Technical Memorandum

An Analysis of the Environmental Concerns Associated with Intertidal Geoduck Clam Aquaculture

Prepared by:

Jeffrey P. Fisher, Ph.D.
Karl Mueller, M.S.;
Scott Luchessa, M.S.
ENVIRON International Corporation
Seattle, WA 98104

Jonathan Davis, Ph.D.
Baywater Research
Bainbridge Island, WA

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1. Introduction and Background

Intertidal geoduck clam (*Panopea abrupta*) farming is a relatively new but significant business in Washington State. Concern has been raised recently regarding the potential environmental significance from this farming practice on marine biota and sediment processes . We have prepared this memorandum on behalf of Taylor Shellfish Farms to summarize known environmental effects from geoduck-specific studies and related shellfish aquaculture practices that result in similar levels of disturbance to the intertidal habitat. The purpose of this summary is to consider whether the effects identified from past and ongoing studies would be considered "adverse" or "significant" in a SEPA context, or whether the effects would be considered benign or beneficial with respect to the changes that could be expected from permitting a geoduck farm. In making a threshold determination, the SEPA responsible official must, among other things, "(b) Determine if the proposal is likely to have a probable significant adverse environmental impact, based on the proposed action, the information in the checklist (WAC 197-11-960), and any additional information furnished under WAC 197-11-335 and 197-11-350." WAC 197-11-330. "Probable" means likely or reasonably likely to occur, as in "a reasonable probability of more than a moderate effect on the quality of the environment" (see WAC 197-11-794). Probable is used to distinguish likely impacts from those that merely have a possibility of occurring, but are remote or speculative. This is not meant as a strict statistical probability test. WAC 197-11-782. "Significant" as used in the SEPA context means a reasonable likelihood of more than a moderate adverse impact on environmental quality (emphasis added).

With respect to geoduck-specific studies, we report on preliminary results from several ongoing studies to evaluate potential environmental changes where the culture of geoduck is practiced. We compare sediment firmness between farmed and unfarmed areas, and within a harvest zone (i.e., pre- vs. post-harvest sediment firmness in the same area). We also examine nutrients and the sediment plume generated during harvest, and discuss the potential significance of the results found with respect to water quality and fisheries resources. Finally, we compare benthic richness and abundance at several farm sites relative to appropriate reference sites. In addition to these studies, we consider the vast literature addressing the environmental effects of shellfish aquaculture, and compare results from some of these studies, often conducted at densities far greater than the culture of geoduck, with metrics for environmental impact that are largely accepted amongst the scientific community to reflect ecological integrity and health.

In developing this memorandum, we have also attempted to address many of the primary concerns raised by Dethier et al. (2007) in their brief to the Pierce County on behalf of Bricklin and Associates. These authors raised concerns regarding the impacts from the deployment and retrieval of shellfish aquaculture equipment in the intertidal zone, the harvesting of intertidal geoduck clams with jets of pressurized seawater, the potential effects of culturing geoduck clams in the vicinity of wild geoduck clams, and the availability of scientifically-relevant information as it pertains to intertidal geoduck clam aquaculture. While they raised several interesting research questions, results from our research on several farms, from extensive work conducted in Canada on geoduck farming impacts, from past environmental documents evaluating the potential effects from geoduck harvesting and farming, from the vast database of relevant studies on other forms of shellfish aquaculture in similar habitats, and from studies documenting the numerous ecological benefits provided by the culture of shellfish around the world, support a determination of environmental insignificance at the scale geoduck culture practiced by Taylor Shellfish Farms in Puget Sound and feasibly permitted at the County level.

2. Impacts of Geoduck Aquaculture Seeding and Grow-out on Biological Resources: the Role of Aquaculture Gear as Habitat for Fish and Invertebrates

a. Overview of Seeding and Growout Processes

During the seeding process of intertidal geoduck clam aquaculture, structured habitat is added to the intertidal zone in the form of protective PVC tubes and mesh netting. The PVC tubes (diameter ~ 4 to 6 inch, length ~ 8 to 12 inches) are sunk into the substrate to protect the out-planted geoduck clam seed from potential predators. Three to four seed clams are placed inside each tube, which extends about eight to 10 cm above the substrate, and the tubes may be positioned as close as 1/sq-ft. The PVC tubes are then netted individually or collectively with a canopy net to provide further protection against predation. Within one to two growing seasons the geoduck clams reach a size at which refuge against predation is no longer needed, and the PVC tubes and predator netting are removed. During this 1 to 2-year period of tube placement, the tube field creates a three dimensional array that is rapidly colonized by marine plants and animals that would otherwise not persist in the dynamic sand flat environments where geoduck are cultured. The text below considers the potential significance of this phase of the geoduck operation on biological resources and physical processes.

b. Potential Impacts to Fisheries Resources

In Puget Sound and elsewhere, the addition of structured habitat, artificial or otherwise, to homogenous marine habitats like sand and mud has long been recognized to increase the types and numbers of colonizing fish, invertebrates, and aquatic plants in a given area (Iversen and Bannerot 1984; Buckley and Hueckel 1985; Hueckel and Buckley 1987; Gregg 1995; Sargent et al. 2006). For example, Buckley (1985) examined the sustained aggregation of *recreationally important* fishes in an artificial reef placed on a sandy bottom off Gedney Island, WA, and examined the rate of development towards a natural temperate reef. In addition to a sustained aggregation of non-game prey fishes (primarily shiner and striped perch), they found that anglers fishing over the reef structure retained 2.4 times more fish/hour than anglers fishing nearby natural waters without the structured habitat. They proferred that the increase in recreationally important fishes was the result of successional biota colonization and development (algae and sessile invertebrates) on the reef structure that provided an alternative prey source when other normal forage fish were in cyclic low abundance. Such reef structures also provide refuge from predation and the enhancement of the availability of food for other marine organisms of no recreational interest—thereby enhancing local Hueckel and Stayton 1982; Hueckel and Buckley 1987). When the habitat provided by such reef structures is otherwise limited in the water body, artificial reefs can result in sustained population increases in recreationally and commercially important fisheries resources through their ultimate effects on the viability of individuals within a broader population; this “halo effect” underpins, in large measure, the initiative behind ‘marine protective areas’ to enhance fishery resources throughout the World Ocean today. In brief, the structures created provide for secure substrate, which in turn facilitates the settlement of ‘epibiota’ (algae, barnacles, etc.) on the surface of the structure, and the consequent development of a more stable biological community that associate with that biota and the food and refuge it provides.

Like artificial reefs composed of concrete blocks, metal lattices, sunken vessels, etc., several studies have shown that the gear associated with shellfish aquaculture provides the basis for similar 'biogenic' habitat services and ecological benefits. For example, Laffargue et al. (2006) demonstrated that the flatfish, *Solea solea*, displays a strong affinity for oyster-rearing structures when resting or seeking refuge during the day. Tallman and Forrester (2007) showed that oyster grow-out cages provided valuable habitat for economically valuable finfishes in Narragansett Bay, Rhode Island and suggested that these structures be considered as part of future habitat restoration programs for the exploited species. Dealteris et al. (2004) found that shellfish aquaculture gear supported more organisms, had higher species richness and higher species diversity than non-vegetated seabed, and was similar or superior to eelgrass (*Zostera marina*) or submerged aquatic vegetation habitat. Likewise, Meyer and Townsend (2000) showed that created oyster reefs had a higher number of fish, and molluscan, and crustacean invertebrate species than adjacent natural reefs. O'Beirn et al. (2004) reported a wide variety and large number of marine organisms associated with the mesh bags of cultured oysters in Virginia. These included worms, mollusks, crustaceans, and fish. And finally, Powers et al. (2007) documented that the macroalgal growth on protective netting placed over hard clam (*Mercenaria mercenaria*) aquaculture sites supported elevated densities of mobile invertebrates and juvenile fishes similar to natural seagrass (*Z. marina* and *Halodule wrightii*) habitats. It is highly notable that in all of these studies, the underlying physical habitat into which the aquaculture gear was placed, was sand flat, which is the preferred habitat for geoduck.

(i) Salmonid Interactions with Geoduck Aquaculture

So, what about salmon, the aquatic species of greatest concern in Puget Sound? Dethier et al. (2007) have asked whether critical corridors for migrating aquatic species, such as Pacific salmon (*Oncorhynchus* spp.), might be disrupted by the addition of PVC tubes to intertidal areas. While an interesting question, there is no evidence to support this contention, and a substantial body of information would suggest this type of impact is extremely unlikely. The fundamental question as to whether salmon could be affected is, "how"? That is, what about a submerged tube field *could* affect a critical corridor, and how exactly? Is such an interruption feasible given what we know about juvenile salmonid behaviors in Puget Sound and elsewhere? We know that nearshore salmonid migrations can be altered by overwater structures that create shade or otherwise obstruct their path along the shore (*see* Nightingale and Simenstad 2001 for review). We also know that piers and vertical bulkheads that extend below the ordinary high water line, such as those prevalent along the shorelines of many of the property owners along Puget Sound, can force juvenile salmonids away from shore where predation may be greater, and disrupt their schooling. In the case of a tube field, the vertical relief created above the sediment surface is, at most, four inches. The tubes, covered with predator netting, are rapidly encrusted with epibiota that creates a reef-type structure and a biogenic source for associated food organisms of use to juvenile salmonids. Based on recent radiotelemetry studies conducted by the USFWS (R. Tabor, pers. Communication, Lacey, WA), juvenile salmonids will orient to their perception of the bottom depth—whether that be relatively featureless sand or the top of a canopy net that is colonized with a diverse algal community 4 inches above that underlying sand layer. Hence, any minor change such as is created by a tube field will not be recognized in the manner that an overwater structure, or a structure that extends through the water column, would be. Because there is no vertical or overwater structure associated with geoduck farming that impedes the path of salmon migration, the argument that their migratory path would be disrupted is tenuous at best.

Specific studies evaluating the use of geoduck farms by salmonids and other fish are ongoing; however, based on shellfish aquaculture studies in similar sand flat habitats, the impacts from a tube field are likely beneficial to salmonids because of the additional food resources obtainable from the biogenic habitat created. Recent studies have shown that small fish are capable of maneuvering through submerged shellfish aquaculture gear (O'Beirn et al. 2004; Tallman and Forrester 2007). Laffargue et al. (2006) reported that the swimming behavior of the flatfish, *Solea solea*, was not impacted by oyster-trestle cultivation installations in France. Furthermore, Ward et al. (1994) concluded that nearshore structures in the lower Willamette River near Portland, Oregon presented few risks to migrating juvenile Pacific salmon. Ostensibly, the greatest effect on the distribution, abundance, and behavior of Puget Sound fishes occurs when shoreline modifications or man-made structures extend from the shoreline through the intertidal zone and *into* the subtidal zone (Toft et al. 2007). Intertidal geoduck clam aquaculture does not span the entire intertidal zone. It generally occurs in a relatively narrow band from approximately + 3 to – 2 ft mean lower low water (MLLW). And while it is true that *overwater* structures, e.g., docks, are known to alter movements of migrating fishes (Toft et al. 2007), there is no evidence that the low vertical relief (eight to 10 cm) of the short PVC tubes inserted into the substrate of the intertidal zone will adversely affect the movement of migrating aquatic species. Depending on the tidal cycle, fish can easily swim over, around, or through these structures, if necessary.

(ii) Forage Fish Interactions

According to Carlson (1980), schooling forage fish like Pacific herring, sand lance and surf smelt use intertidal habitat only for spawning purposes, otherwise the fish forage in deeper nearshore and offshore waters. These foraging habitats are not recognized as limiting to any of the forage species with spatial overlap to intertidal shellfish aquaculture: Pacific herring, sand lance, and surf smelt. However, the intertidal habitats in some of Washington's waters where shellfish culture are used for spawning by these species are increasingly limited because of impacts from shoreline development (bulkheads, etc.) that often extend below the ordinary high water.

These species have distinct requirements for spawning, with sand lance and surf smelt spawning on the beach at tidal heights significantly above where geoduck culture operations occur (http://wdfw.wa.gov/fish/forage/smelt. htm; http://wdfw.wa.gov/fish/forage/lance.htm). In general, these species prefer course sand to pea-gravel sized sediments for their beach spawning. Surf smelt spawn at tidal elevations above + 7 in coarse sand or pea gravel-sized substrate (1 to 7 mm) that is generally larger than the finer sand that predominates in the zones where geoduck are primarily cultured. Sand lance may spawn slightly lower, from +5 to mean high water, and will utilize substrate size of a slightly broader size class—up to 3 cm. Both of these species, based on the tidal height separation, will avoid significant direct interactions with geoduck farming practices. Thus, while farming clearly occurs along beaches also used by these spawning forage fish, direct physical impacts are avoided. Herring, in contrast, will spawn within the intertidal range where geoduck culture occurs (-3 to +2 MLLW) but only when structured habitat such as eelgrass, macroalgae or, ironically, aquaculture gear or similar inert materials are provided. If a geoduck clam aquaculture site is located in a herring spawning area that is documented by WDFW, then, according to Taylor Shellfish Farms' Codes of Practice, Taylor's employees are required to avoid disruptive farming activities during the herring spawning and incubation periods. Taylor's Code of Practice further stipulate that no tubes or netting are to be removed if herring spawn is found on aquaculture gear. In addition it should be recognized that the placement and removal of tubes is an activity that generally occurs during the times of year when daylight minus tides are available for crews to work the beach—late April through September. These time periods inherently miss the predominant February/March spawning period of herring in the Pacific Northwest (Lassuy 1989); thus, the potential from the removal of gear upon which herring have spawned is essentially avoided. In light of these Codes of Practice and timing consideration, it is likely that the overall effects of geoduck farming on herring are positive in that the aquaculture gear provides spawning habitat in sand flat areas where no such habitat previously existed.

Other mechanisms for potential impacts to forage fish that have been suggested include the potential crushing of eggs deposited on the upper beach by workers accessing lower tidal elevations from adjacent uplands, and the effects that geoduck may have in competing with forage fish for food through their filtration of suspended plankton. With the former potential impact, it should be recognized that the vast majority of site access is via boat and impacts to the upper beach are thereby avoided. Where sites are accessed from land, paths are followed to minimize the footprint of this traffic and cumulative impacts are not likely any more significant than typical foot traffic by public and private parties accessing the beach for recreational purposes because the highly infrequent need for growers to access sites over the 4-6 year culture cycle. The subject of impacts from geoduck filtration and associated competition are thoroughly discussed in section 2d, but the beneficial aspects on habitat complexity will also likely benefit forage fishes.

c. Impacts on Benthic Biodiversity

Recent studies of the effects of geoduck clam seeding on the biota of the intertidal zone suggest that impacts are minimal and short-lived. Indeed, benthic communities might well remain unchanged or become enhanced by the process of seeding geoduck clams. For example, in south Puget Sound, Fleece et al. (2004) observed few differences in benthic community structure between geoduck clam aquaculture sites and control sites (Tables 1 – 3). In two of three aquaculture sites surveyed by scuba divers at high tide, Fleece et al. (2004) found that species richness of macroinvertebrates (the number of species observed) was higher in seeded areas compared to the control or unseeded areas. Likewise, Pearce et al. (2007) reported an increase in species richness of benthic infauna in sediment cores two months after geoduck clams were seeded at aquaculture sites in British Columbia, Canada (Figure 1). Within six months of seeding geoduck clams, Pearce et al. (2007) found that species richness of benthic infauna had returned to baseline levels. A similar pattern was observed when the researchers examined the density or number of biota per sediment core at the aquaculture site. Two months after seeding the geoduck clams, the average number of individual organisms found on-site increased dramatically (Figure 2). By six months post-seeding, the number had decreased substantially, but was within the lower range of values observed at the aquaculture site prior to seeding (Pearce et al. 2007). Note that the slight reduction in species density and diversity six weeks post-seeding (Figures 1 and 2) is likely attributable to the different seasons during which the samples were taken (July at seeding, January at six months post-seeding).

Increased densities of benthic infauna at intertidal geoduck clam aquaculture sites may persist even after removing the protective PVC tubes and netting. For example, at one aquaculture site in southern Puget Sound, the average number of infaunal benthic organisms per sediment core from an unprotected seeded area was greater than the density of infaunal benthic organisms found in a reference area located outside of the aquaculture site (Table 4).

Table 1. Diver observations at the Seattle Shellfish Hunter Point intertidal geoduck clam aquaculture site in south Puget Sound on December 3 – 5, 2003. Values are number of organisms per transect (Fleece et al. 2004).

		Transect		
Common name	Species or family	Geoduck bed (150 ft)	Geoduck bed (150 ft)	Control (150 ft)
Red rock crab	*Cancer productus*	1	0	0
Graceful crab	*C. gracilis*	3	4	1
Decorator crab	Majidae	16	12	0
Hermit crab	Paguridae	159	124	179
Flat shrimp	Crangonidae	0	0	0
Humped shrimp	Hippolytidae	0	0	0
Horse clam	*Tresus nuttalli*	0	1	1
Geoduck clam	*Panopea abrupta*	1	0	0
Moon snail	*Polinices lewisii*	1	1	1
Five-arm seastar	*Pisaster* spp.	15	4	1
Sunflower star	*Pycnopodia helianthoides*	2	0	0
Sand dollar	*Dendraster excentricus*	0	0	0
Total individuals		198	146	183
Number of taxa		8	6	5

Table 2. Diver observations at an intertidal geoduck clam aquaculture site in North Bay, Case Inlet (south Puget Sound) on December 3 – 5, 2003. Values are number of organisms per transect (Fleece et al. 2004).

		Transect			
Common name	Species or family	Geoduck bed (150 ft)	Control (150 ft)	Eelgrass below geoduck bed (50 ft)	Eelgrass below control (50 ft)
Red rock crab	*Cancer productus*	0	0	0	0
Graceful crab	*C. gracilis*	3	7	0	3
Decorator crab	Majidae	1	0	0	1
Hermit crab	Paguridae	135	80	50	70
Flat shrimp	Crangonidae	0	0	0	0
Humped shrimp	Hippolytidae	0	0	1	0
Horse clam	*Tresus nuttalli*	0	0	0	0
Geoduck clam	*Panopea abrupta*	0	0	0	0
Moon snail	*Polinices lewisii*	0	3	0	0
Five-arm seastar	*Pisaster* spp.	2	24	0	0
Sunflower star	*Pycnopodia helianthoides*	0	0	0	0
Sand dollar	*Dendraster excentricus*	189	150	3	0
Total individuals		330	264	54	74
Number of taxa		5	5	3	3

Table 3. Diver observations at an intertidal geoduck clam aquaculture site off Stretch Island in Case Inlet, south Puget Sound on December 3 – 5, 2003. Values are number of organisms per transect (Fleece et al. 2004).

		Transect	
Common name	Species or family	Geoduck bed (150 ft)	Control (150 ft)
Red rock crab	*Cancer productus*	0	0
Graceful crab	*C. gracilis*	0	0
Decorator crab	Majidae	0	0
Hermit crab	Paguridae	10	10
Flat shrimp	Crangonidae	0	1
Humped shrimp	Hippolytidae	0	0
Horse clam	*Tresus nuttalli*	0	1
Geoduck clam	*Panopea abrupta*	2	0
Moon snail	*Polinices lewisii*	1	0
Five-arm seastar	*Pisaster* spp.	0	0
Sunflower star	*Pycnopodia helianthoides*	0	0
Sand dollar	*Dendraster excentricus*	0	0
Total individuals		13	12
Number of taxa		3	3

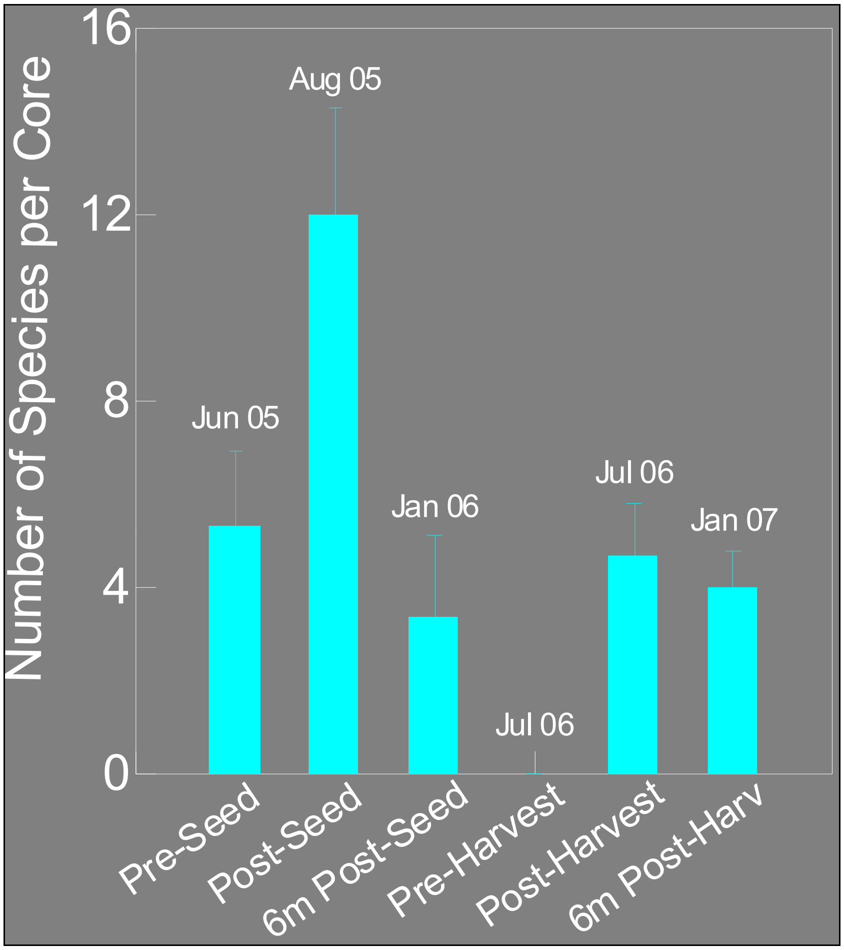

Figure 1. Species richness of benthic fauna at an intertidal geoduck clam aquaculture site in British Columbia, Canada. Data are means (+/- SE) from sediment cores (Pearce et al. 2007). Note, data from July 06 have yet to be processed and therefore show in the graphs as "O."

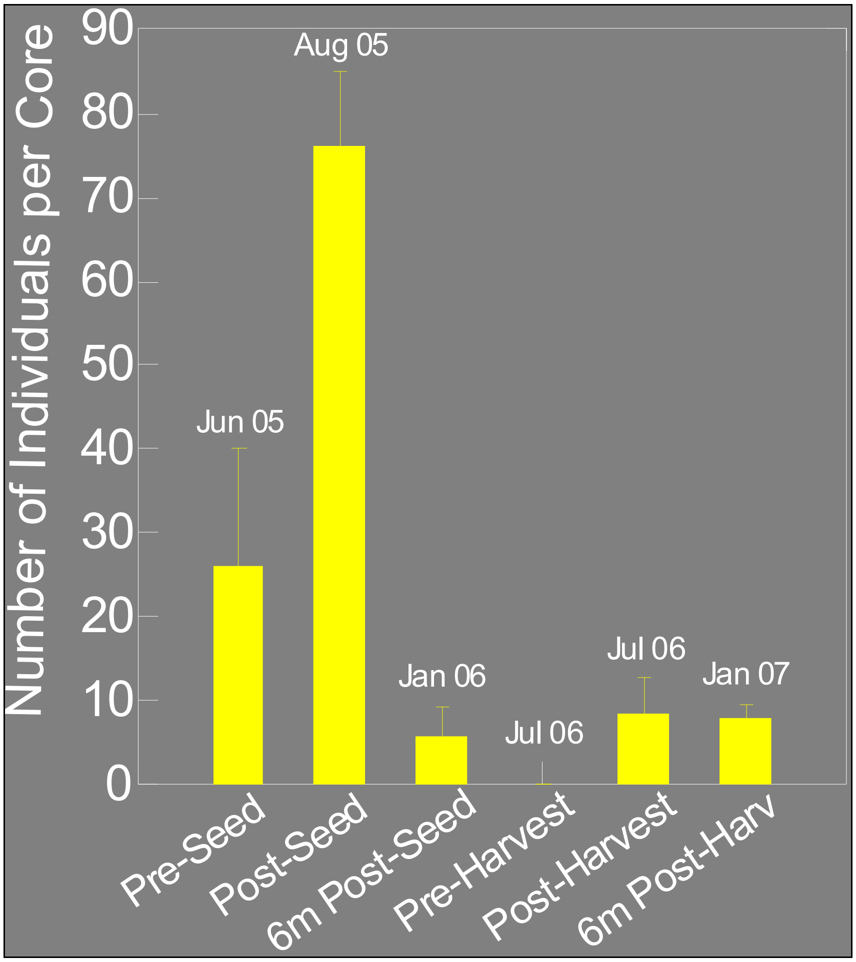

Figure 2. Density of benthic fauna at an intertidal geoduck clam aquaculture site in British Columbia, Canada. Data are means (+/-SE) from sediment cores (Pearce et al. 2007). Note: data from July 06 have not yet been processed and therefore show in the graph as "O."

Table 4. Preliminary results of invertebrate sampling at the Taylor Shellfish Foss Farm intertidal geoduck clam aquaculture site in Case Inlet, south Puget Sound on November 27-28, 2007*.

Type of organism	Seeded area (11 cores)	Harvested area (11 cores)	Reference area (10 cores)
Annelida	7.2	0.9	3.3
Bivalvia	1.3	0.8	0.3
Brachyura	0	0	0.1
Crustacea	0.5	0.2	0.1
Dendraster excentricus	7.9	11.6	8.9
Gastropoda	1.3	0.7	0
Unidentified	3.2	0.4	2.4

*There were no protective PVC tubes or nets in the seeded area. The harvest occurred less than three months before the sample date in this location. The reference area or control was located adjacent to but outside of the aquaculture site. Values are the average number of organisms per core sample (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

d. Potential Impacts of Planting Density During Grow-Out Relationships to Ecological Carrying Capacity

One of the most common criticisms leveled at the practice of geoduck farming has centered around the potential for farmed geoduck to exceed the ecological carrying capacity of Puget Sound’s waters through their direct grazing (filtration) of the phytoplankton biomass, and through the indirect effects of that grazing on biodeposition of feces and pseudofeces in intertidal sediments. Clearly, the densities at which shellfish are cultured in the intertidal zone can affect ecosystem function (Bartoli et al. 2001; Beadman et al. 2004), but this generally occurs at densities considerably higher than those used for intertidal geoduck clam aquaculture.

There is no evidence that existing shellfish aquaculture conditions in the Pacific Northwest are compromising ecosystem function by exceeding carrying capacity. In fact, Whiteley and Bendell-Young (2007) recently demonstrated that there were no significant differences in the densities of 25 clam species sampled among farm and reference sites in Barkley Sound, Baynes Sound, and Desolation Sound, British Columbia, Canada. Furthermore, the bivalve species composition was not significantly different between farm and reference sites. Not surprisingly, the cultured Manila clam (*Tapes philippinarum*) was the only species found in higher abundance at lower intertidal areas on farm sites compared to reference sites. Given the similar abundance and variety of bivalves at farm and reference sites, it is reasonable to interpret Whiteley and Bendell-Young’s (2007) findings as an indication of no loss in ecosystem function at the farm sites during the period they were studied.

Although there are no data that would suggest geoduck farming is affecting ecological carrying capacity at the densities at which they are cultured, for this memo we have attempted to characterize filtration and biodeposition rates in geoduck, and compare these estimates against cultured oyster and manila clam densities in Southern Puget Sound. These specific comparisons are relevant because, like geoduck, the culture of clams and oysters in Southern Puget Sound is sediment-associated, and three dimensional structures for maximizing biomass density (e.g., racks) are not used. Biomass estimates for the species of interest represent values (i.e., number of animals and live weight) typical of commercially farmed intertidal areas in southern Puget Sound and were obtained through discussion with key individuals associated with the shellfish culture industry in Puget Sound over the last thirty to forty years. Values referenced in this discussion represent optimal shellfish densities that are observed at harvest and represent tradeoffs between production potential (i.e., yield as a function of growth and survivorship) and time to harvest. Values for different parts of Puget Sound are likely to be somewhat lower than those reported here due mainly to the very high primary production that characterizes southern Puget Sound and supports the very high shellfish production rates relative to other areas.

Oyster culture in southern Puget Sound is focused almost exclusively on the production of Pacific oysters (*Crassostrea gigas*), a large, fast growing and hardy oyster that forms the mainstay of the west coast shellfish industry. Pacific oysters are typically farmed in southern Puget Sound in one of two ways. The predominant culture technique involves placement of individual Pacific oyster shells or cultch onto inertidal substrates at high density (50-60 per square M). Each piece of oyster cultch harbours typically 2-12 small, juvenile oysters that had previously settled onto the cultch at the conclusion of their larval life cycle, either from wild or hatchery larval production. Cultch containing juvenile oysters is placed onto barges and subsequently sprayed via high pressure hose onto intertidal substrates at high tide. Cultch containing oysters are left for 2-3 years until harvest. Yields at harvest are in the range of 1.2 million oysters per acre with a biomass of 180,000 lbs (81,818 kg).

The second predominant culture method for rearing pacific oysters is single production. In this case, single pacific oyster seed (1-2cm) is spread onto intertidal flats by hand or barge at high water as above but densities of shellfish are maintained at a significantly lower level. Oysters grow for about a year until they are harvested by hand at low tide as extra smalls and smalls for half shell markets. Typical yields at harvest for southern Puget Sound sites are in the range of 240,000 oysters per acre with a biomass of about 36,000 lbs (16,364 kg).

Manila clam culture on southern Puget Sound beaches is relatively intensive with a mix of wild and seed cultivated in beds. Clam seed is spread out over beds by hand and plots are initially covered with predator netting for a period of time that can include the entire growout cycle (in cases where crab, perch and bird predation are intense). There are typically three yearclasses of clams contained on clam beaches. These include young of the year, one year old seed and 2 year clams that are harvested. A high yielding clam beach in southern Puget Sound will contain about 1 million clams per acre with a biomass associated with the annual harvestable portion (2 year old's) of about 50,000 lbs (22,727 kg) per acre per year.

Geoduck culture techniques as practiced on southern Puget Sound beaches, including Taylor's Farms can vary according to site and conditions. Taylor uses a variety of techniques mainly focused on different modes of predator protection and include planting in tubes that are covered with a single area net, or in the case of geoduck farms located in proximity to eagle nests, individual nets over individual tubes. Regardless of the type of predator protection, the density of clams planted is similar across the site and approximated 14 clams per square meter. This density is indeed higher than the average density of geoduck calculated from DNR's subtidal census work (2.2 clams/m^2), but well within the range of densities that have been found in wild geoduck beds (In this case, geoduck clam seed was added to the PVC tubes and predator netting added such that final yield at harvest would approximate at least 35,000 harvestable (800 g) clams per acre over the 4-6 year growout cycle. This represents a biomass at harvest of about 60,000 lbs per acre (27,273 kg).

It should be noted that the biomass estimates described above are based on live weight at harvest. It is clear that the biomass of oysters is biased towards shell production compared to either manila clams or geoducks. Soft tissue production in oysters comprises 10-15% of the live weight, in geoduck clams the ratio is much higher at about 80% of the live weight. On the other hand, in terms of habitat value, the presence of oysters may contribute to epibenthic diversity by contributing biological structure to intertidal habits. As discussed above, this is a function likely shared by the presence of predator protection for geoduck culture.

In an attempt to estimate the potential contribution of cultured shellfish on the water column, we are providing a summary of primary results based on a variety of studies that have directly estimated filtration rates of cultured oysters and geoducks in the field under ambient conditions. Many of these studies are contained in the primary literature and rates generally conformed for oysters conducting feeding studies under in situ conditions. *These values should be treated as preliminary estimates as individual rates for oysters and other bivalves can and will vary greatly based on body mass, temperature, food supply, reproductive condition and a host of other environmental factors. These estimates are made only for the purpose of providing an indication of the likely magnitude of impact or effect on water column dynamics associated with a shellfish farm.*

Estimates of oyster feeding for these studies (J. Davis, unpublished data) were made directly utilizing the quantitative biodeposition approach where the rate of biodeposits is measured over a period of time in concert with simultaneous measurements of the total organic and inorganic seston content in overlying waters. In this case, the inorganic fraction of biodeposits produced (as feces or pseudofeces) is used as a tracer under the assumption that only organic components of the seston are retained by the animal during feeding and digestion and that the inorganic fraction simply passes through the animal. The ratio of organic to inorganic materials in the food is compared to the same ratio in the biodeposits and the difference attributed to uptake and the total inorganic fraction that passes through the animal is then a direct measure of seston uptake.

For geoduck clams, very little information on feeding rates exists in the literature. In 2007, through a contract with the Hood Canal Salmon Enhancement Group, J. Davis made some preliminary estimates of geoduck feeding rates based on the quantitative biodeposition approach. These studies remain underway through 2008 but provide some preliminary information for both filtration and biodeposit production rates as described below.

Medium Pacific Oysters:	70 L filtration per individual/day
	187 mg biodeposit per individual/day
Geoduck Clams (800g live wt)	100 L per individual per day
	500 mg per individual per day

Using these estimates for filtration and biodeposition, a general estimate and comparison for the cumulative effects of oysters and geoducks can be made.

Based on the production estimates made above for oyster and clam aquaculture, the suggested contribution to water column filtration and biodeposition rates equate as follows:

Cluster Oyster Culture	100 million Liters per acre per day 215 kg (dry mass) per acre per day
Single Oyster Culture	20 million Liters per acre per day 43 kg (dry mass) per acre per day
Geoduck Culture	4.6 million Liters per acre per day 17.5 kg (dry mass) per acre per day

What is striking about these generalized estimates is that the effects on surrounding waters in terms of filtration and biodeposition rates generated by cultured shellfish are potentially very different for geoducks and oysters. At biomass levels characteristic of southern Puget Sound, growing oysters in clusters may generate about 20x the filtration activity and about 12x the biodeposit rate than that predicted for geoducks at the commercial densities described above. Part of the explanation for this observation is that the geoduck culture cycle of 4-6 years is significantly longer than that observed for oyster production (1-3 years). In other words, though individual geoducks are large at harvest (800 grams each) and comprised of about 80% soft tissues, it takes significantly longer to reach this harvestable size compared to oysters that take a fraction of the time and may be grown at much higher density overall. The same holds true for manila clams in southern Puget Sound.

In this case, as described above, 50,000 lbs of clams may be harvested annually from good clam ground compared to 60,000 lbs of geoduck harvested once every 4-6 years. The overall result is that oyster and manila clam culture can be considered much more intensive forms of aquaculture compared to geoduck culture, overall. Public perception probably counters these findings simply because of the gear associated with the culture of geoduck and its visual signature, but the associated gear has nothing to do with the potential effects of the animals on filtration and carrying capacity questions raised by some members of the public. The biofouling of the PVC tubes and nets gives the indication that the culture activity is very intensive when this in fact is not the case based on the biomass produced and the biology of the bivalves being cultured.

No significant impact on ecological carrying capacity has been demonstrated over decades of the culture of oysters and clams in the Puget Sound, even though, as demonstrated in this discussion, their potential effects and ecological benefits on plankton populations and water quality (through filtration), and sediment (through biodeposition) are substantially greater than is realized from the marginal scale at which geoduck are cultured in Puget Sound. It is therefore extremely unlikely that geoduck culture has any significant impact on ecological carrying capacity.

e. Impacts of Protective PVC Tubes on Sediment Dynamics

Puget Sound beaches are dynamic places. Wind blowing over the water surface creates and energizes waves. Currents form running parallel to shore. The energy from waves is expended on the shoreline and works to erode, transport, and deposit beach sediment (Terich 1987), the three processes constituting a drift cell. Drift cell dynamics occur over a wide range of spatial and temporal scales. Puget Sound contains hundreds of drift cells ranging from tens of meters in length to tens of kilometers (Finlayson and Shipman 2003). Here, the drift cells are mainly driven by storm events (Terich 1987).

Concerns have been raised over the placement of the protective PVC tubes in the intertidal zone during the geoduck clam seeding process. Dethier et al. (2007) questioned what the effects of the protective PVC tubes might be on drift cell dynamics in nearshore areas. If the PVC tubes were affecting drift cell dynamics in a significant way, one would expect to see changes in sediment composition both inside and outside the PVC tube field. While it is possible that scouring occurs around the PVC tubes (Sumer and Fredsøe 1998), redistributing sediments at the geoduck aquaculture site, and while it is also possible that the PVC tubes scatter some of the energy from incoming waves (Yilmaz 1998) potentially affecting sediment distribution, Pearce et al. (2007) demonstrated that, using grain size analysis, there were no significant differences in the composition of sediments before or up to 12 months after out-planting geoduck clam seed inside protective PVC tubes at one aquaculture site in British Columbia, Canada (Table 5). This was also true of sediments collected up to 50 m away from the PVC tube field. These results suggest that the placement of the protective PVC tubes in the intertidal zone is unlikely to alter sediment distribution, at least at the small spatial scale (tens of meters) used by Pearce et al. (2007).

Sediment chemistry also does not appear to be affected by the placement of the PVC tubes. Pearce et al. (2007) detected no significant differences in percent organics, oxidation-reduction potential, total organic carbon, and total nitrogen immediately before or up to 12 months after the geoduck clam seeding process (Table 6). That significant reductions in [sulphide] occurred in the seeded plot and up to 50 m away from the PVC tube field in research plots (i.e., plots in unseeded reference habitat) suggests that these reductions occurred independently of inserting the PVC tubes into the substrate. In other words, the reduction in sediment [sulphide] ostensibly would have occurred with or without the addition of the PVC tubes.

Table 5. Grain size analysis of sediments at an intertidal geoduck clam aquaculture site in Nanoose Bay, British Columbia, Canada*

Grain Size	Seeded Plot (0 m)	Research Plot (0-50 m)
>2 mm	NS	NS
1-2 mm	NS	NS
500 µm-1 mm	NS	NS
250-500 µm	NS	NS
125-250 µm	NS	NS
63-125 µm	NS	NS
45-63 µm	NS	NS
<45 µm	NS	NS

* NS indicates no significant difference between sediments of given grain sizes sampled before (pre-seed) and after (post-seed) out-planting geoduck clam seed inside protective PVC tubes within two plots (seeded and research). The research plot included samples that were collected up to 50 m away from the PVC tubes (Pearce et al. 2007).

Table 6. Sediment chemistry at an intertidal geoduck clam aquaculture site in Nanoose Bay, British Columbia, Canada*

Variable	Seeded Plot (0 m)	Research Plot (0-50 m)
Percent Organics	NS	NS
[Sulphide] at 2 cm	S, reduction	S, reduction
[Sulphide] at 4 cm	S, reduction	S, reduction
ORP at 2 cm	NS	NS
ORP at 4 cm	NS	NS
Total Organic Carbon	NS	NS
Total Nitrogen	NS	NS

*NS indicates no significant difference in sediment chemistry sampled before (pre-seed) and after (post-seed) out-planting geoduck clam seed inside protective PVC tubes within two plots (seeded and research). S indicates a significant difference in sediment chemistry was detected before and after the seeding process. The research plot included samples that were collected up to 50 m away from the PVC tubes (Pearce et al. 2007). ORP = oxidation-reduction potential.

3. Effects of Geoduck Harvest on the Physiochemical Properties of Sediments and Biological Resources

Perhaps the most visible and ardently questioned aspect of geoduck farming relates to the effects of the harvest technique. To harvest the mature geoduck, water is injected through 1/2 to 5/8" diameter hoses into the sediment at the pressure of a garden hose (~ 40 psi). This procedure loosens the sediment around the clam and permits retrieval of the clam by hand. It also creates a localized source of turbidity and can dislodge benthic infauna—some of which may perish or be consumed by nearby fish and other organisms, some of which may simply be dislodged and re-embed. Temporarily loosening the substrate might also change the vertical distribution of sediment size classes as found in other studies of experimentally disturbed sediments (Sharma et al. 2000; Dernie et al. 2003). Once the sediments become suspended, they can be transported short distances from their original position—with the distance traveled dependent on particle size and local currents. Dethier et al. (2007) have questioned whether the harvest, through this mechanism, would result in the loss of fine silts and clays, leaving only coarse, textured materials.

It should be recognized that the localized sediment disturbance during an intertidal geoduck harvest lasts only during the accessible low tide period—a work window that generally does not exceed 4 hours, and is usually less. Further, given the labor required for this type of harvest, only a small portion of beach can be harvested at a time—generally no more than about a 10th of an acre at most, assuming a *maximum* sized crew of 6 harvesters. Usually the area of beach disturbed is significantly less. This level of sediment disturbance is consistent with many types of shellfish harvest techniques that require sediment movement to dislodge the cultured product (e.g., oyster raking, manila clam digging, etc.). Notwithstanding, studies are ongoing in Canada and in southern Puget Sound to gain a better understanding of these processes and whether any changes that might be found are ecologically relevant or significantly dissimilar from the type of changes to sediment composition that annually occur within a dynamic sand flat environment. To this end, Pearce et al. (2007) reported a statistically significant reduction in large grain sizes and statistically significant increases in intermediate grain sizes after harvesting intertidal geoduck clams in his seeded plot in Nanoose Bay, British Columbia, Canada (Table 7). In unseeded research plots 50 m away from the seeded plots, they conducted mock harvests to compare potential changes in grain size and found no significant differences in the grain sizes amongst all the size classes. At the Foss Site leased by Taylor Shellfish somewhat similar results have been obtained. In this preliminary investigation the ENVIRON team found significantly higher concentrations of fine sand and silt grain sizes (40-63 µm, 63-125 µm and 125-250 µm) and statistically significant reductions in the medium sand and coarse fractions (250-500 µm, 500 µm-1 mm and 1-2 mm) at the harvest site compared with the up-drift control site (Table 8, Figure 3). There were no statistical differences between the nearby reference control site samples (never seeded) and the recently harvested site in the coarse (>2 mm) and silt/clay (<40 µm) grain sizes.

Although the preliminary results from Canada and the Foss Site do not support the Dethier et al. hypothesis of a loss in fine grained sediments from harvest activities, data from these types of studies are inherently varied based on site conditions--irrespective of any intentional manipulations that may be occurring to the intertidal sediments. In our opinion, the question of importance to SEPA is really whether any *statistically* significant change in the size classes of sediments on a sandy beach has probable significant adverse effects on the environment—regardless of the mechanism that might elicit such a change. Because sandy beach environments are subject to change from tidal, wind and wave energy forces, sediment sorting and redistribution occurs constantly. Thus, the size class distributions of sediment collected in the same location on one day may be different the next. Regardless, within the size class range of sediments captured by sand and silt, there is little variation amongst the type of biological communities that would be expected to thrive in these conditions (Ricketts and Calvin 1968). Thus, the question regarding how sediment grain size distribution affects biological communities within a sandy beach environment is relatively insignificant because species richness will be controlled primarily by other physical, biological and climate factors that created the sandy conditions initially and constantly reshape them.

Table 7. Grain size analysis of sediments sampled at an intertidal geoduck clam aquaculture site in Nanoose Bay, British Columbia, Canada*.

Grain Size	Seeded Plot (0 m)	Research Plot (0-50 m)
>2 mm	S, reduction	NS
1-2 mm	NS	NS
500 µm-1 mm	NS	NS
250-500 µm	NS	NS
125-250 µm	S, increase	NS
63-125 µm	S, increase	NS
45-63 µm	S, increase	NS
<45 µm	NS	NS

*NS indicates no significant difference between sediments of given grain sizes sampled before and after a simulated geoduck harvest within two plots (seeded and [unseeded research (reference)] plot). S indicates a significant difference between sediments of a given grain size was detected before and after the harvest. The research plot included samples that were collected up to 50 m away from the harvest area; comparisons were made in this area before and after a simulated harvest without any seeding (Pearce et al. 2007).

Table 8. Summary of Foss Farm sediment grain size distribution sampled at an intertidal geoduck clam aquaculture site in Washington State*.

Grain size	Control	Harvest	T-test ($p \leq 0.05$)	Statistical Result
>2 mm	1.1 ± 1.4	0.6 ± 0.5	0.271	NS
1-2 mm	5.8 ± 2.9	1.1 ± 2.0	0.000	S, reduction
500 μm-1 mm	22.0 ± 5.6	8.4 ± 5.7	0.000	S, reduction
250-500 μm	50.6 ± 5.2	40.7 ± 4.6	0.000	S, reduction
125-250 μm	18.2 ± 7.8	40.7 ± 9.3	0.000	S, increase
63-125 μm	1.1 ± 1.6	6.3 ± 2.2	0.000	S, increase
40-63 μm	0.03 ± 0.09	0.3 ± 0.2	0.000	S, increase
<40 μm	1.0 ± 0.9	1.8 ± 0.8	0.062	NS

*Samples were taken at a reference control site located approximately 400 meters up-drift of the geoduck clam harvest site. Mean values are presented with standard deviation (n = 10 samples per location). A t-test was conducted to determine significant differences for each sediment grain size class. NS indicates no significant difference between sediments of given grain sizes between the control and harvest and S indicates a significant difference between sediments of a given grain size was detected between the control and harvest. (J. Fisher and K. Mueller, ENVIRON International, unpublished data)

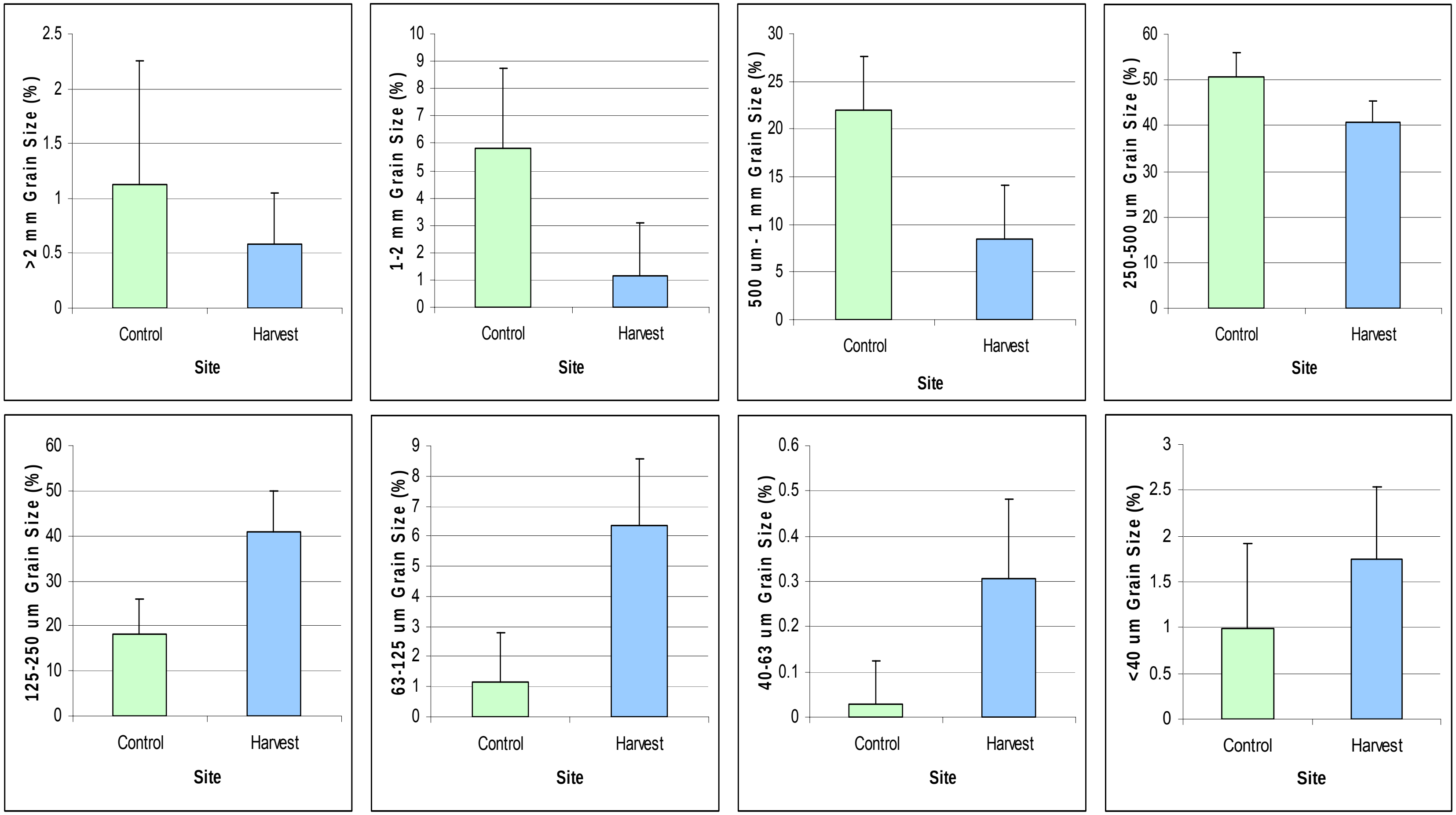

Figure 3. Sediment grain size distribution for Foss Farm intertidal geoduck aquaculture harvest site and a control site with bars representing standard deviation (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

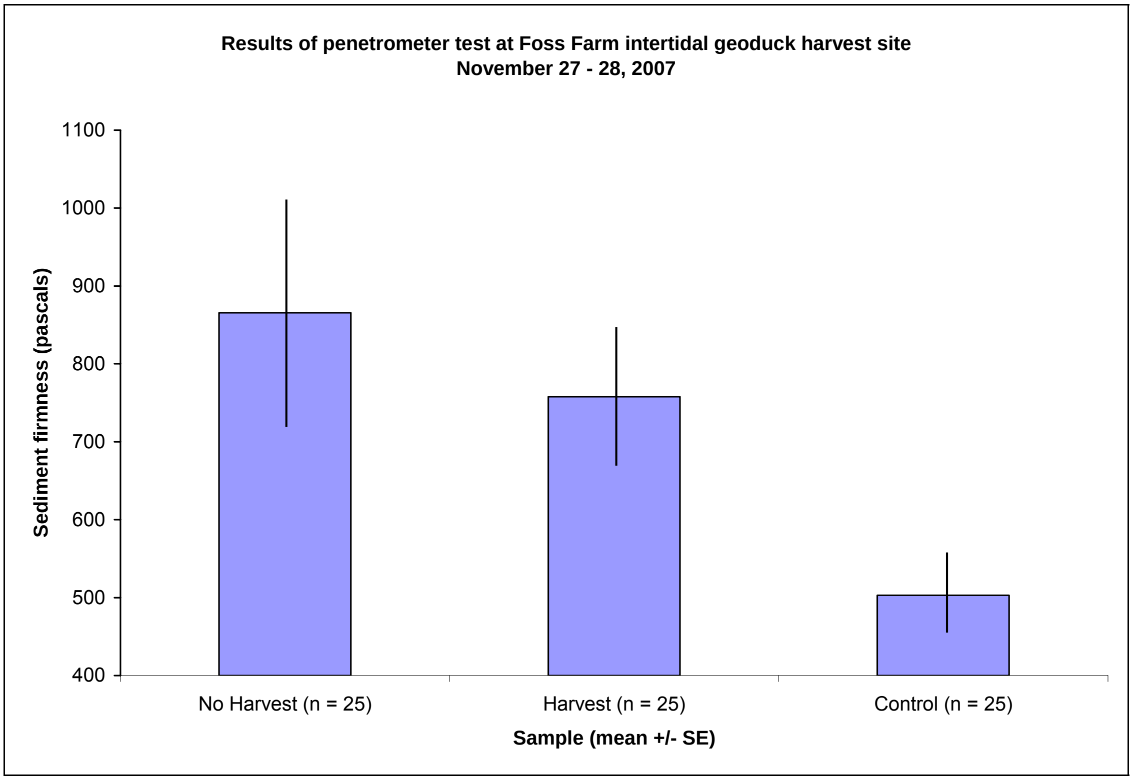

Figure 4. Results of penetrometer testing for sediment firmness at the Foss Farm intertidal geoduck aquaculture site (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

a. Sediment Compaction

The potential for alterations to grain size composition leads to additional concerns regarding compaction and changes to porosity. Compaction effects can lead to increases in the firmness of the substrate and subsequent changes in the benthic community. For example, after measuring firmness of intertidal flats in Willapa Bay, Washington, Gingras et al. (2000) concluded that soft substrates typically had low-diversity assemblages of burrowing worms, whereas soft to moderately firm substrates contained higher diversity assemblages of polychaetes, crustaceans, and bivalves. A similar pattern was observed by the researchers from ENVIRON International (J. Fisher and K. Mueller, unpublished data) when comparing sediment firmness at the Foss Farm aquaculture site and an adjacent control site. Recall that the seeded area at Foss Farm had more species and had higher densities of organisms than the control site (Table 4). Using the modified Brinell hardness test and a penetrometer described by Gingras and Pemberton (2000), the ENVIRON team found that the geoduck aquaculture site had firmer substrate than the reference area (Figure 4), which was consistent with Gingras et al.'s (2000) findings. Testing was also conducted in the field at the Eld Inlet intertidal geoduck harvest site pre- and post-harvest. Results show that sediment firmness was not significantly altered by harvesting (Figure 5).

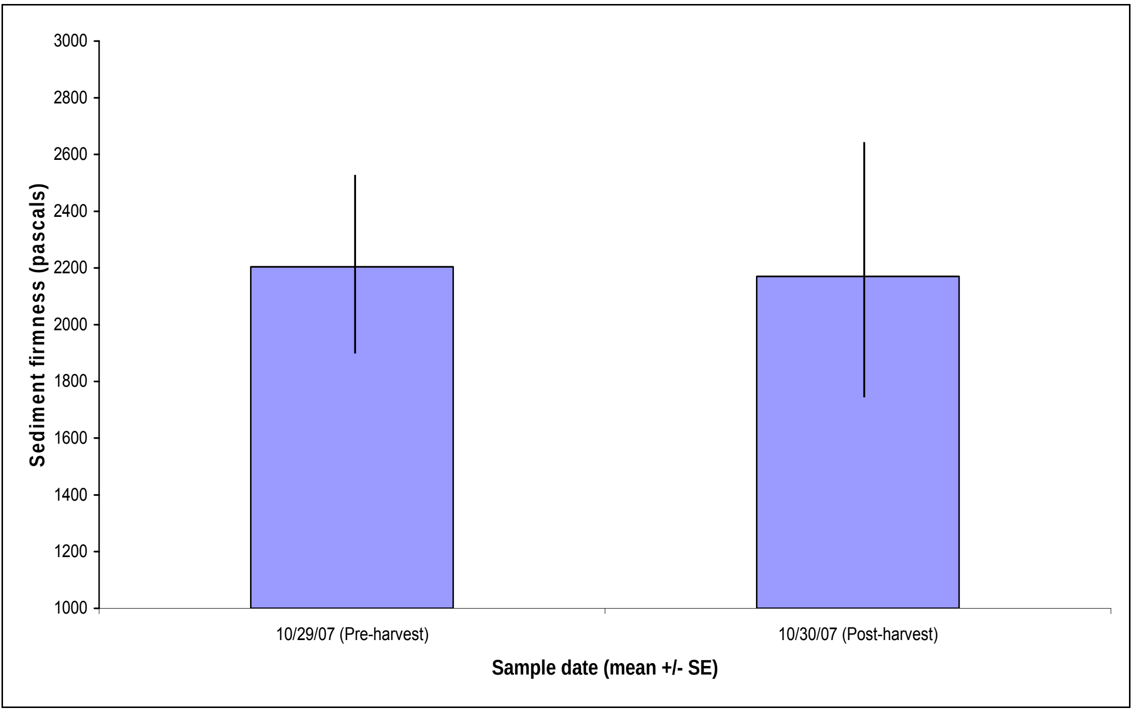

Figure 5. Results of penetrometer test for sediment firmness at Eld Inlet intertidal geoduck harvest site (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

b. Sediment Chemistry

The effect of harvesting intertidal geoduck clams on sediment chemistry was recently studied by Pearce et al. (2007) in British Columbia, Canada. They measured percent organics, [sulphide] at 2 and 4 cm depths, oxidation-reduction potential (ORP) at 2 and 4 cm depths, total organic carbon, and total organic nitrogen. Again, samples were collected from a seeded plot (0 m) and a research plot with samples taken from up to 50 m away from the harvest area. The results showed that, following the harvest, there were statistically significant reductions in ORP at 2 and 4 cm depths. There was also a statistically significant decrease in percent organics from the seeded plot (0 m) relative to pre-harvest levels. In all other categories, [sulphide], total organic carbon, and total organic nitrogen, there were no statistical differences before or after the harvest in the sediment from the seeded plots or the research plots (Table 9).

Table 9. Sediment chemistry at an intertidal geoduck clam aquaculture site in Nanoose Bay, British Columbia, Canada*

Variable	Seeded Plot (0 m)	Research Plot (0-50 m)
Percent Organics	S, reduction	NS
[Sulphide] at 2 cm	NS	NS
[Sulphide] at 4 cm	NS	NS
ORP at 2 cm	NS	S, reduction
ORP at 4 cm	NS	S, reduction
Total Organic Carbon	NS	NS
Total Nitrogen	NS	NS

* NS indicates no significant difference in sediment chemistry sampled before and after harvesting geoduck clams within two plots (seeded and research). S indicates a significant difference in sediment chemistry was detected before and after the harvest. The research plot included samples that were collected up to 50 m away from the harvest area (Pearce et al. 2007). ORP = oxidation-reduction potential.

(i) Potential Effects of Harvest on Water Quality and Sediment Transport

In intertidal systems, the transport of sediments is a dynamic and short-term cyclical morphological process that is largely attributed to the tides (Uchiyama 2007) and local drift cell currents. As discussed, the disturbance of sediments during clam harvest causes sediments to be suspended within the water column and has raised concerns regarding the potential impacts of this disturbance on water quality—both the effects of the suspended sediments, and the potential effects on nutrient redistribution. To address some of the questions on how sediment and nutrient dynamics might be affected from a harvest, ENVIRON researchers measured indicators of water quality associated with the sediment plumes generated in a harvest, including TSS, turbidity and nutrients. The sediment plumes generated from harvest were generally localized and transient, with turbidity most prevalent near the waters edge to approximately 8 m offshore. TSS was measured updrift and downdrift of a Foss Farm harvesting site. They found that updrift of the harvest TSS levels >25 ft offshore and ≤25 ft offshore were similar to background TSS levels. As would be expected, TSS increased at the harvest site and for some distance downdrift of the harvest. It is important to note that although there is an increase in TSS, the trend shows that the concentrated sediment plume is largely limited to the nearshore (≤25 ft) area and concentrations peak at roughly 100ft downdrift. Figure 6, reflects the increase at harvest for both distances from harvest and the trend towards a return to background levels.

These TSS data are consistent with earlier reported data at Seattle Shellfish's Jackowski Farm (Fleece et al. 2004). In that study, turbidities of 1, 35, and 75 NTUs were measured 50 ft, 25 ft, and 5 ft from the shoreline during a December 2003 harvest. Approximately 100 ft downdrift of the harvest area, turbidities 50, 25, and 5 ft from shore were 1.4, 1.5 and 1.0 NTU, respectively. Similar patterns were observed for total suspended solids. Collectively, these data suggest that there can be slight differences among sites, but that turbidity and TSS decline rapidly within a short distance of the shoreline harvest activity and elevated turbidities and TSS are very short lived.

Recent studies at a different Taylor geoduck farm on Eld Inlet revealed a similar trend with phosphorous as observed with the TSS and turbidity sampling at the Foss Farm site. The phosphorous level at distance 0 (i.e., harvest) shows a peak in phosphorous relative to the background level (Figure 7). At roughly 350 ft. downdrift, phosphorous levels returned to background levels. Statistically, phosphorous levels were determined to be statistically different at the harvest site relative to the updrift sampling sites, but no statistically significant differences were found between the updrift and downdrift background sampling sites (Figure 8). Returns to background phosphorous levels downdrift from the harvest site can be attributed to the short time-periods of harvesting, and the rapid tidal and drift-cell driven dissipation. These results suggest that that phosphorous concentrations are tightly tied to sediment.

The concentrations of nitrate and nitrite at the same Eld Inlet site did not reflect the results obtained with phosphorous, in that levels were significantly higher updrift when compared with the harvest site. Further, downdrift samples were determined to be statistcally insignificant from both the updrift and harvest areas (Figure 9). Given that Puget Sound waters are considered nitrogen limited, the lack of any significant increase in dissolved water column nitrogen measurable during a harvest suggests that the effects of the harvest practice are environmentally insignificant with respect to the potential to generate nutrients that could trigger excessive embayment eutrophication.

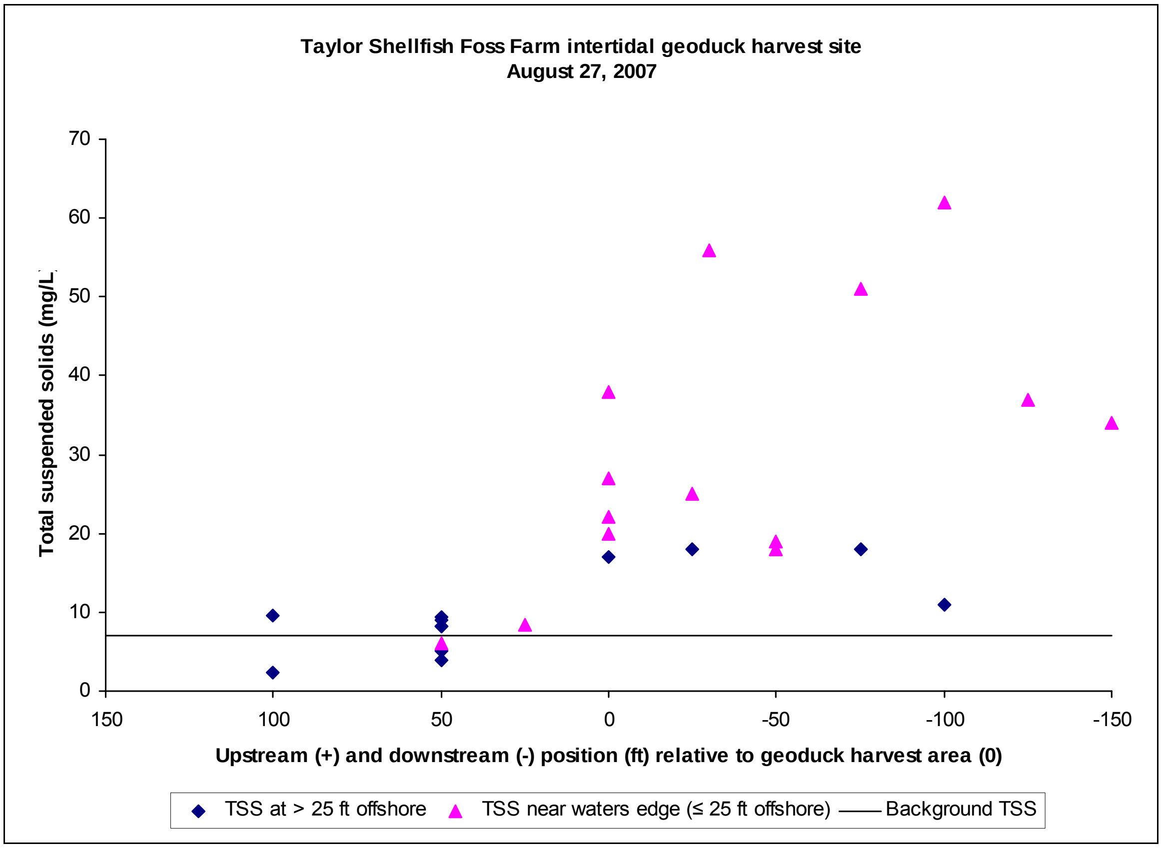

Figure 6. Total suspended solids (TSS) at Taylor Shellfish Foss Farm intertidal geoduck harvest site (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

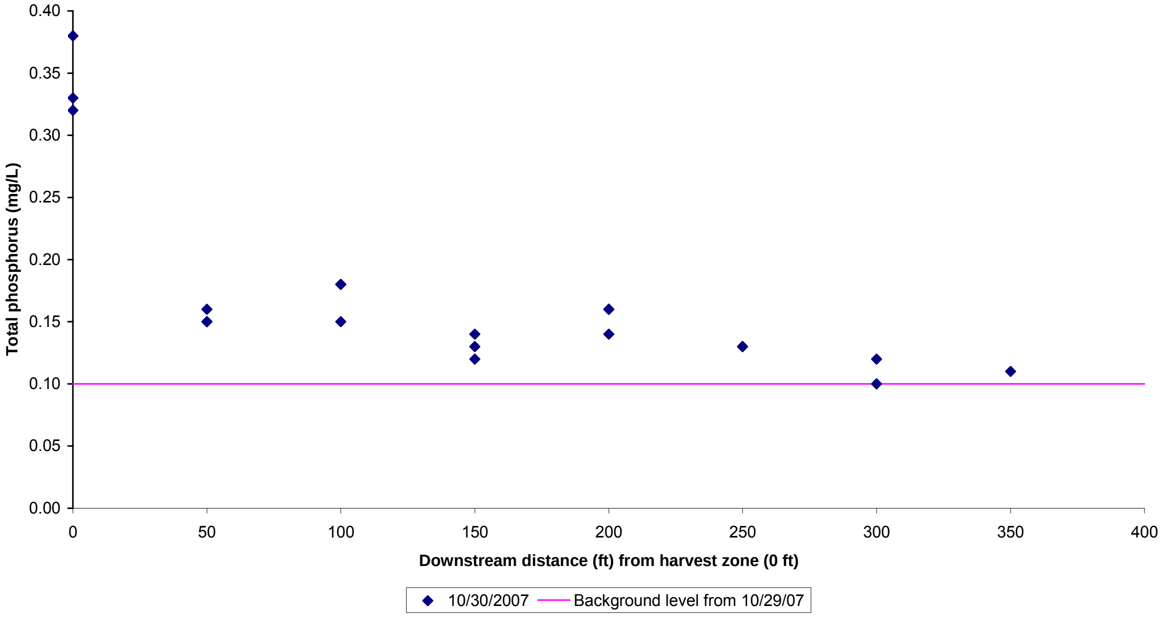

Figure 7. Water quality data for total phosphorous from Taylor Shellfish Eld Inlet intertidal geoduck harvest site (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

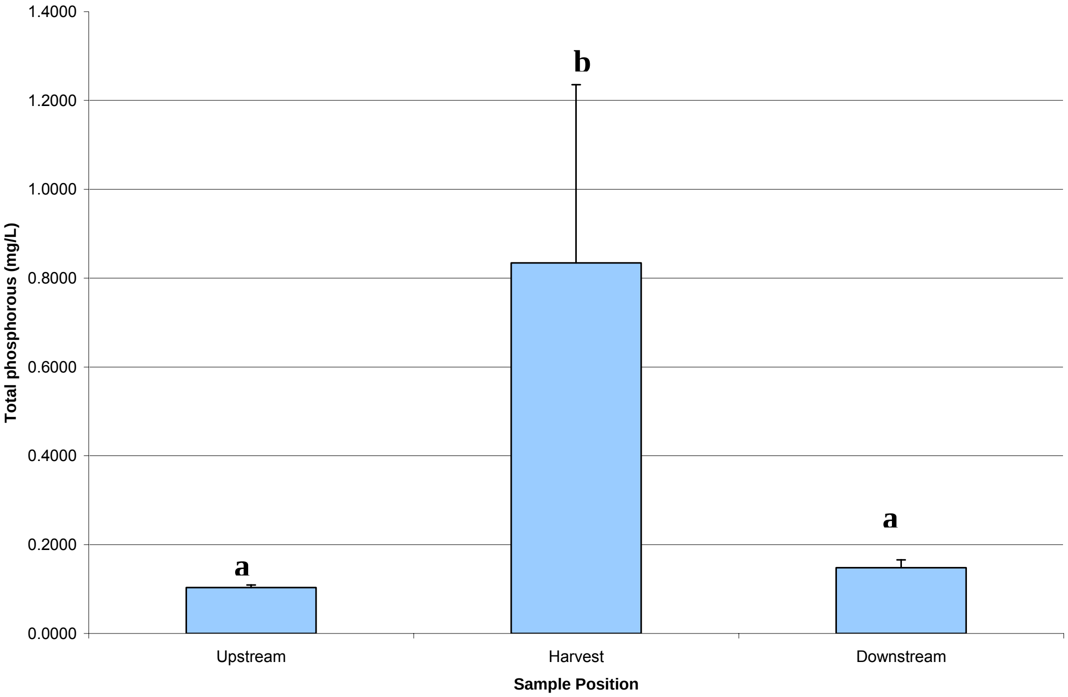

Figure 8. Total phosphorous (mg/L) from samples collected during Eld Inlet intertidal geoduck harvest on 10-29-07 collected at waters edge on incoming tide (J. Fisher and K. Mueller, ENVIRON International, unpublished data). Letters denote significant difference (a, b) using Scheffe's test.

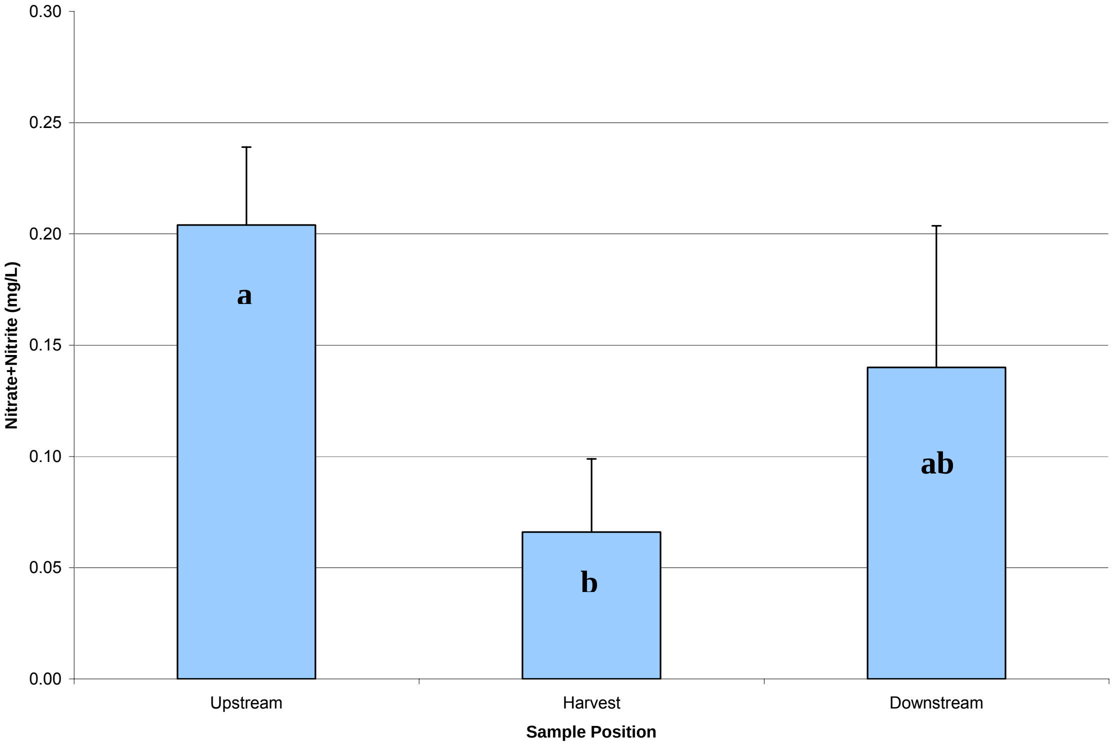

Figure 9. Nitrate+Nitrite (mg/L) results from samples collected during Eld Inlet intertidal geoduck harvest on 10-29-07 collected at waters edge on incoming tide (J. Fisher and K. Mueller, ENVIRON International, unpublished data). Letters denote significant difference (a, b, ab) using Scheffe's test.

(ii) Potential effects of harvest on fishes and benthic diversity

The sediment plume created during harvest by the use of water hoses has also generated concern regarding potential direct effects on fishes, particularly salmonids. These concerns have arisen from past studies where excessive suspended sediment concentrations have been shown to cause ill effects on salmonids and other fishes. Laboratory experiments have shown that survival of wild juvenile Chinook salmon (*Oncorhynchus tshawytscha*) may be jeopardized indirectly by prolonged exposure (48 hours) to extremely high levels of turbidity (e.g., 50,000 NTU) and suspended sediments (e.g., 30,000 mg/L) (Korstrom and Birtwell 2006); however, it should be noted that these concentrations are several orders of magnitude greater than those found at geoduck clam harvest sites during a typical harvest. Furthermore, the sediment plume at an intertidal geoduck clam harvest site dissipates rapidly, disappearing within a single tide cycle.

To further consider the potential effects of the turbidity plume, we modeled the potential severity (SEV) of ill effects of the sediment plume on fishes at the Foss and Eld inlet farm sites in southern Puget Sound, using a standard risk assessment model. The model used was developed by Newcombe and Jensen (1996) to quantify the potential risk of suspended sediments (measured as total suspended solids) on a variety of fishes, but especially salmonids. Newcombe and Jensen (1996) created a concentration-duration response model with a 15-point scale, where a severity (SEV) index range from 0 (nil effect or no behavioral effects) to 14 (> 80–100% mortality) was calculated to screen potential effects. To calculate the SEV number, both the concentration of TSS and the duration of exposure are factored into a logistic equation. In our modeling, we screened conservatively to presume the worst case scenario where fish would remain inside the sediment plume, without ever attempting to avoid or escape it, for periods up to three hours – highly unlikely circumstances given the natural avoidance reaction of many fishes to sharp, sudden increases in turbidity (Wilber and Clarke 2001) and the fate of the sediment plume during a geoduck clam harvest as described above. Still, even with the assumptions above, the SEV was typically below 5 (minor physiological stress) and no more than 6 (moderate physiological stress) (Figures 9 and 10), values that might result in ephemeral changes in fish behavior, or at most, short-term physiological stress such as coughing (Newcombe and Jensen 1996).

Thus, the data shows some increase in turbidity associated with harvest of geoduck. That increase in turbidity in concentrated at the harvest zone, and quickly dissipates downstream from that zone. Salmon and other fish species avoid the harvest zone for the limited duration of harvest activities. However, if salmon did not avoid the area of increased turbidity, the impacts associated with that turbidity would be, at worst, moderate and reversible. In either case, the impacts of harvest activities on fish are unlikely to be "significant" as that term is defined in SEPA.

It is well known that the commercial harvest of shellfish in the intertidal zone results in the immediate loss of some benthic fauna and flora (Hall and Harding 1997; Ferns et al. 2000). Not surprisingly, post-harvest reductions of some taxa have also been observed at intertidal geoduck clam aquaculture sites in southern Puget Sound, as previously depicted (Table 4). The recovery rates of benthic communities following physical disturbance depend on a variety of physical, chemical, and biological factors (Dernie et al. 2003), but in general, they recover fairly quickly. For example, the recovery time for microalgae can be as little as one week (Wulff et al. 1997), whereas the benthic infauna can be restored in about two months (Hall and Harding 1997; Dernie et al. 2003). Regarding the impact of intertidal geoduck clam harvesting, Pearce et al. (2007) report preliminary data that suggests that species richness and relative abundance of benthic fauna at a geoduck aquaculture site in British Columbia, Canada was restored to pre-harvest levels after six months (*see* Figures 1 and 2). Again, these short term impacts to benthic organisms does not rise to the level of significance under SEPA. The isolated and temporally infrequent disturbance to the sandy intertidal zone following a geoduck harvest that might occur every 4 to 6 years, occurs at a much smaller spatial and temporal scale than the broad scale disturbances that cause spatially explicit impacts to the intertidal zone, such as annual winter storms. The level of disturbance created is well within the range of normal sediment processes and the evidence from these disturbances does not indicate an adverse impacts are likely.

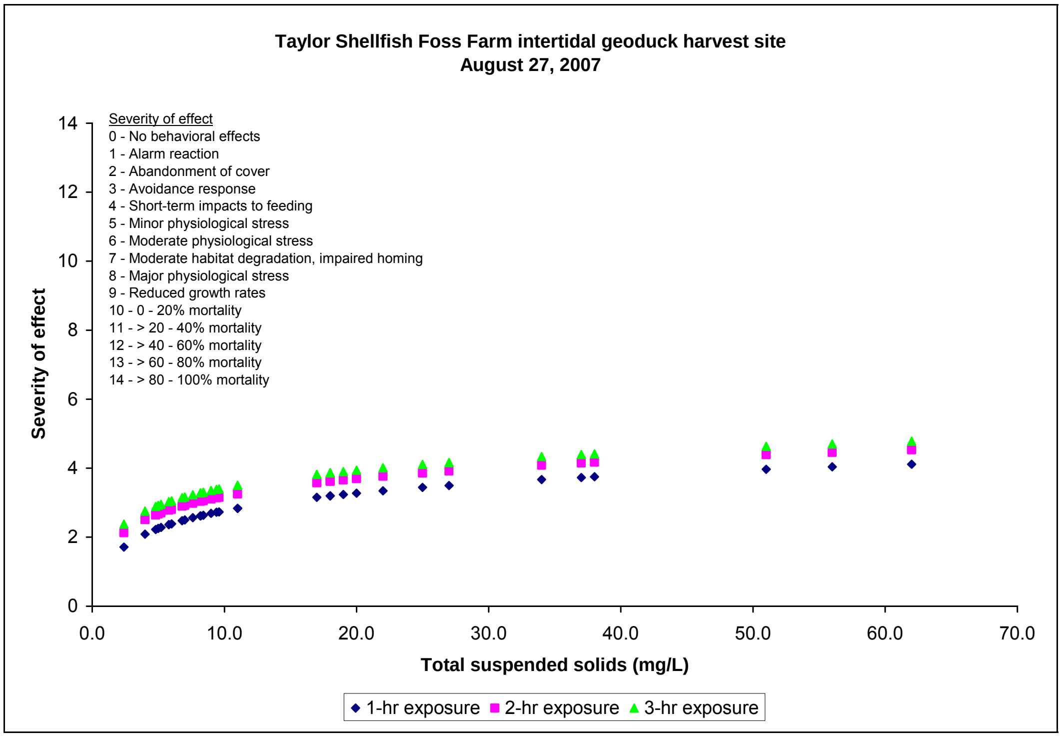

Figure 9. Relationship between the severity of ill effect index and total suspended solids, as measured at the Foss Farm intertidal geoduck clam aquaculture site on August 27, 2007 during a harvest (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

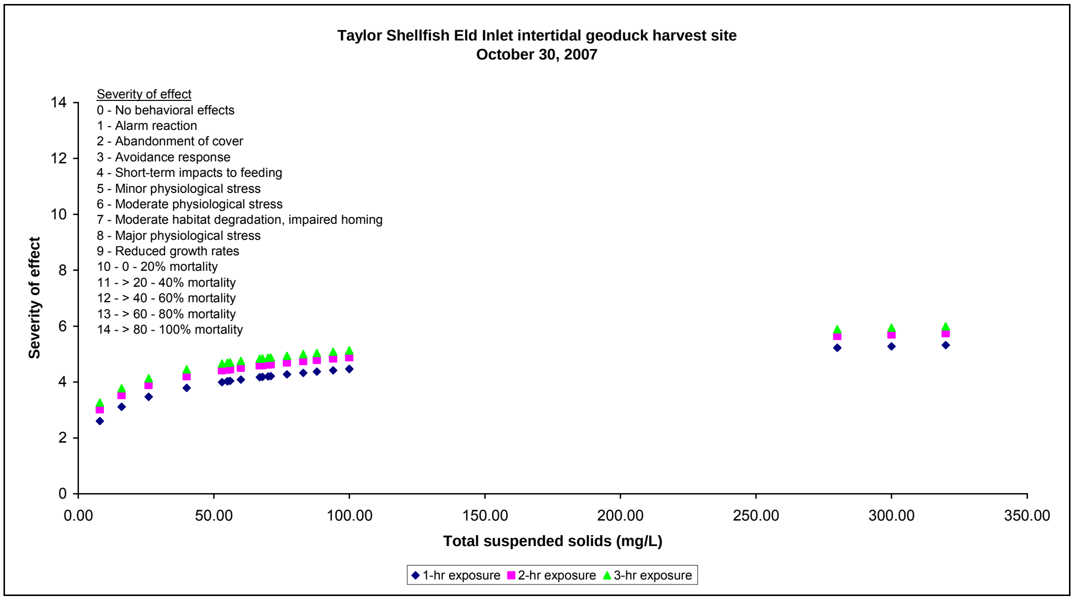

Figure 10. Relationship between severity of ill effect and total suspended solids at the Eld Inlet intertidal geoduck clam aquaculture site on October 30, 2007 (J. Fisher and K. Mueller, ENVIRON International, unpublished data).

4. Potential Impacts from Geoduck Aquaculture on Birds

While ongoing studies are examining bird use at a number of geoduck sites, data from these studies are not yet available. Regardless, a substantial body of information on the bird use of other shellfish aquaculture sites in similar physical habitat provides sufficient information to characterize the potential significance of geoduck aquaculture practices on birds and to examine the question of whether geoduck aquaculture practices on birds using the Nearshore environment. Shellfish, whether cultured or wild, form an important source of food for a wide variety of marine shorebirds (Dankers and Zuidema 1995; Norris et al. 1998; Hilgerloh et al. 2001; Lewis et al. 2007). Through their foraging habits, migrating marine shorebirds can significantly alter the community structure of wild bivalve populations in soft-bottom intertidal areas (Lewis et al. 2007). At shellfish aquaculture sites, some species of marine shorebirds feed directly on the shellfish products themselves (e.g., Dankers and Zuidema 1995), while others feed on the macrofauna and flora that colonize shellfish aquaculture gear (e.g., Hilgerloh et al. 2001).

Changes in the density or biomass of wild shellfish, whether through natural fluctuations or harvesting by people, have been shown to affect the number of marine shorebirds utilizing those shellfish for food (Dankers and Zuidema 1995; Norris et al. 1998). Furthermore, shellfish aquaculture sites influence the abundance of marine shorebirds. For example, Connolly and Colwell (2005) reported that seven of 13 marine shorebirds and three of four wading birds were more abundant on oyster longline plots compared to reference sites. Although marine shorebirds feed at shellfish aquaculture sites, Hilgerloh et al. (2001) found that the aquaculture sites themselves did not necessarily attract larger numbers of birds than non-cultured areas. For instance, Zydelis et al. (2006) found that natural environmental attributes were the primary determinants of densities of wintering surf scoters and white-winged scoters in Baynes Sound, B.C. Moreover, the authors found that shellfish aquaculture variables did not necessarily predict bird densities for both scoter species. According to Zydelis et al. (2006), these findings suggest that winter scoter populations and the shellfish aquaculture industry may be mutually sustainable. In other words, there was no evidence of a negative impact on winter scoter populations at the current level of shellfish farming practiced in Baynes Sound, B.C. Indeed, Connolly and Colwell (2005) found that shellfish aquaculture in Humboldt Bay, California did not negatively affect the foraging behavior of most marine shorebirds studied.

An interesting aside – recently, it was shown that empty oyster shells from commercial shellfish growers in Washington could be used to enhance nearshore nesting habitat for and improve the hatching success of the threatened western snowy plover (Long 2005; Pearson et al. 2007).

In conclusion, the potential for negative interactions to occur for as yet unstudied bird species as a result of intertidal shellfish aquaculture activities cannot be ruled out; however, the collective evidence from a variety of shore and seabird species evaluated suggests there is no significant negative impact. Where impacts have been observed, they have either been positive—increasing avian species richness and abundance due to increased forage opportunities, or benign—eliciting no significant difference in use from natural beds. Although occasions of entrapment in aquaculture gear have been observed, an undeniable negative effect for the individual, these occurrences have been extremely rare and are discountable at the population level, where the evidence suggests the net effect is positive. In addition, after consultation with representatives of the Tahoma Audubon Society, Taylor Shellfish Farms has incorporated a provision in its Codes of Practice requiring the use of individual tube nets, as opposed to canopy nets, on farm areas in the proximity of eagle nests. This practice should estimate the risk of entrapment of eagles.

5. The Potential for Impacts to Eelgrass from the Culture of Geoduck

Eelgrass is widely recognized by natural resource management agencies and the scientific community as an extremely important resource for a variety of biological resources. Among the many ecologically important functions that eelgrass beds provide are nursery areas for many commercially important species of fish and wildlife, including juvenile salmon, Pacific herring, black brant, and Dungeness crab (Phillips 1984; Simenstad 1994; Wilson and Atkinson 1995; Blackmon et al. 2006). In recognition of the various important functions of this resource, the WDFW has identified eelgrass as a priority habitat resource important to maintaining the diversity of Washington's fish and wildlife. The WDFW has developed and adopted specific monitoring protocols to document the presence, absence, areal extent, and characteristics of eelgrass beds (e.g., shoot density) for any proposed activity requiring Hydraulic Project Approval (HPA). Clearly, the culture practices inherent to the culture of geoduck would have the potential to disturb eelgrass *if* it was the practice to culture geoduck within eelgrass. It is not. Thus, while Dr. Jennifer Ruesink from the University of Washington recently presented data to demonstrate such impacts in an experimental plot she and colleagues established in eelgrass, the data collected are not relevant to Taylor's operations because Taylor does not plant geoduck seed in eelgrass beds.

Typically, Taylor conducts an eelgrass survey conforming to WDFW protocols before planting to identify the location and extent of any eelgrass, and avoid planting any shellfish in any eelgrass, and then perform another eelgrass survey following harvest before replanting. Such surveys not only delineate any existing resource changes but also provide the means to avoid eelgrass conflicts to begin with.

Although Taylor's Codes of Practice prevent any direct impacts to eelgrass, potential indirect effects must still be considered in any SEPA determination. According to the scientific literature, eelgrass has very specific and relatively narrow habitat preferences for substrate, wave energy, light, and temperature.

While the effects of the culture of geoducks near eelgrass beds has not been systematically studied, there is a considerable amount of data available on the effects of shellfish aquaculture of other filter-feeding bivalves on these parameters as they pertain to conditions suitable for eelgrass growth and reproduction. Because the basic mechanisms of effect (filteration, biodeposition, etc.) are similar between geoduck and other shellfish, this existing data on other shellfish is likely applicable to geoduck farming. The scientific literature includes a number of examples of the physical effects of filter-feeding bivalves, such as mussels and clams on chlorophyll, nutrients, and water clarity (a few metrics of water quality). Remote sensing data collected around high density blue mussel aquaculture has indicated these animals can reduce chlorophyll and phytoplankton (Grant et al. 2007). Newell (2004) documented reduced turbidity from removal of inorganic particles by filter-feeding bivalves, which can improve conditions for the growth of submerged aquatic vegetation by increasing light for photosynthesis (Newell and Koch 2004. Filter feeders also remove nitrogen and phosphorus from the water column that is ultimately removed from the ecosystem at harvest, which can improve water quality and reduce cultural eutrophication (Lindahl et al. 2005, Newell 2004, Zhou et al. 2006. Such changes are arguably favorable eelgrass growth, which has declined in the Puget Sound region historically, and anecdotal reports from growers suggest shellfish often change local habitat conditions to facilitate eelgrass survival and growth, where they previously did not exist or were present at very low levels.

According to the most recent Puget Sound Submerged Vegetation Monitoring Project Monitoring Report (Gaeckle et al. 2007), there are approximately 20,400 ± 3,300 hectares (50,409 ± 8,200 acres) of native eelgrass (*Zostera marina*) distributed within intertidal and shallow subtidal areas of coastal embayments or linearly along the shorelines of the Puget Sound Basin. The many anthropogenic and natural factors thought to be responsible for declines in eelgrass include dredging, cultural eutrophication (nutrients), industrial waste spills, extreme weather conditions, grazing, and sediment re-suspension (Short and Wyllie-Echeverria 1996; Wright 2002; Wyllie-Echeverria et al. 2003); shellfish farming is not recognized as a factor in the decline.

6. Potential Impacts of Aquacultured Geoduck on Wild Geoduck Genetics

A substantial amount of information has been developed from studies of wild aggregations of geoduck clam (Valdopalas et al. 2004; Miller et al. 2006) that provide important data for evaluating whether the culture of geoduck, as practiced by Taylor, poses a potentially significant and/or adverse impact to wild geoduck populations. Specifically, Valdopalas et al. (2004) found extensive genetic diversity within the geoduck genome (i.e., the total genetic make-up of the species), but few differences *among* geographically separated geoduck populations in their allozyme frequencies and microsattelite arrays. Allozyme frequencies and microsatellite arrays provide measures of population variability or 'relatedness.' By comparison, there can be substantial variation in these distinct genetic metrics among salmon populations originating from different tributaries *within the same watershed* (reference to add). Thus, the geoduck data suggest that there has not been reproductive isolation among populations—in essence, that the geoduck in Puget Sound represent one large 'superpopulation.'

While current data indicate there is a lack of population distinctness among geographically separated geoduck in Washington State waters, several aspects regarding the interactions between wild and hatchery populations are less well understood (Straus et al. 2008). Gene flow, the interbreeding among wild and hatchery-outplanted geoduck clams, is one particular area that has been raised as a potential concern—we do not know if and/or whether a distinct genetic 'signature' of the offspring of such crossings would develop in the event such interbreeding occurs. Similarly, there have been no studies of how traits inadvertently or intentionally selected for in a shellfish hatchery might affect wild populations of geoduck clams in the event that interbreeding occurs.

Given the extensive genetic variability within geoduck and the lack of population distinctness, identifying whether gene flow is occurring among cultured and wild geoduck is challenging. From the standpoint of SEPA, however, the issue is whether these gene flow questions represent environmentally significant data gaps, or simply basic research questions worthy of resolution within academia. Recent information presented at a Sea Grant conference (Valdopalis, B., U. of Washington, pers. communication) suggests that geoduck may reach reproductive maturity within two to three years—well within the 4 to 6- year geoduck farming rotation cycle. It is not known whether the gametes produced from these young clams would be viable, as gamete viability in many other species has been found to increase with age (i.e., until senescence), but the cross-fertilization of gametes from wild and cultured geoduck gametes is certainly possible.

Based on our current understanding of wild geoduck genetics, the lack of distinction among clams from widely divergent geographies, the high individual genetic variation within Washington's geoduck, and the results from studies with other aquatic organisms where genetic drift impacts were found to be insignificant (LeClair et al. 1999; Heggenes et al. 2006), an adverse impact on geoduck genetics from the current culture practices is unlikely. Notwithstanding, Taylor Shellfish, Farms has instituted best management practices in their hatchery protocols to reduce the potential of genetic interactions between wild and hatchery geoduck clams, and the potential loss of genetic variation within the species over time. These measures include: (1) no hatchery-produced geoduck clams are used as brood stock; (2) local, wild brood stock are collected at least twice annually for use as brood stock and brood stock are never used in repeat spawnings (i.e., for multiple seed sets); (3) different aged organisms and multiple males and females are used for each spawning; (4) adequate mixing of all gametes occurs to insure as diverse a genetic product as possible; and (5) progeny are out-planted near the source location of the brood stock (Joth P. Davis, Taylor Resources, Inc., Quilcene, Washington, personal communication). Straus et al. (2008), in their recent SeaGrant funded review of geoduck aquaculture, recognized these measures will maintain population structures, preserve local adaptations in wild populations, help maintain high levels of genetic variation in the progeny, reduce long-term domestication selection, and increase the genetic, effective population size in the hatchery over generations. Based on these conclusions, and the previous discussion, the effects of geoduck aquaculture on wild geoduck genetics does not elevate to the level of a significant impact under the provisions of SEPA.

7. Summary and Conclusions

Shellfish are broadly recognized amongst major non-governmental institutions, independent scientific bodies, and government resource agencies alike as highly beneficial for mitigating the impact of pollution and eutrophication in marine waters. The following quotes demonstrate the broad based recognition of the fundamental benefits provided through bivalve shellfish and bivalve aquaculture.

"Nutrient over-enrichment is a significant problem for the coastal regions of the United States…. Benthic filter feeders such as oysters, mussels, and many species of clams can have a major influence on phytoplankton populations in coastal waters." - National Research Council, the principal operating agency of both the National Academy of Sciences and the National Academy of Engineering. 2000. *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. National Academy Press, Washington, DC.

"One type of aquaculture - mollusk farming – actually reduces nutrient pollution…. Because 35-40% of the total organic matter ingested by a mollusk is used for growth and permanently

removed by harvest of the mollusk." – Environmental Defense. 1997. *Murky Waters: Environmental Effects of Aquaculture in the US.* New York, New York.

"Filter-feeding mollusks can clarify the water by consuming plankton in aquatic systems, significantly improving water quality. Mussel farms can remove nitrogen from water at a 70% higher rate than occurs in surrounding waters." – Pew Oceans Commission. 2001. *Marine Aquaculture in the United Sates.* Arlington, Virginia.

"*Moreover, shellfish farmers are often among the loudest advocates for clean water."* – Pew Oceans Commission. 2001. *Marine Aquaculture in the United Sates.* Arlington, Virginia.

"In the text of the Clean Water Act, Congress plainly and expressly listed the 'protection and propagation of… shellfish' as one of the goals of reduced pollution and cleaner water." United States Court of Appeals for the Ninth Circuit. *APHETI v. Taylor Resources.* 2002. Seattle, Washington.

"These filtering and recycling processes are critical in regulating the health of coastal ecosystems. The processes take on even greater importance as human activities and related pollution discharges increase in shoreline areas. The processes help counteract the potentially damaging effects of excessive nutrient enrichment of coastal waters, a process known as eutrophication." – Puget Sound Action Team. July, 2003. *Keystone Species of the Estuary – Bivalve Basics.* Olympia, WA.

"Oceans have a mighty appetite for swallowing carbon dioxide," and that *"lovely carbon reservoirs-seashells to the layman"* are one place the missing carbon goes. National Geographic. February 2004. *The Case of the Missing Carbon.* National Geographic Society, Washington, DC.

Much of the 'environmental concern' raised by opponents to geoduck farming appears to be based on factors independent of the potential for ecological impact, and stems from visual interpretations by laypersons lacking the scientific background to objectively consider the ecological context of geoduck aquaculture as it is practiced in Puget Sound. Some of the scrutiny of the geoduck aquaculture industry that has been brought forward may stem from studies of high-density (three dimensional) culture operations in Europe, where some shellfish stocks are maintained at average densities of hundreds to thousands of organisms per square meter, and ecosystem functions and carrying capacity have been affected (Sorokin et al. 1999; Bartoli et al. 2001; Nizzoli et al. 2007). It is inappropriate to make comparisons between these Washington apple and European orange extremes.

In this memo we focused particularly on the production of intertidal geoduck clams and its possible effects on the environmental integrity of the intertidal zone and surrounding waters. As discussed in this review, intertidal geoduck clam aquaculture occurs at biomass densities (~ 14 individuals per m^2) that are significantly lower than such shellfish culture operations abroad, and within the range of natural densities of wild geoduck recorded in Puget Sound (up to 22 individuals per m^2) (Strauss et al. 2008). Further, cultured geoduck densities yield significantly lower biomass estimates per unit area than the densities of bottom cultured oysters and manila clams—species that have been cultured here at higher densities and intensity for decades without evidence of significant adverse impact—and arguably with significant ecological benefits to habitat and water quality. While we are not contending that correlation represents causation, it is ironic that the only major nearshore waterbody in the state where salmon populations are *not* listed under the Endangered Species Act is Willapa Bay/Grays Harbor, where oyster farming has been conducted at the highest intensity levels practiced in the state for nearly 100 years.

Similarly, the issues regarding geoduck grow-out gear and harvest techniques are largely visual. We are not aware of any quantitative results that have shown the gear or practices used cause significant adverse impacts on ecological function within the context of SEPA (i.e., more than a moderate chance of an adverse effect) and preliminary data presented here have shown benefits to biodiversity, and insignificant effects on water and sediment quality. When compared with other forms of shellfish aquaculture in the Pacific Northwest (e.g., oysters), or elsewhere for that matter, effects from intertidal geoduck clam aquaculture would appear, at worst, benign, and—by most metrics of ecological function—positive.

The net ecological benefits provided by the filter feeding biomass of the cultured geoduck, and the structured habitat created from the culture operations cannot be discounted simply on the basis of aesthetics. Our data and others have shown that aspects of intertidal geoduck clam aquaculture can indeed have ephemeral impacts on water quality, and there remain questions with respect to benthic responses among different farm sites. Such questions are the nature of scientific research, and we are not contending that research should not continue on some of the questions that have been raised—we certainly will be engaged in addressing some of them. We are contending, however, that the nature of the basic science questions that have been raised are not of such magnitude that the answers to them would alter any SEPA threshold determination of one way or another.

The ephemeral negative impacts of some of the practices involved with geoduck culture are confined primarily to the area cultivated and last on the order of minutes, days or weeks—depending on the variable measured. Within a 4-6-year production cycle, these localized and emphemeral changes are consistent with natural disturbance regimes that are constantly at play in controlling the habitat characteristics that determine biodiversity and abundance in the intertidal environment. Overall, the negative impacts appear to be minimal and insignificant and are largely overshadowed by the long term water quality and habitat benefits created by the culture practice. This conclusion is supported directly by research conducted at several intertidal geoduck clam aquaculture sites in Puget Sound and British Columbia, Canada as previously discussed, and indirectly by accounts of the impacts associated with other types of shellfish aquaculture.

Information gaps exist with any human endeavor, and the culture of geoduck clams in the intertidal zone is no exception (Straus et al. 2008). Narrowing data gaps is always helpful for making informed management decisions (McKindsey et al. 2006) and refining best management practices (BMPs). Although additional research will help resolve unanswered questions concerning the impact of intertidal geoduck clam aquaculture, it is not fundamentally necessary for making a SEPA determination on the farming practice in general. A determination of significance would reflect an acceptance of the precautionary approach to regulating the practice—an industry that has been ongoing now for over 15 years without documentation of any lasting adverse impact on ecological function and with evidence of clear ecological benefit. Further, the industry's Environmental Code of Practices, and the Best Management Practices required by state and federal resource agencies minimize the risks of adverse impacts from occurring from Washington's geoduck industry. New research will continue on the species and the practice regardless of SEPA decisions made at local jurisdictional levels; if this research highlights a better way to do things (i.e., the ‘better mouse trap’), it will be implemented—such is the use and nature of science. The best available science regarding intertidal geoduck clam aquaculture, as outlined in this technical memorandum, indicates that significant[1] adverse effects are unlikely and a SEPA determination of non-significance is fully appropriate for this sustainable industry.

References Cited

Bartoli, M., D. Nizzoli, P. Viaroli, E. Turolla, G. Castaldelli, E. A. Fano, and R. Rossi. 2001. Impact of *Tapes philippinarum* farming on nutrient dynamics and benthic respiration in the Sacca di Goro. Hydrobiologia 455: 203-212.

Beadman, H. A., M. J. Kaiser, M. Galanidi, R. Shucksmith, and R. I. Willows. 2004. Changes in species richness with stocking density of marine bivalves. Journal of Applied Ecology 41: 464-475.

Blackmon, D., Wyllie-Echeverria, T., and Shafer, D. 2006. The role of seagrasses and kelps in marine fish support. WRAP Technical Notes Collection, U. S. Army Engineer Research and Development Center, Vicksburg, MS. ERDC TN-WRAP-06-1 http://el.erdc.usace.army.mil/wrap

Buckley, R. M., and G. J. Hueckel. 1985. Biological processes and ecological development on an artificial reef in Puget Sound, Washington. Bulletin of Marine Science 37: 50-69.

Campbell, A., and M. D. Ming. 2003. Maturity and growth of the Pacific geoduck clam, *Panopea abrupta*, in southern British Columbia, Canada. Journal of Shellfish Research 22: 85-90.

Carlson, H. R. 1980. Seasonal distribution and environment of Pacific herring near Auk Bay, Lynn Canal, southeastern Alaska. Transactions of the American Fisheries Society 109: 71-78.

Connolly, L. M., and M. A. Colwell. 2005. Comparative use of longline oysterbeds and adjacent tidal flats by waterbirds. Bird Conservation International 15: 237-255.

Craig, C. A., K. Rowell, and J. L. Ruesink. 2006. How does geoduck aquaculture affect eelgrass in south Puget Sound? Presented at the annual meeting of the Pacific Estuarine Research Society, February 16-18, 2006, Friday Harbor, Washington.

Dankers, N., and D. R. Zuidema. 1995. The role of the mussel (*Mytilus edulis* L.) and mussel culture in the Dutch Wadden Sea. Estuaries 18 (1A): 71-80.

Dealteris, J. T., B. D. Kilpatrick, and R. B. Rheault. 2004. A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and non-vegetated seabed. Journal of Shellfish Research 23: 867-874.

Dernie, K. M., M. J. Kaiser, and R. M. Warwick. 2003. Recovery rates of benthic communities following physical disturbance. Journal of Animal Ecology 72: 1043-1056.

Dethier, M. N., A. Leitman, and W. Matthews. 2007. Concerns and questions relevant to infaunal and epibenthic impacts of geoduck aquaculture. Draft scoping document prepared by Marine Surveys and Assessments, Port Townsend, Washington.

DNR (Department of Natural Resources). 2006. Best Management Practices (BMPs) for geoduck aquaculture on state-owned aquatic lands in Washington State. Washington Department of Natural Resources, Olympia, Washington (dated November 2006). 7 pp.

Ferns, P. N., D. M. Rostron, and H. Y. Syman. 2000. Effects of mechanical cockle harvesting on intertidal communities. Journal of Applied Ecology 37: 464-474.

Finlayson, D., and H. Shipman. 2003. Puget Sound drift cells: the importance of waves and wave climate. Puget Sound Notes (science news from the Puget Sound Action Team), September, Issue No. 47: 1-4.

Fleece, C., D. Waller, J. Fisher, J. Vanderpham, and G. Reub. 2004. Programmatic biological evaluation of potential impacts of intertidal geoduck culture facilities to endangered species and essential fish habitat. Draft biological evaluation prepared on October 27, 2004 by Entrix, Inc. for Taylor Shellfish, Seattle Shellfish, and Chelsea Farms, Olympia, Washington.

Gaeckle, J., P. Dowty, B. Reeves, H. Berry, S. Wyllie-Echeverria, and T. Mumford. 2007. Puget Sound Submerged Vegetation Monitoring Project 2005 Monitoring Report. Washington Department of Natural Resources, Olympia, WA.

Gingras, M.K., and S.G. Pemberton. 2000. A field method for determining the firmness of colonized sediment substrates. Journal of Sedimentary Research 70(6): 1341-1344.

Gingras, M.K., S.G. Pemberton, and T. Saunders. 2000. Firmness profiles associated with tidal-creek deposits: the temporal significance of *Glossifungites* assemblages. Journal of Sedimentary Research 70(5): 1017-1025.

Grant, J., G. Bugden, E. Horne, M. C. Archambault, and M. Carreau. 2007. Remote sensing of particle depletion by coastal suspension-feeders. Canadian Journal of Fisheries and Aquatic Sciences 64:387-390. [As cited in geoduck lit rev]

Gregg, K. L. 1995. Comparisons of three manufactured artificial reef units in Onslow Bay, North Carolina. North American Journal of Fisheries Management 15: 316-324.

Hall, S. J., and M. J. C. Harding. 1997. Physical disturbance and marine benthic communities: the effects of mechanical harvesting of cockles on non-target benthic infauna. Journal of Applied Ecology 34: 497-517.

Heggenes, J., M. Beere, P. Tamkee, and E. B. Taylor. 2006. Genetic diversity in steelhead before and after conservation hatchery operation in a coastal, boreal river. Transactions of the American Fisheries Society 135: 251-267.

Hilgerloh, G., J. O' Halloran, T. C. Kelly, and G. M. Burnell. 2001. A preliminary study on the effects of oyster culturing structures on birds in a sheltered Irish estuary. Hydrobiologia 465: 175-180.

Hueckel, G. J., and R. L. Stayton. 1982. Fish foraging on an artificial reef in Puget Sound, Washington. Marine Fisheries Review 44: 38-44.

Hueckel, G. J., and R. M. Buckley. 1987. The influence of prey communities on fish species assemblages on artificial reefs in Puget Sound, Washington. Environmental Biology of Fishes 19: 195-214.

Iversen, E. S., and S. P. Bannerot. 1984. Artificial reefs under marina docks in southern Florida. North American Journal of Fisheries Management 4: 294-299.

Kelly, J.R. and J.P. Volpe. 2007. Native eelgrass (*Zostera marina* L.) survival and growth adjacent to non-native oysters (*Crassostrea gigas* Thunberg) in the Strait of Georgia, British Columbia. Botanical Marina 50:143-150.

Korstrom, J. S., and I. K. Birtwell. 2006. Effects of suspended sediment on the escape behavior and cover-seeking response of juvenile Chinook salmon in freshwater. Transactions of the American Fisheries Society 135: 1006-1016.

Laffargue, P., M-L. Bégout, and F. Lagardère. 2006. Testing the potential effects of shellfish farming on swimming activity and spatial distribution of sole (*Solea solea*) in a mesocosm. ICES Journal of Marine Science 63: 1014-1028.McKindsey, C. W., H. Thetmeyer, T. Landry, and W. Silvert. 2006. Review of recent carrying capacity models for bivalve culture and recommendations for research and management. Aquaculture 261: 451–462

Miller, K. M., K. J. Supernault, S. Li, and R. Withler. 2006. Population structure in two marine invertebrate species (*Panopea abrupta* and *Strongylocntrotus franciscanus*) targeted for aquaculture and enhancement in British Columbia. Journal of Shellfish Research 25: 33-42.

Lassuy, D.R.. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest)—Pacific herring. U.S. Fish and Wildlife Service Biological Report 82(11.126).

LeClair, L. L., S. R. Phelps, and T. J. Tynan. 1999. Little gene flow from a hatchery strain of chum salmon to local wild populations. North American Journal of Fisheries Management 19: 530-535.

Lewis, T. L., D. Esler, and W. S. Boyd. 2007. Effects of predation by sea ducks on clam abundance in soft-bottom intertidal habitats. Marine Ecology Progress Series 329: 131-144.

Lindahl, O., R. Hart, B. Hernroth, S. Kollberg, L. O. Loo, L. Olrog, A. S. Rehnstam-Holm, J. Svensson, S. Svensson, and U. Syversen. 2005. Improving marine water quality by mussel farming: A profitable solution for Swedish society. Ambio 34:131-138.

Long, E. 2005. Planted oyster shells appear to be perfect for plover nests. Chinook Observer, September 14, 2005: A1 and A14. Long Beach, Washington.

Meyer, D. L., and E. C. Townsend. 2000. Faunal utilization of created intertidal eastern oyster (*Crassostrea virginica*) reefs in the southeastern United States. Estuaries 23: 34-45.

Miller, K. M., K. J. Supernault, S. Li, and R. Withler. 2006. Population structure in two marine invertebrate species (*Panopea abrupta* and *Strongylocntrotus franciscanus*) targeted for aquaculture and enhancement in British Columbia. Journal of Shellfish Research 25: 33-42.

Newcombe, C. P., and J. O. T. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16: 693-727.

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., 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 27:793-806.

Nightingale, B., C. A. Simenstad. 2001. Overwater structures: marine issues. White Paper Research Project T1803. Washington State Transportation Commission and U.S. Department of Transportation.

Nizzoli, D., M. Bartoli, and P. Viaroli. 2007. Oxygen and ammonium dynamics during a farming cycle of the bivalve Tapes philippinarum. Hydrobiologia 587: 25–36.

Norris, K., R. C. A. Bannister, and P. W. Walker. 1998. Changes in the number of oystercatchers *Haematopus ostralegus* wintering in the Burry Inlet in relation to the biomass of cockles *Cerastoderma edule* and its commercial exploitation. Journal of Applied Ecology 35: 75-85.

O'Beirn, F. X., P. G. Ross, and M. W. Luckenbach. 2004. Organisms associated with oysters cultured in floating systems in Virginia, USA. Journal of Shellfish Research 23: 825-829.

Pearson, S.F., C. Sundstrom, K. Brennan, and M. Fernandez. 2007. Snowy plover distribution, abundance and reproductive success: 2006 research progress report. Washington Department of Fish and Wildlife, Wildlife Science Division, Olympia.

Pearce, C. M., Y. X. An, J. M. Blackburn, L. J. Keddy, D. L. Paltzat, and S. W. Williams. 2007. Geoduck aquaculture: an examination of predator protection methodology and potential environmental impacts. PowerPoint presentation at the October 19 meeting of the B. C. Shellfish Growers Association.

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. FWS/OBS-84/24.

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.

Rumrill, S.S. and V.K. Poulton. 2003. Ecological role and potential impacts of molluscan shellfish culture in the estuarine environment of Humboldt Bay, CA. Journal of Shellfish Research 22: 607.

Sargent, P. S., R. S. Gregory, and D. C. Schneider. 2006. Density responses of subarctic coastal marine fish and crabs to artificial reefs. Transactions of the American Fisheries Society 135: 348-360.

Sharma, R., B. Nagendernath, A. B. Valsangkar, G. Parthiban, K. M. Sivakolundu, and G. Walker. 2000. Benthic disturbance and impact experiments in the central Indian Ocean basin. Marine Georesources and Geotechnology 18: 209-221.

Short, F.T. and S. Wyllie-Echeverria. 1996. Natural and human-induced disturbance of seagrasses. Environ. Conserv. 23:17-27 [As cited by Kelly and Volpe 2007]

Simenstad, C.A. 1994. Faunal associations and ecological interations in seagrass communities of the Pacific Northwest. Pp. 11-18 In: Wyllie-Echeverria, S. A.M. Olson, and M.J. Hershman (eds). Seagrass science and policy in the Pacific Northwest: Proceedings of a seminar series (SMA 94-1). EPA 91-/R-94-004.

Sorokin, I. I., O. Giovanardi, F. Pranovi, and P. I. Sorokin. 1999. Need for restricting bivalve culture in the southern basin of the Lagoon of Venice. Hydrobiologia 400: 141-148.

Sorokin, I. I., O. Giovanardi, F. Pranovi, and P. I. Sorokin. 1999. Need for restricting bivalve culture in the southern basin of the Lagoon of Venice. Hydrobiologia 400: 141-148.

Straus, K. M., L. M. Crosson, and B. Vadopalas. 2008. Effects of geoduck aquaculture on the environment: a synthesis of current knowledge. Washington Sea Grant, Seattle, Washington.

Sumer, B. M., and J. Fredsøe. 1998. Wave scour around a group of vertical piles. Journal of Waterway, Port, Coastal, and Ocean Engineering 124: 248-256.

Tallman, J. C., and G. E. Forrester. 2007. Oyster grow-out cages function as artificial reefs for temperate fishes. Transactions of the American Fisheries Society 136: 790-799.

Terich, T. A. 1987. Living with the shore of Puget Sound and the Georgia Strait. Duke University Press, Durham, North Carolina.

Uchiyama, Y. 2007. Hydrodynamics and associated morphological variations on an estuarine intertidal sand flat. Journal of Coastal Research 23(4): 1015-1027.

Vadopalas, B., L. L. Leclair, and P. Bentzen. 2004. Microsatellite and allozyme analyses reveal few genetic differences among spatially distinct aggregations of geoduck clams (*Panopea abrupta*, Conrad 1849). Journal of Shellfish Research 23: 693-706.

Ward, D. L., A. A. Nigro, R. A. Farr, and C. J. Knutson. 1994. Influence of waterway development on migrational characteristics of juvenile salmonids in the lower Willamette River, Oregon. North American Journal of Fisheries Management 14: 362-371.

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

Wilson, U.W. and J.B. Atkinson. 1995. Black brant winter and spring-stages use at two Washington coastal areas in relation to eelgrass abundance. The Condor 97:91-98.

Wisehart, L., and S. Hacker. 2006. Assessing the importance of early life history stages of eelgrass (*Zostera marina* L.) in response to aquaculture disturbance. Presented at the annual meeting of the Pacific Estuarine Research Society, February 16-18, 2006, Friday Harbor, Washington.

Whiteley, J., and L. Bendell-Young. 2007. Ecological implications of intertidal mariculture: observed differences in bivalve community structure between farm and reference sites. Journal of Applied Ecology 44: 495-505.

Wilber, D. H., and D. G. Clarke. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21: 855-875.

Wright, N. 2002. Eelgrass conservation for the B.C. coast: A discussion paper. B.C. Coastal Eelgrass Stewardship Project. [As cited by Kelly and Volpe 2007]

Wulff, A., K. Sundbäck, C. Nilsson, L. Carlson, and B. Jönsson. 1997. Effect of sediment load on the microbenthic community of a shallow-water sandy sediment. Estuaries 20: 547-558.

Wyllie-Echeverria, S., T. Mumford, J. Gaydos and S. Buffum. 2003. *Zostera marina* declines in San Juan County, WA. Westcott Bay Taskforce Mini-Workshop. [As cited by Kelly and Volpe 2007]

Yilmaz, O. 1998. Hydrodynamic interactions of waves with group of truncated vertical cylinders. Journal of Waterway, Port, Coastal, and Ocean Engineering 124: 272-279.

Zhou, Y., H. S. Yang, T. Zhang, S. L. Liu, S. M. Zhang, Q. Liu, J. H. Xiang, and F. S. Zhang. 2006. Influence of filtering and biodeposition by the cultured scallop Chlamys farreri on benthic-pelagic coupling in a eutrophic bay in China. Marine Ecology-Progress Series 317:127-141. [As cited by geoduck Lit. Rev_final revision 11-29-07]

Zydelis, R., D. Esler, W. S. Boyd, D. Lacroix, and M. Kirk. 2006. Habitat use by wintering surf and white-winged scoters: effects of environmental attributes and shellfish aquaculture. Journal of Wildlife Management 70:1754-1762.