Humboldt Bay
Harbor, Recreation and Conservation District
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June 10, 2015

Dr. Donald McIsaac
Pacific Fishery Management Council
7700 NE Ambassador Place, Suite 101
Portland, OR 97220-1384

Re: Draft Pacific Fishery Management Council letter regarding proposed mariculture projects in Humboldt Bay, California.

Dear Dr. McIsaac and Council Members,

On June 8th I was contacted regarding the draft letter to the Humboldt Bay Harbor District (District) that is on your June 12th agenda. Had I been aware of the Council's interest in the proposed Humboldt Bay mariculture projects, I would have contacted you earlier and potentially arranged to travel to your meeting. Regardless, I have drafted the following comments, please consider them to the extent possible.

PFMC's draft letter pertains to (1) the Mariculture Pre-Permitting Project for which the District is the proponent (District Project) and (2) the Coast Seafoods Permit Renewal and Expansion Project for which Coast Seafoods Company is the proponent (Coast Project). Under the California Environmental Quality Act (CEQA), the District is the Lead Agency for both projects. The formal CEQA process has not begun for the Coast Project, but a Draft Environmental Impact Report has been circulated for the District Project. The District is currently revising our project based on comments received and we will be responding to all comments, including any we receive from PFMC. My understanding is that Coast Seafoods Company is also making project revisions based on comments received prior to the formal CEQA process (i.e., comments on a draft Initial Study).

Although the two Projects have many similarities (i.e., culture of the same species within the same bay) and certainly require a detailed and robust cumulative impact analysis, they are nevertheless separate projects with different timelines and project proponents. As such, the District's role and capacity to respond to comments on each Project is different. Hence, it would be useful, and I respectfully request, a de-coupling of the comments for each Project into separate letters.

Additionally, I urge PFMC staff to contact me and Coast Seafoods Company to gain an understanding of changes to proposed project designs that have occurred in response to previously received comments. As currently drafted, PFMCs letter pertains to outdated project descriptions.

Thank you for your interest in our efforts. I look forward to working with you and continuing to engage with the National Marine Fisheries Service and other agencies to design projects that meet our goals of increasing sustainable seafood production and protecting the bay's ecological resources.

Sincerely,

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June 10, 2015

D.O. McIsaac, Ph.D.
Executive Director
Pacific Fishery Management Council
7700 NE Ambassador Place, Suite 101
Portland, Oregon 97220-1384

RE: Agenda Item C.1: Proposed Letter Concerning Coast Seafoods Aquaculture Project

Dear Dr. McIsaac and Honorable Councilmembers:

Thank you for the opportunity to comment on the Pacific Fishery Management Council ("Council" or "PFMC") Habitat Committee's ("Committee") concerns regarding Coast Seafoods Company's ("Coast") planned Humboldt Bay shellfish culture permit renewal and expansion ("Project"), as expressed in the Committee's draft letter to the Humboldt Bay Harbor, Recreation and Conservation District ("Harbor District").

A. Background and Project Description

Coast has a long history of shellfish cultivation in Humboldt Bay. Coast owns or leases approximately 4,000 acres in Humboldt Bay for the purpose of shellfish cultivation, shown in Figure 1, most of which it has controlled since 1955. Coast began cultivating shellfish in Humboldt Bay in the 1950s and has farmed up to approximately 1,000 acres of its tidelands at the peak of its operations.

Figure 1. Coast Seafoods Company's shellfish culture leases and ownership in Humboldt Bay, California.

Some of the major changes in operation from the 1950s included:

- 1960-1970s: began using various off-bottom methods, including stake, rack-and-bag, and longline culture.

- Early 1970s: started culturing Kumamoto oysters (*C. sikamea*) using longline and rack-and-bag culture methods.

- 1997 to 2006: transitioned more than 500 acres of ground culture to approximately 300 acres of off-bottom culture (e.g., cultch-on-longline, basket-on-longline), with the understanding that Coast may seek to re-permit additional acreage once more was known about off-bottom culture methods.

The proposed Project involves renewing Coast's permits for its existing shellfish beds in Humboldt Bay and re-permitting 622 of the acres it historically farmed, as shown in Figure 2. The Project does not include any areas beyond those that have been leased by Coast for the last several decades and does not involve any changes in species cultured or culture methods. The 622 acres would be comprised of 522 acres of cultch-on-longline culture and 100 acres of either basket-on-longline culture or rack-and-bag culture, although rack-and-bag culture would not be located in areas with eelgrass beds. In total, the Project would expand Coast's operations in Humboldt Bay to 910 acres of shellfish aquaculture (17% of its leased area), all of which falls within its historic operational footprint.

The expansion is essential for Coast to meet its shellfish demand. The Project will provide approximately 50 to 60 living-wage jobs in Humboldt Bay, an area that has faced significant unemployment or underemployment issues in recent years. Without Project implementation, the West Coast will continue to face supply problems associated with a lack of domestic shellfish and will be forced to rely on unregulated international sources for seafood products, further contributing to an $11 billion national seafood deficit and depriving the Pacific Coast of a local, green, and extremely water-efficient food source.

B. Permitting Process

Coast is in the early stages of its permitting process. It has submitted its application to the Harbor District, but must still go through the California Environmental Quality Act ("CEQA") process before the Harbor District issues its determination. As noted in the Committee's draft letter, Coast is preparing a Draft Environmental Impact Report ("DEIR") under CEQA, which will include a detailed analysis of eelgrass and fish impacts, a robust eelgrass monitoring plan, and proposed mitigation compliant with CEQA and State and Federal regulations.

A Notice of Preparation ("NOP") describing the revised project description and proposed mitigation will be circulated for public review within the next two months. We expect that the DEIR will be circulated for public review 30-60 days after the NOP. To the best of our knowledge, the DEIR will be the first such document prepared for a private commercial aquaculture project in California. Coast also plans to file its applications with the California

Coastal Commission and U.S. Army Corps of Engineers concurrently with its release of the DEIR to facilitate concurrent review by those agencies. Coast will also be preparing the appropriate documents to facilitate the Corps' consultation with the National Marine Fisheries Service ("NMFS") and U.S. Fish & Wildlife Service to comply with Endangered Species Act and Magnuson Steven Act requirements, The Biological Assessment and Essential Fish Habitat Analysis will be submitted with Coast's application.

In order to facilitate a more productive conversation based on the revised Project and environmental analysis, Coast requests that the Committee and Council refrain from submitting comments on the Project until the DEIR is published. Coast would be happy to make a presentation to the Committee and/or Council during the DEIR public comment period to update the Committee and Council regarding the revised project and answer any questions.

Figure 2. Areas proposed for continued and expanded shellfish culture.

C. Eelgrass Effects

The Committee's draft letter expresses concern with the Project's potential impacts to essential fish habitat ("EFH"), particularly eelgrass. Because the concerns expressed do not reflect the current status of Project design and may be moot given the extensive modifications made in response to public comment on the draft Initial Study ("IS"), we would recommend that the Committee and Council refrain from commenting on the Project until it has an opportunity to review the changes. However, we provide the following comments which highlight some of the points that will be presented in more detail in the DEIR.

1. Overview of Humboldt Bay Shellfish Aquaculture and Eelgrass Habitat

Coast acknowledges that eelgrass is an important resource that serves several important ecological functions in Humboldt Bay. However, shellfish aquaculture and eelgrass have co-existed in Humboldt Bay (specifically North Bay) for at least the last 60 years of commercial shellfish production and for more than 100 years since the first attempts to introduce shellfish in 1896. During the most recent change in operation (i.e., transition to off-bottom culture), Coast reduced the geographic coverage of its operations by eliminating culture on roughly 200 acres. One of the main researchers that studied the effects of this transition (Steven Rumrill, Ph.D.) noted that, "eelgrass beds and commercial oyster cultivation can coexist in Humboldt Bay, and that implementation of best management practices that include reduced density of oysters (i.e., oyster culture at 5 ft and 10 ft spacing between the longlines) may aid in the conservation of eelgrass communities" (Rumrill 2015).

Based on Dr. Rumrill's research, one of the key modifications Coast has made to the project design that will be in the DEIR will be employing uniform 5-foot spacing between longlines, as opposed to the four lines spaced 2.5 feet apart and an open row of 10 feet described in the IS. The basket-on-longline culture will also be spaced 5 feet apart, with an open row of 20 feet between groups of 3.

There are numerous limitations to eelgrass growth in Humboldt Bay[1], especially in North Bay where growing conditions are not as conducive to eelgrass growth as compared to South Bay (Harding 1973, Gilkerson 2008, Schlosser and Eicher 2012), but shellfish aquaculture does not appear to be a major restriction in eelgrass growth since the transition to off-bottom culture. Bay-wide mapping efforts in Humboldt Bay ranged from a minimum of 840 acres in 1959 to a maximum of 3,577 acres in 2009 (Schlosser and Eicher 2012). While there are limitations to comparing mapped eelgrass between years due to differences in methodology, a review of the data suggests that eelgrass in Humboldt Bay is extensive, relatively stable, and may be increasing in the presence of current sustainable shellfish aquaculture practices (Figure 3).

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[1] See the detailed discussion in the "Coast Seafoods Shellfish Aquaculture and Eelgrass Ecological Review for Humboldt Bay" provided as an appendix in the Initial Study for the Coast Expansion Project.

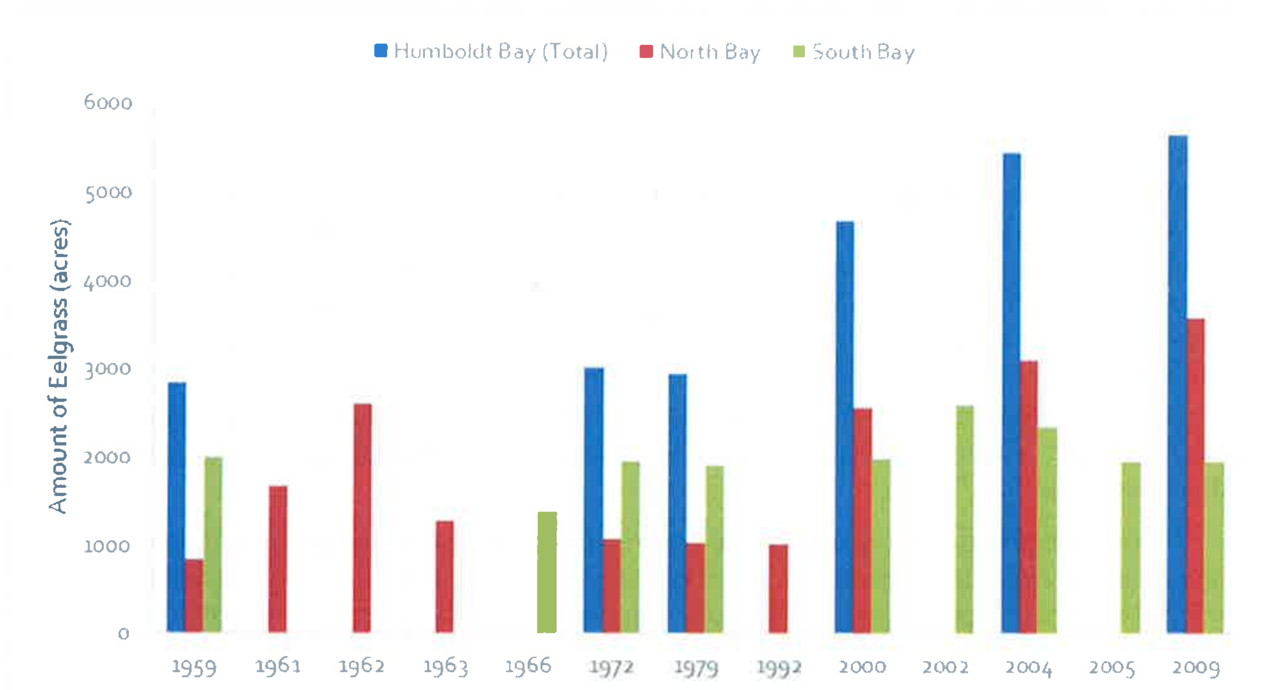

Figure 3 Amount of Eelgrass in North Bay from 1959 to 2009
Source: data presented in Table 22 of Schlosser and Eicher (2012)
Note: one error was noted for the Entire Humboldt Bay; value for 1972 should be 3,017 acres

2. Eelgrass Aquaculture and Shellfish Interactions

The National Oceanic and Atmospheric Administration ("NOAA") Coastal Aquaculture Planning and Environmental Sustainability ("CAPES") program is currently conducting a state of the science assessment on shellfish aquaculture interactions with submerged aquatic vegetation. This effort is expected to be completed by October 2015. Confluence provided a summary of information related to this effort, which is provided as Exhibit A. The document discusses the fact that aquaculture activities can have both positive and negative interactions with eelgrass at a unit scale. This suggests that managers and resource agencies need to look at effects from a broader perspective, such as the landscape or watershed scale, in order to understand the overall effects of shellfish aquaculture in relation to ecosystem health and the ecological functions. This is the effort that is currently underway through the DEIR that will be provided on the Project.

Native eelgrass exhibits a stable and possibly increasing trend in distribution and abundance in areas like Willapa and Humboldt bays, where shellfish have been actively farmed for over 100 years (Barrett 1963, Tallis et al. 2009, Dumbauld et al. 2011). The mechanisms of eelgrass' co-existence, resilience, and recovery related to shellfish aquaculture are not entirely understood. However, processes that interact to create eelgrass resilience depend on both the local habitats and landscape conditions (Thom et al. 2012, Allen et al. 2014). The fact that studies have produced mixed results in terms of the scale of effects and eelgrass recovery after a disturbance from shellfish aquaculture implies that there are conditions at multiple levels that are important to consider (Ruesink et al. 2012). For example, does occasional disturbance enhance growth or reproduction of eelgrass or does shellfish aquaculture reduce desiccation and thereby preserve eelgrass.

In terms of earlier research specific to Humboldt Bay, the best available data is associated with the Western Regional Aquaculture Center study by Rumrill and Poulton (2004). The authors determined that the spatial extent of an eelgrass bed and shoot density were negatively influenced when oyster longline culture was closely spaced (1.5 ft. to 2.5 ft.) but showed no effect compared to control sites when spacing occurred at 5-ft and 10-ft spaces between longlines. Further, recovery areas (e.g., areas that were transitioned from bottom to off-bottom culture) at the 10-ft spacing showed higher eelgrass densities compared to control areas. Both recovery areas and areas within existing longline culture did show a decrease in eelgrass density directly under the longlines but not necessarily in the shellfish aquaculture plot overall (discussed in more detail below). Therefore, there is no support for the language that the draft PFMC letter uses stating that the cumulative impacts could "profoundly damage eelgrass habitat and its ecological role."

Confluence is preparing a technical report for the DEIR to address potential impacts to eelgrass habitat from the Project. The main effects of oyster longlines to eelgrass include the potential to abrade, scour, desiccate, or shade eelgrass blades, although the overall effects to the eelgrass bed can be both positive and negative. For example, Tallis et al. (2009) explored the concept that, while shellfish can break eelgrass blades through abrasion, the reduction in density can release individuals from intraspecific light competition and result in increased growth rates near the aquaculture plots.

Further, one of the major limiting factors in Humboldt Bay is light attenuation due to high suspended sediment loads from freshwater sources and resuspension of material over shallow mudflats in North Bay (Gilkerson 2008, Shaughnessy et al. 2004, Shaughnessy and Hurst 2014). Both eelgrass beds and shellfish aquaculture can enhance water clarity by acting as a suspended sediment sink. Shellfish aquaculture can also result in a reduction in turbidity due to removal of phytoplankton and particulate organic matter through filtration (Peterson and Heck 2001, Newell and Koch 2004, Cranford et al. 2011). By consuming phytoplankton and particulate organic matter, shellfish increase the amount of light reaching the sediment surface that is available for photosynthesis (Dame et al. 1984, Koch and Beer 1996, Newell 2004, Newell and Koch 2004). While there can also be a slight reduction in light from macroalgal growth on the aquaculture gear itself, there is likely a net positive to eelgrass growing conditions through decreased desiccation during low tide.

The calculation in the draft PFMC letter that 17% of eelgrass habitat in Humboldt Bay is proposed for use for shellfish aquaculture is not an accurate understanding of either project. The area where eelgrass is potentially affected includes the area directly under the longlines (i.e., 100 ft. x width of effect) and not the entire proposed acreage associated with shellfish aquaculture. Figure 4 provides a depiction of "width of effect" under the longlines. The total area directly under the longlines, including the area potentially shaded by macroalgal growth, represents about 7% of the proposed expansion area for the Project. However, having an influence on or interaction with eelgrass is not the same as a loss of eelgrass habitat or its ecological function. In fact, a mosaic of habitat types (i.e., eelgrass and shellfish) provides ecological functions and values to a larger suite of species than any other habitat type alone.

Figure 4 Depiction of Width of Effect Directly Under Oyster Longlines
Source: Dale, pers. comm., 2015

Based on recent observations of effects from the existing aquaculture in Humboldt Bay, there are no expected changes to eelgrass bed areal extent (i.e. total acreage of eelgrass beds) but there would likely be a reduction in eelgrass density directly under the longline.[2] Using both recently collected data (2015) and estimates of potential reduction directly under oyster longlines provided by Rumrill (2015), there would likely be a reduction in eelgrass density that equates to a small percent (≤1%) of eelgrass in North Bay overall. While that is a small fraction of available eelgrass in Humboldt Bay, it does represent a potential loss of density for which mitigation is being developed. Coast's proposed mitigation will be circulated with the DEIR.

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[2] It is unclear whether the reduction in density will impact eelgrass function at this time, given that the reduction may be within the natural variability of the eelgrass beds. That effect will be monitored and confirmed as part of the Project's eelgrass monitoring plan.

3. Potential Avoidance of Eelgrass Beds

As noted in the draft PFMC letter, permit applicants must first take all practicable steps to avoid impacts to eelgrass under both state and federal policy. However, avoidance of eelgrass is not an option for this Project due the abundance of eelgrass in Humboldt Bay (including on Coast's tidelands) and overlap between suitable tidal elevations for eelgrass and commercial oyster production. In compliance with its existing Coastal Development Permit, Coast evaluated the feasibility of planting and harvesting oysters at elevations typically considered unsuitable for eelgrass growth. The resulting study, attached here as Exhibit B, measured the differences in growth, biofouling, survival and quality of oysters grown above 1.5 to 2.0 ft. above MLLW and in control plots at industry standard tidal elevations (0.5 – 1.0 ft. above MLLW). The study demonstrated a statistically significant difference in the total weight of Kumamoto and Pacific oysters grown at higher tidal elevations (51% and 65% of control plot values, respectively) and a statistically significant difference in the mean total number of Kumamoto oysters (52% of control). These decreases in production at higher tidal elevations are commercially untenable. For this reason and because eelgrass already populates the majority of the available suitable acreage in the Bay, Coast has no practicable choice but to cultivate shellfish at elevations also suitable for eelgrass growth, using cultivation techniques shown to reduce or eliminate potential impacts to eelgrass.

4. Eelgrass Buffers

Several of the concerns expressed in the draft letter appear to focus on on-bottom culture methods as compared to off-bottom longline aquaculture. The Project calls for a maximum of 522 acres of cultch-on-longline and up to 100 acres of basket-on-longline and rack-and-bag culture techniques. Only cultch-on-longline and basket-on-longline culture will be placed in eelgrass; rack-and-bag culture will not be placed in eelgrass. The 25-30 foot buffer from existing eelgrass referenced in the Council's draft letter is not appropriate for longline aquaculture which, as noted above, has been shown to have little or no impact on eelgrass if planted at appropriate spacing.[3]

Multiple state and federal agencies recognize that longlines are a culture technique that can be employed in eelgrass beds without causing significant impact. NMFS' 2009 Biological Opinion for Nationwide Permit 48 (shellfish aquaculture) distinguished longline and other culture methods and recommended longline spacing at 5 foot intervals over eelgrass beds.[4] NMFS' 2014 California Eelgrass Mitigation Policy ("CEMP") similarly treats shellfish aquaculture longlines as a "Special Circumstance" and states that longlines may not result in measurable net loss of eelgrass habitat within a project area.[5] In such cases, mitigation to

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[3] Appendix A to the PFMC's Pacific Coast Salmon Fishery Management Plan suggest a number of potential conservation measures for artificial propagation of fish and shellfish but recommends that the action agency undertake them on a site-specific basis and acknowledges that more specific or different measures may be implemented based on current science. The recommendations do not suggest a fixed or one-size-fits all approach to conservation. PFMC, Appendix A to the Pacific Coast Salmon Fishery Management Plan 60-61 (Sept. 2014).

[4] NMFS, Endangered Species Act—Section 7 Programmatic Consultation Biological and Conference Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation: Nationwide Permit 48 Washington 64 (April 28, 2009).

[5] NOAA Fisheries, California Eelgrass Mitigation Policy and Implementing Guidelines 28-29 (Oct. 2014).

compensate for local losses under longlines may not be necessary.[6] Imposing a "one-size-fits-all" buffer for all shellfish projects would negate the hard work of Coast and other shellfish companies to design new culture methods that are less impactful on the surrounding environment, including eelgrass.

D. Herring Effects

Pacific herring spawn in both North and South bays, but most spawning occurs in the northern end of the bay (Rabin and Barnhart 1986). This is possibly due to an interaction between herring and freshwater inflows where low-salinity conditions may stimulate herring spawning. Although eelgrass is the principal substrate used for spawning in Humboldt Bay, the densest beds are not used in some years (Barnhart 1988). A typical spawning event involves the deposition of herring eggs on approximately 300 acres of eelgrass in North Bay (Mello and Ramsay 2004), which is less than 10% of available eelgrass used in each spawning event. This is similar to reports from Puget Sound where a large proportion of herring spawning habitat remains unused each year (Shelton et al. 2014). All of this information provides an indication that Pacific herring are not limited by spawning substrates in Humboldt Bay.

Trends in eelgrass abundance and shellfish aquaculture operations also appear to be unrelated to herring spawning biomass in Humboldt Bay. Eelgrass has been stable or expanding since the 1960's with recent estimates more than twice as large as historic estimates (discussed above). Shellfish aquaculture operations were reduced by 200 acres in North Bay starting in 1997 (discussed above), and 83 acres were removed from the East Bay Management Area. Herring biomass estimates have only been recorded for 11 of the past 41 years. The peak observed spawning (950 tons) occurred between 2000 and 2002 (when herring were commercially harvested in the bay), followed by a decline to 7 tons before monitoring was suspended in 2007 (Figure 5).[7] Measurements of herring biomass during the peak of shellfish aquaculture activity prior to 1997 were equivalent to the average biomass reported (Rabin and Barnhart 1986, Mello 2007).

[6] *Id.* at 29.

[7] While herring spawning has been observed since 2007, there are no biomass estimates available since commercial herring harvests were suspended. Recent declines have included truncation of age classes with the loss of older fish (4- to 6-year olds), which is likely an effect of overharvesting (e.g., Anderson et al. 2008, Hsieh et al. 2010). The continued failure to rebuild this age class structure, despite curtailed harvest efforts, suggests that oceanic mortality may be responsible (Bartling 2008).

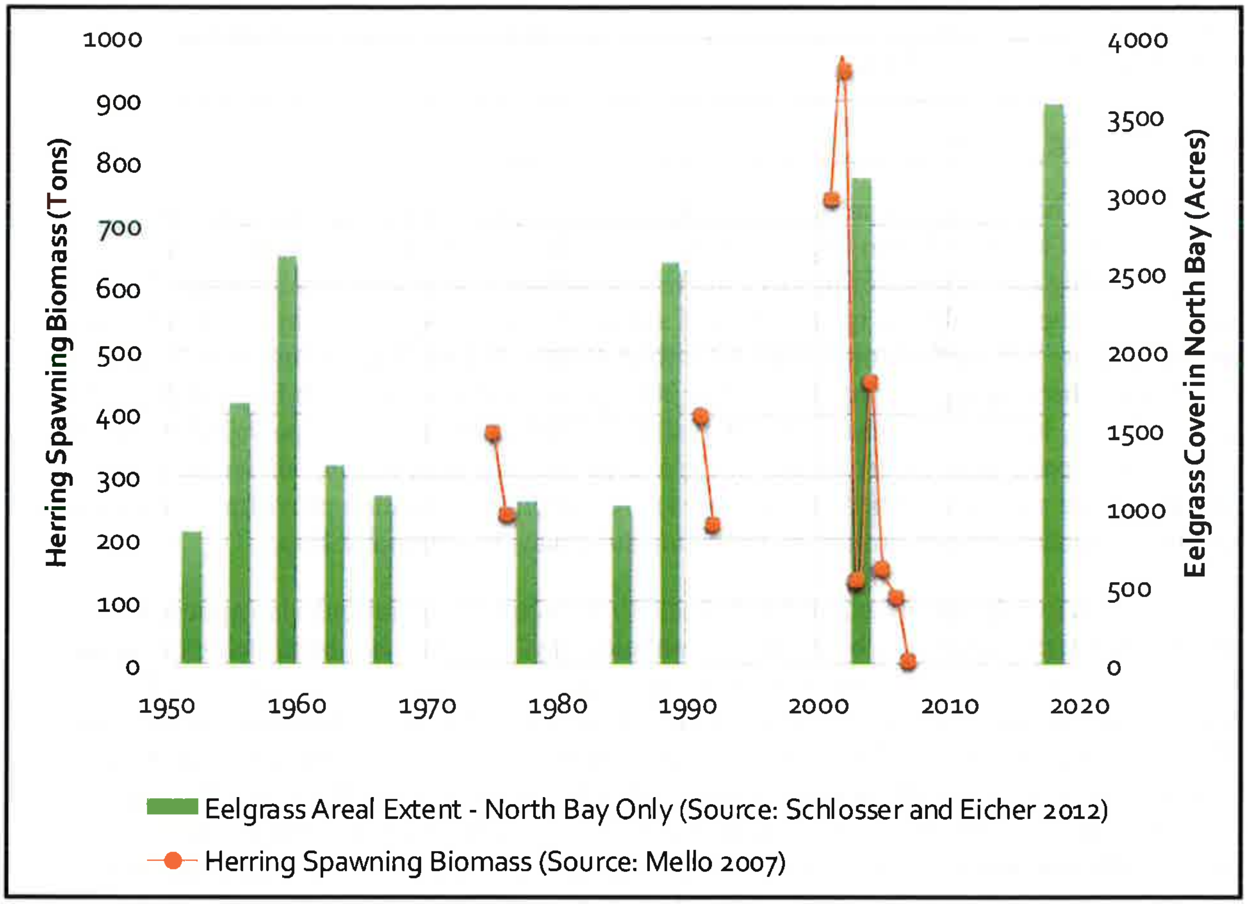

Figure 5 **Herring Spawning Biomass and Eelgrass Areal Extent in North Bay (Humboldt Bay)**

E. Other Fish Effects

Eelgrass beds in Humboldt Bay are known to be very productive areas that provide habitat for refugia, foraging, and spawning for a wide variety of species. Several studies have shown that aquaculture gear can provide similar "biogenic" habitat services and ecological benefits. Shellfish aquaculture can increase the complexity of habitats. Increased habitat complexity typically supports a wider variety of species (Eggleston et al. 1999, Peterson and Heck 1999, Coen et al. 2011).

A study performed by Pinnix et al. (2005) directly addressed the question of difference in use by fish between shellfish aquaculture, eelgrass habitat, and mudflat habitat in Humboldt Bay. The major results of the study indicated that: (1) more fish were collected from oyster culture areas compared to mudflat and eelgrass habitats using trawl and fyke net sampling gear (see Figure 6 as an example of fyke net samples); (2) species richness and diversity of fyke net samples were similar between oyster culture and eelgrass habitat, which were both significantly higher than mudflat habitats; (3) 42% of the total number of fish captured were using these habitats as nursery habitat; and (4) the dominant species collected included English sole and shiner surfperch. Therefore, shellfish aquaculture can provide productive fish habitat (including groundfish) in Humboldt Bay.

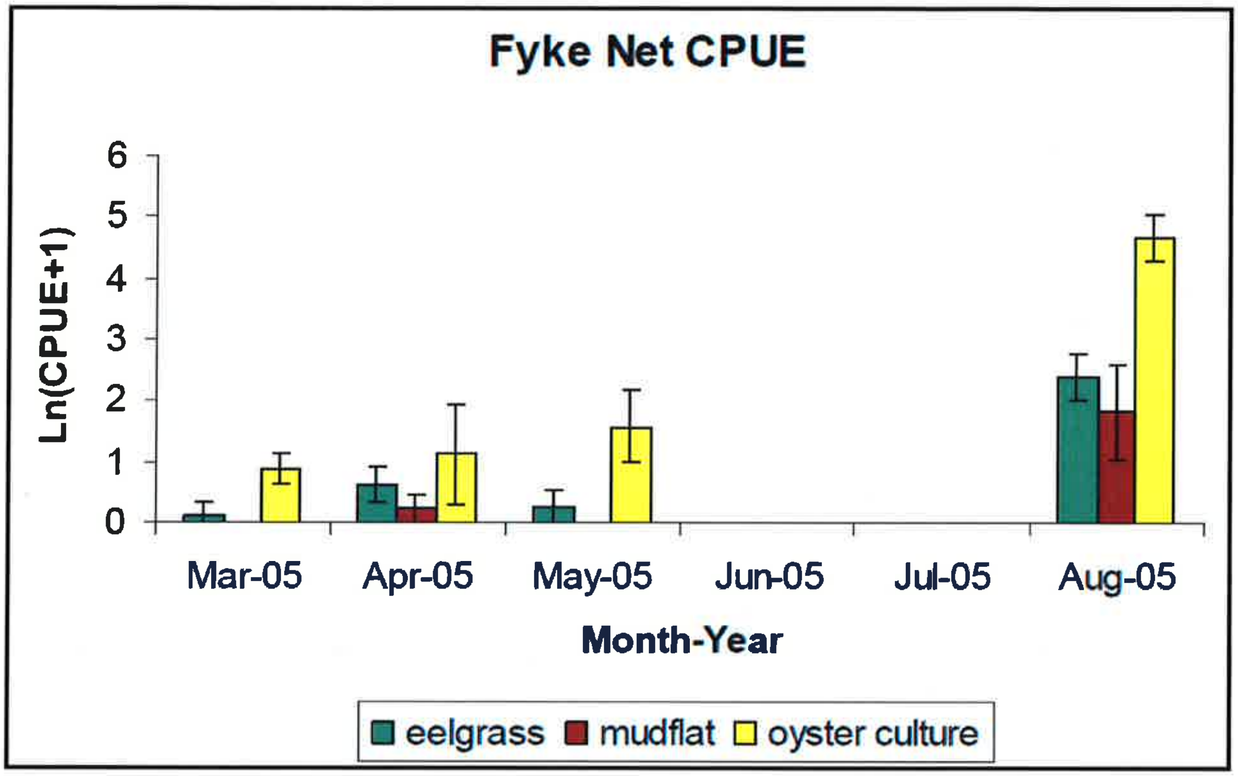

Figure 6 **Mean Monthly Natural Log of Catch per Unit Effort (CPUE+1) of Fyke Net Samples Collected in North Bay, March 2005 to August 2005**

Source: Pinnix et al. (2005)

A more recent study by Pinnix et al. (2013) showed that during their residency in Humboldt Bay, coho smolts primarily used deep channels and channel margins. They were detected near floating eelgrass mats adjacent to the channels, but not over eelgrass beds. The results from this study potentially emphasize the importance of edge habitat and the need for structural heterogeneity during salmonid residency and migration through the estuary. Both shellfish aquaculture and eelgrass habitat provide refugia and foraging potential for mobile species (Hosack 2003, Hosack et al. 2006, Ferraro and Cole 2011, Ferraro and Cole 2012). Therefore, the claim made in the draft PFMC letter that EFH would be significantly impacted by the proposed actions is not supported by the existing literature on species utilization of shellfish aquaculture gear and eelgrass.

Further, the Project includes strong safeguards to minimize or avoid any potential direct impact on Pacific herring. During months when herring spawn in Humboldt Bay (December-February), Coast will perform visual surveys for herring spawn before conducting work on the shellfish beds. The survey will assess eelgrass, culture materials and the substrate for signs of herring spawn. If herring spawning is observed, Coast will postpone any harvesting and planting activities for two weeks (i.e., the amount of time it takes for herring eggs to hatch) and notify the

California Department of Fish and Wildlife's Eureka Marine Region within 24 hours.[8] This practice is consistent with the Council's recommended conservation measures for artificial propagation of fish and shellfish and with best management practices established by the Washington Department of Fish and Wildlife, which trains shellfish managers to recognize herring spawn.

F. Conclusion

Coast is committed to developing an aquaculture project that can meet its needs for shellfish production using appropriate best management practices and sustainable aquaculture methods. As part of this commitment, Coast is preparing the most extensive environmental review prepared to-date for a private commercial aquaculture project, which will include an extensive eelgrass monitoring plan, proposed mitigation, and thorough analysis of eelgrass impacts as evaluated under State and Federal regulations.

PFMC will have several opportunities to comment during this process, including after circulation of Coast's DEIR and upon submittal of Coast's Coastal Commission and Corps applications. Given the number of changes being made to the Project and additional analysis that will hopefully provide useful insight to the Council regarding the Project's effects and mitigation measures, Coast respectfully requests that the Council refrain from commenting on the Project until it has an opportunity to review the DEIR and revised Project description. Thank you for your time and consideration.

Very Truly Yours,

Robert M. Smith

RMS:cml
Enclosures
Cc: Chris Cziesla, Confluence Environmental Company
Marlene Meaders, Confluence Environmental Company
Greg Dale, Coast Seafoods Company

[8] Draft IS at 22.

REFERENCES – PLAUCHÉ & CARR, CONFLUENCE - AGENDA ITEM C.1

Allen, C.R., D.G. Angeler, A.S. Garmestani, L.H. Gunderson, and C.S. Holling. 2014. Panarchy: Theory and Application. Nebraska Cooperative Fish and Wildlife Research Unit – Staff Publications. Paper 127. http://digitalcommons.unl.edu/ncfwrustaff/127

Anderson, C.N.K., C. Hsieh, S.A. Sandin, R. Hewitt, J. Beddington, R.M. May, and G. Sugihara. 2008. Why fishing magnifies fluctuations in fish abundance. Nature 452:835-839.

Barnhart, R.A. 1988. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest) – Pacific herring. U.S. Fish and Wildlife Service Biol. Rep. 82(11.79). U.S. Army Corps of Engineers, TR EL-82-4. 14 pp.

Barrett, E.M. 1963. The California oyster industry. Scripps Institution of Oceanography Library. http://scilib.ucsd.edu/sio/

Bartling, R. 2008. Pacific herring. As reported in *Status of the Fisheries Report – an Update Through 2006*. Prepared by California Department of Fish and Game. 153 pp.

Coen L.D., B.R. Dumbauld, and M.L. Judge. 2011. Expanding shellfish aquaculture: a review of the ecological services provided by and impacts of native and cultured bivalves in shellfish dominated ecosystems. *In*: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Dale, G. 2015. Personal communication regarding photographs of Coast Seafoods operations in Humboldt Bay. Coast Seafoods Company. May 4, 2015. GDale@coastseafoods.com

Dumbauld, B.R., B.E. Kauffman, A.C. Trimble, and J.L. Ruesink. 2011. The Willapa Bay Oyster Reserves in Washington State: fishery collapse, creating a sustainable replacement, and the potential for habitat conservation and restoration. Journal of Shellfish Research 30(1):71-83.

Eggleston, D.B., E.E. Ward, L.L. Etherington, C.P. Dahlgren, and M.H. Posey. 1999. Organism responses to habitat fragmentation and diversity: habitat colonization by estuarine macrofauna. Journal of Experimental Marine Biology and Ecology 236:107-132.

Ferraro, S.P. and F.A. Cole. 2011. Ecological periodic tables for benthic macrofaunal usage of estuarine habitats in the US Pacific Northwest. Estuarine, Coast and Shelf Science 94:36-47.

Ferraro, S.P. and F.A. Cole. 2012. Ecological period tables for benthic macrofaunal usage of estuarine habitats: Insights from a case study in Tillamook Bay, Oregon, USA. Estuarine, Coast and Shelf Science 102-103:70-83.

Gilkerson W. 2008. A spatial model of eelgrass (*Zostera marina*) habitat in Humboldt Bay, California. Master's Thesis. Humboldt State University, Arcata, California.

Harding Jr., L.W. 1973. Primary Production in Humboldt Bay. Master's Thesis. Humboldt State University, Arcata, California.

Hosack, G.R. 2003. Does habitat structure influence low intertidal communities in Willapa Bay, Washington? Presented at Pacific Estuarine Research Society (PERS) Conference Vancouver British Columbia. April 3-4, 2003.

Hosack, G.R., B.R. Dumbauld, J.L. Ruesink, and D.A. Armstrong. 2006. Habitat associations of estuarine species: comparisons of intertidal mudflat, seagrass (*Zostera marina*), and oyster (Crass*ostrea gigas*) habitats. Estuaries and Coasts 29:1150-1160.

Hsieh, C., A. Yamauchi, T. Nakazawa, and W. Wang. 2010. Fishing effects on age and spatial structures undermine population stability of fishes. Aquat. Sci. 72:165-178.

Koch, E.W. 1999. Sediment resuspension in a shallow Thalassia testadium banks ex. König bed. Aquatic Botany 65:269-280.

Mello, J.J. and J. Ramsay. 2004. Summary of 2003-2004 Pacific herring spawning-ground surveys and commercial catch in Humboldt Bay and Crescent City. California Department of Fish, Wildlife and Game.

Mello, J.J. 2007. Summary of 2006-2007 Pacific herring spawning-ground surveys and commercial catch in Humboldt Bay and Crescent City. California Department of Fish, Wildlife and Game.

NMFS (National Marine Fisheries Service). 2014. California eelgrass mitigation policy and implementing guidelines. NOAA Fisheries, West Coast Region. http://www.westcoast.fisheries.noaa.gov/publications/habitat/california_eelgrass_mitigation/Final%20CEMP%20October%202014/cemp_oct_2014_final.pdf

Peterson, B.J. and K.L. Heck, Jr. 1999. The potential for suspension feeding bivalves to increase seagrass productivity. Journal of Experimental Marine Biology and Ecology 240:37-52.

Pinnix, W.D., P.A. Nelson, G. Stutzer, K.A. Wright. 2013. Residence time and habitat use of coho salmon in Humboldt Bay, California: an acoustic telemetry study. Environ. Biol. Fish. 96:315-323.

Pinnix, W.D., T.A. Shaw, K.C. Acker, and N. J. Hetrick. 2005. Fish communities in eelgrass, oyster culture, and mudflat habitat of North Humboldt Bay, California, Final Report. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Arcata Fisheries Program Technical Report Number TR2005-02. Arcata, California.

Rabin and Barnhart. 1986. Population characteristics of Pacific herring, *Clupea harengus pallasi*, in Humboldt Bay, California. California Fish and Game 72(1):4-16.

Ruesink, J.L., J.P. Fitzpatrick, B.R. Dumbauld, S.D. Hacker, A.C. Trimble, E.L. Wagner, and L.M. Wisehart. 2012. Life history and morphological shifts in an intertidal seagrass following multiple disturbances. Journal of Experimental Marine Biology and Ecology 424-425:25-31.

Rumrill, S. 2015. Letter to Korie Schaeffer regarding eelgrass and shellfish aquaculture interactions from Humboldt Bay WRAC study. Oregon Department of Fish and Wildlife. April 5, 2015. steven.s.rumrill@state.or.us

Rumrill, S. and V. Poulton. 2004. Ecological role and potential impacts of molluscan shellfish culture in the estuarine environment of Humboldt Bay, CA. Western Regional Aquaculture Center Annual Report November 2004. 79 p.

Schlosser, S., and A. Eicher. 2012. The Humboldt Bay and Eel River Estuary Benthic Habitat Project. California Sea Grant Publication T-075. 246 p.

Shelton, A.O. T.B. Francis, G.D. Williams, B. Feist, K. Stick and P.S. Levin. 2014. Habitat limitation and spatial variation in Pacific herring egg survival. Marine Ecology Progress Series 514:231-254.

Tallis, H.M., J.L. Ruesink, B.R. Dumbauld, S.D. Hacker, and L.M. Wisehart. 2009. Oysters and aquaculture practices affect eelgrass density and productivity in a Pacific Northwest estuary. Journal of Shellfish Research 28:251-261.

Thom, R.M., H.L. Diefenderfer, J. Vavrinec, and A.B. Borde. 2012. Restoring resiliency: Case studies from Pacific Northwest estuarine eelgrass (*Zostera marina* L.) ecosystems. Estuaries and Coasts 35:78-91.

Exhibit A

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FINAL

State of the Science Assessment: Shellfish Aquaculture Interactions with Submerged Aquatic Vegetation

Prepared for:

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and

Coast Seafoods Company

April 17, 2015

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STATE OF THE SCIENCE ASSESSMENT: SHELLFISH AQUACULTURE INTERACTIONS WITH SUBMERGED AQUATIC VEGETATION

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STATE OF THE SCIENCE ASSESSMENT: SHELLFISH AQUACULTURE INTERACTIONS WITH SUBMERGED AQUATIC VEGETATION

National Oceanic and Atmospheric Administration (NOAA) Coastal Aquaculture Planning and Environmental Sustainability (CAPES) program specializes in "understanding environmental interactions of aquaculture with marine ecosystems" and is currently requesting information to conduct a state of the science assessment on shellfish aquaculture interactions with submerged aquatic vegetation (SAV). This document provides scientific and technical information related to interactions between shellfish aquaculture and SAV according to an outline developed by Dr. James Morris. Specifically, the information provided focuses on two topics:

- Shellfish Aquaculture and SAV Interactions
- Habitat Equivalency of Shellfish Aquaculture and SAV

The species of SAV discussed in this document is almost exclusively the native eelgrass, *Zostera marina*, which is the predominant native SAV in the Pacific Northwest. It should be noted that many kelp species (in the order Laminariales) are also an important SAV with positive relationships to shellfish and aquaculture gear that should be considered, although kelp are not discussed below.

The information presented supports the concept that shellfish aquaculture and eelgrass can be mutually sustainable, and many of the ecosystem functions provided by eelgrass are also provided by shellfish. It is fully recognized that there are important management considerations related to shellfish aquaculture practices aimed at reducing potential negative interactions. However, based on a growing body of evidence, it is an obtainable goal to conduct shellfish aquaculture operations in the presence of eelgrass resulting in no net loss of ecosystem functions.

Shellfish aquaculture and eelgrass have co-existed in Pacific coastal estuaries and in Puget Sound since commercial aquaculture began along the U.S. West Coast. For example, in Willapa Bay, Washington, oyster aquaculture has been part of the mosaic of habitats for over 100 years, and eelgrass is abundantly present in areas where aquaculture historically occurred and continues to occur (Figure 1). The fact that aquaculture activities can have both positive and negative interactions with eelgrass at a unit scale is suggestive of looking at effects from a broader perspective, such as the landscape or watershed scale, in order to understand the overall effects of shellfish aquaculture in relation to ecosystem health. This would allow the resource to be managed in terms of the overall ecological function of a region, estuary, bay, or watershed.

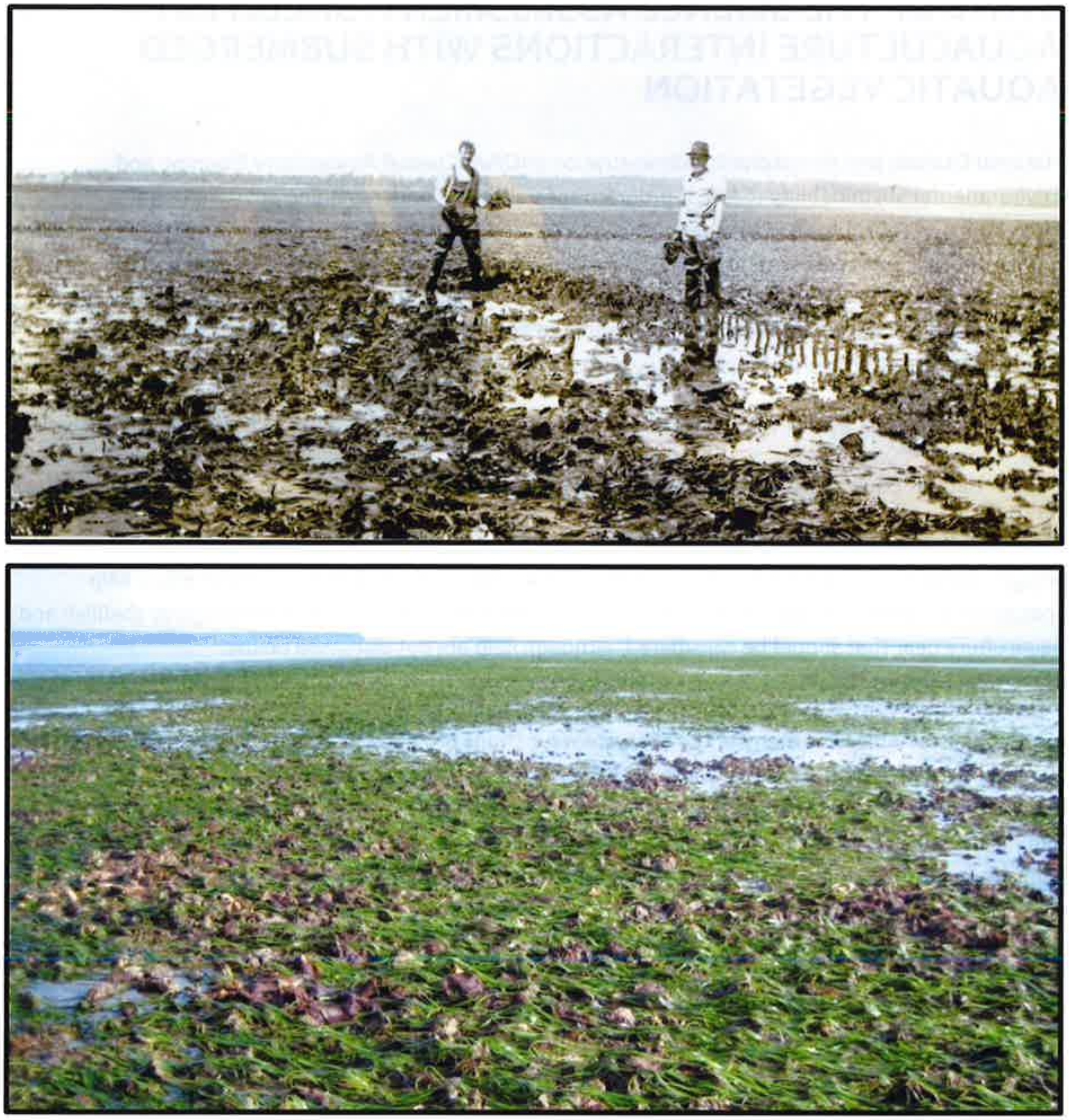

Figure 1 Historical and Current Pacific Oyster (*Crassostrea gigas*) Bottom Culture in Dense Native Eelgrass (*Zostera marina*) in Willapa Bay, Washington

Notes: Top Photograph: early/mid 1900 picture acquired from Washington State History Museum of a Pacific oyster bottom culture bed in Willapa Bay in dense eelgrass; Bottom Photograph: 2009 picture of Pacific oyster bottom culture in dense eelgrass in Willapa Bay.

Source: Dewey (pers. comm., 2015)

One stated focus of CAPES is developing science-based decision tools for planning, scoping, authorizing, and mitigating habitat use to allow for aquaculture development. The overall goal of this document is to provide relevant scientific information and a conceptual framework for evaluating aquaculture projects proposed for new areas in order to determine net effects to ecosystem functions. Concepts from habitat equivalency analyses (HEA) provide a good framework upon which to build. A primary assumption used in HEA is that comparable ecosystem functions can be provided by created habitat for areas of lost or displaced habitat. This concept is applicable for eelgrass and shellfish aquaculture. It does not presume a 1:1 relationship between eelgrass and shellfish aquaculture in terms of the ecosystem functions provided, but recognizes that displacement of eelgrass is not a complete loss of relevant ecosystem functions. Eelgrass may provide more value for some functions while shellfish and/or aquaculture gear may provide more value for other functions. Furthermore, evaluating losses and gains in ecosystem functions should take into account a watershed or regional approach to focus and prioritize which functions are most important for maintaining ecological health in any given setting (similar to a limiting factors analysis).

1.0 SHELLFISH AQUACULTURE AND SAV INTERACTIONS

Eelgrass abundance and distribution varies over time and space, and although beds are often perceived as static, the edges tend to extend and contract over time (Duarte and Sand-Jensen 1990, Robbins and Bell 2000, Gaeckle et al. 2011). The seabed is also dynamic, and sediment movement may bury plants, expose roots, or uproot plants (Kirkman and Kuo 1990, Preen et al. 1995). Biomass increases and decreases based on seasonal and annual growing conditions. In addition, eelgrass biomass may potentially follow 18.6-year tidal cycles and 8.8-year tidal modulations[1] (Jay, pers. comm., 2014). The superficial stability of eelgrass beds conceals the underlying balance between the continuous loss and replacement of shoots (Duarte 1989, Olesen and Sand-Jensen 1994).

Shellfish aquaculture operations also vary seasonally and annually in terms positive and negative interactions with the surrounding habitat. In the case of negative interactions, there are two types of disturbance that will be discussed below (Figure 2). Short-term disturbances are known in ecology as pulse disturbances because their temporary nature allows the affected biota to recover to the previous equilibrium state. Contrasted with press disturbances, which are long-term in both duration and effect, and require the system to reach a new equilibrium (Bender et al. 1984). In general, shellfish aquaculture represents pulse disturbances that are often within the natural disturbance regime of a system.

[1] Dr. David Jay at Portland State University has been analyzing tide data and found that "estuaries change over time, and tides are generally getting larger in the NE Pacific for reasons that are not well understood." There seems to be a periodicity in terms of the cycle of tidal variation (18.6 and 8.8-year cycles). We are currently in a period of less extreme lowest seasonal tides. This means that eelgrass may be at historical highs in certain areas because there is shorter exposure time at the lowest tide levels. Dr. Jay indicated that tides may be maximal about 2024 or 2025 and then will start to decline again.

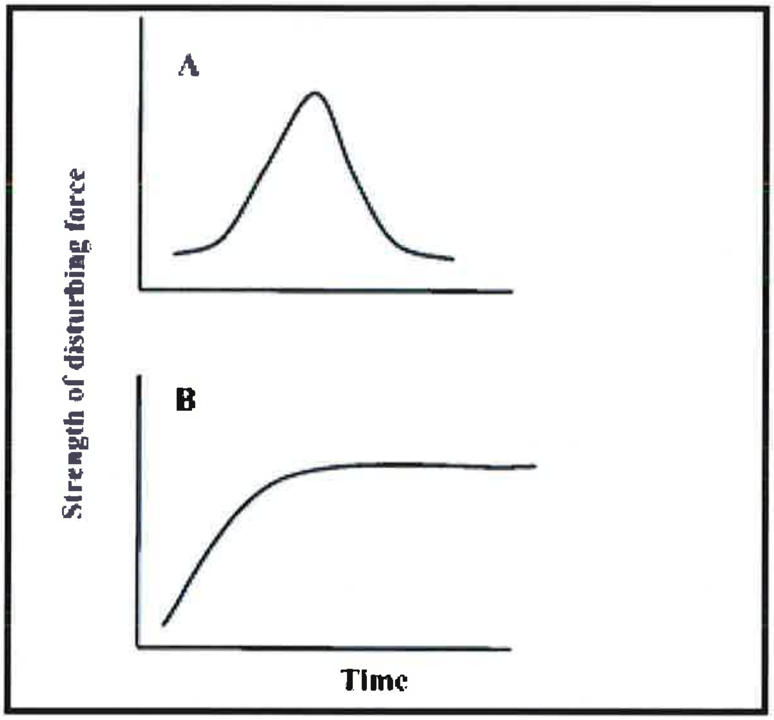

Figure 2 Two Types of Ecological Disturbance: Pulse and Press

Notes: A. Pulse Disturbance; B. Press Disturbance

Source: modified from Lake (2000)

At the present scale of operations, shellfish aquaculture along the West Coast and in Puget Sound appears to be more sustainable than most other human activities occurring in estuaries (e.g., shoreline development), which can degrade and eliminate estuarine function (Dumbauld et al. 2009, Coen et al. 2011). Management decisions for the regulation of shellfish aquaculture should, therefore, consider both temporal and spatial scales. In addition, the functional value of a mosaic of habitats, including shellfish beds with edges and corridors, should be considered at the landscape scale in terms of the potential to add ecosystem function. According to Coen et al. (2011), this concept of a habitat mosaic "may be an area where innovative practices and best management practices (BMPs) developed by growers in association with scientists can be applied to conserve and even enhance the functional value" of estuarine habitats.

1.1 *Negative Interactions (Pulse and Press Disturbance)*

Disturbance by shellfish aquaculture to eelgrass varies from simple space competition (or press disturbance if gear/shellfish remains in the same location long-term) to removal of entire plants and rhizomes via harvest (or pulse disturbance). However, disturbance does not necessarily equate to an impact. There are three interactions described below that can potentially result in a direct loss of eelgrass biomass, including: (1) shellfish aquaculture gear, (2) working practices, and (3) cumulative effects with other stressors. In addition, a concept that is not explored in the outline by Dr. Morris is the potential for consistent disturbances within the natural variability of the system to potentially result in increased resiliency of eelgrass.

1.1.1 Shellfish Aquaculture Gear

Because eelgrass requires light for photosynthesis, light availability is considered to be one of the most influential factors on its distribution and health (Dennison 1987, Zimmerman et al. 2001, Thom et al. 2003, Thom et al. 2011). Equipment associated with the culture of shellfish (e.g., nets, racks, bags) can lead to shading, which may affect the spatial extent and density of eelgrass beds in the immediate vicinity of the shellfish gear. The type and concentration of equipment can influence the level of this effect. For example, Everett et al. (1995) found that oyster racks could lead to a total loss of shoots directly under the racks. Comparatively, Rumrill and Poulton (2004) determined that the spatial extent of an eelgrass bed and shoot density were negatively influenced when oyster longline culture was closely spaced (1.5 ft to 2.5 ft) but showed no effect compared to control sites when spacing occurred at 5-ft and 10-ft spaces between longlines.

Predator exclusion netting used in Manila clam and geoduck clam aquaculture can be detrimental to eelgrass if it is placed directly on top of eelgrass. Predator exclusion netting is generally not placed over shellfish in areas with eelgrass. However, in at least one example in Washington State (Fisk Bar, Samish Bay), eelgrass recruited into a geoduck clam culture area after the placement of aquaculture gear stabilized the sediment and protected the new eelgrass shoots from erosive wave energy (see Section 1.2.1 below). Later in the culture cycle, predator exclusion nets were placed on the bed and Horwith (2013) reported a total loss of eelgrass under predator exclusion netting due to the shading effects from *Ulva* sp. growth on the net surface. It should be noted that the eelgrass was shown to recover within two years of net removal (Horwith, pers. comm., 2014). Under such a scenario, the functions of eelgrass may have been temporarily offset by the development of a macroalgal community, as described by Powers et al. (2007). Predator exclusion nets or culture gear often gets colonized with macroalgae (Figure 3), which was quantified by Powers et al. (2007) as providing comparable ecosystem functions (e.g., nursery habitat, epibiota biomass) as seagrass[2] beds.

Other effects of shellfish aquaculture gear to eelgrass include the potential to abrade, scour, or desiccate eelgrass blades, although the overall effects to the eelgrass bed can be both positive and negative. For example, Tallis et al. (2009) explored the concept that, while shellfish can break eelgrass blades through abrasion, the reduction in density can release individuals from intraspecific light competition and result in increased growth rates near the aquaculture plots. Similarly, scouring or dredging can result in higher eelgrass seedling density and seed production in the disturbed areas (Wisehart et al. 2007). The creation of "edge" habitat can increase productivity as long as the operation does not result in a disruption of the connectivity between habitats. Alternatively, eelgrass blades can

[2] Note that eelgrass is a type of seagrass. and the two terms will be used throughout this document.

desiccate on shellfish or aquaculture gear, which can eventually lead to a reduction in shoot size (Wisehart et al. 2007, Tallis et al. 2009). These tradeoffs are explored in more detail in Section 2.0.

Figure 3 Macroalgae Colonizing Oyster Lines in Blue Heron Bay

Source: Dewey, pers. comm., 2015

1.1.2 Working Practices

Working practices during harvesting and general access of shellfish plots can lead to pulse disturbances to eelgrass beds. These concepts are detailed below.

Harvesting

Shellfish harvest can cause localized and temporary increases in suspended sediments, physical damage and/or removal of eelgrass shoots, as well as changes to other eelgrass metrics (e.g., biomass, seed germination, growth). The two practices that likely generate the most suspended sediment are mechanical harvest of oysters and geoduck clam harvest. Shellfish aquaculture typically occurs in areas with sand or gravel substrate, which are substrate sizes that have high settling velocities. For example, Mercaldo-Allen and Goldberg (2011) reported that suspended sediments may take 30 minutes to 24 hours to resettle in areas typical of oyster and clam aquaculture operations. Suspended sediment effects of shellfish harvest are generally short-lived and recovery is rapid (Short and Walton 1992, Liu and Pearce 2011). Additionally, shellfish culture often occurs in shallow estuarine embayments where freshwater runoff, currents, and wind waves can lead to naturally high background levels of suspended sediments. Therefore, pulse disturbances of suspended sediment by shellfish harvesting often fall within baseline measurements and the natural variability of the system.

Tallis et al (2009) compared eelgrass densities in areas that were harvested by hand, using a dredge, or on longlines. Although there was a 70% reduction in eelgrass productivity at all aquaculture sites when averaged together, Tallis et al. (2009) pointed out that effects to eelgrass from aquaculture occurred in both directions (positive and negative), and a magnitude of effects observed were dependent on the site and type of harvest methods used. While the authors suggested that it is impossible to incur no effect when oyster aquaculture takes place in an eelgrass bed, they also indicated that there are ample opportunities for decreased impact with tailored culture methods and timing.

Wisehart et al. (2007) also examined the effects of different aquaculture techniques on eelgrass biomass, density, and growth rates in Willapa Bay. As discussed above, the authors reported that shellfish aquaculture may facilitate increased growth rates due to a reduction in intraspecific competition by surrounding plants (e.g., increased light availability), increased seed supply and germination, and a more open seed dispersal setting. In addition, while oysters grown on longlines caused some minor reduction in eelgrass density and cover, the highest eelgrass growth rates occurred at the longline culture and reference sites. These areas also had the greatest eelgrass biomass, density, and percent cover. The study reported statistically significant site and culture type interactions for most variables, suggesting that site-specific conditions may be just as influential as aquaculture techniques in determining eelgrass parameters.

General Access

The two potential types of disturbance to eelgrass from general access include trampling and propeller scarring. Depending on site conditions, access to shellfish aquaculture plots sometimes requires that growers walk on eelgrass beds or ground their skiffs at low tide in areas with eelgrass. Based on the low tide cycle, crews typically work for a period of 4 to 8 hours in one area for about 1 week out of every several months. Therefore, these disturbance events would be considered infrequent and of short duration relative to the time that the beds remain submerged and undisturbed. Nonetheless, eelgrass rhizomes may be uprooted or shoots broken from trampling or moving around gear.

Boat access can also result in potential negative impacts to eelgrass shoots. Ruesink et al. (2012) conducted experimental treatments in Willapa Bay where they imposed two disturbance types: shoot damage and shoot removal. For the most part, the extent of damage from boat propellers would be taking off the ends of the shoots (i.e., shoot damage), but not removing the entire shoot. Regrowth for eelgrass that is only damaged on the surface requires branching of the plant to replace the lost biomass. Growth rates of eelgrass affected by shoot damage for less than 4 weeks recovered within 2 months following a single cutting event when the rhizome was still rooted. There would be no long-term damage in terms of eelgrass density for this type of action.

Potential longer term impacts were calculated based on an accumulation of shoot removal over a year or more (e.g., consistent access routes). If the shoot is removed, the removal area can be repopulated by rhizome extension from shoots at the edge (asexual reproduction) or germination of seeds (sexual reproduction). Ruesink et al. (2012) reported that recovery of eelgrass after complete shoot removal (6.6 x 6.6 ft gaps) could occur after 2 years. Based on this rate of recovery, a conservative estimate of a propeller scar width of 3 ft could replace the lost biomass in approximately 0.9 years. If regrowth occurs at a rate faster than removal, it can be assumed that there would be no significant loss in biomass from this type of activity unless an area is not allowed to regrow through continuous disturbance.

1.1.3 Cumulative Effects with Other Stressors

Boström et al. (2006) conducted a meta-analysis of the scientific literature associated with plant-animal interactions in seagrass landscapes. According to this analysis, the authors indicated that, "The growth and recruitment dynamics of seagrasses as well as man-made and/or natural disturbances create complex spatial configurations of seagrass over broad (metres to kilometres) spatial scales. Hence, it is important to identify mechanisms maintaining and/or threatening the diversity-promoting function of seagrass meadows and to understand their effects on benthic populations and communities." It is well recognized that there are a variety of natural and anthropogenic stressors on eelgrass (Dennison 1987, Fonseca and Bell 1998, Shaughnessy et al. 2004, Boese et al. 2005, Mumford 2007, Thom et al. 2011, Stevens and Lacy 2012). What is not well understood is the extent to which shellfish aquaculture contributes to the overall understanding of cumulative effects.

There are a few studies that address landscape scale changes to eelgrass relative to shellfish aquaculture. The most comprehensive analysis of factors that drive the changes observed in eelgrass density was conducted by Dumbauld and McCoy (*in press*). This study modeled eelgrass (*Z. marina*) density in Willapa Bay and compared a number of predictors, including: (1) distance to estuary mouth, (2) distance to channel, (3) salinity, (4) elevation, (5) cumulative wave stress, and (6) shellfish aquaculture. The model results indicated that eelgrass cover was lower in oyster aquaculture beds, but the impact directly associated with aquaculture represented less than 1.5% of the total predicted eelgrass density in any year.

Aside from the overall low amount of impact to eelgrass at the landscape scale, the Dumbauld and McCoy (*in press*) work also indicated that the harvest method was a significant predictor in explaining eelgrass loss. For example, mechanically harvested beds had a significantly lower amount of eelgrass compared to beds harvested by hand or with a mixed harvest technique (similar to the results reported by Tallis et al. 2009). Comparatively, the type of aquaculture (e.g., longline, seed bed, fattening ground) was not a significant contributor to the variation of eelgrass predicted versus actually observed. The authors suggested that, overall, aquaculture resulted in a minor change to eelgrass at the landscape scale because the effect of culture was variable enough at smaller spatial scales as to eliminate a significant effect at the landscape scale.

As stated throughout this report, the landscape scale is very important to consider when trying to protect for mobile species, such as fish and crabs. This sentiment was stated within Semmens (2008), where the author indicated that, "From a management perspective, it may therefore be tempting to downplay the importance of fine-scale benthic habitats in favor of larger-scale estuarine features such as deep tidal channels and salinity gradients for smolt-sized fish." In other words, although it is important to understand small-scale effects in order to effectively manage potential effects to eelgrass, it is the landscape scale that drives how species will use the habitat.

1.1.4 Eelgrass Resilience

Holling (1973) defined resilience as "a measure of the ability of these systems to absorb change of state variables, driving variables, and parameters, and still persist." Native eelgrass exhibits a stable and possibly increasing trend in distribution and abundance in areas like Willapa and Humboldt bays where oysters have been actively farmed for over 100 years and are currently operated by commercial growers (Barrett 1963, Tallis et al. 2009, Dumbauld et al. 2011). The mechanisms of eelgrass resilience related to shellfish aquaculture are not entirely understood. However, processes that interact to create resiliency depends on both the local habitats and landscape conditions (Thom et al. 2012, Allen et al. 2014). The fact that studies have produced mixed results in terms of the scale of effects and eelgrass recovery after a disturbance from shellfish aquaculture implies that there are conditions at multiple levels that are important to consider (Ruesink et al. 2012). One topic for further research is whether shellfish aquaculture increases sexual reproduction (e.g., flowering, seed production, and germination), which can be a factor in faster recovery rates of eelgrass following a disturbance event (Ruesink et al. 2012, Thom et al. 2012). Another important question is whether the scale of shellfish aquaculture and culture methods affect the resilience of eelgrass, and if these effects are more important than limiting factors at the landscape scale (e.g., water quality conditions).

1.2 Positive Interactions

Many effects of shellfish aquaculture can be considered a positive interaction with eelgrass. According to Forrest et al. (2009), "the acceptability of aquaculture operations or new developments should recognize the full range of effects, since adverse impacts may be compensated to some extent by the nominally 'positive' effects of cultivation." There are five effects described below that can potentially result in a direct gain of eelgrass biomass, including: (1) sediment stabilization and eelgrass colonization, (2) increased water clarity and light penetration, (3) sediment enrichment or fertilization, (4) eelgrass recruitment, germination, and seedling retention, and (5) reduced desiccation. In addition, there is the ancillary benefit of having an industry present in the ecosystem with the incentive to maintain high water quality conditions. These positive interactions are described below in more detail.

1.2.1 Sediment Stabilization and Eelgrass Colonization

Shellfish have been labeled "ecosystem engineers" because of the ecological roles that they play in coastal habitat processes (Jones et al. 1994, Lenihan 1999). For example, the presence of shellfish can protect shorelines from erosion by stabilizing sediments and dampening waves (Meyer et al. 1997, Scyphers et al. 2011, Spalding et al. 2014). This same function provided by shellfish can benefit eelgrass. Eelgrass has been known to expand into areas after sediments are stabilized. There are numerous examples along the West Coast where eelgrass expanded into shellfish aquaculture plots. The information for each of these areas is primarily anecdotal with notable exceptions (e.g., Ward et al. 2003), and the cause has not been directly linked to the aquaculture operation. However, shellfish aquaculture was not associated with detectable loss in the spatial extent of eelgrass. On the contrary, there was an apparent increase in eelgrass coincident with an expansion (or at the very least consistent presence) of shellfish aquaculture operations.

A case study that highlights the potential for eelgrass to colonize into an area previously devoid of eelgrass is for a sand bar (Fisk Bar) in the center of Samish Bay, Washington. Prior to geoduck aquaculture on Fisk Bar, seeds from the surrounding eelgrass beds would occasionally result in ephemeral shoots on the sand bar that would get eroded during winter storms. In 2002, geoduck nursery tubes (6-inch diameter polyvinyl chloride tubes) for planting geoduck seed were placed to establish the first geoduck crop on Fisk Bar. Shortly after nursery tubes were placed, eelgrass began to fill in and establish a dense bed around the tubes (Dewey, pers. comm., 2015). For this first crop of geoduck, individual net caps were placed on each tube. When the tubes were removed 2 years after seeding, eelgrass was well enough established that it remained and thrived on the sand bar. In 2008, when the geoducks were harvested, eelgrass was significantly reduced but not eliminated. After this first harvest event, nursery tubes were reinstalled, seeded with geoducks, and the entire tube field was covered by predator exclusion nets. Horwith (2013) reported a total loss of eelgrass in areas where the predator exclusion netting was placed. The eelgrass loss was attributed to shading effects from *Ulva* sp. growth on the net surface. Recovery of eelgrass began one year after removal of tubes and nets. In July 2014, Dr. Horwith (pers. comm., 2014) indicated that, "there is no longer any significant difference in eelgrass coverage or density between the farmed and unfarmed areas." Figure 4 provides an overview of the geoduck culture cycle at Samish Bay between 2002 and 2014. Overall, the Fisk Bar eelgrass appears to be resilient and thriving in an area where it could not previously establish.

Fisk Bar, Samish Bay, 2002: Planting Geoduck Seed with Culture Tubes and Individual Net Caps. No eelgrass bed present prior to planting activities.

Fisk Bar, Samish Bay, 2014: Eelgrass Bed prior to Harvest Activities

Figure 4 **Eelgrass at Fisk Bar, Samish Bay, between 2002 and 2014**

Source: Dewey (pers. comm., 2015)

1.2.2 Increased Water Clarity and Light Penetration

Shellfish aquaculture can result in a reduction in turbidity due to removal of phytoplankton and particulate organic matter through filtration (Peterson and Heck 2001, Newell and Koch 2004, Cranford et al. 2011). By consuming phytoplankton and particulate organic matter, shellfish increase the amount of light reaching the sediment surface that is available for photosynthesis (Dame et al. 1984, Koch and Beer 1996, Newell 2004, Newell and Koch 2004). Improvements to water clarity and light penetration can improve habitat conditions that promote the growth of eelgrass.

The removal of nutrients (especially nitrogen) through filtration can also benefit eelgrass growth by reducing epiphytes and macroalgae (Figure 5). Epiphytes (primarily diatoms) can form thick layers on eelgrass blades. This is a natural processes, and important in the food chain because this layer of epiphytes is grazed by aquatic invertebrates (van Montfrans et al. 1984, Nelson and Waaland 1997). However, overproduction of epiphytes is a result of nutrient water column pollution (Williams and Ruckelshaus 1993, Hauxwell et al. 2001, Nielsen et al. 2004). Shellfish aquaculture can provide mitigation of these conditions due to water filtration and control of nutrients that promote the growth of epiphytes.

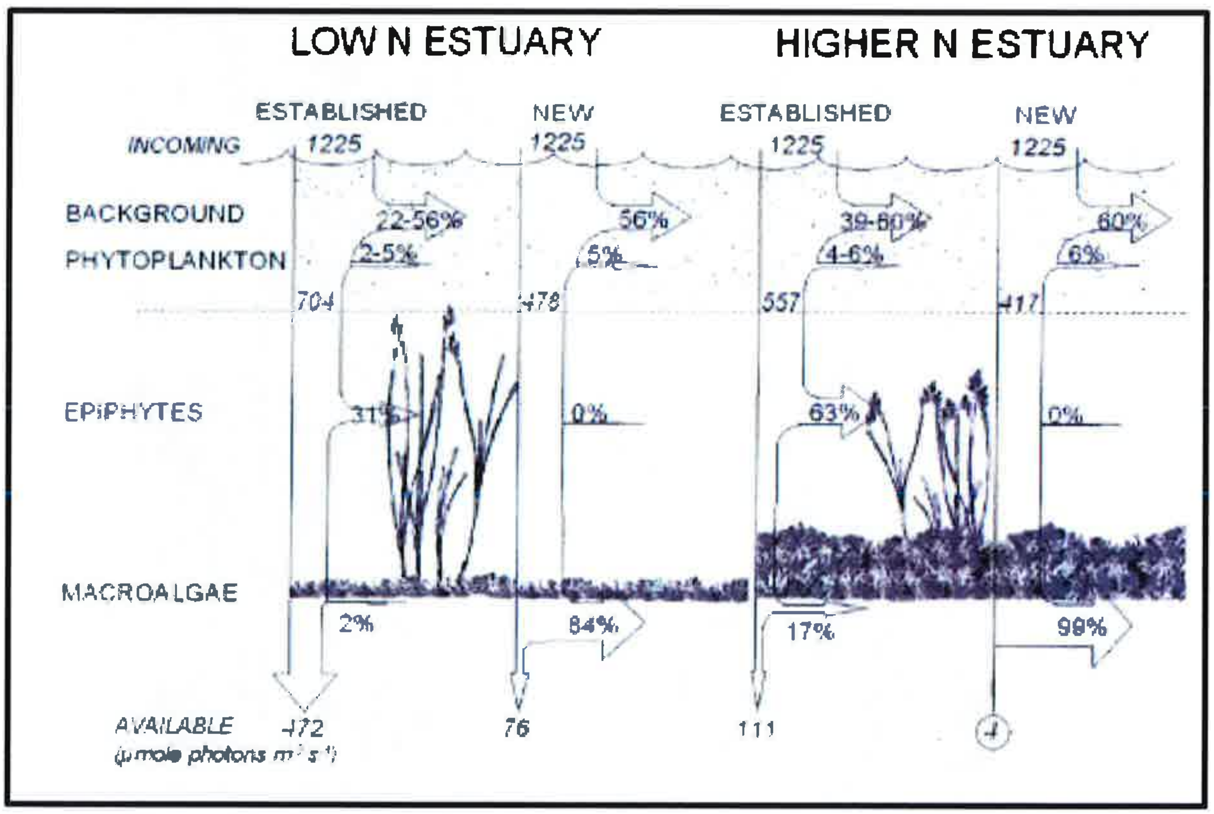

Figure 5 **Illustration of Mean Summer Light Intensity Effects to Eelgrass**

Source: Hauxwell et al. (2001)

Another service potentially provided by shellfish related to epiphytes was explored by Peterson and Heck (2001). The authors observed a significantly reduced epiphytic load on seagrass leaves when mussels were present. Spaces between shells of adjacent mussels were thought to provide a predation refuge for epiphytic grazers (e.g., small gastropods and amphipods). Increased densities of epiphytic grazers could then lead to an increased amount of grazing, which consequently might lead to an increase in leaf light absorption. This study also noted that the mussels themselves may potentially reduce epiphytic loads by consuming the epiphyte propagules before recruitment to the leaves. Although likely a benefit to eelgrass, the shellfish would need to be in the eelgrass bed to provide this service for epiphytic grazers.

Macroalgae does not colonize eelgrass shoots in the way that epiphytes do, but changes in the amounts of nutrients in the aquatic environment can shift the competitive balance between aquatic vegetation species, allowing plants that can respond quickly to nutrients to dominate (Taylor et al. 1995, Schramm and Nienhuis 1996, Taylor et al. 2001, Cardoso et al. 2004, Nielsen et al. 2004, Mumford 2007, Smetacek and Zingone 2013). The mechanism for loss of eelgrass is likely related to a combination of light competition (Nienhuis 1996), smothering by macroalgal blooms (den Hartog and Phillips 2000), and competition for nutrients (Nienhuis 1996).

1.2.3 Sediment Enrichment

Bivalve filter-feeding also serves an important role in improving water quality conditions through benthic-pelagic coupling, which is the consumption of nutrients (via filtration of phytoplankton) and creation of biodeposits (feces and pseudofeces). Nitrogen and phosphorous that are not digested and incorporated into tissue are processed through the bivalves and excreted as soluble ammonia and biodeposits. When these biodeposits become incorporated into aerobic surficial sediments, microbially-mediated processes facilitate nitrification-denitrification coupling to permanently remove sediment-associated nitrogen as nitrogen gas (Newell 2004, Kellogg et al. 2013). According to Newell et al. (2005), "the species of bivalves that can exert the greatest influence on benthic-pelagic coupling are those, such as oysters and mussels, which maintain high clearance rates and reject relatively large amounts of POM [particulate organic matter] as pseudofeces."

Peterson and Heck (1999) suggested that by increasing sediment nutrient levels, shellfish may create new habitat areas for colonization of seagrasses, or maintain sufficient nutrient levels for the continued existence of seagrasses in stressful environments. Eelgrass can derive nutrients from both the sediments and the water column. The interstitial water (or sediment porewater) contains relatively higher concentrations of dissolved inorganic and organic nutrients than the water column, and eelgrass obtains most macronutrients from sediments. Sediment reservoirs of nutrients can become depleted when biogeochemical regeneration rates cannot meet plant demands (Short 1983, 1987). However, in the course of removing water column particulates, shellfish also alter sediment characteristics. Positive impacts occur because the shellfish move carbon and nutrients from the water column to the benthos. Although studies related to sediment "fertilization" from bivalve deposition have shown enhanced eelgrass growth along the East Coast (e.g., Peterson and Heck 1999), similar studies in the Pacific

Northwest appear to show no effect on eelgrass growth (Wagner et al. 2012, Ruesink and Rowell 2012, Wheat and Ruesink 2013). Studies in the Pacific Northwest indicate that eelgrass is not generally nutrient limited or that sediment porewater nutrients are naturally high.

1.2.4 Eelgrass Recruitment, Germination, and Seedling Retention

Positive effects of shellfish to eelgrass recruitment, germination, and seedling stages are likely to be driven by three main mechanisms. First, by providing a larger boundary layer and slowing water current speed, shellfish may increase recruitment of floating seeds as they travel singly or within detached reproductive shoots. Retention of seedlings could also be facilitated by the structure shellfish provide. Seed dispersal is typically limited outside of an eelgrass bed; approximately 80% of seeds travel within 10 m of parent plants (Orth et al. 1994, Ruckelshaus 1996). Therefore, this effect is only important when eelgrass beds are nearby. Second, by filtering seawater and increasing sediment organic content, bivalves provide superior conditions for seed germination. Eelgrass seed germination is dependent on burial depth, with the highest germination occurring at the anaerobic/aerobic interface (Bigley 1981). Seeds buried below this depth have very low germination and are essentially lost from the population. Shellfish may act to bury and fertilize seeds at a depth that is appropriate for germination. Third, shellfish may increase the survival of seedlings, which have very high mortality rates by increasing light levels, nutrients, and protecting against erosion and herbivory (Orth et al. 1994, Ruckelshaus 1996).

1.2.5 Reduced Desiccation

As the tide recedes, shellfish retain seawater as they shut down filter feeding to wait for the returning tide. The water that is retained in the mantle cavity is expelled prior to the tide returning, creating a spray of water that is released into the surrounding environment. One of the species that can expel a significant amount of water is the geoduck clam (*Panopea generosa*). Figure 6 shows geoduck clams at Fisk Bar expelling water onto the surrounding eelgrass habitat. Water retention and release from other shellfish species may act in a similar fashion. Although this is likely a minor ecosystem function, it potentially reduces desiccation pressure when eelgrass is exposed during a low tide.

1.2.6 Ancillary Benefits

In addition to direct beneficial interactions between shellfish aquaculture and eelgrass, the presence of aquaculture within an embayment or watershed may provide indirect benefits to SAV through a variety of mechanisms. The aquaculture industry is inherently reliant on the maintenance of good water quality conditions to ensure the safety and survival of their product. Because of this incentive, there are numerous examples of actions taken by aquaculture companies and their supporters that result in improvements to water quality and/or the prevention of anthropogenic activities threatening water quality and habitat function in areas were aquaculture occurs (Dewey et al. 2011), as described below.

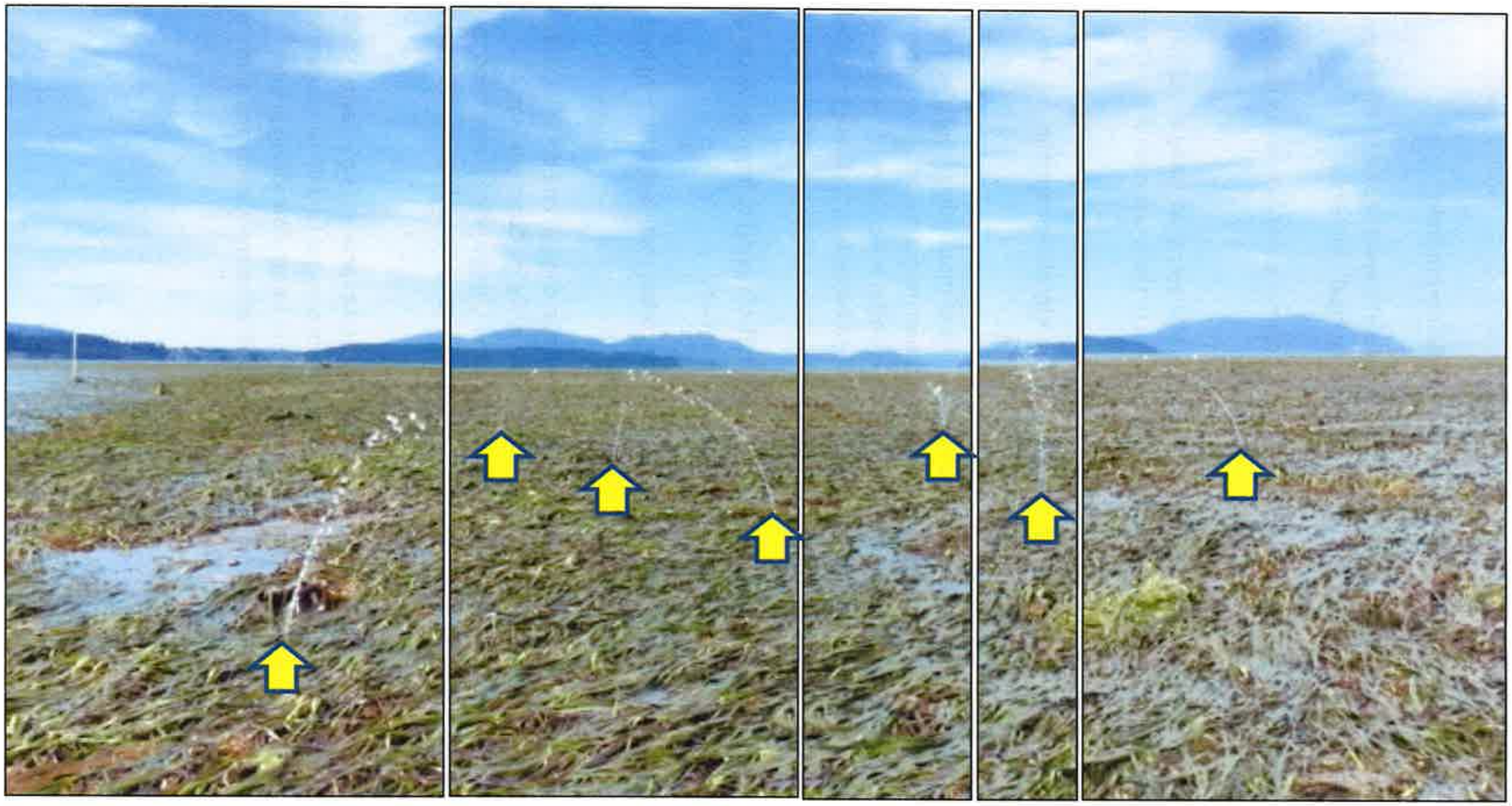

Figure 6 **Potential Ecosystem Service (Reduced Desiccation) Provided by Geoduck Aquaculture to Native Eelgrass**

Notes: Arrows indicate geoduck expelling water onto the adjacent eelgrass bed.

Source: Dewey (pers. comm., 2015)

Examples of some ancillary benefits of the shellfish aquaculture industry include:

- Working with local jurisdictions and regulators to identify and eliminate point and non-point source pollution, including agricultural, industrial, and municipal discharges.
- Participating and providing input on regulatory updates to ensure that high water quality standards are included in local, state, and federal policies.
- Lobbying state and federal legislatures for improvements to water quality and developing water quality standards (e.g., shellfish industry contribution to the enactment of the Clean Water Act in 1972).
- Maintaining ownership or leases of large aquatic areas, thereby eliminating the risk of development or other environmentally deleterious uses.
- Participating in and collecting water quality samples as part of monitoring programs with federal and state agencies (e.g., National Shellfish Sanitation Program) to track water quality trends and identify areas targeted for improvement. These efforts have directly resulted in numerous areas now being determined suitable for shellfish harvesting and have provided data for other target areas with opportunities for improvement.
- Donating to local and state organizations (e.g., Humboldt Baykeeper) to improve water quality conditions within the estuaries that shellfish aquaculture occurs.
- Organizing and participating in beach cleanup events that collect marine debris from both shoreline development and shellfish aquaculture operations.
- Actively engaging in efforts to quickly remediate and clean up oil spills and other hazardous waste sites to protect water quality and the health of shellfish (e.g., Coast Seafood's partnership with the Humboldt Bay Harbor, Recreation, and Conservation District and the Environmental Protection Agency to remediate and remove toxic hazardous wastes from a former pulp mill site).
- Encouraging shellfish gardening through sponsored seed and gear sales (e.g., Taylor Shellfish annual events in Washington State). Shellfish gardening encourages land owners to learn about the importance of maintaining properly functioning septic systems, controlling pet and domestic animal wastes, and the fate of herbicides and pesticides from lawns and gardens.

1.3 Summary of Shellfish Aquaculture and SAV Interactions (Net Impact Assessment)

A net impact assessment assumes that development will result in a change, which may be ecologically positive (e.g., improvement in water quality) or negative (e.g., reduction in aquatic habitat for fish), but adequate measures are used to mitigate negative effects such that the post-development conditions are at least no worse overall than the pre-development conditions. Dumbauld et al. (2009) indicated that it is important to not only consider disturbance in terms of a degradation from baseline functions, but also how disturbance can influence the resilience of the system to withstand or recover from additional disturbances.

As discussed above, shellfish aquaculture can have both positive and negative interactions with eelgrass habitat. Dr. Steve Rumrill (2011) produced a simple conceptual model associated with these interactions (Figure 7). There are also numerous efforts in Washington State to develop an understanding of how shellfish aquaculture may be influencing the ecological carrying capacity of the system using the Farm Aquaculture Resource Management (FARM) model (e.g., PSI 2015). Ferreira et al. (2011) describes the applicability of the FARM model and other models in understanding the role of shellfish farms in providing ecosystem services. These models provide a way to "assess sustainability and trade-offs in the context of marine spatial planning" (Ferreira et al. 2011).

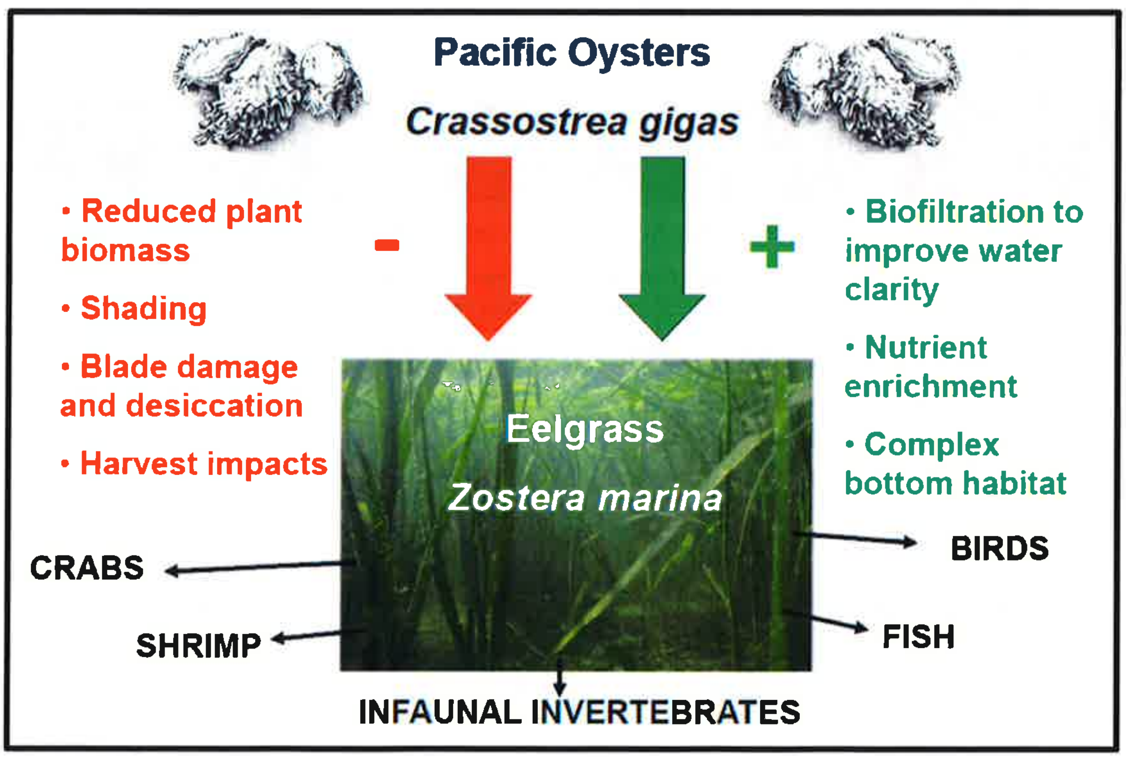

Figure 7 Biotic Interactions between Pacific Oysters and Eelgrass

Source: Rumrill (2011)

2.0 HABITAT EQUIVALENCY OF SHELLFISH AQUACULTURE AND SAV

It is widely recognized that both shellfish and SAV form unique, productive, and highly diverse ecosystems that provide physical structure and ecological functions beyond found within uniform substrates such as bare sand or mud. Furthermore, shellfish and eelgrass areas are typically distributed in units described as "beds," "meadows," "patches," or "reefs," indicating that their presence is discontinuous and contributes to the heterogeneity and ecological complexity of marine ecosystems. The following section describes some of the most prominent ecological functions provided by shellfish aquaculture and eelgrass, and discusses the similarities and differences of the ecological functions provided by each.

2.1 *The Case for Habitat Equivalency*

Eelgrass is an ecosystem engineer that provides a wealth of ecosystem functions. Although shellfish aquaculture does not necessarily provide identical functions to eelgrass, there are a wide variety of ecosystem functions provided by aquaculture that are comparable to eelgrass (Table 1). In addition, there are some functions that shellfish aquaculture provides that eelgrass does not necessarily provide as well as functions provided by eelgrass that are not provided by shellfish aquaculture, depending on density of eelgrass and life history of the organisms utilizing the habitat.

Table 1 Ecosystem Functions and Values Provided by Eelgrass and Shellfish Aquaculture

Category	Function	Value	Provided by	
			Native Eelgrass	Shellfish Aquaculture
Water Quality	Nutrient and contaminant filtration	Improve water quality and support fisheries	X	X
	Sediment filtration and trapping	Improve water quality, counter sea level rise, support fisheries, and expand eelgrass growth	X	X
	Nutrient regeneration and recycling	Support primary production and fisheries	X	X
	Oxygen production	Improve water quality and support fisheries	X	
	Remove nitrogen from the system	Improve water quality and support fisheries	X*	X
	Enhanced benthic-pelagic coupling	Improve nutrient cycling, improve water quality, and support secondary production and fisheries	X	X
	Buffering capacity	Improve pH conditions and support shellfish production	X	
Habitat Structure**	Canopy and three-dimensional structure	Habitat, refuge, nursery, settlement, and support fisheries	X	X
	Mosaic of habitat and edge effect	Habitat, refuge, nursery, settlement, and support fisheries	X	X
	Consistent presence	Provides habitat year-round to support fisheries and wildlife		X

Category	Function	Value	Provided by	
			Native Eelgrass	Shellfish Aquaculture
Circulation and Energy	Wave and current energy dampening	Prevents erosion/ resuspension and increases sedimentation	X	X
	Stabilize sediments/ protect from erosion	Support eelgrass growth/expansion and fisheries	X	X
Prey Resources	Primary production	Food for herbivores and support fisheries and wildlife	X	
	Epibenthic and benthic production	Support of food web and fisheries	X	X
	Epiphyte and epifaunal substratum	Support of secondary production and fisheries	X	X
Species Utilization	Self-sustaining ecosystem	Recreation, education, and landscape level biodiversity	X	X
Other Ecological Functions	Bioturbation/ fertilization of sediment	Support eelgrass growth/expansion and fisheries		X
	Organic production and export	Support of estuarine offshore food webs and fisheries	X	
	Organic matter accumulation	Support of food webs and counter sea level rise	X	X
	Seed production/ vegetative expansion	Self-maintenance of habitat and support of wildlife	X	

**Although eelgrass sequesters nitrogen, a harvest is one of the few ways to permanently remove nitrogen from the system.*
***Eelgrass can be dramatically reduced in the winter, so can be more limited in terms of what it provides for structure during this time-period. Density is also a consideration, and all eelgrass and shellfish aquaculture does not provide the same level of ecosystem functions (discussed in Section 2.5).*

General sources: Eelgrass functions/value = Short et al. (2000); Shellfish aquaculture functions/value = Shumway (2011)

Notes: (1) Actual value provided may vary. Additional literature and discussion is provided within the text.
(2) Highlighted cells represent ecosystem functions unique to either native eelgrass or shellfish aquaculture.

2.2 *Ecosystem Functions Not Provided by Shellfish Aquaculture*

The main ecosystem functions provided by eelgrass that are not provided by shellfish aquaculture in the same way are related to support of the food web and buffering capacity for ocean acidification (Table 1). These ecosystem functions are summarized below.

2.2.1 Support of the Food Web

Eelgrass is a common perennial aquatic plant that creates three-dimensional habitat structure and forms extensive intertidal and subtidal beds in estuaries and coastal areas. Eelgrass beds are an important component of coastal ecosystems because they stabilize coastal sediments, provide direct and indirect food sources for marine species, and provide nursery habitat for many marine species (e.g., Phillips 1984, Short et al. 2000). Because eelgrass is an autotroph, which means that it produces complex organic compounds (e.g., carbon-based solid) from simple substances present in its surroundings (e.g., light and nutrients), many ecosystem functions of eelgrass cannot be replaced by higher trophic organisms, such as shellfish. That said, many of the cultured shellfish crops, with the exception of triploid oysters, spawn several times before reaching harvest size. These events produce billions of planktonic larvae, the vast majority of which are consumed by predators.

Primary producers provide the foundation upon which the rest of the ecosystem consumer species rely, and eelgrass beds may rank among the most productive ecosystems in the ocean (Thayer et al 1975). All other consumers derive their energy from primary producers, either directly (herbivory) or indirectly (predation). However, as noted in the Boström et al. (2006) meta-analysis, the type and amount of ecosystem services provided by eelgrass varies with the density and size of the actual bed in a positive or negative direction depending on the species or function in question.

Another manner by which eelgrass influences the health and productivity of ecosystems is through the input of detrital drift. When eelgrass shoots are broken free by tidal currents or wave energy, the resulting "wrack" may be relocated by the wind and/or tidal currents to habitats that are less productive. The input of wrack provides a potentially significant nutrient subsidy to intra-tidal communities (Duggins et al. 1989, Bustamante et al. 1995). Wrack has been shown to beneficially support shorebirds by supporting large populations of wrack-associated invertebrates (Dugan and Hubbard and 2002). This may represent an increase in an important prey resource to avian predators by making prey available at a wider range of tidal elevations relative to beaches without wrack inputs (Orr et al 2005). Tarr and Tarr (1987) showed a direct correlation in the increase of shorebirds to an increase in the availability of wrack.

2.2.2 Buffering Capacity

Ocean acidification is a progressive increase in the acidity (or decrease in pH) of the ocean over an extended period of time caused primarily by the uptake of CO_2 from the atmosphere by the ocean (WSBRP 2012). According to Waldbusser et al. (2011), "estuarine waters are more susceptible to acidification because they are subject to multiple acid sources and are less buffered than marine waters." Seagrasses have been shown to raise pH values and provide some buffering capacity to the surrounding waters (Beer et al. 2006, Horwith 2014). Dr. Horwith, a researcher at the Washington Department of Natural Resources, measured carbonate chemistry dynamics across three benthic habitats (eelgrass beds, Pacific oyster plots, and bare sand) in five Washington state embayments. Preliminary results showed that the pH in surface water over eelgrass beds increased significantly (pH increase of 0.05). In the same study, there was no significant difference between the change in water chemistry over either oyster plots or bare sand. Therefore, eelgrass may be able to provide some buffering to shellfish and other pH sensitive organisms from ocean acidification. As ocean chemistry changes, eelgrass and macroalgae are being considered as potential refuges for calcifying organisms.

2.3 Ecosystem Functions Not Provided by SAV

The main ecosystem function provided by shellfish aquaculture that is not provided by eelgrass is related to the removal of nitrogen at harvest and denitrification of biodeposits (Table 1). A number of other ecosystem functions are also provided by shellfish aquaculture, as discussed in the positive interaction section above (Section 1.2). In addition, there are density considerations that should be made in terms of understanding the value provided by either eelgrass or shellfish aquaculture, as discussed in Section 2.5 below. The following information is focused on the sequestration of nitrogen and bioextraction during a harvest event.

While eelgrass can remove nitrogen from the immediate intertidal and subtidal area when the blades decay, become dislodged, and are transported away on tidal currents (Rumrill, pers. comm., 2015), nitrogen is permanently removed from coastal marine waters through harvest of shellfish and denitrification processes (Figure 8). As shellfish filter large quantities of organic matter from the water column, they assimilate nitrogen and phosphorus into their shells and tissue (Newell et al. 2002). When shellfish are harvested, the sequestered nutrients are permanently removed from the system. According to Newell (2004), bioextraction is one of the only methods available that removes nutrients after they have entered a system, which can then make that system more resilient to nutrient loading.

The amount of nutrients sequestered in shellfish shells and tissue is dependent on species-specific filtration rates, which may be influenced by local water quality conditions that affect physiological parameters of the shellfish (e.g., water temperature, plankton abundance). The amount of benefit that filtration provides also depends on the physical mixing of nutrient sources (e.g., oceanic vs. riverine), residence time in the estuary, and grazing pressure of farmed shellfish (Dumbauld et al. 2009). Although not currently recognized as a direct benefit on the West Coast, bivalve filtration may become more valuable as nutrient input increases within coastal communities (Shumway et al. 2003, Burkholder and Shumway 2011, Kellogg et al. 2013).

An example of the potential benefits offered by shellfish filtration and nutrient sequestration, Kellogg et al. (2013) partially quantified the removal of nutrients from the water column at a subtidal oyster reef restoration site compared to an adjacent control site in the Choptank River within Chesapeake Bay, Maryland. The authors indicated that denitrification rates at the oyster reef in August were "among the highest ever recorded for an aquatic system." In addition, a significant portion (47 and 48% of total standing stock) of the available nitrogen and phosphorous were sequestered in the shells of live oysters and mussels. An ancillary benefit of the shellfish reef structure, which is also true for shellfish aquaculture gear and shellfish, was that the structure and faunal composition provided ample microhabitats for communities of nitrifying microbes. One of the conclusions by Kellogg et al. (2013) was that oyster reef restoration could be considered a "safety net" to reduce additional downstream impacts to water quality. Because shellfish aquaculture provides many of the same benefits, with the added benefit of the total removal of nutrients at harvest, the shellfish industry can be considered a net benefit to water quality ecosystem functions.

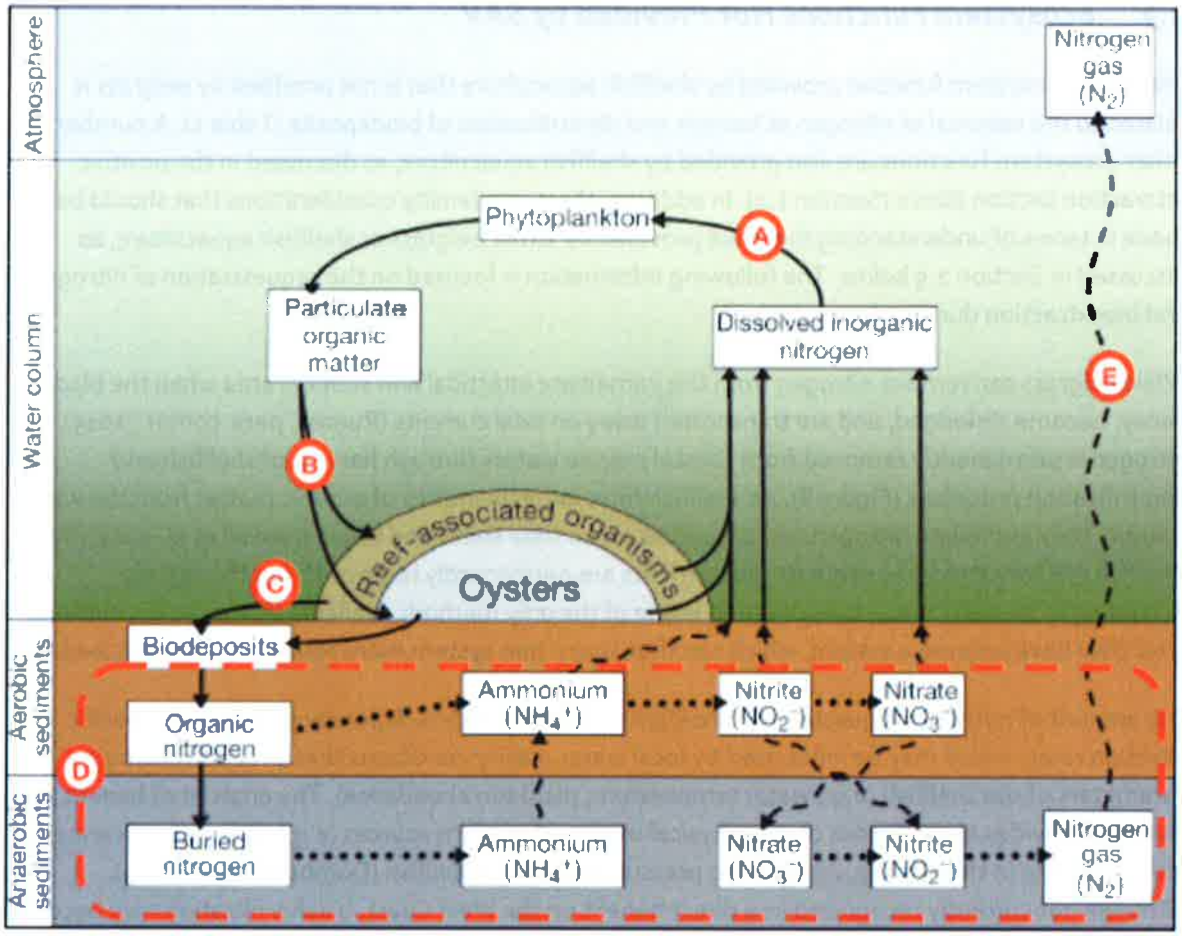

Figure 8 Primary Nitrogen Pathways Associated with Oysters and Reef-Associated Organisms

Notes: Phytoplankton use dissolved inorganic nitrogen for their growth (A), oysters and other reef-associated organisms filter phytoplankton and other particulate organic matter from the water column (B), some of the associated nitrogen is incorporated into organisms and some is deposited on the surface of the sediments (C), and, given the right conditions, a portion of the nitrogen in these biodeposits is transformed into nitrogen gas (D) which diffuses out of the sediments back to the atmosphere I where it is no longer available to phytoplankton for growth (diagram adapted from Newell et al. 2005).

Source: Kellogg et al. (2013)

2.4 *Habitat Structure (Density-Dependent)*

Estuaries are important nursery and feeding areas for many species. Use of these habitats may be controlled by habitat structure which influences the supply of food and abundance of predators. An individual habitat type rarely captures all life stages for the species found there, which emphasizes the importance of connectivity and movement between habitats (Sheaves 2009). While habitats can be mapped, the importance is often characterized in relation to ecological processes such as refuge from predation and foraging that are harder to quantify. In considering habitat protection targets for estuarine systems, Sheaves et al. (2006) cautioned that "the basic complexity of natural systems means we have little clear idea of the likely consequences of concentrating on [protecting] those habitats at the expense of others."

One of the most important benefits offered by shellfish aquaculture is the ability to provide nursery habitats that create transitional zones between mudflats and eelgrass. One of the most comprehensive analyses of the attributes relevant to identifying the role of nursery habitat was performed by Heck et al. (2003). The authors conducted a meta-analysis of more than 200 papers that compared seagrass beds to other habitats, and examined the data using the attributes suggested by Beck et al. (2001) for defining the ecological processes operating in nursery habitats, including: density, growth, survival, and migration to adult habitat. The results indicated that few significant differences existed between the relevant attributes when seagrass meadows were compared to other structured habitats, such as oyster reefs, cobble reefs, or macroalgal beds. The most important determinant of nursery value was the presence of structure rather than the type of structure.

What does appear to be an important determinant in terms of the quality of habitat provided is density and diversity. Optimal foraging/movement and fitness strategies depend on a mosaic of different habitats, and edges or transitional zones between two habitat types often represent areas with increased biological diversity (Holt et al. 1983, Orth et al. 1984, Boström et al. 2006). For example, several species of fish are found in higher densities in patchy eelgrass beds versus continuous dense beds (Orth et al. 1984). Holt et al. (1983) suggested that some species of fish require open feeding areas and refuge areas in the same location, and that patchy vegetation with a high percentage of edges may support higher densities of mobile foraging species.

The observations of edge effect are partially supported by a recent study in Humboldt Bay, California, by Pinnix et al. (2013). The study used acoustic transmitters that were surgically implanted into out-migrating coho salmon (*Oncorhynchus kisutch*) smolts. During their residency in Humboldt Bay, coho smolts primarily used deep channels and channel margins. They were detected near floating eelgrass mats adjacent to the channels, but not over eelgrass beds. The results from this study potentially emphasize the importance of edge habitat and the need for structural heterogeneity during salmonid residency and migration through the estuary.

In terms of prey resources, Tanner (2005) found epifauna, such as tanaids and gammaridean amphipods (i.e., typical salmonid prey items), exhibited significantly higher abundances at sand/seagrass edges versus seagrass bed interiors. Similarly, Hirst and Attrill (2008) determined that eelgrass patches, regardless of size or number of plants, were found to support a higher level of biodiversity than surrounding sand habitats. Thus, it could be argued that modest displacement of eelgrass resulting in some patchiness may be neutral or beneficial for certain species, such as salmonids, provided that an abundance of eelgrass was present in the surrounding environment to ensure that none of the other ecological functions provided by eelgrass were significantly reduced.

Structured habitat is often associated with higher species diversity for benthic invertebrates but not directly for mobile species. Hosack et al. (2006) reported that benthic invertebrates were strongly associated with habitat type, and structured habitats (oyster beds and eelgrass) had higher species abundance. This concept is consistent with what was reported above in terms of an increase in potential prey resources associated with eelgrass patches and along edges. However, Hosack et al. (2006) indicated that, "Fish and decapod species richness and the size of ecologically and commercially important species, such as Dungeness crab (*Cancer magister*), English sole (*Parophrys vetulus*), or lingcod (*Ophiodon elongatus*), were not significantly related to habitat type." This is important because these mobile species are using a mosaic of habitats, and one is not necessarily more important than others as long as there is a diversity of habitat provided.

Additional work by Dr. Hosack and others can help to further illustrate this point. Hosack (2003) reported that important fish prey organisms, such as harpacticoid copepods, exhibited an inverse trend with higher densities in both dense eelgrass and oyster habitats. These observations parallel those of Ferraro and Cole (2011, 2012), who studied oyster bottom culture in Yaquina Bay, Oregon, Willapa Bay, Washington, and Grays Harbor, Washington. The authors reported similar species abundance and richness in benthic macrofaunal communities between native eelgrass and oyster habitat in the three areas studied. Both eelgrass and oyster habitats had significantly more prey resources than mudflat or sandy habitats. This serves to illustrate the relative importance of eelgrass *and* shellfish-rich habitat in coastal estuaries as refugia and a source of prey for foraging nekton and other marine life.

A recent manuscript that ties these concepts together is one by Dumbauld et al. (*in review*[3]). The study objective was to identify whether intertidal oyster aquaculture in Willapa Bay effects the distribution and feeding ecology of juvenile salmonids. The study identified no significant differences in the density of juvenile salmonids caught in the four habitat types analyzed (undisturbed open mudflat, seagrass, channel habitats, and oyster aquaculture), and few significant associations with the prey items that the fish consumed. In other words, the majority of salmon found over low intertidal habitats were not dependent on structured habitat (e.g., eelgrass or oyster aquaculture) for prey items. Chum salmon was the possible exception, which is typically a smaller fish during estuarine residency. The final conclusion by Dumbauld et al. (*in review*) was that: "Permanent or 'press' disturbances like diking marshes, dredging and filling shallower estuarine habitats and even hardening shorelines would be expected to have significant impacts for other stocks and life history variants with smaller juveniles that utilize upper intertidal areas (Fresh 2006; Bottom et al. 2009), but our research suggests that short term 'pulse' disturbances like aquaculture which alter the benthic substrate in lower intertidal areas used primarily by larger juvenile salmon outmigrants may pose a less significant threat to maintaining resilience of these fish populations."

[3] Although the information presented was taken from the manuscript, it is also discussed in the Western Regional Aquaculture Center (WRAC) project termination report that supported the manuscript (Dumbauld 2006).

2.5 *Summary of Habitat Equivalency of Shellfish Aquaculture and SAV*

There is an important difference between habitat fragmentation and increasing a mosaic of habitats available for aquatic species. According to Boström et al. (2006), seagrass fragmentation can result in impacts to aquatic organisms but that is not necessarily the case if it is replaced by some other structured habitat that offers similar ecosystem functions. Given the potential benefits of increased complexity (i.e., habitat mosaic) for aquatic organisms, both eelgrass and shellfish aquaculture may be better than either alone. This could provide the full suite of ecological services, such as increased edges and migration corridors, diversity of habitat structure, prey resources, refugia, and water quality benefits. Furthermore, this provision would be even better if shellfish aquaculture siting included areas without SAV, with SAV, preserving areas of SAV without shellfish, and preserving areas without SAV and shellfish. In other words, a diverse array of habitat types available within estuaries.

3.0 EFFECTS ANALYSIS FRAMEWORK

As discussed throughout this document, the interactions between shellfish aquaculture and eelgrass are complex with both mutually beneficial and competitive aspects. Ultimately, it is important to evaluate these interactions at the ecosystem level, taking into account site-specific interactions in the context of the landscape scale. For example, the displacement of a modest amount of eelgrass by shellfish aquaculture may have more relevance in a watershed with limited eelgrass resources and few eelgrass ecosystem services. In contrast, the displacement of a modest amount of eelgrass by shellfish aquaculture may be of limited significance in a watershed with abundant eelgrass that occurs at levels similar to historical or pre-developed conditions where eelgrass ecosystem services are adequately represented. Limiting factors within the watershed should also be taken into account when assessing interactions between shellfish aquaculture and eelgrass. For example, in watersheds with limited saltwater marsh habitat from diking that results in high natural suspended sediments in the estuary, increasing depositional environments may provide more benefit to eelgrass habitat than simple protection of the eelgrass itself. The decision framework presented in Figure 9 is a preliminary effort to account for both the site-specific and landscape scale when evaluating shellfish aquaculture and eelgrass interactions.

The conceptual framework presented is intended to help decision-makers organize and synthesize existing information and professional judgement to create working hypotheses about the sensitivity of likely environmental changes that may arise from implementation of aquaculture operations near or within eelgrass beds. This framework offers an opportunity to measure ecosystem function trade-offs resulting from losses of SAV density or areal extent and increases of aquaculture and related structures. For ecosystem functions about which little is known, professional knowledge and assumptions about the direction of ecosystem services effect may serve until additional scientific knowledge comes to light. For ecosystem functions about which much is known, sensitivity curves relating loss of SAV to loss of ecosystem function could provide a more quantitative method of accounting. The increase or decrease in each ecosystem function can be accounted for in light of effects arising from both changes in SAV and changes in shellfish aquaculture.

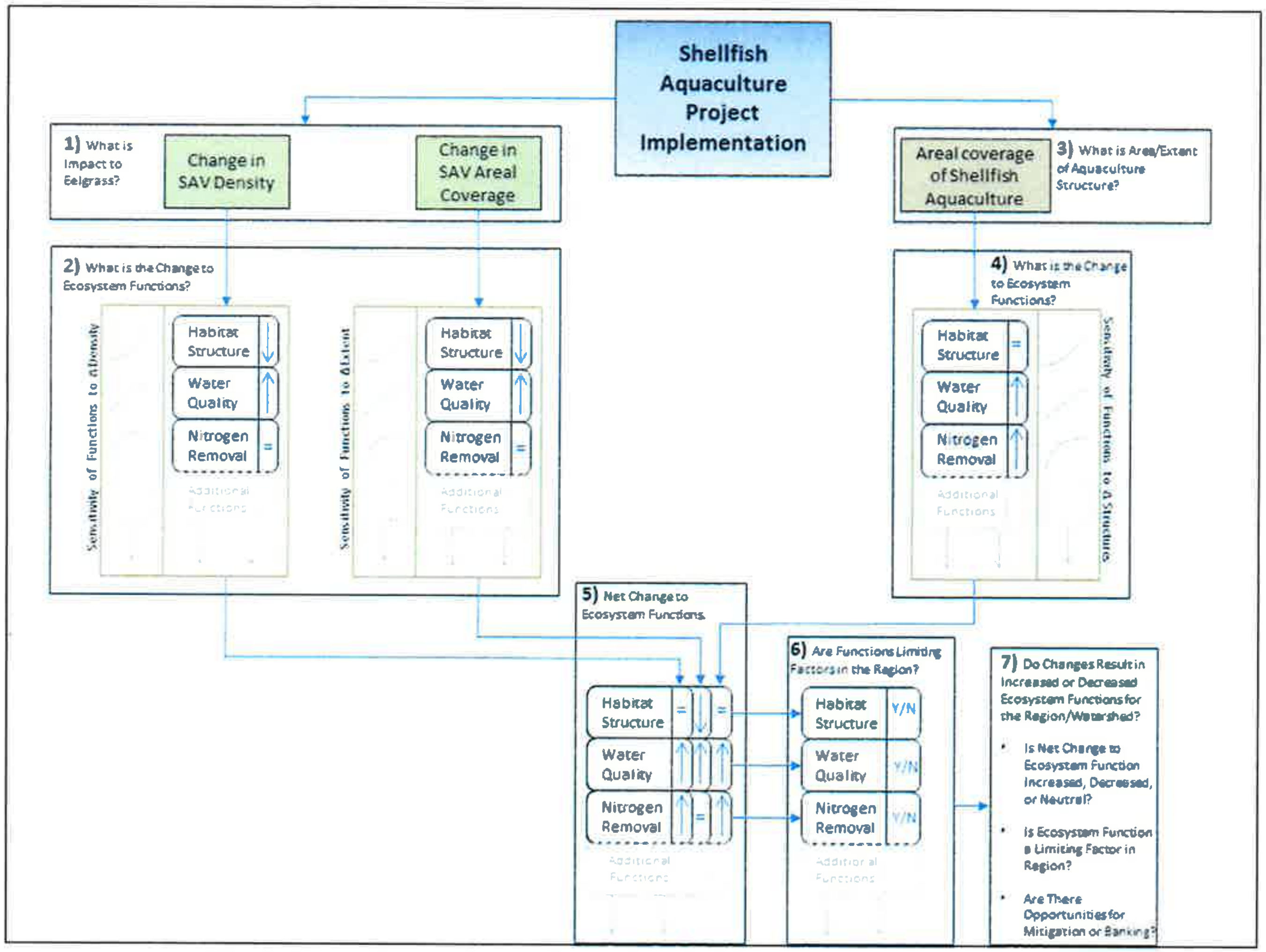

Figure 9 Conceptual Method to Account for Changes to Ecosystem Functions Due to Shellfish Aquaculture

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As mentioned previously, whether an ecosystem function is a limiting factor within the region or watershed is an important detail in the broader assessment of effects to the ecosystem. This is not to say that when a function is not limiting within a region that detrimental impacts should be ignored, but rather, there may be the opportunity to prioritize ecosystem functions and apply increased or reduced multipliers or out-of-kind mitigation solutions based on the needs of the region/watershed setting.

The value of conceptual frameworks lies in their ability to clearly define linkages between professional knowledge and insights and scientific understanding of natural process interactions. Conceptual frameworks provide a useful methodology for guiding balanced decision-making in the face of incomplete information.

4.0 SUMMARY

This document supports the CAPES effort to conduct a state of the science assessment on shellfish aquaculture interactions with SAV. The overall goal was to provide relevant scientific information and a conceptual framework for evaluating aquaculture projects proposed for new areas in order to determine net effects to ecosystem functions. The existing scientific literature supports the understanding that shellfish aquaculture operations can be conducted in the presence of eelgrass and still result in no net loss of ecosystem functions.

Shellfish aquaculture and eelgrass have co-existed along the West Coast and in Puget Sound for over 100 years, and eelgrass is still abundantly present in areas with active culture operations. Although there are pulse disturbances to eelgrass from shellfish aquaculture, the effects are generally localized and temporary and are considered minor when viewed from a landscape scale. Further, considering the combined ecosystem services provided by shellfish aquaculture and eelgrass, a mosaic of habitat could be the most beneficial ecological option to support aquatic organisms.

Finally, aquaculture activities need to be managed at the landscape or watershed scale in addition to the project scale. The concepts of HEA provide a good framework upon which to build a management tool that is directly applicable to shellfish aquaculture. A primary assumption used in HEA is that comparable ecosystem functions can be provided by created habitat for areas of lost or displaced habitat. By evaluating the limiting factors for eelgrass directly, and within the watershed more broadly, managers can create a regional approach to focus and prioritize which ecosystem functions are most important for maintaining the ecological health of that system.

5.0 REFERENCES

Allen, C.R., D.G. Angeler, A.S. Garmestani, L.H. Gunderson, and C.S. Holling. 2014. Panarchy: Theory and Application. Nebraska Cooperative Fish and Wildlife Research Unit – Staff Publications. Paper 127. http://digitalcommons.unl.edu/ncfwrustaff/127

Barrett, E.M. 1963. The California oyster industry. Scripps Institution of Oceanography Library. http://scilib.ucsd.edu/sio/

Beck, M.W., K.L. Heck, Jr., K.W. Able, D.L. Childers, D.B. Eggleston, B.M. Gillanders, B. Halpern, C.G. Hays, K. Hoshino, T.J. Minello, R.J. Orth, P.F. Sheridan, and M.P. Weinstein. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates: A better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. Bioscience 51(8):633-641.

Beer, S., L. Axelsson, and M. Björk. 2006. Modes of photosynthetic bicarbonate utilisation in seagrasses, and their possible roles in adaptations to specific habitats. Biologia Marina Mediterranea 13:3-7.

Bender, E.A., T.J. Case, and M.E. Gilpin. 1984. Perturbation experiments in community ecology: Theory and practice. Ecology 65:1-13.

Bigley, R.E. 1981. The population biology of two intertidal seagrasses, *Zostera marina* and *Ruppia maritima*, at Roberts Bank, British Columbia. Thesis, University of British Columbia, Vancouver British Columbia, Canada.

Bodin, P. and B.L.B. Wiman. 2004. Resilience and other stability concepts in ecology: Notes on their origin, validity, and usefulness. ESS Bulletin 2:33-43.

Boese, B.L., B.D. Robbins, and G. Thursby. 2005. Desiccation is a limiting factor for eelgrass (*Zostera marina* L.) distribution in the intertidal zone of a northeastern Pacific (USA) estuary. Botanica Marina 48(4):274-283.

Boström, C., E.L. Jackson, and C.A. Simenstad. 2006. Seagrass landscapes and their effects on associated fauna: A review. Estuarine Coastal and Shelf Science 68:383-403.

Bottom, D.L., K. Jones, C.A. Simenstad, and C.L. Smith. 2009. Reconnecting social and ecological resilience in salmon ecosystems. Ecology and Society 14:5.

Burkholder, J.M. and S.E. Shumway. 2011. Bivalve shellfish aquaculture and eutrophication. *In*: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Bustamante, R. H., and G. M. Branch. 1996. The dependence of intertidal consumers on kelp-derived organic matter on the west coast of South Africa. Journal of Experimental Marine Biology and Ecology 196:1–28.

Cardoso, P.G., M.A. Pardal, A.I. Lillebø, S.M. Ferreira, D. Raffaelli, and J.C. Marques. 2004. Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. Journal of Experimental Biology and Ecology 302:233-248.

Coen L.D., B.R. Dumbauld, and M.L. Judge. 2011. Expanding shellfish aquaculture: a review of the ecological services provided by and impacts of native and cultured bivalves in shellfish dominated ecosystems. *In*: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Cranford, P.J., J.E Ward, and S.E. Shumway. 2011. Bivalve filter feeding: Variability and limits of the aquaculture biofilter. In: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Dame, R.F., R.G. Zingmark, and E. Haskin. 1984. Oyster reefs as processors of estuarine materials. J. Exp. Mar. Biol. Ecol. 83:239-247.

den Hartog, C. and R.C. Phillips. 2000. Seagrasses and benthic fauna of sediment shores. *In*: K. Reise (ed). Ecological Comparisons of Sedimentary Shores. Springer, Berlin, pp. 195–212.

Dennison, W.C. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27(1):15-26.

Dewey, B. 2015. Personal communication regarding shellfish aquaculture operations in Willapa Bay and Samish Bay. Taylor Shellfish Farms. April 16, 2015. BillD@taylorshellfish.com

Dewey, W., J.P. Davis, and D.C. Cheney. 2011. Shellfish aquaculture and the environment: An industry perspective. Pages 33-50. *In*: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Duarte C. M. and K. Sand-Jensen. 1990. Seagrass colonization: Patch formation and patch growth in *Cymodocea nodosa*. Marine Ecology Progress Series 65:193-200.

Duarte, C. M. 1989. Temporal biomass variability and production/biomass relationships of seagrass communities. Marine Ecology Progress Series 51:269-276.

Dugan, J. E., Hubbard, D. M., McCrary, M., & Pierson, M. (2003). The response of macrofauna communities and shorebirds to macrophyte wrack subsidies on exposed sandy beaches of southern California. Estuarine, Coastal and Shelf Science 58S, 133–148.

Duggins, D. O., C. A. Simenstad, and J. A. Estes. 1989. Magnification of secondary production by kelp detritus in coastal marine ecosystems. Science 41:170–173.

Dumbauld, B.R. 2006. WRAC Project Termination Report: The Ecological Role and Potential Impacts of Molluscan Shellfish Culture in the Estuarine Environment. Western Regional Aquaculture Center. 42 pp

Dumbauld, B.R. and L.M. McCoy. *in press*. The effect of oyster aquaculture on seagrass (*Zostera marina*) at the estuarine landscape scale in Willapa Bay, Washington (USA). Aquaculture Environment Interactions.

Dumbauld, B.R., B.E. Kauffman, A.C. Trimble, and J.L. Ruesink. 2011. The Willapa Bay Oyster Reserves in Washington State: fishery collapse, creating a sustainable replacement, and the potential for habitat conservation and restoration. Journal of Shellfish Research 30(1):71-83.

Dumbauld, B.R., G.R. Hosack, and K.M. Bosley. *in review*. Association of juvenile salmon and estuarine fish with intertidal seagrass and oyster aquaculture habitats in a northeast Pacific estuary. Transactions of the American Fisheries Society.

Dumbauld, B.R., J.L. Ruesink, and S.S. Rumrill. 2009. The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries. Aquaculture 290:196-223.

Dumbauld, B.R., K.M. Brooks, and M.H. Posey. 2001. Response of an estuarine benthic community to application of the pesticide carbaryl and cultivation of Pacific oysters (*Crassostrea gigas*) in Willapa Bay, Washington. Mar. Pollut. Bull. 42, 826–844.

Everett, R.A., G.M. Ruiz, and J.T. Carlton. 1995. Effects of oyster mariculture on submerged aquatic vegetation: An experimental test in a Pacific Northwest estuary. Marine Ecology Progress Series 125:2205-2217.

Ferraro, S.P. and F.A. Cole. 2011. Ecological periodic tables for benthic macrofaunal usage of estuarine habitats in the US Pacific Northwest. Estuarine, Coast and Shelf Science 94:36-47.

Ferraro, S.P. and F.A. Cole. 2012. Ecological period tables for benthic macrofaunal usage of estuarine habitats: Insights from a case study in Tillamook Bay, Oregon, USA. Estuarine, Coast and Shelf Science 102-103:70-83.

Ferreira, J.G., A.J.S. Hawkins, and S.B. Bricker. 2011. The role of shellfish farms in provision of ecosystem goods and services. Pages 3-32. *In*: S.E. Shumway (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Fonseca, M.S. and S.S. Bell. 1998. Influence of Physical Setting on Seagrass Landscapes near Beaufort, North Carolina, USA. Marine Ecology-Progress Series 171:109-121.

Forrest, B.M., N.B. Kelley, G.A. Hopkins, S.C. Webb, and D.M. Clement. 2009. Bivalve aquaculture in estuaries: Review and synthesis of oyster cultivation effects. Aquaculture 298:1-15.

Fresh, K.L. 2006. Juvenile salmon in Puget Sound. Puget Sound Nearshore Partnership, Seattle, Washington. Technical Report 2006-6. Website: http://www.pugetsoundnearshore.org/technical_papers/pacjuv_salmon.pdf

Gaeckle, J., P. Dowty, H. Berry, and L. Ferrier. 2011. Puget Sound Submerged Vegetation Monitoring Project 2009 Report. Washington State Department of Natural Resources, Nearshore Habitat Program, Olympia, Washington.

Hauxwell, J., J. Cebrian, C. Furlong, and I. Valiela. 2001. Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. Ecology 82:1007-1022.

Heck, K.L., G. Hays, and R.J. Orth. 2003. Critical evaluation of the nursery role hypothesis for seagrass meadows. Marine Ecology Progress Series 253:123-136.

Hirst, J.A. and M.J. Attrill. 2008. Small is beautiful: An inverted view of habitat fragmentation in seagrass beds. Estuarine, Coastal and Shelf Science 78:811-818.

Holling, C.S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4:1-23.

Holt, S.A., C.L. Kitting, and C.R. Arnold. 1983. Distribution of young red drums among different seagrass meadows. Transactions of the American Fisheries Society 112(2B):267-271.

Horwith, M. 2013. Changes in seagrass (*Zostera marina*) and infauna through a five-year crop cycle of geoduck clams (*Panopea generosa*) in Samish Bay, WA. Pacific Science

Horwith, M. 2014. Effects of benthic intertidal organisms on marine water properties. Washington State Department of Natural Resources. Pacific Coast Shellfish Growers Association and National Shellfisheries Association, Pacific Coast Section. 68th Annual Shellfish Growers Conference and Trade Show. Vancouver, Washington. September 21-25, 2014.

Horwith, M. 2014. Personal communication regarding eelgrass status on Fisk Bar, Samish Bay, WA. Washington State Department of Natural Resources. Micah.horwith@dnr.wa.gov (actual communication with Bill Dewey, Taylor Shellfish)

Hosack, G.R. 2003. Does habitat structure influence low intertidal communities in Willapa Bay, Washington? Presented at Pacific Estuarine Research Society (PERS) Conference Vancouver British Columbia. April 3-4, 2003.

Hosack, G.R., B.R. Dumbauld, J.L. Ruesink, and D.A. Armstrong. 2006. Habitat associations of estuarine species: comparisons of intertidal mudflat, seagrass (*Zostera marina*), and oyster (*Crassostrea gigas*) habitats. Estuaries and Coasts 29:1150-1160.

Jay, D. 2014. Personal communication regarding reduction in extreme low tides. Portland State University. September 4, 2014. djay@pdx.edu (actual communication with Bill Dewey, Taylor Shellfish).

Jones C.G., J.H. Lawton, and M. Shachak. 1994. Organisms as ecosystem engineers. Oikos 69:373-386.

Kellogg, M.L., J.C. Cornwell, M.S. Owens, and K.T. Paynter. 2013. Denitrification and nutrient assimilation on a restored oyster reef. Marine Ecology Progress Series 480:1-19.

Kirkman, H. and J. Kuo 1990. Pattern and process in southern Western Australia seagrasses. Aquatic Botany 37:367-382.

Koch, E.W., and S. Beer. 1996. Tides, light and the distribution of *Zostera marina* in Long Island Sound, USA. Aquatic Botany 53(1-2):97-107.

Lake, P.S. 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society 19(4):573-592.

Lenihan, H.S. 1999. Physical-biological coupling on oyster reefs: How habitat structure influences individual performance. Ecological Monographs 69: 251-275.

Liu, W. and C. Pearce. 2011. Assessing potential benthic impacts of subtidal geoduck clam harvesting. Pacific Biological Station, Fisheries and Oceans Canada, Nanaimo, British Columbia, Canada. In: Abstracts of Technical Papers Presented at the 103rd Annual Meeting, National Shellfisheries Association Baltimore, Maryland. March 27-31, 2011. Journal of Shellfish Research. 30: 459-568.

Mercaldo-Allen, R. and R. Goldberg. 2011. Review of the ecological effects of dredging in the cultivation and harvest of mulluscan shellfish. NOAA National Marine Fisheries Service. NOAA Technical Memorandum NMFS-NE-220.

Meyer, D.L., E.C. Townsend and G.W. Thayer. 1997. Stabilization and erosion control value of oyster cultch for intertidal marsh. Restoration Ecol. 5:93-99.

Mumford, T.F. 2007. Kelp and Eelgrass in Puget Sound. Puget Sound Nearshore Partnership Report No. 2007-05. Published by Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Nelson, T.A. and J.R. Waaland. 1997. Seasonality of eelgrass, epiphyte, and grazer biomass and productivity in subtidal eelgrass meadows subjected to moderate tidal amplitude. Aquatic Botany 56(1):51-74.

Newell, R. I. E. 2004. Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. Journal of Shellfish Research 23:51-61.

Newell, R.I.E, T.R. Fisher, R.R. Holyoke and J.C. Cornwell. 2005. Pages 93 - 120. *In*: R. Dame and S. Olenin (eds.). The Comparative Roles of Suspension Feeders in Ecosystems. Vol. 47, NATO Science Series: IV - Earth and Environmental Sciences. Springer, Netherlands.

Newell, R.I.E., and E.W. Koch. 2004. Modeling seagrass density and distribution in response to changes in turbidity stemming from bivalve filtration and seagrass sediment stabilization. Estuaries 27(5):793-806.

Newell, R.I.E., J.C. Cornwell, and M.S. Owens. 2002. Influence of simulated bivalve biodeposition and microphytobenthos on sediment nitrogen dynamics: A laboratory study. Limnology and Oceanography 47(5):1367-1379.

Nielsen, S.L., G.T. Banta, and M.F. Pedersen. 2004. Estuarine nutrient cycling: The influence of primary producers. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Nienhuis, P.H. 1996. The North Sea coasts of Denmark, Germany and the Netherlands. *In*: W. Schramm and P.H. Nienhuis (eds.). Marine Benthic Vegetation. Recent Changes and the Effects of Eutrophication. Springer, Berlin, pp. 187–222.

Olesen, B. and K. Sand-Jensen. 1994. Patch dynamics of eelgrass *Zostera marina*. Marine Ecology-Progress Series 106:147-156.

Orr, M., M. Zimmer, D.E. Jelinski, and M. Mews. 2005. Wrack Deposition on Different Beach Types: Spatial and Temporal Variation in the Pattern of Subsidy. Ecology, 86(6) pp 1469-1507.

Orth, R. J., M. Luckenbach and K.A. Moore (1994). Seed dispersal in a marine macrophyte: Implications for colonization and restoration Ecology 75(7):1927-1939.

Orth, R.J., K.L. Heck, Jr., and J. van Montfrans. 1984. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predator-prey relationships. Estuaries 7(4A):339-350.

Peterson, B.J. and K.L. Heck, Jr. 1999. The potential for suspension feeding bivalves to increase seagrass productivity. Journal of Experimental Marine Biology and Ecology 240:37-52.

Peterson, B.J., and K.L. Heck, Jr. 2001. Positive interactions between suspension-feeding bivalves and seagrass—a facultative mutualism. Marine Ecology Progress Series 213: 143-155.

Phillips, R.C. 1984. The ecology of eelgrass meadows in the Pacific Northwest: a community profile. U.S. Fish and Wildlife Service. Washington, D.C.

Pinnix, W.D., P.A. Nelson, G. Stutzer, K.A. Wright. 2013. Residence time and habitat use of coho salmon in Humboldt Bay, California: an acoustic telemetry study. Environ. Biol. Fish. 96:315-323.

Powers, M.J., C.H. Peterson, H.C. Summerson, and S.P. Powers. 2007. Macroalgal growth on bivalve aquaculture netting enhances nursery habitat for mobile invertebrates and juvenile fishes. Marine Ecology Progress Series 339:109-122.

Preen, A.R., W.J. Lee Long, and R.G. Coles. 1995. Flood and cyclone related loss, and partial recovery, of more than 1000 km^2 of seagrass in Hervey Bay, Queensland, Australia. Aquatic Botany 52:3-17.

PSI (Pacific Shellfish Institute). 2015. Shellfish carrying capacity. PSI, Olympia, Washington. http://www.pacshell.org/carrying-capacity.asp (assessed on April 13, 2015).

Robbins, B.D. and S.S. Bell 2000. Dynamics of a subtidal seagrass landscape: Seasonal and annual change in relation to water depth. Ecology 81(5):1193-1205.

Ruckelshaus, M.H. 1996. Estimation of genetic neighborhood parameters from pollen and seed dispersal in the marine angiosperm *Zostera marina*. Evolution 50(2):865-864.

Ruesink, J.L. and K. Rowell. 2012. Seasonal effects of clam (*Panopea generosa*) on eelgrass (*Zostera marina*) density but not recovery dynamics at an intertidal site. Aquatic Conservation Marine and Freshwater Ecosystems. 22: 712-720.

Ruesink, J.L., J.P. Fitzpatrick, B.R. Dumbauld, S.D. Hacker, A.C. Trimble, E.L. Wagner, and L.M. Wisehart. 2012. Life history and morphological shifts in an intertidal seagrass following multiple disturbances. Journal of Experimental Marine Biology and Ecology 424-425:25-31.

Rumrill, S. 2011. Native eelgrass (*Zostera marina*) in Oregon estuaries: Ecology, threats, and mitigation for damage and loss of eelgrass (*Zostera marina*) in estuarine habitats. Oregon Department of State Lands, South Slough National Estuarine Research Reserve. http://www.oregon.gov/dsl/SSNERR/docs/eelgrass_threats.pdf

Rumrill, S. 2015. Personal communication regarding eelgrass and shellfish aquaculture interactions from Humboldt Bay WRAC study. Oregon Department of Fish and Wildlife. April 5, 2015. steven.s.rumrill@state.or.us

Rumrill, S. and V. Poulton. 2004. Ecological role and potential impacts of molluscan shellfish culture in the estuarine environment of Humboldt Bay, CA. Western Regional Aquaculture Center Annual Report November 2004. 79 p

Schlosser, S., and A. Eicher. 2012. The Humboldt Bay and Eel River Estuary Benthic Habitat Project. California Sea Grant Publication T-075. 246 p.

Schramm, W. and P.H. Nienhuis (eds.). 1996. Marine Benthic Vegetation. Recent Changes and the Effects of Eutrophication. Springer, Berlin. 465 pp

Scyphers, S.B., S.P. Powers, K.L. Heck, Jr., and D. Byron. 2011. Oyster reefs as natural breakwaters mitigate shoreline loss and facilitate fisheries. PLoS ONE 6(8):e22396.

Semmens, B.X. 2008. Acoustically derived fine-scale behaviors of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) associated with intertidal benthic habitats in an estuary. Can. J. Fish. Aquat. Sci. 65:2053-2062.

Shaughnessy, F.J., C.L. McGary, A.J. Frimodig, C. Witte, and G.B. Roberts. 2004. Known and unknown aspects of bottom-up and top-down regulation of eelgrass in Humboldt Bay, California. *In* S.C. Schlosser and R. Rasmussen (eds). Proceedings of the Symposium: Current perspectives on the physical and biological processes of Humboldt Bay, March 15, 2004, Eureka, California pp. 51-64. California Sea Grant Publication T-063, 270 pages.

Sheaves, M. 2009. Consequences of ecological connectivity: the coastal ecosystem mosaic. Marine Ecology Progress Series 391:107-115.

Sheaves, M., R. Baker, and R. Johnston. 2006. Marine nurseries and effective juvenile habitats: An alternative view. Marine Ecology Progress Series 318:303-306.

Short, F.T., D.M. Burdick, C.A. Short, R.C. Davis, and P.A. 2000. Developing success criteria for restored eelgrass, salt marsh and mud flat habitats. Ecological Engineering 15:239-252.

Short, K.S. and R. Walton. 1992. The transport and fate of suspended sediment plumes associated with commercial geoduck harvesting, Final Report. Prepared for the State of Washington Department of Natural Resources. Prepared by Ebasco Environmental, Bellevue, Washington.

Shumway, S.E. (ed). Shellfish Aquaculture and the Environment. Wiley-Blackwell, West Sussex, UK.

Shumway, S.E., C. Davis, R. Downey, R. Karney, J. Kraeuter, J. Parsons, R. Rheault, and G. Wikfors. 2003. Shellfish aquaculture – In praise of sustainable economies and environments. World Aquaculture 34(4):15-18.

Smetacek, V. and A. Zingone. 2013. Green and golden seaweed tides on the rise. Nature 504:84-88.

Spalding, M.D., S.Ruffo, C. Lacambra, I. Meliane, L. Zeitlin Hale, C. C. Shepard, and M.W. Beck. 2014. The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. Ocean & Coastal Management 90:50-57.

Stevens, A.W. and J.R. Lacy. 2012. The influence of wave energy and sediment transport on seagrass distribution. Estuaries and Coasts 35(1):92-108.

Tallis, H.M., J.L. Ruesink, B.R. Dumbauld, S.D. Hacker, and L.M. Wisehart. 2009. Oysters and aquaculture practices affect eelgrass density and productivity in a Pacific Northwest estuary. Journal of Shellfish Research 28:251-261.

Tanner, J.E. 2005. Edge effects on fauna in fragmented seagrass meadows. Austral. Ecology 30:210-218.

Tarr, J. G., & Tarr, P. W. (1987). Seasonal abundance and the distribution of coastal birds on the northern Skeleton Coast, south west Africa/Namibia. Madoqua 15, 63–72.

Taylor, D., S. Nixon, S. Granger, and B. Buckley. 1995. Nutrient limitation and the eutrophication of coastal lagoons. Marine Ecology Progress Series 127:235-244.

Taylor, R., R.L. Fletcher, and J.A. Raven. 2001. Preliminary studies on the growth of selected 'green tide' algae in laboratory culture: Effects of irradiance, temperature, salinity and nutrients on growth rate. Botanica Marina 44:327-336.

Thayer, G.W., D.A. Wolfe, and R.B. Williams. 1975. The impact of man on seagrass systems. American Scientist 8:288-296.

Thom, R.M., A.B. Borde, S. Rumrill, D.L. Woodruff, G.D. Williams, J.A. Southard, and S.L. Sargeant. 2003. Factors influencing spatial and annual variability in eelgrass (*Zostera marina* L.) meadows in Willapa Bay, Washington, and Coos Bay, Oregon, estuaries. Estuaries 26(4B):1117-1129.

Thom, R.M., H.L. Diefenderfer, J. Vavrinec, and A.B. Borde. 2012. Restoring resiliency: Case studies from Pacific Northwest estuarine eelgrass (*Zostera marina* L.) ecosystems. Estuaries and Coasts 35:78-91.

Thom, R.M., K.E. Buenau, C. Judd, and V.I. Cullinan. 2011. Eelgrass (*Zoster marina* L.) stressors in Puget Sound. Washington State Department of Natural Resources, Nearshore Habitat Program, Olympia, Washington.

van Montfrans, J., R.L. Wetzel, and R.J. Orth. 1984. Epiphyte-grazer relationships in seagrass meadows: Consequences for seagrass growth and production. Estuaries. 7(4 part A): 289-309.

Wagner, E., B.R. Dumbauld, S.D. Hacker, A.C. Trimble, L.M. Wisehart, and J.L. Ruesink. 2012. Density-dependent effects of an introduced oyster, *Crassostrea gigas*, on a native intertidal seagrass, *Zostera marina*. Marine Ecology Progress Series 468:149-160.

Waldbusser, G.G., E.P. Volgt, H. Bergschneider, M.A. Green, and R.I.E. Newell. 2011. Biocalcification in the eastern oyster (*Crassostrea virginica*) in relation to long-term trends in Chesapeake Bay pH. Estuaries and Coasts 34:221-231.

Ward, D.H., A. Morton, T.L. Tibbitts, D.C. Douglas, and E. Carrera-Gonzalez. 2003. Long-term change in eelgrass distribution at Bahía San Quintín, Baja California, Mexico, using satellite imagery. Estuaries 26(6):1529-1539.

Wheat, E. and J.L. Ruesink. 2013. Commercially cultured oysters (*Crassostrea gigas*) exert top-down control on intertidal pelagic resources in Willapa Bay, Washington, USA, Journal of Sea Research 81:33-39.

Williams, S.L. and M.H. Ruckelshaus. 1993. Effects of nitrogen availability and herbivory on eelgrass (*Zostera marina*) and epiphytes. Ecology 74: 904-918.

Wisehart, L.M., J.L. Ruesink, S.D. Hacker, and B.R. Dumbauld. 2007. Importance of eelgrass early life history stages in response to oyster aquaculture disturbance. Marine Ecology Progress Series 344:71-80.

WSBRP (Washington State Blue Ribbon Panel on Ocean Acidification). 2012. Ocean acidification: From knowledge to action, Washington State's Strategic Response. H. Adelsman and L. Whitely Binder (eds). Washington Department of Ecology, Olympia, Washington. Publication no. 12-01-015.

Zimmerman, R.C., D.L. Steller, D.G. Kohrs, and R.S. Alberte. 2001. Top-down impact through a bottom-up mechanism. *In situ* effects of limpet grazing on growth, light requirements and survival of the eelgrass *Zostera marina*. Marine Ecology Progress Series. 218: 127-140.

Exhibit B

H.T. HARVEY & ASSOCIATES
Ecological Consultants

Memorandum

Project# 3225-01

3 March 2015

To: Greg Dale, Southwest Operations Manager, Coast Seafoods Company

From: Neil Kalson, Fisheries Ecologist, and Ken Lindke, Quantitative Ecologist, H. T. Harvey & Associates

Subject: 1.5-Foot-Elevation Oyster Culture Feasibility Study—Final Report

Summary

This report presents the methods, results, and conclusions of a study carried out by H. T. Harvey & Associates (HTH), on behalf of Coast Seafoods Company, to characterize the commercial feasibility of cultivating oysters in Humboldt Bay at tidal elevations above (i.e., separate from) those suitable for eelgrass (*Zostera marina*) habitat. The study's results indicate that there was no significant difference in oyster growth, biofouling, or quality between the higher- and lower-elevation study plots. However, for one type of oyster, numbers and total weight per oyster cluster were significantly lower in the higher-elevation study plots. For another type of oyster, the total weight per cluster was significantly lower in the higher plots.

Background

On 11 June 2006, the California Coastal Commission (CCC) approved Coast Seafoods Company's Coastal Development Permit E-06-003 (Permit) to continue oyster culture operations in the coastal zone of northern Humboldt Bay. Permitted activities include "planting, growing and harvesting off-bottom oyster culture on approximately 255 acres; completing conversion (from bottom culture) and planting, growing and harvesting off-bottom oyster culture on approximately 45 acres; and operating a nursery area, floating upwelling system (FLUPSY), and wet storage floats" (CCC 2006).

Special Condition #5 of the Permit requires that Coast Seafoods Company "evaluate the feasibility of cultivating oysters at depths typically unsuitable for eelgrass (*Zostera marina*) growth (i.e., 1.5 feet above mean lower low water (MLLW)) in Humboldt Bay." Current commercial harvest plans in northern Humboldt Bay rely on planting and harvesting oysters at elevations at which eelgrass also grows; however, it is the National Marine Fisheries Service's [NMFS's] policy to recommend that there be no net loss of eelgrass habitat function (National Oceanic and Atmospheric Administration [NOAA] 2014). Eelgrass beds, and shallows that may support eelgrass, are considered special aquatic sites under the Clean Water Act Section 404(b)(1) guidelines (Title 40, Code of Federal Regulations, Part 230.43). Eelgrass also is considered an essential fish habitat area of particular concern under the Magnuson-Stevens Fishery Conservation and Management Act for some fish species that are managed under the Pacific Coast Groundfish Fishery Management Plan (Pacific Fishery Management Council [PFMC]) 2008).

Cultivation of oysters is technically possible at a wide range of elevations in Humboldt Bay, but some locations and elevations are preferred because they produce consistently high-quality oysters. Although it is accepted that oysters can be grown at higher elevations (e.g., 1.5 feet above MLLW), the extent to which oysters grown at such elevations would meet commercial expectations for growth, biofouling, survival, and quality has not been documented.

HTH (2011) developed a study plan titled *Coast Seafoods Company +1.5' Elevation Oyster Culture Feasibility Study* to evaluate the feasibility of cultivation options and thus help Coast Seafoods Company fulfill the conditions of Special Condition #5. CCC approved the study plan on 7 June 2011. In the study plan, we posed four research questions to address the feasibility of culturing oysters 1.5 feet above MLLW:

1. Is there a difference in oyster growth rates when oysters are grown 1.5 feet above MLLW versus 1.5 feet below MLLW?
2. Is there a difference in the amount of oyster biofouling when oysters are grown 1.5 feet above MLLW versus below 1.5 feet MLLW?
3. Is there a difference in oyster quality (measured as the ratio of tissue weight to tissue volume) when oysters are grown 1.5 feet above MLLW versus 1.5 feet below MLLW?
4. Is there a difference in oyster survival when oysters are grown 1.5 feet above MLLW versus 1.5 feet below MLLW?

Each of these questions addresses a critical component of commercially viable oyster culture. Growth rates directly correspond with harvest rates and production schedules. Biofouling organisms colonizing oyster clusters can affect growth rates and survival by competing with oysters for food, or may suffocate oysters. Oyster quality can affect the marketable yield from an oyster bed. Survival (related to productivity) also directly affects yield, and thus economic feasibility.

Methods

Planning Test and Control Plots

In 2012, three quarter-acre oyster beds were planted at industry-standard elevations (0.5 feet–1.0 feet above MLLW)—these served as the study's *control plots*. Three quarter-acre beds were planted at 1.5 feet–2.0 feet above MLLW to serve as *test plots*. These six plots had been randomly selected from an initial pool of 12 potential plots identified as having the correct characteristics for the study (Figure 1).

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H.T. HARVEY & ASSOCIATES
Ecological Consultants

Figure 1: Study Area

Coast Seafoods Company 1.5' Elevation Oyster Culture Feasibility Study
Northern Humboldt Bay, California (3225-02)
March 2015

In each plot, 20 longlines of Pacific oysters, (*Crassostrea gigas*), alternating with 20 longlines of Kumamoto oysters (*Crassostrea sikamea*) were planted by Coast Seafoods Company staff, using industry-standard methods. Each longline contained approximately 100 shell *cultch* (i.e., oyster shells with juvenile oysters [*spat*] attached). Each cultch had a similar number of spat attached, as determined by counting random samples and calculating the average spat count for each cultch. As spat grow on cultch, they become clusters of marketable-size oysters.

Modifications to the Study Plan

Although the study plan stated that we would estimate survival rates by monitoring the number of oysters on individual cultch/clusters over the study period, final results indicated that these data were inappropriate for analysis of survival. The data were not usable because there were several clusters that had more oysters present at the end of the study than had been counted at the beginning of the study. This circumstance resulted in estimates of survival greater than one, which is invalid. When clusters are set on lines, oysters are very small and can be difficult to see. Thus, undercounting at the beginning of the study is the most likely explanation for the spurious results; natural recruitment of oysters was considered as a possible explanation, but natural recruitment is unlikely or uncommon in Humboldt Bay (Dale pers. comm.). As a substitute for survival estimates, we used the total number of oysters present on individual clusters at the end of the study as a measure of productivity. This metric is expected to provide information similar to survival rates because the number of oysters per cluster should be strongly and positively related to survival. Also, the total number of oysters per cluster is important for evaluating production, because oysters may be sold individually.

One additional deviation from the study plan was to subsample some of the clusters for weighing and measuring individual oysters, in order to work within time constraints. Subsampling was not done systematically, but rather occurred on the rare occasion (5 out of 129 clusters) that HTH staff measuring oysters could not keep up with the speed at which Coast Seafoods personnel were processing clusters. We believe that the subsampling had no appreciable effect on the results of this study because it occurred so rarely, and most of the oysters were measured in each case.

One additional metric was not originally considered in the study plan, but was incorporated later into the study: total tissue weight per cluster. This, in addition to individual oyster weight, was measured to provide another indicator of productivity. Total weight per cluster is an important metric for commercial oyster production because oysters may be sold by weight as well as individually.

Sampling Methods

Sample Collection

Commercially grown oysters are typically harvested 18 to 30 months after planting. Accordingly, oysters were harvested after 24 months from the study plots. Ten to 17 longlines were randomly selected from each of the six study plots, and one to four clusters were randomly sampled from each of these longlines (Table 1). Each cluster was labeled and transported to Coast Seafoods Company's plant for processing.

Table 1. Numbers of Longlines and Clusters Sampled

Plot	Number of Longlines Sampled per Plot	Total Number of Clusters Sampled per Plot
Control-1	17	22
Control-2	17	22
Control-3	11	21
Test-1	10	21
Test-2	12	21
Test-3	16	22

Sample Processing

The following methods were used to measure oyster growth, biofouling, quality, and productivity:

- **Growth** was evaluated by weighing individual oysters without their shells.
- **Biofouling** was measured by visually estimating the percent of an individual cluster that was covered with biofouling organisms. Individual oysters were then separated from their cluster, and all oysters were shucked for the growth and quality measurements.
- **Quality** was defined as the ratio of tissue weight to tissue volume. The volume of tissue of individual oysters was determined by submersing the tissue in a water-filled graduated cylinder sized appropriately to the volume of the oyster meat, then measuring the volume of water that was displaced (in milliliters). The oyster tissue was then drained in a sieve and weighed to the nearest 0.1 gram.
- **Productivity** was measured as the total number of oysters per cluster and the total weight per cluster. Only live oysters were counted to find the total number of oysters per cluster. Total weight per cluster was defined as the sum of the weights (without shells) of all oysters on a cluster. For 5 of the 129 clusters that were sampled, not all oysters in the cluster were processed, owing to time constraints. Instead, they were randomly subsampled. For these five clusters, the total weight per cluster was calculated as the weight of all the processed oysters in the cluster, plus the mean weight of the processed oysters in that cluster multiplied by the number of unprocessed oysters.

Statistical Analysis

We used generalized linear mixed-effects models (GLMMs), the second-order bias adjusted version of Akaike's Information Criterion (AICc) (Burnham and Anderson 2002), and the likelihood ratio test (LRT) to determine if there were significant differences in oyster growth, biofouling, quality, total oysters per cluster, and total weight per cluster between control plots and test plots. Mixed-effects models allow us to define experimental blocking variables (e.g., plot and/or cluster) as random variables, which avoids pseudoreplication (Zuur et al. 2009). When pseudoreplication is unaccounted for, standard errors are underestimated, and predictive power is overestimated.

Data collected on individual oysters and clusters at the end of the 2-year study were used as dependent variables in model development. Prior to model fitting, quantile-quantile plots for each dependent variable were examined for each species of oyster (Kumamoto and Pacific). When data were not normally distributed, we applied data transformations to see if approximation to normality improved. When normality improved, the transformation that resulted in the closest approximation to normality was used in subsequent modeling. In a few cases, we considered error distributions other than the normal distribution. For example, a GLMM with gamma error was used for evaluating the quality of Kumamoto oysters because it fit the observed data better than a GLMM with normal error. Details of the transformation and model type (e.g., gamma) used for each dependent variable and oyster species can be found in Tables 2 and 3.

For each dependent variable and species, we defined a model with test/control as an independent variable and either plot number (for biofouling, total oysters per cluster, and total weight per cluster), or plot number and cluster number (for growth and quality) as random effects. This model was compared with a null model that excluded the test/control variable. Differences in AICc values between these two models, and *p*-values obtained from the LRT, were used to assess whether the test/control variable was significant, and thus whether there was a significant difference in the dependent variable between test and control plots. The model with the lowest AICc value fits the data best, so if the null model has a lower AICc score, then the test/control variable does not improve the fit, and we conclude that there is no difference between test and control plots. For the test/control variable to be considered important, the model that includes this variable must have the lower AICc value, and the null model should have an AICc value that is at least 1.5 lower (Burnham and Anderson 2002). This is not a definitive cut-off as in traditional hypothesis testing, so *p*-values from the LRT help us to interpret our results. *P*-values less than 0.05 indicate that there is a significant difference between test and control plots at the 95% confidence level.

Finally, model residuals for each top model were examined to ensure that model assumptions were adequately met. Plots of model residuals versus fitted values were examined to evaluate the assumption of independence for all models, and normal quantile-quantile plots of residuals were examined for models that assumed normally distributed error. All models were fit via maximum likelihood using either the function *glmer()* or *glmer.nb()* (for the number of oysters per cluster only) in package *lme4* (Bates et al. 2014) in the statistical computing environment R (R Core Team 2014).

Additional model assessment was necessary to analyze the number of oysters per cluster. A poisson mixed-effects model was initially used because it is the preferred model for count data. However, we found (using methods from Bolker et al. 2009) that the poisson mixed-effects model was overdispersed, so we instead used a negative-binomial GLMM. This is the preferred model for overdispersed count data (Bolker 2008), and it eliminated overdispersion for both species of oyster.

Results

Kumamoto Oysters

Individual oyster growth (p=0.914) and quality (p=0.440), and percent biofouling per cluster (p=0.463) did not differ significantly between test and control plots for Kumamoto oysters. AICc values differed between the test/control model and the null model by 2.07 for growth, 1.71 for quality, and 1.49 for percent biofouling. The null models had lower AICc values in all cases (Table 2). Also, there was little difference in mean growth, quality, or percent biofouling between test and control plots (Figure 2).

The number of live oysters per cluster (p=0.009) and the total weight per cluster (p=0.014) were significantly greater at control plots than at test plots at the 95% confidence level (Table 2). The AICc value for the test/control model was 4.58 lower than the null model for total oysters per cluster, and 3.82 lower than the null model for total weight per cluster. Mean total weight per cluster at control plots was 20.1 grams greater than at test plots—cluster weight in test plots thus averaged 51% of the cluster weight in control plots (Figure 2, panel E). The mean number of oysters per cluster in control plots was 5.2, versus 2.7 in the test plots; in other words, the average number of oysters in the test plots was 52% of the average number in the control plots (Figure 2, panel D). Residual analysis did not reveal any substantial violations of model assumptions, and there was no overdispersion for the negative-binomial model of the number of oysters per cluster.

Pacific Oysters

Individual oyster growth (p=0.191) and quality (p=0.588), percent biofouling (p=0.457), and the total number of oysters per cluster (p=0.078) did not differ significantly between test and control plots for Pacific oysters. AICc values differed between the test/control model and null model by 0.38 for growth, 1.81 for quality, and 1.75 for percent biofouling, with the null model having a lower AICc value in all cases. The AICc value for the test/control model was 0.81 lower than for the null model for the total number of oysters per cluster (Table 3). The mean values for control and test plots were very similar, and mean values of oyster quality were identical to the first decimal place (Figure 3, panel C).

The total weight per cluster was significantly greater at control plots at the 95% confidence level (p=0.039), and the AICc value was 1.99 lower for the test/control model. Mean total weight per cluster at control plots was 65.6 grams greater than at test plots; in other words, the cluster weight in test plots averaged about 65% of the average cluster weight in control plots (Figure 3, panel E). Residual analysis did not reveal any substantial violations of model assumptions, and there was no overdispersion for the negative-binomial model of the number of oysters per cluster.

Table 2. Modeling Results for Kumamoto Oysters Grown at High (Test) and Low (Control) Elevation Plots in Humboldt Bay, California

Attribute	Measured Variable	Transformation	Error Distribution	Random Effects	Fixed Effect	Fixed Effect Coefficient (SE)	AICc	*P*-value
Growth	Individual oyster weight (grams)	n/a	Normal	Plot, cluster	Test/control	-0.042 (0.388)	1239.88	0.914
					Null	n/a	1237.81	
Biofouling	Percent biofouling per cluster	Arcsine-square root	Normal	Plot	Test/control	-0.036 (0.049)	-14.47	0.463
					Null	n/a	-16.18	
Quality	Individual oyster weight/volume (grams/milliliters)	n/a	Gamma	Plot, cluster	Test/control	0.009 (0.012)	-429.40	0.44
					Null	n/a	-430.89	
Total oysters per cluster	Total oysters per cluster	Log	Negative-binomial	Plot	Test/control	-0.648 (0.188)	314.02	0.009
					Null	n/a	318.60	
Total weight per cluster	Sum of individual weights for all oysters in cluster (grams)	n/a	Normal	Plot	Test/control	-20.168 (6.295)	647.49	0.014
					Null	n/a	651.31	

Note: All models are generalized linear mixed-effects models, fit using maximum likelihood; p-values are based on the likelihood ratio test.

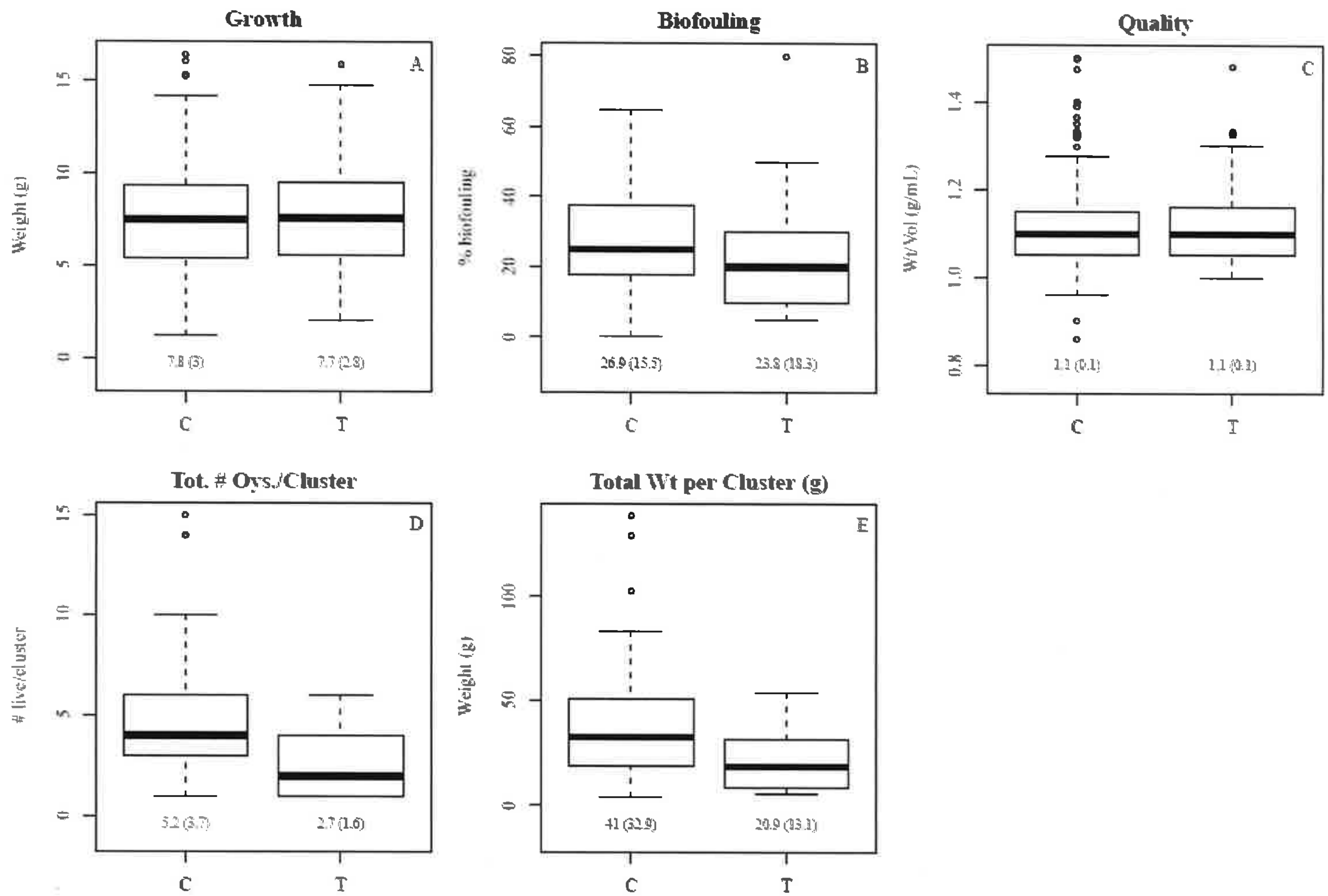

Figure 2. Box and Whisker Plots of Growth (A), Biofouling (B), Quality (C), the Number of Oysters per Cluster (D), and Total Tissue Weight per Cluster (E), for Kumamoto Oysters Grown in Humboldt Bay, California

C indicates control plots and *T* indicates test plots. Means are presented below each boxplot with standard deviations in parentheses. Thick horizontal bars represent medians, lower and upper edges of boxes represent first and third quartiles, whiskers extend to the smallest and largest observed values within 1.5 times the interquartile range, and open circles represent outliers.

Table 3. Modeling Results for Pacific Oysters Grown at High (Test) and Low (Control) Elevation Plots in Humboldt Bay, California

Attribute	Measured Variable	Transformation	Error Distribution	Random Effects	Fixed Effect	Fixed Effect Coefficient (SE)	AICc	*P*-value
Growth	Individual oyster weight (grams)	n/a	Normal	Plot, cluster	Test/control	-4.701 (3.283)	1771.59	0.191
					Null	n/a	1771.21	
Biofouling	Percent biofouling per cluster	Arcsine-square root	Normal	Plot	Test/control	-0.053 (0.071)	23.80	0.457
					Null	n/a	22.05	
Quality	Individual oyster weight/volume (grams/milliliters)	n/a	Log-normal	Plot, cluster	Test/control	-0.012 (0.021)	-365.96	0.588
					Null	n/a	-367.77	
Total oysters per cluster	Total oysters per cluster	Log	Negative-binomial	Plot	Test/control	1.323e-6 (0.162)	265.08	0.078
					Null	n/a	265.89	
Total weight per cluster	Sum of individual weights for all oysters in cluster (grams)	n/a	Normal	Plot	Test/control	-65.62 (31.15)	754.13	0.039
					Null	n/a	756.12	

Note: All models are generalized linear mixed-effects models, fit using maximum likelihood; p-values are based on the likelihood ratio test.

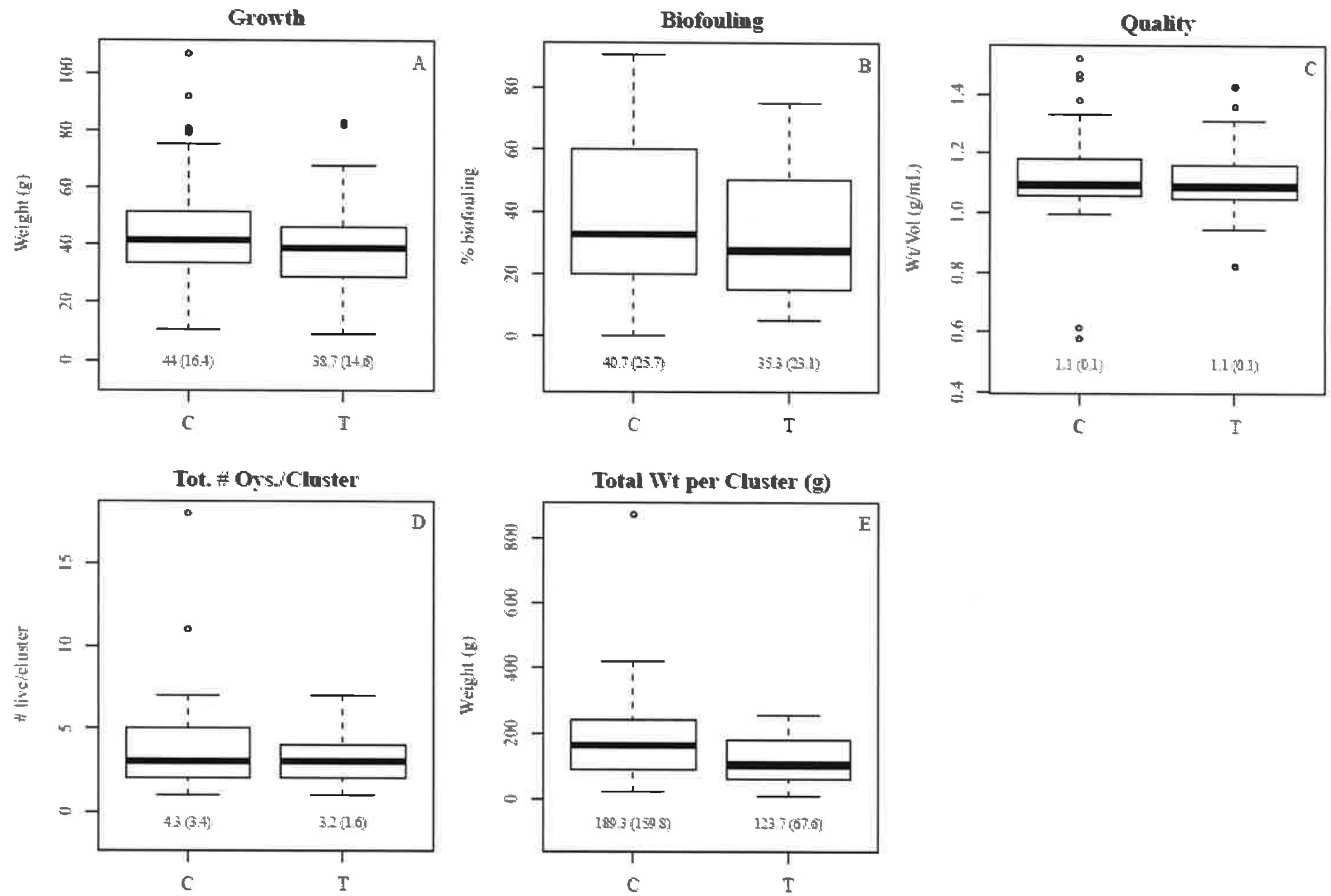

Figure 3. Box and Whisker Plots of Growth (A), Biofouling (B), Quality (C), the Number of Oysters per Cluster (D), and Total Tissue Weight per Cluster (E) for Pacific Oysters Grown in Humboldt Bay, California

C indicates control plots and *T* indicates test plots. Means are presented below each boxplot with standard deviations in parentheses. Thick horizontal bars represent medians, lower and upper edges of boxes represent first and third quartiles, whiskers extend to the smallest and largest observed values within 1.5 times the interquartile range, and open circles represent outliers.

Conclusion

This study demonstrated that cultivating oysters at elevations lower than 1.0 foot above MLLW produced more Kumamoto oysters by number and total weight, and more Pacific oysters by total weight, than cultivation 1.5 feet above MLLW. The difference observed in total weight per cluster was greater for Kumamoto oysters than for Pacific oysters. Specifically, the total weight per cluster of Kumamoto oysters in test plots was 51% of the total weight per cluster in control plots. For Pacific oysters, the total weight per cluster in test plots was 65% of the total weight per cluster in control plots. The mean total number of Kumamoto oysters per cluster in test plots was 52% of the mean total number in control plots.

Individual oyster weight did not differ between the two elevations for either species. For Kumamoto oysters, the difference in total weight per cluster can be attributed to the greater number of oysters per cluster at the low-elevation control plots. For Pacific oysters, the combination of a slightly greater average oyster weight and a slightly greater average number of oysters per cluster in the control plots (even though these effects were not statistically significant) is likely responsible for the significantly greater total weight per cluster observed in the control plots.

Other characteristics relevant to commercial cultivation (growth, biofouling, and quality) did not differ significantly between the test and control plots for either species.

References

Bates, D., M. Maechler, B. Bolker, S. Walker, R. H. B. Christensen, H. Singmann, and B. Dai. 2014. lme4: Linear mixed-effects models using Eigen and S4. R package version 1.1-7. [online]: http://CRAN.R-project.org/package=lme4.

Bolker, B. M. 2008. Ecological Models and Data in R. Princeton University Press, Princeton, New Jersey.

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J. S. White. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology and Evolution 24:127–135.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. Second edition. Springer, New York, New York.

[CCC] California Coastal Commission. 2006. Staff Report and Recommendation Coastal Development Permit Application. California Coastal Commission, San Francisco, California. CDP Application No. E-06-003. [online]: http://documents.coastal.ca.gov/reports/2006/5/Th5a-5-2006.pdf. Accessed 25 February 2015.

[HTH] H. T. Harvey & Associates. 2011. Coast Seafoods Company +1.5' Elevation Oyster Culture Feasibility Study. Coast Seafoods Company, Eureka, California.

[NOAA] National Oceanic and Atmospheric Administration. 2014. California Eelgrass Mitigation Policy and Implementing Guidelines. NOAA Fisheries, West Coast Region.

[PFMC] Pacific Fishery Management Council. 2008. Pacific Coast Groundfish Fishery Management Plan for the California, Oregon, and Washington Groundfish Fishery as Amended through Amendment 19 Including Amendment 15. NOAA Award No. NA05NMF441008. Pacific Fishery Management Council, Portland, Oregon.

R Core Team. 2014. R: A language and environment for statistical computing, v 3.1.1. R Foundation for Statistical Computing, Vienna, Austria. [online]: http://www.R-project.org/.

Zuur, A. F., E. N. Ieno, J. J. Walker, A. A. Saveliev, and G. M. Smith. 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York, New York.

Personal Communications

Dale, Greg. Southwest Operations Manager. Coast Seafoods Company. January 2015—Meeting with Adam Wagschal, Neil Kalson, and Ken Lindke of H. T. Harvey & Associates, regarding results of oyster feasibility study, project #3225-01.