hurdle shared by participants is that the NISMP has a large lag time between identification of a non-native species and a designation as invasive. Manager and researcher participants alike shared a genuine desire to understand how the new species fits into the ecology of a region before making that determination. Researcher participants stated that multiple years may be needed to examine a non-native species within a new ecosystem to assess effects on native biota and ecosystem function, particularly when a species has never before been seen outside its native range. By the time research has made its determinations and passed them on to managers for assessment and determination of management action, too much time may have passed to initiate effective eradication efforts. The structure of NISMP and NISA, and the biological systems that are being examined operate across highly variable timescales.

Neither environmental variability nor the difficulty initiating management are reflected in NISMP or invasive species management as a whole. This is particularly true in the case of novel introductions, where the species or functional type has not been observed in an invasive capacity before.

3.2.2.5 Standardization of Invasive Species Management

Standardizing the listing process was a common critique of interview participants. Many participants expressed a feeling that the current process is piecemeal and often confusing, which leads to gaps in enforcement and lags in management. The current federal system for managing invasive species is a coordinated effort of 35 agencies, in addition to state cooperation. This system, at times, can be labyrinthine to navigate for stakeholders wishing to engage managers, as well as for managers to find the appropriate agency that has priority in managing any single case. In addition, the independent listing process for each species that must occur for each state was felt to be an impediment to effective management by interviewees.

Political consistency was another issue brought up in interviews. Manager participants stated that compliance is more obtainable when there are not multiple sets of regulations either in different agencies or geographic locations. Several participants believed that consistency gives an appearance of solidarity, support, and open communication not only to the public at large but also state and federal legislatures. Participants also reported a belief that public education is more accessible when the regulatory environment is not littered with caveats. Multiple interviewees stated that

consistency does not preclude different levels of management action, particularly under environmental or spatial conditions where one set of rules is impractical. They felt if multiple management regimes were necessary for a species, like selective removal of *Z. japonica* in one bay and eradication in another, it is particularly important to convey to the public why there is a necessity for multiple management strategies. Without a clear understanding of why control regimes are different, the public could perceive management of the issue as inconsistent or ill-conceived.

3.2.2.6 Outreach & Education

Interview participants also indicated that public outreach, education, and professional training were all seen as critical to effective management of invasive species. Public and professional understanding of several fundamental concepts in invasion biology have been found to be commonly misunderstood (Selge et al. 2011). It is possible that both political compliance and involvement in conservation issues may improve with more transparent communication of core concepts of invasion biology (i.e., vectors, damage, impacts) to the general public and stakeholders.

Increasing the availability of outreach and engagement opportunities targeting stakeholders also has the potential to frame issues in such a way that concerns are addressed and hasty decisions are bypassed. Outreach and engagement can also do more than just educate about the harm invasive species are doing. They also provide an opportunity to discuss the actions being undertaken to restore, conserve, or respond to incidents of non-native species, providing increased understanding of management

mechanisms and potentially facilitating greater compliance and involvement.

Public outreach was also seen as critical for the effective management of invasive species. Educating the public about the cost of invasive species, not only to the government, but to industry and the general public, was one suggestion. By creating a sense of individual investment with an issue, one participant believed that members of the public would be more open to not only following management suggestions, but also getting involved. Providing opportunities for the public to help control the spread or removal of invasive species was seen as invaluable. It was also suggested that attempts to engage the public via social media be attempted as well, because there is currently little utilization of the technology by natural resource managers.

The public and stakeholders are not the only ones who may benefit from outreach, education, and engagement. One interview participant expressed a desire for more professional development regarding the characteristics, monitoring techniques, and management strategies used to control invasive species. Development of professional development programs also has the desired side effect of standardizing monitoring techniques, clarifying the ecology of an invasive species, and reiterating the management objectives and position to those who need to know it most.

3.2.2.7 Social Context of Natural Resource Management

All participants had strong opinions that the management of natural resources should reflect society's views of an issue, and that management must be fluid enough to change if social pressures shift. A management approach must also not suffer from

political or legislative inertia that would limit its ability to change objectives or methods if social pressures emphasize different conditions than what managers are aiming for. In order for management to be relevant and accepted by the general public, re-examination of current practices and standards are absolutely necessary. Managers and scientist participants alike felt that managers need to be responsive to concerns of the public.

Multiple participants stated that there is not always an internal mechanism that facilitates changing management strategies to reflect changes in social pressures. Rather, change is typically initiated by an external party that wants to see a certain change in goals that the management actions are emphasizing. Sometimes, however, there are political or legislative constraints limiting managers' ability to respond to changing social values. Even if no external party actively argues for change, there is room in some management strategies for passive changes to occur.

For example, in Washington State, if stakeholders decided management of *Z. japonica* was no longer an issue, then applications for herbicide permits would drop off and management of it would not occur because of a lack of demand. Participants emphasized that scientific information should not define desired future conditions, but rather provide a mechanism to achieve societally desired conditions. As scientists and managers learn more about a given subject, it is their duty to educate the public so that the public's perceptions and values are based on reality rather than an abstraction or distortion of it.

3.3 Future Studies

Recent studies have greatly expanded our understanding of the role of *Z. japonica*

in the PNW. However, significant research gaps still exist. While the study described here provides baseline observations about the nekton community during spring tidal exchanges, it does not explore variation in use across seasonal or diurnal time scales. Inference on the mechanism structuring the observed patterns is also limited. Future studies should examine how the intertidal habitats are being used by their inhabitants, as this will improve our understanding of the mechanisms structuring community composition. This can be done through either an assessment of time spent within the video frame or a categorization of behavior for each observation.

One of the goals of this research was to examine how these habitats are used by species of interest across multiple life stages or year classes. However, this remains unexamined due to an inability to quantitatively compare organism size across habitats or replicates. Continuation of this work should identify patterns of utilization across multiple life stages or year classes of species utilizing *Z. japonica* and other intertidal habitats by having an effective means of comparing size in video footage. Examination of the data collected in this study, and those of similar methods, would glean much more information about habitat use by examining temporal trends in habitat use by referencing when individual species move into each habitat and relating this to physical variables such as water temperature, dissolved oxygen, and tide height.

Additional certainty could be added to identification of individuals by adding in an oblique angle to the video footage. Stitching of multiple camera angles together may allow for both assessment of the community utilizing the epibenthos as well as identification of fauna that are difficult to distinguish. Rare taxa may also be more

effectively sampled by either increasing the sample area of the video frame, increasing the number of cameras, or by towing cameras within habitats.

To provide information that would better inform management of *Z. japonica* in the region, broader questions must also be addressed. One subject of study that is necessary would be an understanding of how competitive interactions change between *Z. japonica* and *Z. marina* under different climate regimes. Managers require a clearer understanding of the impacts of *Z. japonica* expansion on clam and oyster aquaculture. A large scale accounting for edge effects and scale is essential in addressing the contrasting observations observed between Willapa Bay and Yaquina Bay (Patten 2014).

3.4 Concluding Remarks

Utilizing the information generated by this study could lead marine resource managers to closely examine site and relative distribution of *Z. japonica* and *Z. marina* if there are certain aspects of community composition they wish to preserve or alter by managing *Z. japonica*. Managers highlighted a number of suggestions based on their experience working with *Z. japonica*. The view that all levels – federal down to local – cooperate and explicitly communicate on issues of invasive species was shared by all interviewees. Requests that all agencies explicitly state a management position for every invasive species and share that position was called for in an effort to simplify the regulatory framework that the public must comply with and provide an image of solidarity. It was also suggested that a more centralized approach to invasive species management similar to that taken in New Zealand would be effective. Finally,

participants also called for increased public outreach and education on invasive species, as well as more professional development opportunities for managers and stakeholders.

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Appendices

Appendix 1

Background Questions Provided to Interview Participants

The goals of the interview are as follows: 1) Provide a cursory background to historical and current management decisions regarding Z. japonica in the PNW region, 2) discuss contrasting regional responses to Z. japonica in the PNW, and 3) highlight suggestions for continued management of Z. japonica in the PNW.

1. What characteristics of *Z. japonica*'s ecology and expansion do you think **were** most important in determining management strategies? How do these characteristics continue to play into management of *Z. japonica?*

2. One of the tenants of invasive species management is early detection and rapid response. Do you think that rapid eradication efforts were the most appropriate response to *Z. japonica*'s expanding range?

3. Given that there is some evidence that *Z. japonica* provides a number of ecosystem services in the PNW, are eradication efforts still necessary?

4. Do you think that management decisions should or could change depending on how society's views towards environmental issues change?

5. (CA & WA) Your state has a management plan for *Z. japonica*. How do you see management in adjacent states affecting management in your state?

6. (OR) Your state doesn't have a management plan regarding *Z. japonica*. How do you see management strategies in your state affecting attempts in adjacent states?

7. What are some changes that you think would make management and determination of invasive species in the US more robust and effective?

8. What are some changes that would make management of aquatic invasive species in the US more robust and effective?