Assessing potential benthic impacts of harvesting the Pacific geoduck clam, *Panopea generosa* (Gould, 1850), in intertidal and subtidal sites in British Columbia, Canada

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ABSTRACT: The Pacific geoduck clam (*Panopea generosa*) is the largest burrowing clam in the world and adults can live up to a meter below the sediment surface. In order to extract these clams, harvesters use pressurized water jets to liquefy the surrounding sediments. This type of disturbance could have profound effects on the local benthic environment, but little experimentation has examined this issue. The present research was conducted on both intertidal and subtidal plots to assess potential effects of commercial-scale geoduck harvest on sedimentary benthic environments and nearby eelgrass populations. Sediment and/or eelgrass samples were collected within the harvest plots, at various distances from the harvest plots, and at various times before and after harvest to assess the spatial and temporal scales of potential impact. Sediment qualities examined included: grain size, percent organics, total nitrogen, total organic carbon, sulphide content, redox potential, and infaunal community structure. Eelgrass parameters studied included: shoot length, shoot density, and biomass. Sedimentation rates during the harvest were examined and compared to those of natural occurrence. No significant impacts of harvesting on any of the measured sediment qualities were indicated in the harvest plot, nearby area, or eelgrass bed. No significant effects on eelgrass parameters were observed. Suspended sediments generated during the harvest were generally limited to within the harvest plot and the levels were not greater than those during wind/storm conditions. This study and previous intertidal and subtidal studies in British Columbia and Washington state indicate that commercial geoduck harvesting does not appear to cause significant negative impacts to the benthic environment beyond the borders of the immediate harvest area, including nearby eelgrass beds.

KEY WORDS: Benthic impact · Eelgrass · Geoduck · Harvest · *Panopea generosa*

INTRODUCTION

The Pacific geoduck clam [*Panopea generosa* (Gould, 1850) – erroneously referred to as *P. abrupta* (Conrad, 1849) in most recent publications (see Vadopalas et al. (2010)] is distributed from Alaska to Baja California (28–58^{o}N) (Bernard 1983). It lives in the low intertidal zone and subtidally to as deep as 110 m, buried in sand, silt, gravel, and other soft substrates (Goodwin & Pease 1989, Bureau et al. 2002, Zhang & Hand 2006). It is the largest infaunal clam in the world, growing up to 3.25 kg whole weight and living up to a meter below the sediment surface (Goodwin & Pease 1987). This species is also very long-lived – the oldest geoduck on record being approximately 168 years old (Bureau et al. 2002).

Panopea generosa currently supports the most valuable dive fishery on the west coast of North America, with 1,963 metric tons (MT), worth USD $36.2 million, being landed in Washington state (WA), USA in 2010 (Washington Department of Fish and Wildlife 2012) and 1,600 MT, worth CAD $40.9 million, in British Columbia (BC), Canada in the same year (BC Seafood Industry Year in Review 2010). Aquaculture production of geoduck started intertidally in WA in the mid 1990s and has increased at a relatively rapid rate to a point where approximately 613 MT of cultured clams, worth USD $18.5 million, were harvested in 2010 (Washington Department of Fish and Wildlife 2012). There has been widespread interest in the culture of geoduck in BC for many years, but the commercial-scale development has been hindered until fairly recently by a lack of governmental policy/legislation and concerns about how geoduck culture will impact the environment [despite these issues, 52 MT of farmed geoduck, worth CAD $1.1 million, were harvested in 2010 in BC (BC Ministry of Agriculture 2012)]. These environmental concerns are generally focused on the harvest process as pressurized water jets (called stingers in industry vernacular) are used to liquefy the soft-bottom substrate around the clams in order to extract them. A stinger comprises high-pressure water pumped through approximately 2” hose that runs through an elbow joint and a long metal pipe which the harvester holds. The harvester extracts geoducks individually by inserting the stinger into the substrate around each geoduck to liquefy the substrate with a burst of water and remove the geoduck live. It should be noted that this technique is not just isolated to aquaculturists, as it is also the harvest technique used in the wild geoduck fishery and considered to be the most environmentally benign method available (Palazzi et al. 2001).

Geoduck harvest by water jets appears to be highly disruptive of the substrate (Goodwin 1978, Breen & Shields 1983). During the harvest, sediments are re-suspended in the water column. While large particles will settle fairly rapidly into the harvest vicinity, finer particles will be carried away by the local water currents, forming turbid plumes, and subsequently re-deposited some distance away (Short & Walton 1992). After a geoduck is removed, a shallow hole about 0.5 m in diameter, partially filled with an emulsion of loose substrates and water, is created (Goodwin 1978, Breen & Shields 1983). The potential ecological implications of geoduck harvest, however, seem to extend much beyond these purely physical effects. As the substrates are disturbed, both abiotic and biotic conditions of the sediments may also be altered. The harvest is expected to have the potential to impact the benthic environment in a number of ways: 1) alteration of sediment grain size due to loss of fine particles and loose compaction of re-deposited substrates that are more susceptible to removal by water currents (Goodwin 1978); 2) loss of organic matter, minerals, and heavy metals associated with the loss of fine particles, as the fines (< 63 μm) tend to accumulate or bond such materials more than other grain size fractions, mainly because of their higher surface area (Horowitz & Elrick 1987, Tam & Wong 2000); 3) exposure of anoxic sediments and oxygenation of sediment pore water, affecting sediment chemistry (Palazzi et al. 2001, Straus et al. 2008); 4) release of materials back into the water column, including nutrients, toxic planktonic eggs or cysts, contaminants, and pollutants (Pilskaln et al. 1998, Tengberg et al. 2003, Straus et al. 2008), subsequently affecting water quality and animal and plant growth; 5) reduction in infaunal abundance due to damage, burial, and exposure to currents and predators (Goodwin 1978, Breen & Shields 1983, Currie & Parry 1996); and 6) impact on nearby aquatic communities in areas outside the immediate harvest bed due to turbid plumes and deposition of materials from these plumes (Short & Walton 1992).

The nearby areas of the harvest plots can be important near-shore marine habitats such as open sand/mud flats and eelgrass (seagrass) meadows, both hosting diverse animal and plant communities (Cain & Bradbury 1996, Short & Wyllie-Echeverria 1996, Vermaat et al. 1997, Chambers et al. 1999, Rossi et al. 2007). Deposition of materials from turbid harvest plumes onto the nearby areas may lead to changes in sediment grain size and infaunal communities through burying, smothering, and crushing, subsequently affecting feeding (welfare) of benthic filter/deposit feeders and altering benthic chemical microenvironments (Miller et al. 2002, Airoldi 2003). Furthermore, decreased light levels due to shading, as a result of increased turbidity from sediment plumes and deposition of sediments on eelgrass leaf surfaces, may reduce eelgrass growth and survival (Moore et al. 1997, Cabello-Pasini et al. 2002, Tamaki et al. 2002).

The potential impact of geoduck harvest on benthic environments has been evaluated for commercial subtidal fisheries in both WA and BC, using small experimental plots (Goodwin 1978, Breen & Shields 1983, Short & Walton 1992). Goodwin (1978) compared non-harvested and harvested plots (30 x 3 m each) sampled before harvesting and again seven months after the disturbance and found that: 1) the harvest did not significantly affect sediment grain size distribution in the harvest plot as a whole; 2) the harvest did not create dramatic decreases in the major infaunal species present; 3) the holes created during the disturbance had completely lost their identity by the end of seven months. He also found, however, that there were significant decreases in the percentage of fine and coarse sediments within the harvest holes immediately after the harvest. Breen and Shields (1983) compared non-harvested and harvested plots (6 x 5 m each) where geoducks were completely harvested 10 months prior to the sampling. They found no significant difference in sediment grain size distribution and no simple relationship in infaunal community structure between the two plots (some species decreased and some increased due to the harvest), but an increase in species diversity in the disturbed plot. Short and Walton (1992) examined the transport and fate of suspended plumes resulting from subtidal geoduck harvest through a modeling approach. Their study concluded that most suspended materials settled within 1 m of the harvest holes and that the transport and fate of suspended sediments associated with commercial geoduck harvest would have minimal impacts on the physical environments in the harvest bed and adjacent areas (Short & Walton 1992).

Despite the prevalence of intertidal geoduck aquaculture in WA and the burgeoning commercial interest in BC, few studies have examined the potential harvest affects in the intertidal zone. DFO (2012b) found that the harvest impact to the benthos (*i.e.* grain size, percent organics, total organic carbon, total nitrogen, sulphide concentration, redox levels) was relatively limited in terms of scale and duration with a relatively small (20 x 3 m) harvest plot. Price (2011) compared harvested and non-harvested plots (2,500–4,500 m^2) in each of three sites, concluding that the harvest did not cause any distinctive response patterns in infaunal communities within the harvest plot and that the effect of harvest on infauna was within the range of natural variation experienced by the community and was not of long-term ecological significance. Regarding infaunal community structure, Price (2011) also found that the harvest did not cause any "spillover" effects in areas adjacent (up to 60 m outside) to the harvest plots.

To date, no published studies have examined the potential effects of subtidal or intertidal geoduck harvest on nearby aquatic vegetation (*e.g.* eelgrass) and very few published studies examining the effects of geoduck harvest have been subjected to peer review (DFO 2012b, Ruesink & Rowell 2012). Since cultured or enhanced geoduck densities are generally higher than those of wild stock, aquaculture harvest impacts may be amplified as compared to the wild fisheries. The objective of the present study was to evaluate the spatial and temporal extents of the potential impact of large-scale subtidal and intertidal geoduck harvest on the benthic environment. The evaluation was based on a gradient sampling design, as the disturbance is likely to attenuate with distance from a point of source (Ellis & Schneider 1997). Periodical samplings were used to address temporal variability as with the before-after and control-impact (BACI) sampling design (Stewart-Oaten & Bence 2001). Samples for benthic environments were taken in the harvest plot, nearby area, and eelgrass bed over two years. The present study aims to create a base of evidence to inform government's decision and policy making for the management of geoduck aquaculture in BC and elsewhere.

MATERIALS AND METHODS

Study sites and site layouts

This study was carried out between Oct 2008 and Oct 2010 at two sites located in the Strait of Georgia, BC, Canada, both comprising a harvest plot, a nearby (non-harvest) area, and an eelgrass bed (Fig. 1). The Cortes Island site (CI, 50°02'N, 124°58'W, approximate) was located in the northern Strait of Georgia on a subtidal sandy strip 3.5–7.8 m below chart datum on a portion of a wild commercial geoduck bed in DFO statistical area 15. The harvest plot (100 x 60 m) was a geoduck fisheries enhancement area placed within the commercial bed, previously seeded and ready for harvest during the course of the present study. This enhancement area was seeded with geoducks between 1999 and 2000 (Bruce Clapp, West Coast Geoduck Research

Corporation, personal communication). In 2008 the harvest plot had a surveyed geoduck density of 1.58 ind m^{-2}. The mean density on wild geoduck beds in DFO statistical area 15 is 0.19 ind m^{-2} with a range of 0.03–0.32 ind m^{-2} (DFO 2012a). The nearby area had never been seeded or harvested. The Nanoose Bay site (NB, 49°16'05.68"N, 124°10'43.74"W, center of harvest plot) was located on a shellfish tenure on an intertidal sand flat (3.6–5.1 m above chart datum at high tide). The entire study site, including the harvest plot (30 x 15 m), had not been used for aquaculture operations for many years prior to this study and no geoduck clams were present (currently, there are no commercial-scale intertidal geoduck farming within BC and hence a mimic harvest was conducted). It should be noted that there was a small eelgrass bed in the north east corner of the harvest plot at NB (Fig. 1).

At the start of the project current profiles were conducted at both sites using an Acoustic Doppler Current Profiler (Teledyne RD Instruments, San Diego, CA, USA) set centrally in the harvest plots. Current direction and velocity were recorded every 10 min for a period of 6 and 7 d for CI and NB, respectively. Data from three depth bins were then extracted for both study sites (0.3, 2.8, and 5.7 m above sea bed for CI; 0.2, 0.6, and 1.1 m above sea bed for NB) and averaged to determine the major current directions and velocities for both study sites. The data were then used to establish the transect lines (not physically laid) of the study sites, which ran through the centers of the harvest plots and parallel with the major current direction. As a result, the nearby sampling area was in the down-current direction of the harvest plot (CI and NB) while the nearby eelgrass bed was in the direction paralleling the current (CI and NB) and down-current (NB) (Fig. 1). Typical current speed was 6–18 cm s^{-1} at CI and 0–12 cm s^{-1} at NB during the period of measurements.

In the nearby area, five sampling distances were allocated for CI and six for NB along the transect line. These were 5, 10, 25, 50, and 75 m from the edge of the harvest plot for CI and 1, 5, 10, 25, 50, and 75 m for NB. The harvest plot was considered as 0 m for both study sites. The gradient sampling design assumed that maximum impact occurred at or adjacent to the harvest plot with impact intensity decreasing with distance, dropping to nil at a certain distance from the area of harvest (Borja et al. 2009). The maximum sampling distance covered both potentially impacted and non-impacted areas (*i.e.* 75–100 m; Short & Walton 1992, Price 2011). For the eelgrass bed at CI, four sampling distances from the edge of the harvest plot (5, 10, 25, and 50 m) were assigned (Fig. 1). The eelgrass bed at NB had two directions (shoreward and seaward); three sampling distances (1, 5, and 10 m) being used for each direction (Fig. 1). The maximum eelgrass-bed sampling distances approximated the eelgrass boundary or the access limit during low tides (*i.e.* the seaward direction at NB) of the study sites.

Sampling schedules

Samples were taken in the harvest plot, nearby area, and eelgrass bed over a two-year period (Table 1). At each time, samples were taken at each sampling distance in the nearby area and eelgrass bed from five sampling points, which were spaced approximately evenly across the length or width of the harvest plot. Samples were also taken from five random sampling points within the harvest plot at each sampling time for both study sites (Fig. 1).

The benthic environments examined and samples collected included:

- Harvest plot and nearby area
    - Sediment physics: sediment grain size

- Sediment chemistry: percent organics, total nitrogen, total organic carbon, sulphide content, and redox potential
    - Infaunal community structure
    - Sedimentation during harvest
- Eelgrass bed
    - Sediment physics: sediment grain size
    - Infaunal community structure
    - Eelgrass population
    - Sedimentation during harvest

Note that eelgrass samples were not taken immediately after the harvest at both study sites, since the harvest was not directly done on the eelgrass bed, except for the small corner of the eelgrass bed in the harvest plot at NB (see Fig. 1 and Discussion). Any indirect harvest effect on eelgrass will not be detected until after a prolonged period of time. Additional samplings were added to monitor seasonal eelgrass variations.

Sample collection and processing

Sediment physics and chemistry

At each sampling point and time, the top 2-cm layer of sediments was collected [using a sample corer (6.5 cm diameter x 20 cm height)], transported to the laboratory on ice, and frozen at -20°C. After samples were thawed and overlying seawater was removed, sub-samples were taken and freeze dried to determine percent organics, total nitrogen, and total organic carbon. The remaining samples were dried at 60°C to constant weight to determine sediment grain size. Percent organics were determined as a percentage of sample dry weight loss after combustion at

500°C for 5 h. Total nitrogen and organic carbon were determined by high-temperature combustion in a Carlo Erba CHN analyzer (NA-1500) and expressed as percentages of sample dry weight. Sediment grain size was determined by sifting samples through a series of nested 203-mm diameter sieves on a sediment shaker. Particle compositions were calculated as percentages of total sample dry weight for gravel (> 2,000 μm), very coarse/coarse sand (2,000–500 μm), medium sand (500–250 μm), fine/very fine sand (250–63 μm), and silt/clay (< 63 μm), according to the Wentworth (1922) scale.

Sulphide content and redox potential were measured for sediments collected at 2 and 6 cm depth. At CI, a sample corer (6.5 cm diameter x 20 cm height) with two small holes (1.7 cm diameter, 4 cm apart vertically) was pushed into the seabed at each sampling point to position the two holes right at the 2 and 6 cm depths. A sediment sample was then taken from each hole using a 10-ml cut-off plastic syringe. The syringe was sealed air-tight, stored on ice, and transported to the laboratory. At NB, a sample corer as above (but with the two holes sealed with duct tape) was pushed into the seabed at each sampling point. The whole corer, with sediments filled inside, was then capped (sealed air-tight) at the two ends and brought back on ice to the laboratory, as the presence of gravels in sediments made it difficult to apply the syringes on site (due to time limitation). Samples were analyzed within several hours after collection for both study sites. Prior to analysis, samples were left in the dark and kept at room temperature for 1 h. Sulphide content was measured with a silver/sulphide electrode and redox potential with a platinum redox electrode after the method of Wildish et al. (1999). The redox potential readings were corrected to the standard hydrogen reference electrode.

Infaunal community structure

A sediment core (6.5 cm diameter x 10 cm height) was collected at each sampling point and time. After overnight storage at 4^{o}C, the cores were washed on a 1.0-mm sieve and the resultant material was preserved in 8% phosphate-buffered formalin for at least one week and then in 70% ethanol for longer-term storage. Observed organisms were classified to the lowest taxomic levels by an infaunal taxonomy specialist (one person). The number of species, number of individuals, and Shannon-Wiener's index were calculated for each sample core (Crawford et al. 2003).

Eelgrass parameters

Eelgrass samples were taken from a 40 x 40 cm sample quadrat at each eelgrass-bed sampling point and time. All above-ground shoots in these quadrats were severed and stored at -20^{o}C until analysis. The thawed samples were sorted to determine maximum shoot length (for CI) and shoot density (for CI), and then cleaned of any visible epifauna and dried at 60℃ to constant weight to determine per quadrat biomass (for CI and NB) for each sampling point.

Harvesting and sedimentation during harvest

At CI, a total of 1,554 geoducks, with an average weight of 0.82 kg, were harvested in two work days by a commercial dive-harvest crew using standard commercial harvest gear (high-pressure water and a stinger). This represented a harvest intensity of 0.26 ind m^{-2} on the 6,000 m^{2} harvest plot. This harvest intensity illustrates how potential impacts from cultured/enhanced geoduck harvesting may be amplified compared to the wild geoduck fishery. The upper end of the densities on wild geoduck beds in the vicinity of the harvest plot is 0.03–0.32 ind m^{-2} (DFO 2012a), where the wild fishery operates on a three year rotation at a harvest rate of 1.8% estimated biomass per year or a maximum of 5.4% estimated biomass every three years (DFO 2012a). Therefore, the wild fishery would target an overall removal rate of 0.02 ind m^{-2} every three years at the upper end of the densities of wild geoduck beds near the study site. Individual clams were identified by their show (siphon tip protruding from the sediment surface) and harvested one by one. At NB, a mimic harvest was performed as there were no geoduck clams present. This was done by inserting a pressurized water jet (standard stinger powered by a 5.5 hp Honda WH29 water pump) repeatedly into the substrate across the 450-m^2 harvest plot during a low tide, creating approximately 9 holes m^{-2} (the whole plot was essentially disturbed). A small corner of the eelgrass bed at NB was also disturbed (Fig. 1)

Deposition of suspended materials created by the harvest was determined using sediment traps. For both study sites, three sediment traps were used in the harvest plot (along the central line perpendicular to the transect line) and at each sampling distance in the nearby area and in the eelgrass bed (Fig. 1). Each trap was 40 cm high and 7.7 cm in diameter with an aspect ratio of 5:1 (Ongley 2006). Prior to harvest, the traps were deployed for 2 d, to collect background suspended sediment data, and then redeployed just before the harvest and collected 2–3 d later when the harvest was completed. It should be noted that for harvest-related sediment collection, the subtidal traps collected both sediments created during the harvest and those re-deposited by water currents after the harvest was completed. The intertidal traps, however, only collected sediments re-deposited by water currents after the harvest was done as the tide came in. It should also be noted that, for both study sites, it was quite windy before the harvest, but very calm during/after the harvest.

At each sampling point, the trap was placed in a larger PVC pipe, embedded in the seabed, to minimize disturbance of the surrounding sediments during the setup and removal of the traps. At

NB, sediments inside the larger PVC pipes were carefully dug out, so that the openings of the traps placed inside were about 15 cm above the seabed, to increase submersion time of the traps as the tide came in. However, no sediments in the larger PVC pipes were removed at CI and the openings of the traps were 40 cm above the seabed.

After recovery, the traps sat in the dark for at least 12 h to allow suspended material to settle. The overlying seawater was then siphoned off as much as possible. The trapped materials were transferred into pre-weighed 50-ml plastic tubes and centrifuged for 10 min at 3,000 rpm or 1,509 g. The resultant solids were washed with distilled water, centrifuged again with the same conditions as above, and dried at 60℃ to constant weight. Sedimentation rates were determined as dry sediment weight collected per trap per day (g $trap^{-1}$ d^{-1}) for each sampling point.

Additional sampling for sedimentation

Sedimentation during a winter-storm event at CI

Sediments were collected at CI during a winter storm event in Feb 2011. Six sediment traps (three in the nearby area and three in the eelgrass bed) were deployed just before the storm (Feb 11) and retrieved after the storm (Feb 16). The wind speed was 9.8/20 km h^{-1} (average/maximum hourly) on Feb 11, 19.7/33 km h^{-1} on Feb 12, 13.4/28 km h^{-1} on Feb 13, 20.7/35 km h^{-1} on Feb 14, 7.0/19 km h^{-1} on Feb 15, and 6.3/15 km h^{-1} on Feb 16, as recorded by the closest weather station at Campbell River, BC (Climate ID: 1021261; Meteorological Service of Canada 2012). The wind direction came mostly from the southeast, which would have the highest impact at CI. Background data on suspended sediments for a calm sea were not collected until Mar 20–24, 2011 as various storm events passed through the area for a prolonged period of time.

Annual sedimentation at NB

Winter storm sampling at NB was not possible as storm events never occurred at a suitable low tide during the study period (in order to sample when the tide was out). Instead, annual sedimentation rate was monitored for this study site every 2–3 months for one year (Apr 2009–2010). At each sampling time, nine sediment traps (three in the nearby area and three in both directions of the eelgrass bed) were deployed for 11–14 d during a full tidal cycle. For both study sites, the setup of sediment traps and processing of sediment samples were the same as previously described.

Statistics

Statistical analysis was facilitated using the software NCSS 2007 (Kaysville, Utah, USA). Data were analyzed using two-way fixed ANOVA, with sampling distance and time set as the main factors and each sampling point as a replicate (n=5). The two study sites were analyzed separately. Within each study site, the harvest plot and nearby area were grouped together (0–75 m) and analyzed separately from the eelgrass bed (5–50 m for CI and 1–10 m for NB). The two directions of eelgrass bed at NB were also analyzed separately. Additionally, for sediments collected during the harvest, the harvest plot was grouped with the eelgrass bed for analysis (0–50 m for CI and 0–10 m for NB). One-way ANOVA was used to examine the temporal pattern of sediments collected during the additional sampling. Data were log-transformed, where applicable, to satisfy conditions of normality and homogeneity (Underwood 1997), as confirmed by the Kolmogorov-Smirnov test and Levene's test, respectively. Some very high, sporadic, outlier sediment values collected at some sampling points during the harvest were removed from the analyses in order to satisfy normality and homogeneity. Data in the text are presented as the range from the lowest to the highest means observed across the different distances over the study period for each variable examined, unless otherwise specified.

Interpretations of the potential harvest effect are based on concepts of the BACI design (Green 1979, Steward-Oaten et al. 1986, 1992): the affected distance (site) will show a different response pattern from the unaffected distance (control) after the harvest (disturbance), as manifested by the significant interaction between sampling distance and time. This is irrespective of the main effects due to the likely heterogeneity across space and the considerable natural variability over time. If the interaction between sampling distance and time is insignificant ($P > 0.05$), this suggests that each distance (including the harvest plot) shows the same pattern of variation in response to time, therefore indicating that the harvest effect is likely none. If, however, the interaction is significant ($P < 0.05$), this does not necessarily mean that the harvest effect is also significant. Two-way ANOVAs followed by post-hoc analyses (Newman Keuls, NK) are used to identify where and when the significance occurs for correct indication of any harvest effect, because of the serial sampling distances and times adopted here.

RESULTS

Harvest plot and nearby area

Sediment physics and chemistry

Sediments of the harvest plot and nearby area at CI were mainly composed of medium sand (48.0–58.8%), followed by very coarse/coarse and fine/very fine sands (17.5–26.5 and 18.9–26.9%, respectively). Silt/clay accounted for only < 0.3% of the sediments and no gravel was encountered (Fig. 2). Percent organics varied in the range of 0.42–0.64%, total nitrogen 0.015–0.025%, and total organic carbon 0.078–0.169%. Sulphide contents were 12.5–326.4 μM at 2 cm depth and 45.4–273.0 μM at 6 cm depth. Redox potential at the respective depths was 188.5–

334.8 mV and 186.5–323.7 mV (Fig. 3). ANOVAs did not reveal any significant ($P > 0.05$) interactions between sampling distance and time for all the above sediment characteristics, except for redox potential at 2 cm depth (Fig. 2, 3). A NK test revealed that this significance was related to time sequence only (significant differences between -12 and -8 at 10 m, between -12 and +6 at 10 m, between -12 and +12 at 25 m, between -0 and +6 at 50 m, and between -3 and +6 at 75 m). There is no consistent pattern to relate this significance to the harvest.

Sediments of the harvest plot and nearby area at NB were mainly composed of fine/very fine sand (41.8–82.2%). The site was also presented with a wide range of gravels (0.1–36.5%), suggesting a relatively heterogeneous sediment composition. Percentages of very coarse/coarse sand, medium sand, and silt/clay were relatively low (2.8–13.3, 8.7–25.6, and 2.5–7.5%, respectively) (Fig. 4). There were significant ($P < 0.05$) interactions between sampling distance and time for all the sediment grain sizes, except for silt/clay (Fig. 4). These significant interactions, however, were all related to the +18 sampling (Apr 30, 2010) and they no longer existed when data at this sampling time were removed from each analysis ($F_{30,\ 168}$ = 1.17, 1.37, 1.44, and 1.50, respectively, all $P > 0.05$). In fact, at the +18 sampling, a recent land-water runoff event had swept away more finer sediments at 50 and 75 m, but done the opposite to the other distances (Fig. 4). Percent organics varied in the range of 0.80–1.54%, total nitrogen 0.034–0.074%, and total organic carbon 0.27–0.56% (Fig. 5). Sulphide contents were 34.7–445.7 and 152.9–492.5 μM at the 2 and 6 cm depths, respectively, and redox potential 120.3–262.9 and 91.1–257.0 mV, respectively. None of the interactions between sampling distance and time were significant ($P > 0.05$) for any of the sediment chemistry variables examined at NB (Fig. 5).

Infaunal community structure

The number of species per core at CI was 7.6–25.2, the number of individuals 11.2–61.6, and Shannon-Wiener's index 1.6–2.8. None of the interactions between sampling distance and time were significant ($P > 0.05$) for these variables (Fig. 6). At each sampling time, annelids, arthropods, and mollusks (predominately bivalves) were the most presented infauna, accounting for 20.0–44.3, 20.4–49.7, and 12.0–46.4% of the respective total individuals enumerated over the entire harvest plot and nearby area.

At NB, the numbers of species and individuals per core were 5.2–16.6 and 10.2–98.0, respectively. No significant ($P > 0.05$) interaction was found between sampling distance and time. Shannon-Wiener's index was 1.0–2.2 and the interaction between sampling distance and time was significant ($P < 0.05$) (Fig. 7). A NK test revealed that the significance was related to time sequence only (significant differences between +6 and +24 at 0 m and between +3 and +6 at 5 m). Again, there is no consistent pattern to relate the significance to the harvest. Annelids, arthropods, and mollusks (predominately bivalves) were the most abundant fauna observed at each sampling time, accounting for 38.1–59.6, 17.7–50.4, and 6.3–20.8%, respectively, of the total individuals counted in the entire harvest plot and nearby area. The top five species observed by number of individuals in each of the three most presented infaunal groups are listed in Table 2 (a) and (b) for the harvest plot and nearby area of CI and NB, respectively.

Sedimentation during harvest

At CI, sediments collected at each distance (0–75 m) varied in the range of 0.22–0.69 g trap^{-1} d^{-1} before the harvest, but were lower (0.04–0.09 g trap^{-1} d^{-1}) during the harvest except for the harvest plot (0.88 g trap^{-1} d^{-1}) and the 5-m distance (5.72 g trap^{-1} d^{-1}) (Fig. 8). The much higher value at 5 m was caused by one large replicate value (16.86 g trap^{-1} d^{-1}), which was likely due to direct "spill" from the harvest. After this larger value was removed from the analysis, ANOVA showed that the interaction between sampling distance and time was significant ($P < 0.05$) (Fig. 8). A NK test revealed that there was no significant ($P > 0.05$) difference among all distances in the background before-harvest data. During the harvest, sediment levels collected in the harvest plot (0 m) were significantly ($P < 0.05$) higher than those at all the other distances except for 5 m, yet comparable ($P > 0.05$) to those before the harvest. When compared to the before-harvest data, although generally less sediment was collected at each distance from 5 to 75 m during the harvest than before the harvest, the differences were significant ($P < 0.05$) only for 75 m.

At NB, sediments collected at each distance (0–75 m) ranged between 0.78 and 1.47 g trap^{-1} d^{-1} before the harvest, but were lower (0.09–0.62 g trap^{-1} d^{-1}) during the harvest (Fig. 8). After removal of a relatively large replicate value at 5 m during the harvest (1.37 g trap^{-1} d^{-1}) from the analysis, ANOVA found that the interaction between sampling distance and time was significant ($P < 0.05$) (Fig. 8). A NK test found that significantly ($P < 0.05$) less sediment was collected during the harvest than before the harvest at each distance (1–75 m) except for the harvest plot (0 m).

Eelgrass bed

Sediment physics

At CI, sediment compositions of the eelgrass bed were similar to those of the harvest plot and nearby area, being 13.1–28.2% for very coarse/coarse sand, 43.3–58.5% for medium sand, 18.9–40.7% for very fine/fine sand, and < 0.5% for silt/clay (Fig. 2). There were no significant ($P > 0.05$) interactions between sampling distance and time for any of the grain size fractions (Fig. 2).

Sediment compositions of the eelgrass beds at NB were predominately fine/very fine sand (63.5–84.6 and 71.1–88.3% for the seaward and shoreward beds, respectively), followed by medium sand (7.2–18.6 and 6.5–18.0%), very coarse/coarse sand (3.3–12.2 and 1.0–5.8), and silt/clay (2.6–6.8 and 2.0–9.4%). Gravels were generally low (< 4.0%). No interactions between sampling distance and time were significant ($P > 0.05$) for any of the grain sizes classified in both eelgrass beds at NB (Fig. 4).

Infaunal community structure

At CI, the number of species, the number of individuals, and Shannon-Wiener's index were 6.6–20.2, 13.4–95.0, and 1.4–2.6 per core, respectively (Fig. 6). There were no significant ($P > 0.05$) differences in the interactions between sampling distance and time for any of these three variables (Fig. 6). At each sampling time, mollusks (bivalves) were the more observed infaunal group, accounting for 37.5–63.7% of the total number of individuals counted over the entire eelgrass bed, followed by annelids and arthropods (13.6–30.7 and 16.1–42.2%, respectively).

Infaunal community structure at NB was similar between the seaward and shoreward eelgrass beds (number of species per core: 7.2–17.0 and 6.2–15.6; number of individuals per core: 14.0–85.2 and 13.4–80.8; Shannon-Wiener's index: 1.7–2.3 and 1.4–2.4) (Fig. 7). There were no significant ($P > 0.05$) interactions between sampling distance and time for any of the variables assessed (Fig. 7). At each sampling time, annelids, arthropods, and mollusks (predominately bivalves) were the most common infaunal taxa, accounting for 30.5–62.8, 3.1–44.4, and 11.3–41.1%, respectively, of the total number of individuals enumerated over the entire eelgrass beds. The top five species observed by number of individuals in each of the three most abundant infaunal groups are listed in Table 2 (c) and (d) for the eelgrass bed of CI and NB, respectively.

Eelgrass parameters

At CI, maximum shoot length of eelgrass ranged from 45.4 to 76.8 mm, shoot density from 3.5 to 16.5 quadrat^{-1}, and biomass from 1.28 to 7.83 g quadrat^{-1} (Fig. 9). None of the interactions between sampling distance and time were significant ($P > 0.05$) (Fig. 9). The eelgrass species present was exclusively *Zostera marina*.

The eelgrass biomass at NB was in the range of 0.57–9.23 g quadrat^{-1} for the seaward bed and 0.97–12.58 g quadrat^{-1} for the shoreward bed (Fig. 9). The interactions between sampling distance and time were not significant ($P > 0.05$) (Fig. 9). The eelgrass species present were *Z. marina* and *Z. japonica*. The inconsistent distribution of the two eelgrass species over space and time made it difficult to compare such variables as shoot length and density.

Sedimentation during harvest

At CI, the amounts of sediments collected at each distance (0–50 m) were 0.28–0.83 g trap^{-1} d^{-1} before the harvest. Lower amounts of sediment were collected at each distance during the harvest (0.02–0.04 g trap^{-1} d^{-1}), except for the harvest plot (0 m) (0.88 g trap^{-1} d^{-1}) (Fig. 8). ANOVA results showed that the effects of sampling distance, time, and the interaction were all significant ($P < 0.05$, Fig. 8). A NK test revealed that there were no significant ($P > 0.05$) differences among all the distances in the background before-harvest data. During the harvest, significantly ($P < 0.05$) more sediments were collected within the harvest plot (0 m) than at any of the eelgrass distances (5–50 m). The lower amounts of sediment collected during the harvest were also significantly ($P < 0.05$) different from those before the harvest at each eelgrass distance (5–50 m).

At NB, the amounts of sediments collected at each distance (0–10 m) before the harvest were 0.65–1.08 g trap^{-1} d^{-1} in the seaward bed and 1.12–4.34 g trap^{-1} d^{-1} in the shoreward bed. During the harvest, the amounts were lower at 1 and 10 m (0.26 and 0.59 g trap^{-1} d^{-1}) of the seaward bed, 5 and 10 m (0.36 and 0.26 g trap^{-1} d^{-1}) of the shoreward bed (Fig. 8), and the harvest plot (0 m) as well (0.45 g trap^{-1} d^{-1}). Higher amounts of sediments were observed during the harvest at 5 m of the seaward bed (2.92 g trap^{-1} d^{-1}) and at 1 m of the shoreward bed (2.22 g trap^{-1} d^{-1}), caused by two (2.87 and 5.64 g trap^{-1} d^{-1}) and one (5.64 g trap^{-1} d^{-1}) larger replicate value(s), respectively. When these larger values were excluded from the analysis, ANOVA revealed that for both eelgrass beds, the effects of sampling distance and the interaction between time and distance were not significant ($P > 0.05$), but significantly ($P < 0.05$) less sediment was collected during than before the harvest at each distance (0–10 m) (Fig. 8).

Sedimentation from additional sampling

The amount of sediment collected during the winter storm event at CI was 0.36 ± 0.02 g trap^{-1} d^{-1} (mean ± SE, $n = 6$) which was significantly ($F_{1,10} = 69.95$, $P < 0.01$) higher than that collected during a calm sea (0.02 ± 0.00 g trap^{-1} d^{-1}).

The annual sedimentation rates at NB stayed relatively low in Apr, Jun, and Aug (0.48 ± 0.09, 0.22 ± 0.06, and 0.10 ± 0.07 g trap^{-1} d^{-1}, respectively; mean ± SE, $n = 9$), elevated in Nov (2.07 ± 1.48 g trap^{-1} d^{-1}), and peaked in Jan (9.04 ± 2.35 g trap^{-1} d^{-1}), after which the rates decreased (1.92 ± 0.58 g trap^{-1} d^{-1} in next Apr). The amount of sediment collected in Jan was significantly ($P < 0.05$) higher than that at any other time of the year. Nov to Mar is usually the heavy precipitation season in the study areas (Environment Canada 2012).

DISCUSSION

Experimental design

A major problem to overcome in assessing anthropogenic impact on environments is that there is usually only one potentially impacted site, precluding the choice of randomization in experimentation. This is made complex by the considerable natural variability over time and the likely heterogeneity across space (Steward-Oaten et al. 1986, Underwood 1992). Several experimental designs and statistical analyses have been proposed for detecting the environmental impact of such kind, although it appears that there are no simple solutions (Underwood 1992, Stewart-Oaten et al. 1992, Stewart-Oaten & Bence 2001). Green (1979) proposed a sampling design in which an impact site and a control site are sampled once before and once after the disturbance (the BACI design). The impact site will show a different pattern after the disturbance from the control site and, therefore, the impact can be tested using the null hypothesis that there is no interaction between site and time. The difficulty is that the results may be spatially confounded, because neither site is replicated (Hurlbert 1984), and that the interaction is not interpretable in situations where the two sites vary through time in different ways even when there is no disturbance (Underwood 1997). The BACI version of Steward-Oaten et al. (1986) compares an impact site and a control site by sampling several times before and after the disturbance. This design covers that of Green (1979) and provides a proper temporal resolution that allows interpretation of differences from before to after as being more sustained than simple noise in time between the two sites (Underwood 1992). However, Underwood (1992) pointed out that this is still insufficient because any site-specific temporal difference between the two sites will be interpreted as an impact even if there is none. By comparing an impact site and a set of randomly-chosen control sites over multiple times, the beyond BACI design of Underwood (1992) is believed to offer a satisfactory solution to the problem due to different time courses between two sites (an impact and a control site), which the BACI design cannot overcome (but see Stewart-Oaten & Bence 2001). Ellis and Schneider (1997) stated that there are many circumstances in which a disturbance attenuates with distance from a point source and, in such circumstances, it would seem more appropriate to sample with distance from the disturbance, as with the case of the present study. Indeed, such an approach has been adopted to evaluate the environmental consequences of aquaculture farming practices (Crawford et al. 2003, Borja et al. 2009). However, it is important to note that the distance effect itself is not a clear indication whether or not this is due to disturbance because of the potential confounding from spatial correlation with distance (location) or pseudoreplication as raised by Hurlbert (1984). In reality, there might be a reason why a particular site is chosen for use as a farm, making interpretations of any site difference from surroundings due to the impact effect *per se* very difficult.

Stewart-Oaten and Bence (2001) discussed in detail, based on experimental principles (randomization), why BACI design is far more reliable for detecting impacts than those with only control or reference sites. They stated that the goal of BACI is to detect change at the specific impact site, so no controls are needed. The controls of BACI are not experimental controls to measure the impact effects but covariates, deliberately chosen to be correlated with the impact site [*i.e.* the control and impact sites should not be directly compared and this is consistent with concerns raised by Hurlbert (1984) over the spatial confounding when neither site is replicated]. The requirements for a control of the BACI are that it should be close enough to the impact site to share the same natural processes, and yet far enough away so that it is not affected by the potential disturbance.

However, before an experiment, it is often not known whether the impact and the control sites are comparable with respect to various natural processes (even if the control site is chosen so that it looks similar to the impact site) and how far the impact may extend. Using a control site and a fishing site to examine the effect of commercial geoduck fishing on Dungeness crab catch per unit effort in Hood Canal, WA, Cain and Bradbury (1996) proposed a series of steps to test if the two sites were equally affected by natural (non-fishing) processes. The essence is that if the two sites did not show correlation over time for the pre-fishing samples then the control site would not have been a reliable analog of the fishing site in terms of natural effects. Without being able to "tease out" natural effect at the fishing site, one would be unable to determine if fishing effects had occurred and the experiment would be ended. It is, therefore, reasonable that several control sites are chosen at the same time, and only the ones having been demonstrated to show similar natural effects as the impact site be used for comparison purposes.

Our distance-time sampling strategy appears to have been appropriate in assessing the impact of geoduck harvest towards our research goals. All the distances sampled were located within a limited area, increasing the likelihood of sharing the same natural processes among each other. The maximum sampling distance (75 m) covered both potentially impacted and non-impacted areas (Short & Walton 1992, Price 2011), therefore avoiding complete auto-correlation across the entire study sites. By examining the interactive patterns of sampling distance and time using ANOVA, it is possible to make informative interpretations on the potential benthic impact of harvest of geoduck clams.

Of the various benthic parameters examined for the harvest plots, nearby areas, and eelgrass beds in the present two-year study, the interactions between sampling distance and time were mostly insignificant at both study sites (intertidal and subtidal), except for redox potential at 2 cm depth at CI, Shannon-Wiener's index at NB, and several sediment grain size fractions in the harvest plot and nearby area at NB. For redox and the species indices, the significances were due to time sequence only and seem not to have been directly related to harvest activities. The sediment grain size significance was related to a natural process (large land-water runoff) at the +18 sampling (Apr 30, 2010, Fig. 4), causing different distances to show different variations in sediment grain size at this particular sampling time. Therefore, results of the present study suggest that the overall benthic impacts of harvest of geoduck clams were not perceived in either study site – including the harvest plots, nearby areas, and eelgrass beds – though it did create visible harvest holes during the harvest.

Harvest plot

For both study sites, the insignificant harvest effect on the various benthic parameters measured in the harvest plots over the study period is likely due to the nature of the harvest process itself. Geoduck harvesting by water jets tends to create small, scattered holes but not to disturb the entire seabed (though depending on the harvest intensity), and the harvest leaves most disturbed materials in place (Goodwin 1978, Breen & Shields 1983, Short & Walton 1992). This suggests that the harvest may not necessarily alter much of the overall integrity of the harvest plot. As the results of the present study revealed, there were no obvious changes due to harvest even when a more intensive harvest was applied in the intertidal NB. In reality, due to variations in geoduck show factors over time (*i.e.* not all geoduck siphons are visible at a given moment), the harvest on a target area may occur several times before its final completion (Goodwin 1978, Breen &

Shields 1983). If the harvest occurs on a much larger tract (Price 2011), it is likely that only a portion of the target area will be disturbed at any given time, due to the harvesting capacity using water jets (possibly in excess of 100 ind h^{-1}; Palazzi et al. 2001, Fleece et al. 2004, Dominique Bureau, Fisheries and Oceans Canada, personal communication). As a result, any immediate harvest effect would be spread over space and time and alleviate the overall effect over the entire target area for sampling, as compared to if all geoduck on it were harvested at once.

It is, however, possible that the maximum impact of geoduck harvest, if any, may not be immediate because of some indirect benthic change – an example being infaunal organisms that are exposed by the harvest becoming more vulnerable to subsequent predation (Goodwin 1978, Breen & Shields 1983, Currie & Parry 1996). No indirect benthic changes were perceived in the harvest plots of either study site over the present study period (with two years post-harvest sampling in the intertidal and one year in the subtidal sites). In other subtidal studies, no dramatic changes in sediment size distribution and no major change or simple relationship in infaunal community structure were found in harvest plots 7 or 10 months after the disturbance (Goodwin 1978, Breen & Shields 1983). Species diversity (Shannon Index) actually increased as a result of harvesting in the Breen & Shields (1983) study. Similarly, intertidal harvests did not appear to significantly negatively affect various benthic parameters, including infaunal community structure, over time in harvest plots (Price 2011). Although, in contrast to the present work, some of the above studies mentioned previously did observe significant changes in certain benthic characteristics immediately after harvest, such as sediment composition in the harvest plots/holes or in infaunal community structure, these were in general short-lived (*i.e.* disappeared within several months; Goodwin 1978, Price 2011, DFO 2012b) or did not extend very far outside the area of harvest (< 10 m, DFO 2012b). This is probably because geoduck harvesting has the potential to displace and yet preserve benthic fauna so that they can recolonize the disturbed areas immediately after the harvest (Price 2011) and because small disturbed patches can be recolonized more quickly by movement of fauna across sediments due to their higher edge/surface area ratios (Guerra-García et al. 2003).

Table 3 summarizes geoduck harvest (using water jets) intensities in various subtidal and intertidal studies in WA and BC. Despite these studies varying in harvest intensities (*e.g.* harvest plot size, harvest duration, and number of harvest holes per unit area) and likely in site-specific conditions (*e.g.* depth, tidal current, sediment composition, infaunal community structure, and productivity), the collective results suggest that geoduck harvesting has very limited impact on the benthic environment, any significant harvest effect being generally short-lived or near-field, as discussed above. This contrasts to such commercial shellfish harvest activity as suction-dredging cockles, where a large area could be disturbed intensively within a relatively short period of time (*i.e.* a trench of 0.5–1.15 m wide and up to 8 km long per hour per boat), causing long-lasting negative effects, up to 8 years, in sediment composition and bivalve stock in the fished area (Piersma et al. 2001). The published literature has indicated that recovery of benthic environments after various forms of shellfish harvest activities can often take days to months (Hall et al. 1990, Currie & Parry 1996, Kaiser et al. 1996, Ferns at al. 2000, Tuck et al. 2000, Kaiser et al. 2001, Constantino et al. 2009), although in extreme cases it can take years (Piersma et al. 2001). Given the lengthy grow-out period (7–10 years) of *P. generosa*, repeat harvest of any given geoduck culture bed would only occur after a minimum of perhaps 7 years, reducing the likelihood of compounded effects due to repeated harvesting of the same area.

Nearby area

Outside the harvest plot, no significant benthic changes were detected in the down-current nearby areas over the study period for either study site. Geoduck harvest by water jets places sediments into suspension and this may result in effects encountered within slightly broader areas than the region of direct disturbance (ENVIRON 2009). Short and Walton (1992) found that suspended solids generated in the water column by a subtidal harvest were the highest near the harvest diver. Depending on the current speed (0.05–1.00 m s^{-1}), small quantities of suspended materials may be deposited down-current, up to 100–200 m, but most materials settle within 1 m of the harvest hole (Short & Walton 1992). Intertidal harvests cause overland flow by water used for the harvest, transporting suspended sediments over the exposed intertidal area to the water's edge (Fleece et al. 2004). In both scenarios, it is the fines (< 63 µm) that are the most relevant to transportation by water current and redeposition away from the source substrate, as they settle much more slowly and remain in the water column for longer periods (Short & Walton 1992, Palazzi et al 2001). Therefore, knowledge of sediment composition and sedimentation rate during the harvest is important to understand the potential impact of geoduck harvest on down-current nearby areas outside the immediate harvest bed.

Based on a simulation model using a fine content of 8% in the sediments, Short and Walton (1992) predicted that deposition of all suspended materials by a subtidal harvest would be 0.4 cm thick (including all grain sizes) in the affected down-current area, if 2,500 holes were made per ¼ acre bed [*i.e.* 2.5 holes m^{-2}, typical of high-density geoduck fisheries beds in WA (Palazzi et al. 2001)]. They concluded that the transport and fate of suspended sediment associated with such fisheries harvest would have minimal impacts on the physical environment in the harvest and adjacent areas. Palazzi et al. (2001) estimated a layer of 0.2 cm sediment for just the fines if 10,000 holes were dug per acre (with a fine content of 3.5%) and if all the fines were settled within that acre, suggesting that the actual sediment thickness of just the fines would be much smaller down current from the harvest area.

In the subtidal site (CI) in the present study, the fines accounted for only < 0.3% of the sediments. Such a low fine content, usually associated with a high-energy environment, is not uncommon in commercial geoduck fisheries beds in BC (and likely future geoduck aquaculture tenures). Under such conditions, little fine material would be available for suspension and subsequent redeposition due to harvesting. This is supported by the sedimentation data compiled with the sediment traps in the down-current nearby area of CI. Sediments collected during the harvest at 5–75 m (except for one large replicate value at 5 m) were 0.04–0.09 g trap^{-1} d^{-1}, representing a layer of 0.001–0.002 cm thick over the whole nearby area during the 2-d harvest [estimated using a sediment density of 1.84 g cm^{-3} (Short & Walton 1992)]. Even if the present harvest intensity were increased by 10 times to 2.6 holes m^{-2} within the 6,000 m^{2} harvest plot, the accumulation of sediments at the various distances would be 0.01–0.02 cm thick (note that this estimation does not take into effect out natural sedimentation), well below the estimations of Short & Walton (1992) and Palazzi et al. (2001). Furthermore, sediment amounts collected during the harvest at CI were similar to those during a calm sea (0.02 g trap^{-1} d^{-1}), but much lower than those during a rough sea (just before the harvest) and during the winter storm at this study site (0.22–0.69 g and 0.36 g trap^{-1} d^{-1}, respectively). In the intertidal study site (NB), the fines accounted for 2.5–7.5% of the sediments (Fig. 4). The amount of sediments collected during the harvest at 1–75 m (except for one large replicate value at 5 m) was 0.09–0.30 g trap^{-1} d^{-1} or a layer of 0.002–0.007 cm thick over the 1-tidal cycle harvest (estimated as above). The annual sedimentation rates at NB varied in the range from 0.10 to 9.04 g trap^{-1} d^{-1}, including those during windy conditions (just before the harvest), and could be much higher than rates during harvest.

It can be concluded that sediments deposited in the down-current nearby areas during harvest for both study sites CI and NB are more likely the result of natural sedimentation than the harvest process itself. In other words, the harvest did not cause any significant overall material changes down-current on top of the natural background sedimentation. It is, therefore, not surprising that the present research did not find any significant benthic changes in the down-current nearby areas at either study site. Furthermore, as commercial geoduck harvest is unlikely to occur in contaminated areas, there is little risk that water quality will be significantly deteriorated by the release of contaminants or pollutants from the harvest.

The present study did not examine the issue of overland flow, caused by water used for intertidal harvest, carrying suspended sediments into the water column. Fleece et al. (2004) and ENVIRON (2009) found that the increased turbidity from intertidal harvesting was limited to the near-shore area (< 25 ft from shoreline), peaked at 100±50 ft downstream of the harvest site, and declined rapidly within a short distance. The total distance that a turbid plume may travel is dependent on a number of factors including the proximity of the water's edge to the harvest site, strength and direction of near-shore currents, sediment characteristics on the culture beach, and local weather during the harvest. Natural turbidity generated along the shoreline during windy days is generally not discernible from that created via a harvest and turbidity generated from a harvest is only visible on calm days (ENVIRON 2009). It seems probable therefore that any effect of overland flow into the nearby water column by intertidal harvest would be confined to a relatively limited area close to the harvest site, would not exceed that generated by natural force, and would dissipate quickly as the tide comes in. It should be noted that this limited area potentially affected by the overland flow during harvest is not the same as the down-current nearby area as targeted by the present study. The latter was subject to the redeposition of sediments from the harvest plot after the harvest was done and the tide came in. The harvest would generate more materials available for subsequent redeposition from the harvest plot.

Eelgrass bed

In Canada, eelgrass beds or meadows are considered as sensitive aquatic vegetation (critical as fish habitat) and protected from harmful alteration, disruption, and destruction, unless authorized under Section 35 of the federal Fisheries Act. This actually precludes the possibility that future geoduck aquaculture (and present/future wild fisheries) will be permitted within any eelgrass bed in BC. Clam digging within eelgrass beds has been reported to significantly reduce plant shoot density and total biomass, particularly when the harvest effort is higher (Cabaço et al. 2005, Ruesink & Rowell 2012). Although geoduck harvesting within eelgrass beds in BC is prohibited, and direct disturbance due to harvests within beds is unlikely to occur, there may be certain indirect effects resulting from the deposition of materials from turbid plumes and increased turbidity due to the harvest as discussed in the Introduction. However, no significant benthic changes in the eelgrass beds and no significant eelgrass parameter alterations were detected over time for either study site. Although results of the present study might be site specific, some generality can probably be made for potential culture sites of similar site layouts regarding possible effects of geoduck harvests on eelgrass beds, as discussed below.

The depth limit of eelgrass distribution is largely regulated by light availability underwater (Duarte 1991). This suggests that a local eelgrass bed may not extend below a certain depth contour. For example, eelgrass surveys in Puget Sound, WA have shown that eelgrass rarely occurs deeper than the -5.5 m mean lower low water contour (Palazzi et al. 2001). Similarly, in the present study, the lower boundary of the eelgrass bed at the subtidal CI site occurred along the depth contour of approximately 3.5 m below chart datum. Presently, harvesters in the geoduck wild fishery in BC are not allowed to fish shallower than 3.0 m below chart datum, placing them deeper than most eelgrass beds (DFO 2012a). Accordingly, it is very likely that future subtidal geoduck culture in BC will only be permitted on seabeds deeper than where eelgrass beds exist. Since the near-shore major current direction typically parallels the shoreline or depth contour (*e.g.* Fig. 1), it is expected that deposition of materials from turbid plumes and increased turbidity from the harvest would be minimal in the shallower eelgrass beds which would not be subject to the direct down-current influence from the harvest. Findings from the present study at CI are consistent with this notion as sediment amounts collected in the eelgrass bed through the harvest were comparable to those during a calm sea, but much lower than those during a rough sea (just before the harvest) and winter storm at this site. The DFO Integrated Fisheries Management Plan, Geoduck and Horse Clam (DFO 2012a) states that it is believed that (subtidal harvest) activities are unlikely to negatively impact eelgrass beds if they occur at least 10 m away from the edge of the bed. This is likely the case.

For the intertidal study site at NB, the shoreward eelgrass bed paralleled the major current direction. Despite the seaward eelgrass bed having been located in the minor down-current direction, materials available for redeposition from the harvest would first have been carried in the opposite direction to the nearby area as the tide came in, leaving less materials available for subsequent redeposition on the seaward eelgrass bed during ebbing. In both cases, redeposition of materials from the harvest on the eelgrass beds would be expected to be low. Indeed, amounts of sediments collected in both shoreward and seaward eelgrass beds were much lower during the harvest than during windy conditions (just before the harvest), except for a few large replicate value(s) at 1 m (shoreward) and 5 m (seaward). Therefore, as with the down-current nearby areas, the low levels of sediments caused by the harvest on the eelgrass beds would be inconsequential at both study sites when compared to natural variations. This is consistent with our research findings that no significant changes were found in grain size, infaunal community, or eelgrass parameters in the eelgrass beds at either study site.

It is worth noting some observations made during the present study regarding the extent of natural variability. Eelgrass (*Z. marina*) shoots in the shoreward bed at NB were burnt out when exposed to air at mid-day low tides during a summer heat wave in Jul 2010. Two weeks later, however, the burnt shoots were replaced by new ones. The eelgrass bed looked normal as if the event had never occurred. Boese (2002) found that *Z. marina* recovered in two weeks after large numbers of the shoots and some rhizomes were removed by recreational clam raking in Yaquina Bay, Oregon USA. These results show that *Z. marina* can recover rapidly during summer growing seasons. Ruesink & Rowell (2012) reported a longer recovery time of 2 years for *Z. marina* in 1-m^2 treatment plots where all shoots and rhizomes were previously removed, but they did mention that the recovery was notably faster at the plot edge. As noted earlier, the harvest in the present study at NB included a small corner of the eelgrass bed (see Fig. 1). Although the potential harvest effect was not examined, the harvest apparently did not uproot all eelgrass shoots in this disturbed area and no visible difference was apparent between this small harvested eelgrass area and the adjacent non-harvested eelgrass bed at later samplings. Nor did we see clear quadrat patches from previous samplings, where all above-ground eelgrass shoots were severed (leaving rhizomes). These observations suggest that remnant eelgrass shoots and rhizomes are critical for fast recovery after disturbance.

At the +18 sampling at NB, there was a recent land-water runoff which swept away fine sediments at 50 and 75 m, depositing them at other distances in the nearby area (*i.e.* about 2 cm thick sediments, W. Liu, personal observations). Accordingly, the number of species and number of individuals of infauna were greatly reduced at this sampling time (Fig. 7), likely due to the flush of fresh water and/or sediment burial. At CI, the winter-storm sampling nevertheless did not reveal any significant difference before the storm versus after the storm in sediment grain size, eelgrass parameters, and infaunal community structure (unpublished data), other than the significantly higher rate of sedimentation observed during the storm event. Commercial geoduck harvest is unlikely to cause such magnitudes of impact on the benthic environments in nearby areas and eelgrass beds, which are not disturbed directly. Thus, in the context of natural variability and based on results of the present study (and others), it can be concluded that commercial geoduck harvesting does not appear to cause significant negative impacts to the benthic environment beyond the borders of the immediate harvest area, including nearby eelgrass beds. It must be noted, however, that changes in habitat, size of the culture plot, frequency of culture, and seasonal timing of out-planting and harvest may alter the degree of impact on, and rate of recovery of, the marine environment.

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Table 1 Sampling and harvest schedules at Cortes Island and Nanoose Bay.

Cortes Island		Nanoose Bay	
Date	Time point	Date	Time point
Oct 9–10, 2008	-12	Oct 16, 2008	-0
Feb 12–13, 2009	-8	Oct 18, 2008	Harvest
Jul 6–7, 2009	-3	Oct 20, 2008	+0
Oct 2–3, 2009	-0	Jan 7–8, 2009	+3
Oct 4–5, 2009	Harvest	Mar 31–Apr 1, 2009	+6
Oct 6–7, 2009	+0	Nov 3, 2009	+13
Feb 7–8, 2010	+4	Apr 29–30, 2010	+18
May 4–5, 2010	+7	Oct 10, 2010	+24
Oct 5/27, 2010	+12		

-: months before harvest; +: months after harvest; -0: immediately before harvest; +0: immediately after harvest.

Table 2 (a) Top five species observed by number of individuals (in descending order) in each of the three most presented infaunal groups in the harvest plot and nearby area at Cortes Island. Harvest was done on Oct 4–5, 2009.

	Oct 10, 2008	Feb 13, 2009	Jul 7, 2009	Oct 2, 2009	Oct 7, 2009	Feb 8, 2010	May 5, 2010	Oct 5, 2010
	Annelids							
1	*Nereis procera*	*Nereis procera*	*Axiothella sp.*	*Euclymene sp.*	*Nephtys caeca*	*Nereis procera*	*Praxillella sp.*	*Owenia collaris*
2	*Armandia brevis*	*Syllidae Indet.*	*Leitoscoloplos pugettensis*	*Nereis procera*	*Euclymeninae Indet.*	*Nephtys caeca*	*Podarkeopsis glabrus*	*Nereis procera*
3	*Ophelia limacina*	*Glycera sp.*	*Phyllodoce groenlandica*	*Nephtys caeca*	*Nereis procera*	*Syllidae Indet.*	*Capitella capitata Cmplx*	*Praxillella sp.*
4	*Prionospio steenstrupi*	*Nephtys caecoides*	*Nereis procera*	*Leitoscoloplos pugettensis*	*Leitoscoloplos pugettensis*	*Euclymeninae Indet.*	*Euclymene sp.*	*Pectinaria californiensis*
5	*Platynereis bicanaliculata*	*Scoloplos nr. acmeceps*	*Pectinaria californiensis*	*Ophelia limacina*	*Ophelia limacina*	*Euclymene sp.*	*Mediomastus sp. Cmplx.*	*Prionospio (Minuspio) lighti*
	Arthropods							
1	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Photis brevipes*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*
2	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*	*Euphilomedes carcharodonta*	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*	*Leptochelia savignyi*	*Photis brevipes*	*Photis brevipes*
3	*Leptochelia savignyi*	*Leptochelia savignyi*	*Leptochelia savignyi*	*Monocorophium acherusicum*	*Leptochelia savignyi*	*Aoroides sp.*	*Protomedeia sp.*	*Protomedeia sp.*
4	*Photis brevipes*	*Photis brevipes*	*Aoroides sp.*	*Leptochelia savignyi*	*Photis brevipes*	*Protomedeia sp.*	*Aoroides sp.*	*Americhelidium shoemakeri*
5	*Caprellidae Indet.*	*Caprellidae Indet.*	*Americhelidium shoemakeri*	*Photis sp.*	*Aoroides sp.*	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*	*Leptochelia savignyi*
	Mollusks							
1	*Tellina modesta*	*Tellina modesta*	*Nutricola lordi*	*Tellina modesta*	*Tellina modesta*	*Tellina modesta*	*Tellina modesta*	*Tellina modesta*
2	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Nutricola lordi*	*Nutricola lordi*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*
3	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*	*Clinocardium nuttallii*	*Rochefortia tumida*	*Rochefortia tumida*	*Parvilucina tenuisculpta*	*Nutricola lordi*	*Nutricola lordi*
4	*Nutricola lordi*	*Nutricola lordi*	*Tellina modesta*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*	*Olivella baetica*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*
5	*Olivella baetica*	*Turbonilla sp.*	*Parvilucina tenuisculpta*	*Olivella baetica*	*Olivella baetica*	*Nutricola lordi*	*Olivella baetica*	*Olivella baetica*

Table 2 (b) Top five species observed by number of individuals (in descending order) in each of the three most presented infaunal groups in the harvest plot and nearby area at Nanoose Bay. Harvest was done on Oct 18, 2008.

	Oct 16, 2008	Oct 20, 2008	Jan 8, 2009	Mar 31, 2009	Nov 3, 2009	Apr 30, 2010	Oct 10, 2010
	Annelids						
1	*Armandia brevis*	*Armandia brevis*	*Glycera nana*	*Pygospio elegans*	*Armandia brevis*	*Notomastus lineatus*	*Notomastus lineatus*
2	*Notomastus lineatus*	*Notomastus lineatus*	*Armandia brevis*	*Notomastus tenuis*	*Notomastus lineatus*	*Rhynchospio glutea*	*Armandia brevis*
3	*Glycinde armigera*	*Glycinde armigera*	*Pygospio elegans*	*Spiophanes berkeleyorum*	*Pygospio sp.*	*Nephtys caeca*	*Glycinde armigera*
4	*Nereis procera*	*Pseudopolydora kempi*	*Glycinde armigera*	*Armandia brevis*	*Platynereis bicanaliculata*	*Glycinde armigera*	*Nephtys caeca*
5	*Spiophanes berkeleyorum*	*Nereis procera*	*Platynereis bicanaliculata*	*Nereis procera*	*Glycinde armigera*	*Nereis procera*	*Nephtys ferruginea*
	Arthropods						
1	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Cumella vulgaris*	*Monocorophium acherusicum*	*Cumella vulgaris*	*Monocorophium acherusicum*
2	*Cumella vulgaris*	*Cumella vulgaris*	*Cumella vulgaris*	*Monocorophium acherusicum*	*Cumella vulgaris*	*Anisogammarus pugettensis*	*Hemigrapsus nudus*
3	*Hemigrapsus nudus*	*Hemigrapsus nudus*	*Ampithoe lacertosa*	*Leptochelia savignyi*	*Hemigrapsus nudus*	*Monocorophium acherusicum*	*Cumella vulgaris*
4	*Ischyrocerus anguipes*	*Ampithoe lacertosa*	*Americhelidium shoemakeri*	*Harpacticoida*	*Photis brevipes*	*Hemigrapsus nudus*	*Ampithoe lacertosa*
5	*Aoroides sp.*	*Ischyrocerus anguipes*	*Anisogammarus pugettensis*	*Hemigrapsus nudus*	*Ampithoe lacertosa*	*Protomedeia sp.*	*Crangon nigricauda*
	Mollusks						
1	*Macoma nasuta*	*Macoma nasuta*	*Macoma nasuta*	*Macoma nasuta*	*Macoma nasuta*	*Rochefortia tumida*	*Macoma nasuta*
2	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Macoma nasuta*	*Rochefortia tumida*
3	*Protothaca staminea*	*Protothaca staminea*	*Protothaca staminea*	*Protothaca staminea*	*Protothaca staminea*	*Protothaca staminea*	*Protothaca staminea*
4	*Tellina modesta*	*Odostomia sp.*	*Venerupis philippinarum*	*Venerupis philippinarum*	*Venerupis philippinarum*	*Venerupis philippinarum*	*Macoma spp.*
5	*Venerupis philippinarum*	*Venerupis philippinarum*	*Macoma spp.*	*Macoma spp.*	*Nassarius mendicus*	*Parvilucina tenuisculpta*	*Venerupis philippinarum*

Table 2 (c) Top five species observed by number of individuals (in descending order) in each of the three most presented infaunal groups in the eelgrass bed at Cortes Island. Harvest was done on Oct 4–5, 2009.

	Oct 9, 2008	Feb 12, 2009	Jul 6, 2009	Oct 3, 2009	Oct 6, 2009	Feb 7, 2010	May 4, 2010	Oct 27, 2010
	Annelids							
1	*Nereis procera*	*Nereis procera*	*Prionospio steenstrupi*	*Nereis procera*	*Nephtys caeca*	*Nephtys caeca*	*Podarkeopsis glabrus*	*Owenia collaris*
2	*Nephtys caecoides*	*Nephtys caecoides*	*Axiothella sp.*	*Leitoscoloplos pugettensis*	*Nereis procera*	*Nereis procera*	*Mediomastus sp. Cmplx.*	*Nereis procera*
3	*Prionospio steenstrupi*	*Aphelochaeta sp.*	*Owenia collaris*	*Prionospio steenstrupi*	*Euclymeninae Indet.*	*Mediomastus sp. Cmplx.*	*Nephtys caeca*	*Prionospio (Minuspio) lighti*
4	*Scoloplos nr. acmeceps*	*Cirratulidae Indet.*	*Pectinaria californiensis*	*Axiothella sp.*	*Prionospio steenstrupi*	*Syllidae Indet.*	*Pholoe glabra*	*Praxillella sp.*
5	*Armandia brevis*	*Mediomastus sp. Cmplx.*	*Leitoscoloplos pugettensis*	*Nephtys caeca*	*Leitoscoloplos pugettensis*	*Leitoscoloplos pugettensis*	*Prionospio (Minuspio) lighti*	*Leitoscoloplos pugettensis*
	Arthropods							
1	*Leptochelia savignyi*	*Americhelidium shoemakeri*	*Photis brevipes*	*Leptochelia savignyi*	*Leptochelia savignyi*	*Leptochelia savignyi*	*Leptochelia savignyi*	*Leptochelia savignyi*
2	*Americhelidium shoemakeri*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Photis brevipes*	*Photis brevipes*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*
3	*Euphilomedes carcharodonta*	*Photis brevipes*	*Leptochelia savignyi*	*Euphilomedes carcharodonta*	*Euphilomedes carcharodonta*	*Photis brevipes*	*Photis brevipes*	*Photis brevipes*
4	*Photis brevipes*	*Leptochelia savignyi*	*Americhelidium shoemakeri*	*Aoroides sp.*	*Aoroides sp.*	*Protomedeia sp.*	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*
5	*Aoroides sp.*	*Aoroides sp.*	*Aoroides sp.*	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*	*Americhelidium shoemakeri*	*Protomedeia sp.*	*Protomedeia sp.*
	Mollusks							
1	*Tellina modesta*	*Tellina modesta*	*Rochefortia tumida*	*Rochefortia tumida*	*Tellina modesta*	*Tellina modesta*	*Tellina modesta*	*Tellina modesta*
2	*Rochefortia tumida*	*Rochefortia tumida*	*Tellina modesta*	*Tellina modesta*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*
3	*Nutricola lordi*	*Parvilucina tenuisculpta*	*Nutricola lordi*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*	*Parvilucina tenuisculpta*
4	*Parvilucina tenuisculpta*	*Nutricola lordi*	*Parvilucina tenuisculpta*	*Nutricola lordi*	*Nutricola lordi*	*Nutricola lordi*	*Nutricola lordi*	*Nutricola lordi*
5	*Clinocardium nuttallii*	*Clinocardium nuttallii*	*Lyonsia californica*	*Olivella baetica*	*Olivella baetica*	*Astyris gausapata*	*Gastropteron pacificum*	*Protothaca staminea*

Table 2 (d) Top five species observed by number of individuals (in descending order) in each of the three most presented infaunal groups in the eelgrass bed at Nanoose Bay. Harvest was done on Oct 18, 2008.

	Oct 16, 2008	Oct 20, 2008	Jan 7, 2009	Apr 1, 2009	Nov 3, 2009	Apr 29, 2010	Oct 10, 2010
	Annelids						
1	*Armandia brevis*	*Armandia brevis*	*Armandia brevis*	*Armandia brevis*	*Notomastus lineatus*	*Notomastus lineatus*	*Notomastus lineatus*
2	*Notomastus lineatus*	*Notomastus lineatus*	*Platynereis bicanaliculata*	*Spiophanes berkeleyorum*	*Platynereis bicanaliculata*	*Glycinde armigera*	*Owenia collaris*
3	*Spiophanes berkeleyorum*	*Platynereis bicanaliculata*	*Notomastus tenuis*	*Platynereis bicanaliculata*	*Armandia brevis*	*Owenia collaris*	*Platynereis bicanaliculata*
4	*Glycinde armigera*	*Nereis procera*	*Pygospio elegans*	*Notomastus tenuis*	*Glycinde armigera*	*Prionospio (Minuspio) lighti*	*Glycinde armigera*
5	*Alvania compacta*	*Spiophanes berkeleyorum*	*Spiophanes berkeleyorum*	*Pygospio elegans*	*Nephtys caeca*	*Rhynchospio glutea*	*Nephtys ferruginea*
	Arthropods						
1	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Cumella vulgaris*	*Hemigrapsus nudus*	*Cumella vulgaris*	*Hemigrapsus nudus*
2	*Cumella vulgaris*	*Ischyrocerus anguipes*	*Cumella vulgaris*	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Monocorophium acherusicum*	*Monocorophium acherusicum*
3	*Ischyrocerus anguipes*	*Cumella vulgaris*	*Caprellidae*	*Leptochelia savignyi*	*Heptacarpus sp.*	*Pagurus sp.*	*Pagurus sp.*
4	*Ampithoe lacertosa*	*Hemigrapsus nudus*	*Ampithoe lacertosa*	*Hemigrapsus nudus*	*Crangon nigricauda*	*Ampithoe lacertosa*	*Crangon nigricauda*
5	*Aoroides sp.*	*Leptochelia savignyi*	*Pleustidae Indet.*	*Harpacticoida*	*Telmessus cheiragonus*	*Harpacticoida*	*Hippolytidae*
	Mollusks						
1	*Macoma elimata*	*Rochefortia tumida*	*Rochefortia tumida*	*Rochefortia tumida*	*Macoma nasuta*	*Rochefortia tumida*	*Rochefortia tumida*
2	*Rochefortia tumida*	*Macoma nasuta*	*Macoma nasuta*	*Macoma spp.*	*Rochefortia tumida*	*Macoma nasuta*	*Protothaca staminea*
3	*Macoma nasuta*	*Tellina modesta*	*Alvania compacta*	*Macoma nasuta*	*Tellina modesta*	*Tellina modesta*	*Macoma nasuta*
4	*Tellina modesta*	*Protothaca staminea*	*Tellina modesta*	*Alvania compacta*	*Haminoea sp.*	*Parvilucina tenuisculpta*	*Macoma spp.*
5	*Alvania compacta*	*Odostomia sp.*	*Protothaca staminea*	*Alvania rosana*	*Alvania compacta*	*Alvania compacta*	*Tellina sp.*

Table 3 Summary of publications reporting subtidal and intertidal geoduck clam (*Panopea generosa*) harvest (by water jets) intensities in Washington state, USA and British Columbia, Canada.

Harvest plot size (m^2)	Total duration when harvest occurred (days)	Duration of actual harvest (days)	Number of harvest holes (m^{-2})	Type of harvest	Reference
90	29	5	4.3	S, F	Goodwin (1978)
30	6	–	8.4	S, F	Breen & Shields (1983)
60	1	1	Swath harvest	I, A	DFO (2012b)
2,500–4,500	2–5 (months)	–	–*	I, A	Price (2011)
6,000	2	2	0.26	S, A/F	Present study
450	1	1	9	I, A	Present study

I: intertidal plot; S: subtidal plot; F: fisheries plot; A: aquaculture plot; –: not specified in the study. *: the number of harvest holes is expected to be relatively higher on these aquaculture plots.

Note that an estimation of 2.5 holes m^{-2} is assumed for high-density commercial geoduck fisheries beds in Washington state (Palazzi et al. 2001).

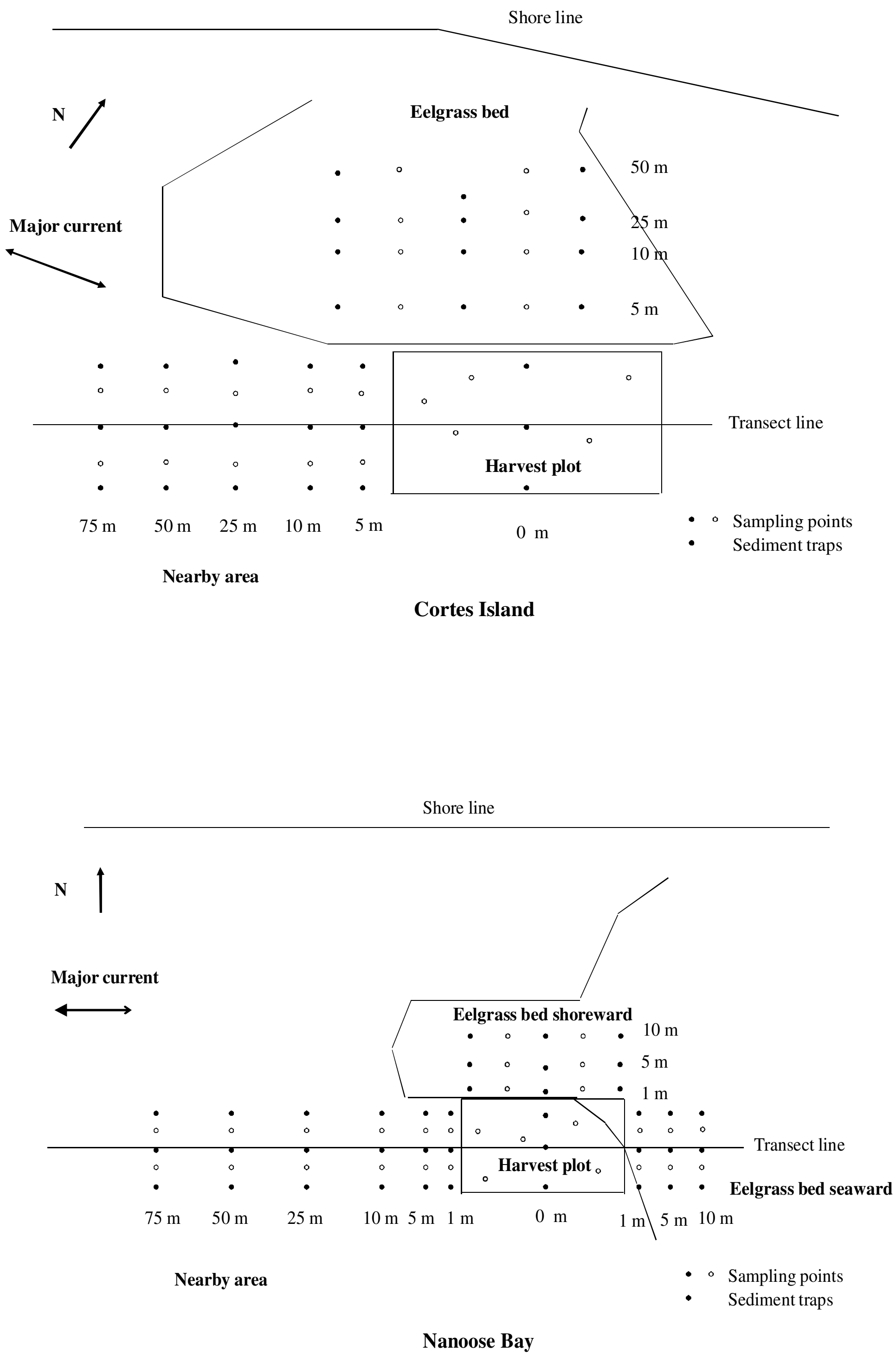

Fig. 1 Experimental layouts of subtidal study site at Cortes Island (harvest plot: 100 x 60 m) and intertidal study site at Nanoose Bay (harvest plot: 30 x 15 m). See text for details.

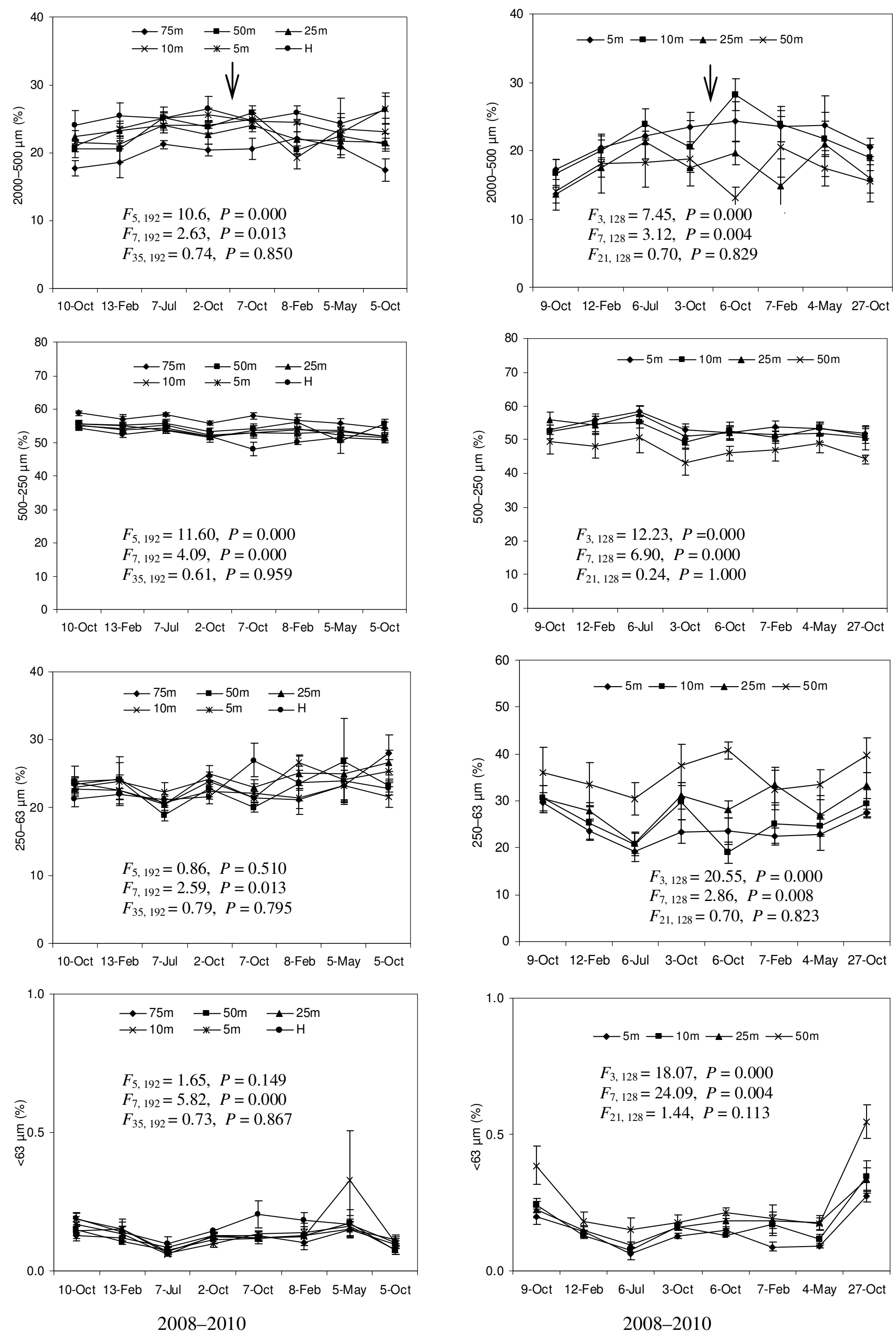

Fig. 2 Sediment grain size compositions in the harvest plot (H) and nearby area (left column) and eelgrass bed (right column) at Cortes Island. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 4–5, 2009). Error bars are SE and *n*=5.

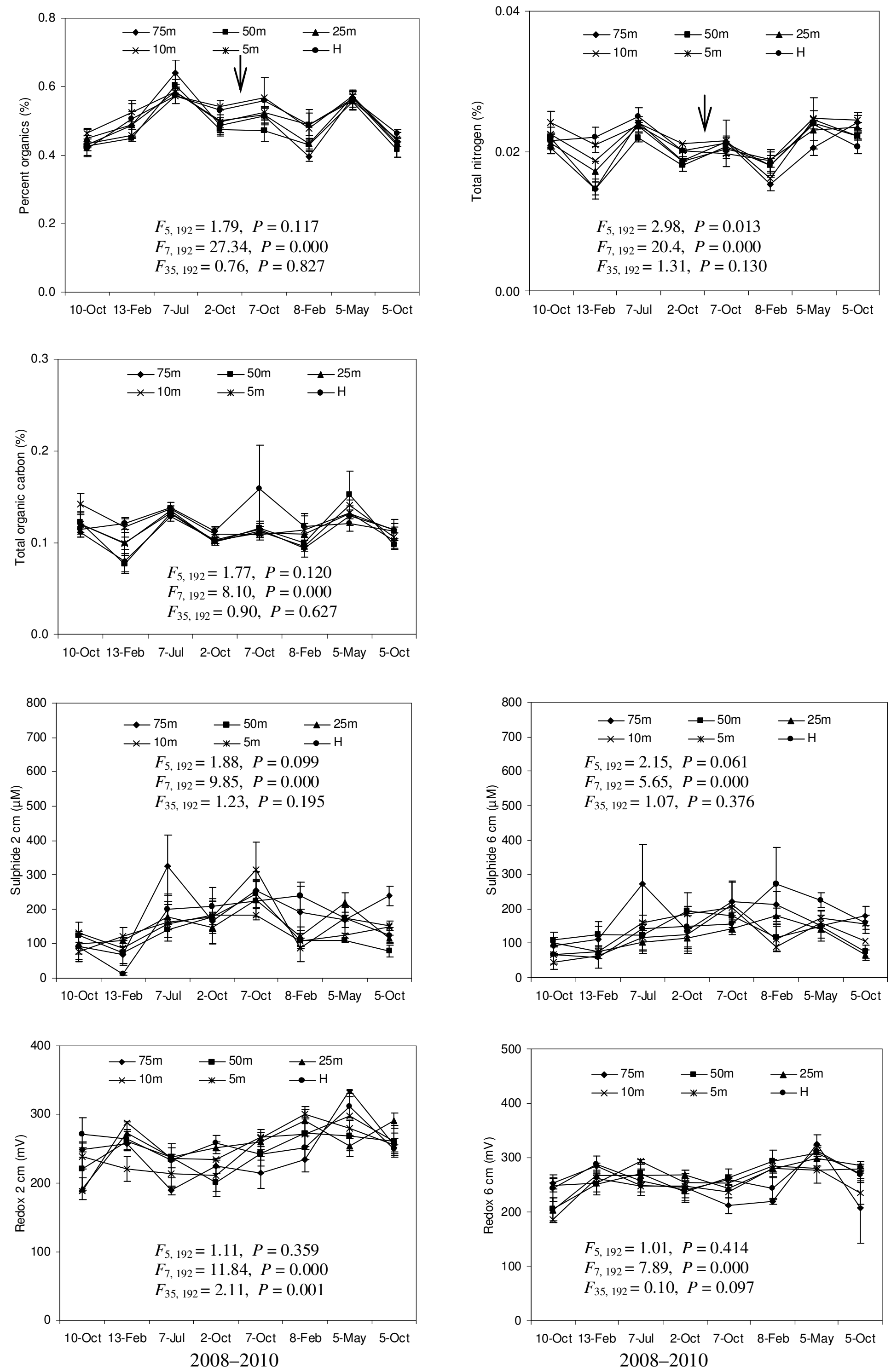

Fig. 3 Percent organics, total nitrogen, total organic carbon, sulphide content, and redox potential in the harvest plot (H) and nearby area at Cortes Island. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 4–5, 2009). Error bars are SE and *n*=5.

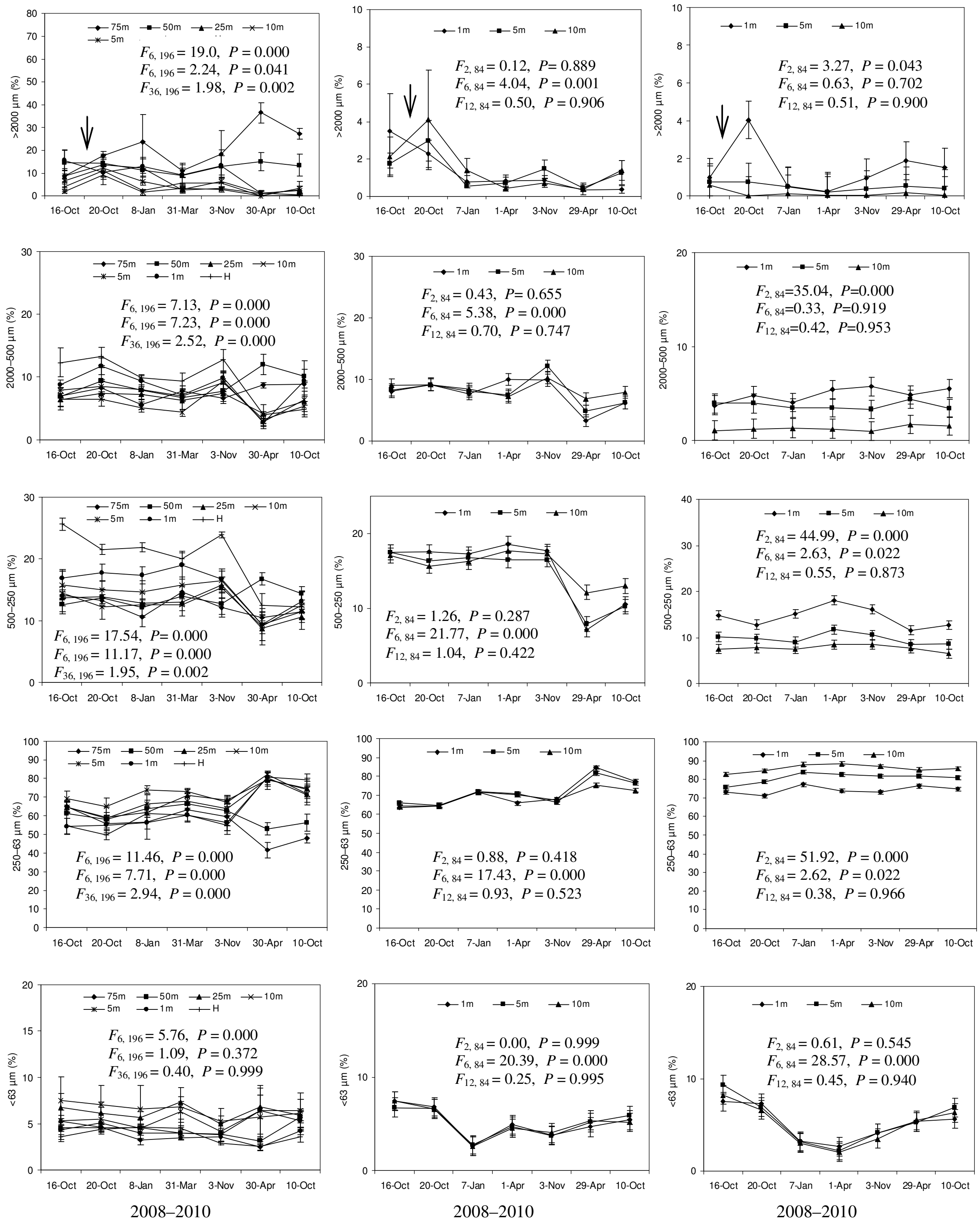

Fig. 4 Sediment grain size compositions in the harvest plot (H) and nearby area (left column) and eelgrass beds (middle column for seaward and right column for shoreward) at Nanoose Bay. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 18, 2008). Error bars are SE and *n*=5.

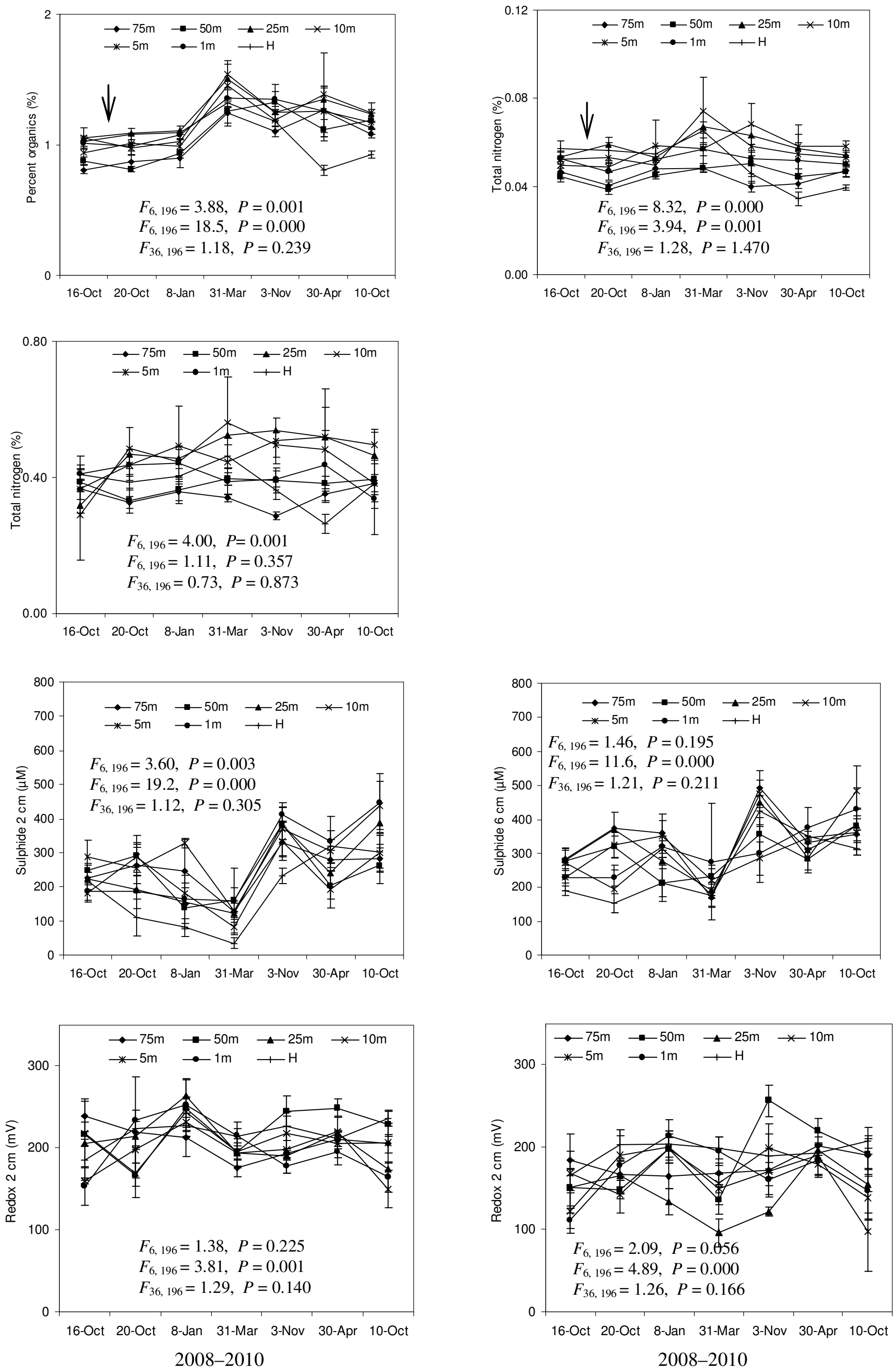

Fig. 5 Percent organics, total nitrogen, total organic carbon, sulphide content, and redox potential in the harvest plot (H) and nearby area at Nanoose Bay. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 18, 2008). Error bars are SE and *n*=5.

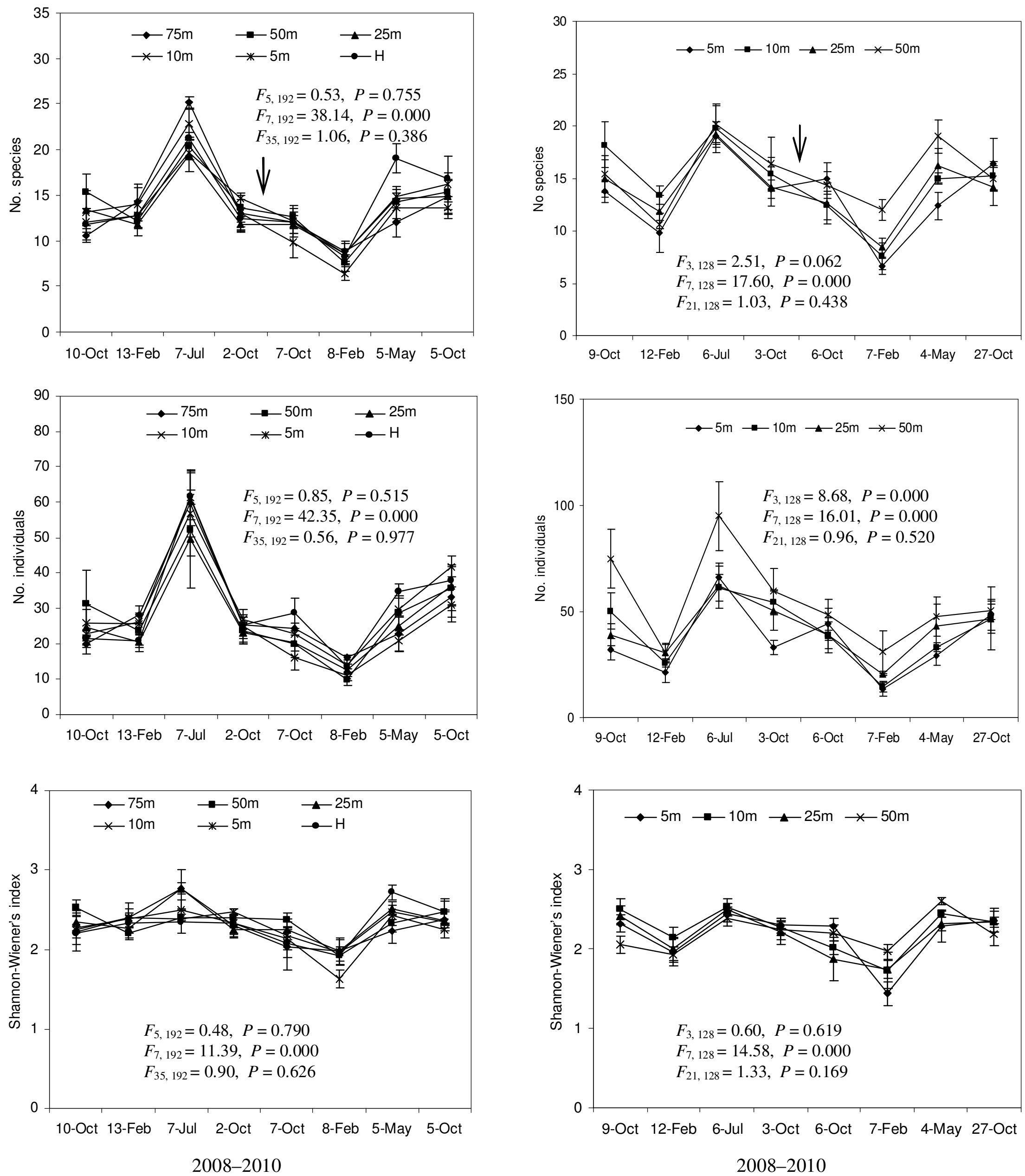

Fig. 6 Infaunal community structure in the harvest plot (H) and nearby area (left column) and eelgrass bed (right column) at Cortes Island. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 4–5, 2009). Error bars are SE and n=5.

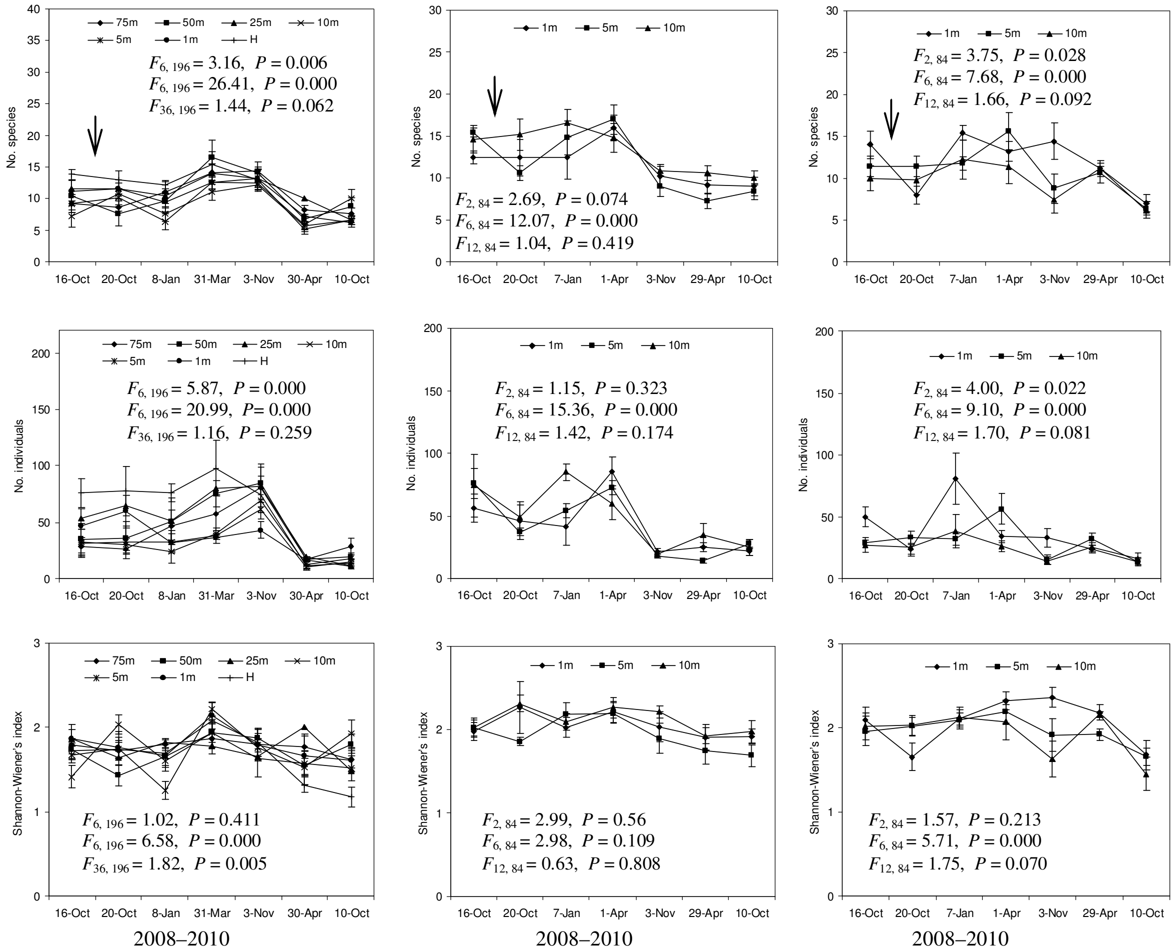

Fig. 7 Infaunal community structure in the harvest plot (H) and nearby area (left column) and eelgrass beds (middle column for seaward and right column for shoreward) at Nanoose Bay. ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 18, 2008). Error bars are SE and n=5.

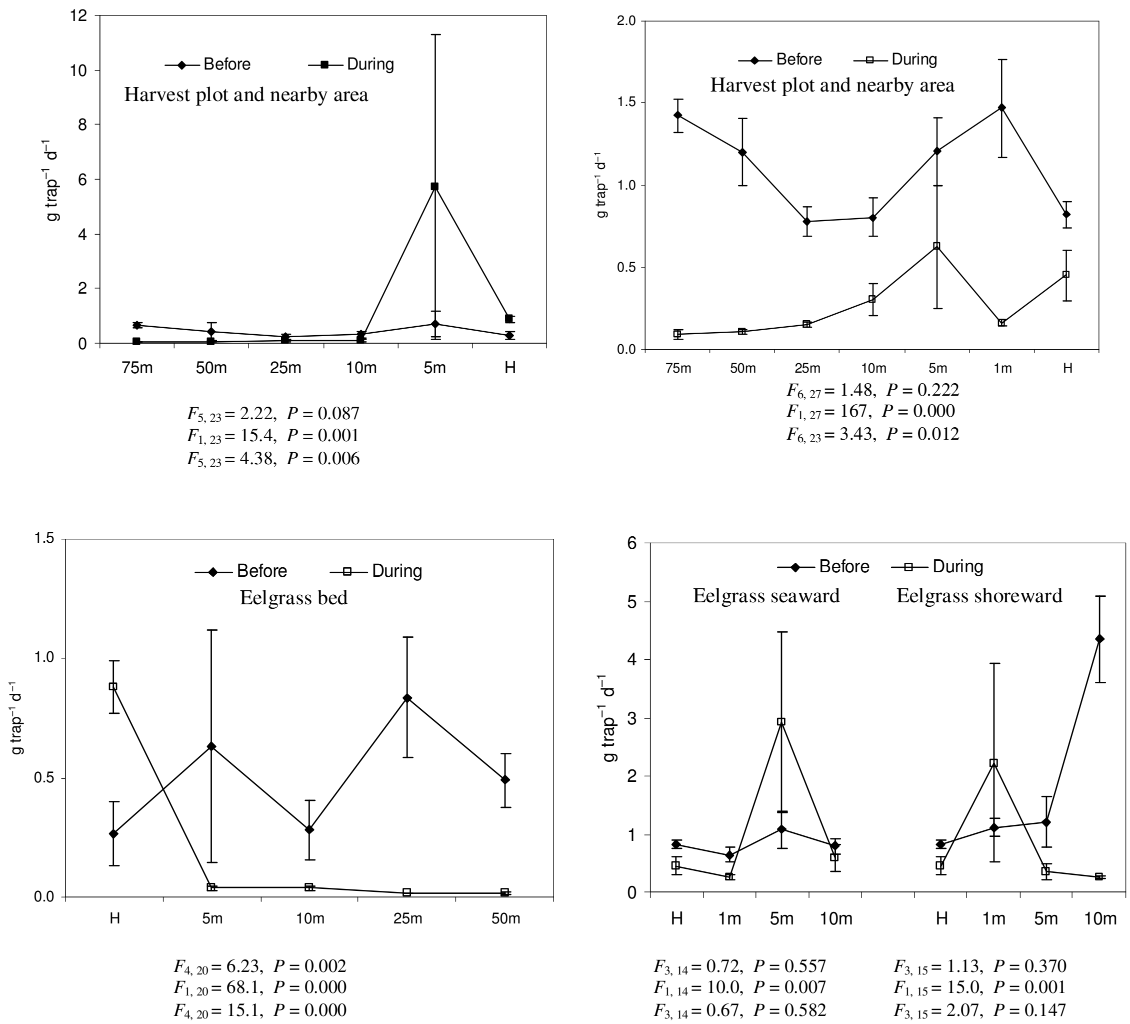

Fig. 8 Sedimentation before and during the harvest (H) for Cortes Island (left column) and Nanoose Bay (right column). ANOVA results are presented in each figure in the order of sampling distance, time (before versus during harvest), and the interaction. Error bars are SE and n=3.

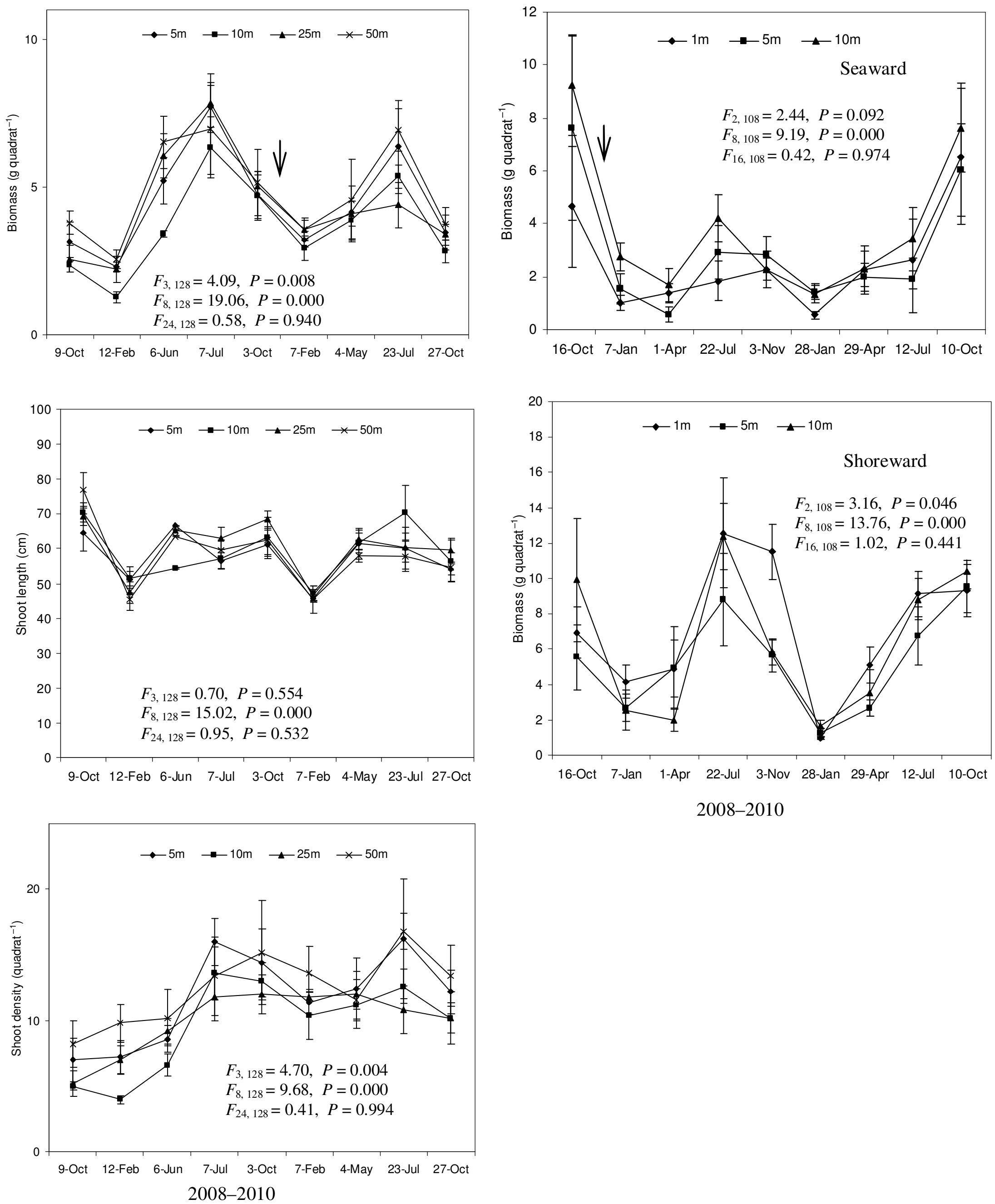

Fig. 9 Eelgrass parameters for Cortes Island (left column) and Nanoose Bay (right column). ANOVA results are presented in each figure in the order of sampling distance, time, and the interaction. Arrows indicate harvest (Oct 4–5, 2009 for Cortes Island and Oct 18, 2008 for Nanoose Bay). Error bars are SE and n=5.