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Long-term seafloor monitoring at an open ocean aquaculture site in the western Gulf of Maine, USA: Development of an adaptive protocol

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ABSTRACT

The seafloor at an open ocean finfish aquaculture facility in the western Gulf of Maine, USA was monitored from 1999 to 2008 by sampling sites inside a predicted impact area modeled by oceanographic conditions and fecal and food settling characteristics, and nearby reference sites. Univariate and multivariate analyses of benthic community measures from box core samples indicated minimal or no significant differences between impact and reference areas. These findings resulted in development of an adaptive monitoring protocol involving initial low-cost methods that required more intensive and costly efforts only when negative impacts were initially indicated. The continued growth of marine aquaculture is dependent on further development of farming methods that minimize negative environmental impacts, as well as effective monitoring protocols. Adaptive monitoring protocols, such as the one described herein, coupled with mathematical modeling approaches, have the potential to provide effective protection of the environment while minimize monitoring effort and costs.

1. Introduction

Marine aquaculture is a rapidly growing industry worldwide, with projections for continued growth and expansion (WHOI, 2007; Lucas and Southgate, 2012). Long-term success of marine aquaculture is in part dependent on further development of culture methods that minimize negative environmental impacts. Open ocean finfish cage culture is a relatively new endeavor that is being pursued in part because the potential for negative impacts are generally lessened in deeper offshore waters compared to shallower nearshore sites (Pearson and Black, 2001; Price and Morris, 2013). However, there have been relatively few published studies of open ocean aquaculture facilities, especially those reporting environmental monitoring data, which has hindered further development (Holmer, 2010).

The major effects on the seafloor from fish cage culture result from deposition of organic materials, including biodeposits from the culture organisms and excess food material. In general, studies to date on organic wastes indicate if deposition rates do not exceed the ability of benthic organisms to assimilate the organic matter, benthic communities are not detrimentally affected (Pearson and Black, 2001). The Pearson and Rosenberg (1978) conceptual model for the response of benthic infauna to organic inputs remains generally valid for mud-bottom habitats (Rhoads and Germano, 1982; Grizzle and Penniman, 1991; Nilsson and Rosenberg, 2000; Wildish et al., 2004). And models have been developed that are capable of predicting impacts to the benthos for various combinations of environmental conditions and fish biomass (Rensel et al., 2006; Kiefer et al., 2011). However, there still is a pressing need move beyond the heavy reliance on traditional and costly environmental monitoring programs commonly required by regulatory agencies to adaptive approaches that include predictive modeling.

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The University of New Hampshire (USA) initiated an experimental open ocean aquaculture (OOA) site in the western Gulf of Maine in 1993 and completed grow-out of a variety of species until the project ceased in 2010. Permitting and environmental monitoring requirements evolved over the term of the project but consistently included seafloor monitoring. Here, we describe the results of monitoring sediments and benthic macrofauna from 1999 to 2008 (the time period when similar methods were used), and how the persistent finding of 'no negative impacts' resulted in an adaptive approach to environmental monitoring that was implemented in 2008.

2. Materials and methods

2.1. Study site

The farm site was located 10 km offshore of Portsmouth, New Hampshire, USA in the southwestern Gulf of Maine in ~50 m of water (Fig. 1). The seafloor in the general vicinity is relatively heterogeneous, including bedrock outcrops, gravel, and muddy sands. However, the area most likely impacted by the aquaculture operation consisted primarily of poorly sorted, muddy sands.

Two major fish cage and mooring system deployments were conducted at the OOA site over the duration of the project. The first occurred in the summer of 1999 with the installation of two 600 m^3 SeaStation™ fish cages, each with a submerged, grid-type mooring system (design and deployment details can be found in Tsukrov et al., 2000; Fredriksson et al., 2000; Baldwin et al., 2000). With this mooring system, the cages could either be kept at the surface for better access and greater dispersal of wastes or submerged to reduce loads from winds, waves and currents. While the deployment of the two small SeaStation™ fish cages and moorings proved to be an engineering success (see Fredriksson et al., 2003), it was clear that this configuration had limited biomass capacity. Thus, in the summer of 2003, all gear was replaced with a new submerged grid mooring system capable of holding four SeaStation™ fish cages (in a 2 × 2 configuration) each with an internal volume of 3000 m^3. Though many of the design attributes were similar to the earlier cage moorings (Fredriksson et al., 2004), a much greater biomass capacity was achieved.

2.2. Monitoring history and study design

Concurrent with these changes in cage configurations there were also variations in fish species, densities and biomasses deployed to the cages over the study period (Table 1). Although the field and laboratory methods (see below) remained consistent from year to year, the number of sampling sites, their locations, and sampling frequency varied substantially over time, mainly in response to feedback from the permitting agencies as the program evolved. Sampling frequency was on an approximately monthly basis for 1999–2001, quarterly from 2002 to 2004, and semi-annually from 2005 to 2007. Finally in 2008, a single set of samples was taken and an adaptive monitoring protocol was implemented (see detailed section below). In order to minimize variability and directly compare the data from all 10 years of the study, the overall dataset was averaged to a series of annual means, one for reference (or control) sites and another for impact sites.

Determination of the spatial extent of the impact and reference areas was accomplished initially using a collection of unpublished environmental data (e.g., currents) and the application of a simple settling model (Gowen et al., 1989). This process identified a 700 m × 450 m rectangular "impact area" (Fig. 2). Though the calculations supporting this decision are not shown herein, it was evident that a limited amount of detailed current velocity information existed. The process also verified the need to establish a long-term observational platform for which more accurate currents at the site could be measured and therefore waste movement better estimated. Therefore, as part of the overall environmental monitoring program, an oceanographic buoy was installed at the site from 2001 to 2008, with a typical deployment period between 2 weeks and 3 months. Instrumentation on the platform included wave measurement sensors, temperature,

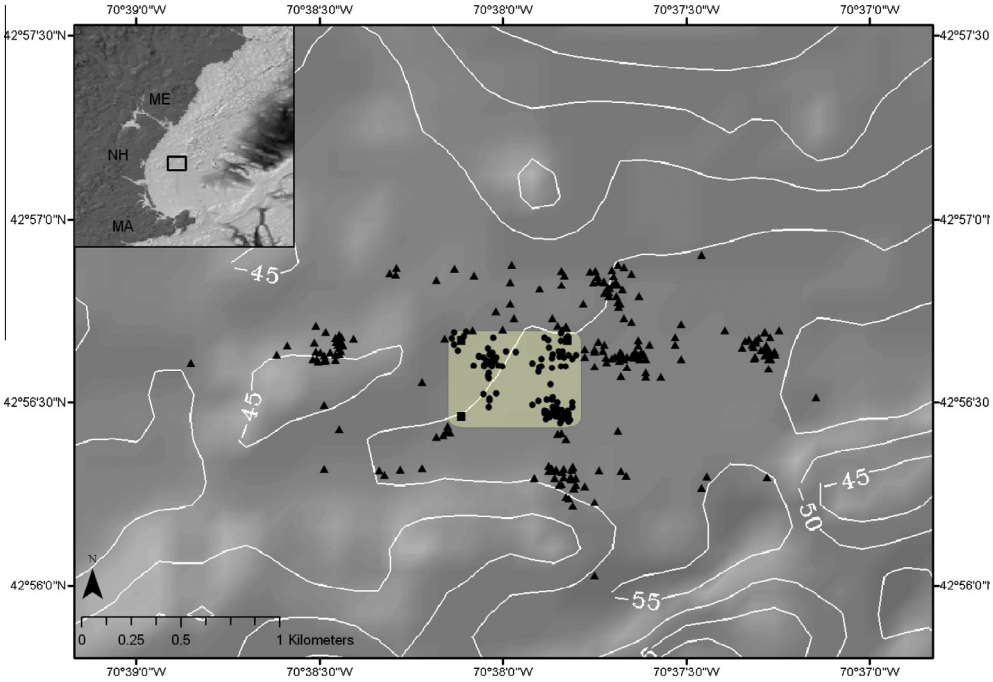

Fig. 1. Box corer sample locations in impact (light colored rectangle; sample locations shown as circular dots) and reference (sample locations shown as triangles) areas. Water depth contours in meters. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

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Table 1
Fish stocking abundance and biomass over the duration of the project.

Species	Initial stocking				Harvest			
	Date	Individual weight (g)	Number	Total biomass (kg)	Date	Individual weight (g)	Total biomass (kg)	Survival (%)
Summer flounder (*Paralichthys dentatus*)	14-05-1999	280	6000	1680	15-11-1999	360	13,132	45
Summer flounder (*Paralichthys dentatus*)	28-06-2000	400	560	224	(n/a)	(n/a)	0	0
Atlantic halibut (*Hippoglosus hippoglosus*)	15-09-2001	100	1900	190	10-06-2004	3442	1,31,305	70
Haddock (*Melanogrammus aeglefinus*)	08-12-2002	77	2900	223	25-02-2005	1087	41,749	95
Atlantic cod (*Gadus morhua*)	12-05-2003	30	27,000	810	28-11-2005	1500	58,026	55
Atlantic cod (*Gadus morhua*)	24-04-2006	30	48,000	1440	14-02-2009	2200	87,688	60

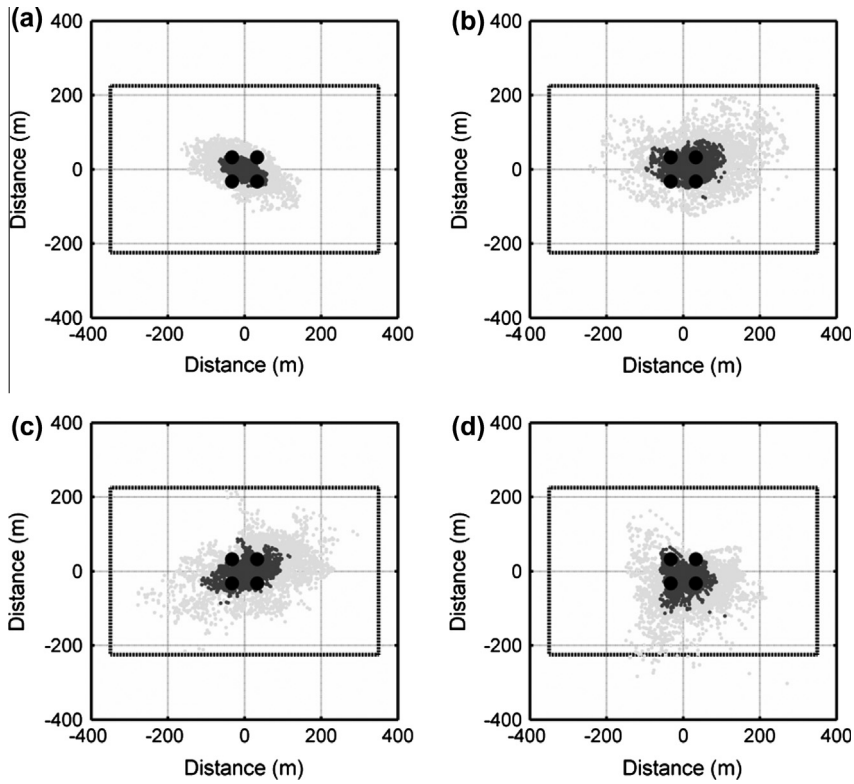

Fig. 2. Estimated deposition areas for waste feed (dark gray dots) and fecal matter (light gray dots) within the designated 450 m × 700 m impact zone (black rectangle) for each of the seasons of summer (a), fall (b), winter (c) and spring (d); see Table 2 for dates corresponding to each season. Locations of the four fish cages are denoted by black circles.

salinity, dissolved oxygen, turbidity, chlorophyll fluorescence and current profiles. The oceanographic buoy details are provided in Irish et al. (2004). The current profiles were measured with an Acoustic Doppler Current Profiler (ADCP), which was configured to sample every 15 min at 2 m bins from a depth of 13–47 m (the nominal depth at the site is 52 m). One of the major aims of these additional studies was to assess the adequacy of the originally determined impact area.

2.3. Benthic field and laboratory methods

Sampling cruises were conducted using the University of New Hampshire Research Vessel *Gulf Challenger*. Bottom samples were taken with a Wildco box corer with a designed sampling area of 0.0625 m^2 (25 × 25 cm opening). The location of each sampling site was determined by shipboard GPS. Only samples with a minimum of 5 cm sediment penetration and relatively undisturbed condition overall were accepted for further processing. The sediment inside the corer was subsampled for infauna with a 10.4 cm inside diameter (0.0085 m^2) acrylic core tube, and an aliquot of the upper 2–3 cm of the remaining sediment taken for organic content and texture analyses (see below). The contents of the 10.4 cm core tube were washed on a 0.5 mm mesh sieve in the field, fixed in 3% unbuffered formalin with rose Bengal stain for 3–5 days, then transferred to 70% isopropanol for preservation. In the laboratory, all invertebrates were removed under 3× magnification, sorted by major taxa, identified to family level in most cases based on Weis (1995) and Pollock (1998), counted, and weighed (wet weight of preserved specimens).

All sediment samples were analyzed for total organic content by loss on ignition (LOI; 450 °C for 4 h). Grain size distribution was measured on subsets of the samples. Not every sediment sample was analyzed for complete grain size as the monitoring program showed the bottom sediment changed very little over the study period. The grain size distribution was determined using standard sieve and pipette analytical techniques as described in Folk (1980). Sediment grain size statistics and classifications were computed using Gradistat (Blott and Pye, 2001).

2.4. Benthic data analysis

Benthic data were analyzed as two separate datasets: annual means assessed across all years sampled, and years 2006 and 2007 only. This was done because 2006 and 2007 were the last two years of the project with complete datasets available and would likely have represented the cumulative effect of the latter years when fish biomass was highest (Table 1). The following methods were used for both datasets.

Assessment of the univariate data (total community density, wet weight biomass, and taxonomic richness) consisted of plots of the means of each of the three dependent variables over time to inspect for possible long-term trends and differences between the impact and reference sites. Statistical analyses were performed using JMP (SAS Institute Inc. 2006, 6.0.2). A two-factor analysis of variance (ANOVA, $\alpha < 0.05$) was used to examine the effects of the aquaculture activities (impact vs. control samples), year, and their interaction on benthic community measures, which were considered fixed factors. The relationship between benthic community measures and sediment characteristics were examined using linear regression. For all datasets, residuals were examined and data were transformed (log), if needed, to produce homogenous, normally distributed error terms.

To evaluate multivariate community characteristics, the density and biomass of all benthic taxa were each compared using non-metric multi-dimensional scaling (MDS), analysis of similarity (ANOSIM), and contributions to similarity analysis (SIMPER) using PRIMER statistical software (Clarke and Gorley, 2001). Data were entered into a Bray-Curtis similarity matrix for all PRIMER analyses. Stress values, shown on the MDS plots, indicate how well the Bray-Curtis similarity matrix matches up with the dimensional relationships among samples in the ordination. A one-factor ANOSIM was conducted to test for significant differences in benthic communities between reference and impact sites. The output of the ANOSIM test includes a global R statistic in addition to a p value. Values for the global R range from -1 to 1, where a value of 0 suggests random grouping and a value of 1 suggests that groups differ in community composition. A SIMPER (similarity percentages) routine was calculated to determine the species contributing most to any differences detected between groups of samples. Biomass and density data were averaged among sites and entered into an average linkage hierarchical cluster analysis to further evaluate similarity in benthic communities between reference and impact sites.

3. Results

3.1. Determination of theoretical "impact" area under the cages

Fully understanding the current profile structure at the OOA site in the western Gulf of Maine was nontrivial due to large seasonal variations. Therefore, it was decided to examine only depth-averaged currents, but for each of the summer, fall, winter and spring seasons. The depth-averaged currents were then used to investigate waste feed and fecal matter settling distances for a range of low to high-energy weather conditions. Though the oceanographic buoy was deployed for several years, it was found that in 2003 and 2006, data sets were collected that represented the four seasons (Table 2). The four current velocity data sets were depth-averaged and separated into East–West (x-direction) and North–South (y-direction) components. With the depth-averaged components the current speed was obtained and the maximum, minimum, average and standard deviation statistics calculated (Table 2). The basic statistics showed that the current velocities tended to be smaller during the spring and summer months and more energetic during the fall and winter months.

With a more detailed representation of the seasonal currents at the OOA site, the characteristics of the deposition area was re-examined using the same Gowen et al. (1989) approach (initially used in this study). In this case, each of the velocity components were used in the simple settling model to estimate the corresponding distance components (d) that the waste feed and fecal matter voided would travel:

$$d_{(x \text{ or } y)} = \frac{h \cdot u_{(x \text{ or } y)}}{V_{(1 \text{ or } 2)}}. \quad (1)$$

In Eq. (1), h is the vertical distance the waste traveled, calculated from some point in the cage down to the bottom (50 m), u is the x- and y-components of the velocity vector and V is the particle settling velocity (1 for the fecal matter and 2 for the food; see below). While the seasonal current velocity components and the depth can be measured, fecal and food settling velocities still needed to be estimated. Numerous studies have been conducted that examined the fecal settling rates of marine aquaculture finfish species including Atlantic salmon (*Salmo salar*), gilthead seabream (*Sparus aurata*), and sea bass (*Dicentrarchus labrax*) as summarized by Magill et al. (2006). Mean settling rates in these studies ranged from 0.48 to 6 cm/s. Since the fecal settling rates of the fish raised at the OOA site were not determined, an intermediate value of 4 cm/s was used in the preliminary analysis. Food settling rate was obtained from observational results described in Panchang et al. (1997). They used a value of 0.1 m/s, which compared reasonably well with previous studies conducted by Warren-Hansen (1982).

With depth, current velocity components, and settling velocities, the simple settling model described by Eq. (1) was applied to examine the deposition area for the fecal matter and waste food as if a point source was in the center of each of the four cages (Fig. 2). As expected, the estimated deposition area for the fecal waste was greater than that calculated for the waste food and smaller areas for each was found during the lower velocities of the summer months (Fig. 2a). The summer months are also the time of increased growth, feed and discharge rates which will result in higher waste concentration on the bottom. Larger deposition areas were also estimated for the fall and winter months (Fig. 2b and c). The largest current velocity was measured in the spring (Fig. 2d) and therefore the largest distance calculated. It should be noted that water column turbulence and bottom resuspension were not considered in this analysis. The results do show, however, that the original 450 m × 700 m impact area (Fig. 1) provided a reasonable estimate of the bottom area most likely to have been affected by waste deposition from the fish cages.

Table 2
Depth-averaged current speeds for four seasonal data sets; maximum, minimum, and depth-averaged current statistics were calculated from speed time-series (see text for details).

Season	Dates	Max (m/s)	Min (m/s)	Ave (m/s)	Std (m/s)
Summer	6/17/2006–7/17/2006	0.1380	0.0010	0.0453	0.0254
Fall	11/4/2003–12/4/2003	0.2319	0.0015	0.0816	0.0436
Winter	1/21/2006–2/20/2006	0.2482	0.0014	0.0814	0.0415
Spring	4/18/2006–5/18/2006	0.3248	0.0018	0.0722	0.0390

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3.2. Univariate analyses of benthic and sediment data

The means for all three univariate measures showed very similar values for the impact sites compared to the reference sites for all years, and two-factor ANOVAs showed no significant effects (all $p > 0.05$; Fig. 3). Focusing only on years 2006–2007 when fish biomass was highest, however, indicated a significant difference ($p = 0.0029$) in community taxa with the reference sites being greater than the impact both years, and significantly higher ($p = 0.0031$) biomass during 2007. Thus, for most years there wereno differences in overall community characteristics of the benthos at the impact and reference areas.

The organic content (LOI) of all samples was uniformly low, typically with a maximum value <3% for most years and most annual means <2% (Table 3). Inspection of the data indicate an apparent gradual increase in mean LOI from 1999 to 2005 at both impact and reference sites but then a decrease during 2006 and 2007 when fish biomass was highest. From 1999 to 2005, a subset of the cores taken for benthic infauna and LOI analysis was analyzed for grain size characteristics. In total, 79 samples were taken and analyzed: 26 within the impact area and 53 in reference areas. The bottom sediments within the impact area were poorly sorted, muddy very fine sands. The graphic mean grain sizes computed after Folk (1980) ranged from 3.87φ (0.07 mm) to 3.43φ (0.09 mm); all within the very fine sand range. The percent sand in the samples ranged from 76% to 88%, while the mud content was 12–24%. Only one sample contained gravel (1.2%). The reference samples had a larger range of sediment sizes than the impact sites due to the wider geographic distribution of samples and some nearby rock outcrops. However, the samples were predominantly poorly to very poorly sorted, muddy very fine sands (of the samples). Five samples were gravelly muddy sands (1–11% gravel) and one sample was a sandy mud (81% mud). The graphic mean grain sizes all of the samples but one were between 3.10φ (0.12 mm) and 4.83φ (0.04 mm). The exception had a mean grain size of 7.90φ (0.0042 mm). Of the samples between 3.10φ (0.12 mm) and 4.83φ (0.04 mm), were very fine sands (3φ to 4φ), similar to the samples from the impact sites and 6 fell within the very coarse silt range (4φ to 5φ). Overall, with very few exceptions, the grain size of the samples from the impact sites and the reference sites were very similar in both the grain size and LOI distribution with the exception of the sandy mud which had a LOI of 8.2% and was clearly an outlier.

3.3. Multivariate analyses

In the overall dataset, four polychaete families (Spionidae, Paraonidae, Oweniidae, and Thyasiridae) made up about 60% of the total abundances from all samples collected from both impact and reference areas (Table 4). The rankings for impact and reference samples differed little; there were nine shared taxa among the top ten from both areas. Other dominant taxa included several polychaete families, bivalves and crustaceans. Focusing only on the dataset for 2006–2007 when fish biomass was highest, the 2-D MDS plots suggested some differences in community composition for the impact compared to reference sites based on density, but less so for biomass (Fig. 4). Results from the ANOSIMs confirm this pattern, indicating no significant differences for density or biomass. Cluster analysis also showed some community separation between impact and reference for density, but not for biomass. Overall, these data suggest similar taxonomic composition (at the family level) between samples from the impact and reference areas, with differences in relative abundance, but not biomass. Finally, no discernable relationships were found between benthic communities and sediment characteristics; regression models explained on average 1% of the variability and a maximum of 3%.

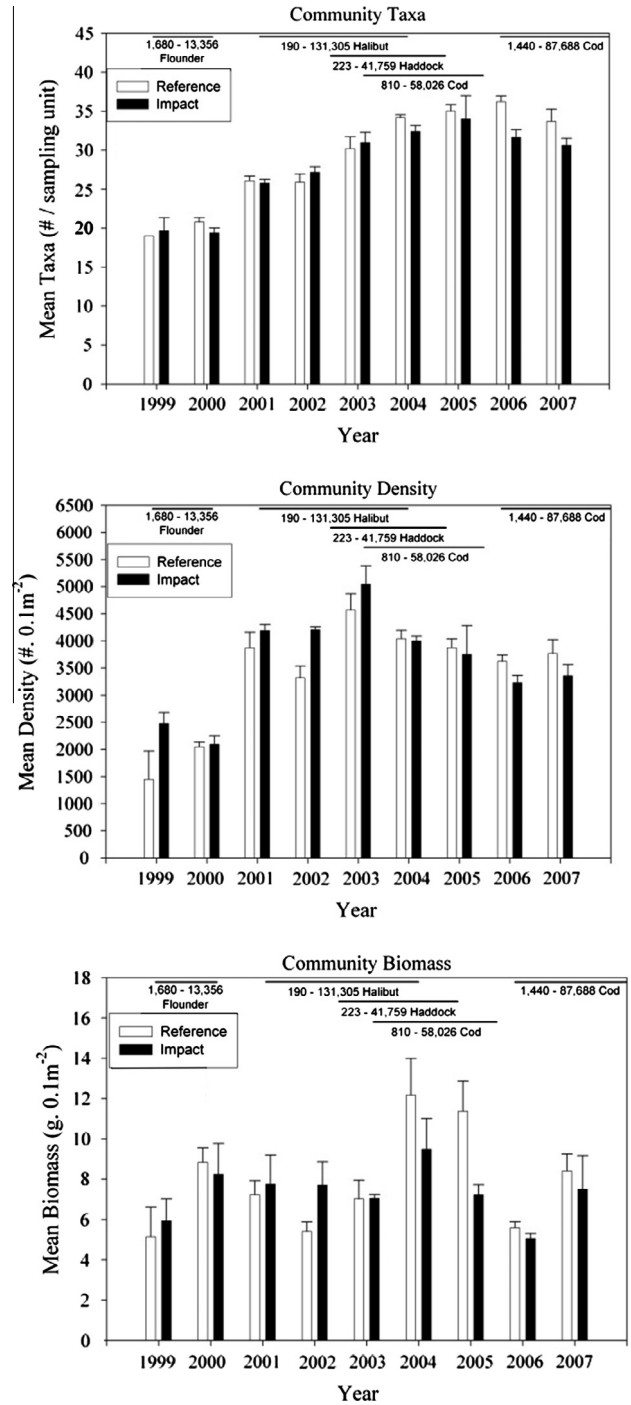

Fig. 3. Benthic community univariate means comparing impact and reference samples on an annual basis; 1 SE shown.

3.4. Development of adaptive monitoring protocol

The sediment and benthic monitoring data, including both univariate and multivariate analyses described above, consistently indicated no significant differences between the impact and reference areas for nearly all annual assessments prior to 2008 (see Grizzle et al. (2003) for early data). This resulted in approval by the permitting agencies of an adaptive monitoring protocol designed to decrease the cost and effort involved in the routine monitoring, as well as development of an "early warning system" should negative impacts occur. Therefore, during 2008 an adaptive

Table 3
Summary of percent loss on ignition (LOI) data on an annual basis. 1 SE given for each mean: *n* values show total number of measurements for duration of study.

LOI	*n*	1999	2000	2001	2002	2003	2004	2005	2006	2007
Control	199	1.52 ± 0.12	1.58 ± 0.05	1.68 ± 0.05	1.71 ± 0.07	1.76 ± 0.07	1.86 ± 0.21	1.84 ± 0.08	1.92 ± 0.43	1.37 ± 0.07
Impact	135	1.47 ± 0.05	1.41 ± 0.03	1.58 ± 0.03	1.54 ± 0.05	1.85 ± 0.14	2.12 ± 0.62	1.83 ± 0.23	1.48 ± 0.08	1.19 ± 0.04

Table 4
Ranking of top twenty taxa from impact and reference sites.

Taxa	Impact		Reference	
	Total number collected	% of total	Total number collected	% of total
F. Spionidae	20,656	50.6	23,648	41.0
F. Paraonidae	2470	6.1	3792	6.6
F. Oweniidae	1761	4.3	2221	3.9
F. Thyasiridae	1612	3.9	2405	4.2
F. Ampharetidae	1479	3.6	2356	4.1
F. Maldanidae	1111	2.7	2385	4.1
F. Sabellidae	1040	2.5	1016	1.8
F. Nuculidae	981	2.4	1848	3.2
P. Nematoda	749	1.8	2836	4.9
C. Oligochaeta	743	1.8	985	1.7
F. Tellinidae	732	1.8	1123	1.9
F. Sternapsidae	622	1.5	861	1.5
F. Sigalionidae	622	1.5	644	1.1
O. Cumacea	481	1.2	922	1.6
F. Orbiniidae	425	1.0	629	1.1
F. Phyllodocidae	417	1.0	602	1.0
F. Epitoniidae	341	0.8	583	1.0
P. Rhynchoela	331	0.8	864	1.5
F. Cirratulidae	319	0.8	752	1.3
F. Mytilidae	202	0.5	644	1.1

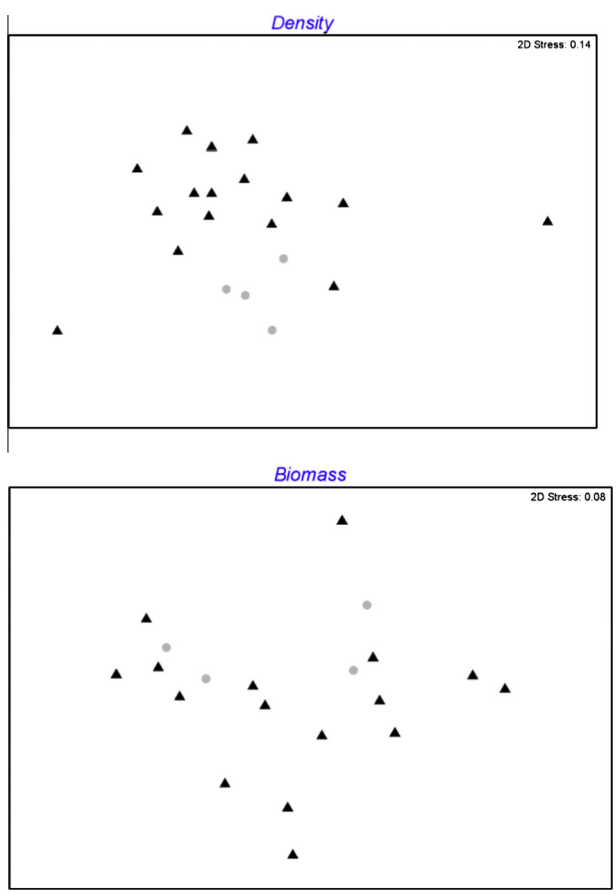

Fig. 4. ANOSIM analysis of family level density and biomass data for 2006–2007 data for impact and reference sites from a Bray-Curtis similarity matrix. Circles represent impact sites, triangles are reference sites.

monitoring protocol consisting of the following three major components was implemented (Fig. 5): (1) sampling the sediments and infauna with box corer (using methods described above) at multiple locations inside and outside of the impact area; (2) recording video imagery at the same sites sampled by box corer; and (3) analyzing the resulting data using a phased, adaptive approach.

All ten sediment samples in the 2008 dataset had LOI values well below the 5% threshold that would have indicated potential organic contamination from aquaculture activities, and signaled the need for further actions to characterize the impacts (Fig. 5). Real-time observations using a towed underwater video system and later complete analysis of the video imagery also indicated no evidence of excessive organic loading to the seafloor (e.g., fish food accumulations, excess densities of worm or amphipod tubes). These two sets of findings resulted in going to the "no further action" step in the new monitoring protocol (Fig. 5).

4. Discussion

The rationale behind the pre-2008 monitoring protocols was periodic comparisons of data from the impact and reference areas, as reported in annual reports to the permitting agencies (see Grizzle et al., 2003 for summary of early data). The modeling assessment discussed above indicates that the modeled impact zone (Figs. 1 and 2) reasonably represented the portion of the seafloor under the fish cages likely to be most-affected by waste discharges for the range of hydrodynamic conditions occurring at the study site most of the time. Thus, the findings of no or minimal differences between impact and reference areas based on univariate and multivariate benthic analyses (Figs. 3–5; Table 4) and sediment data (Table 3) suggest that impacts to the seafloor during the history of operation of the facility were minimal. Comparisons of our data to previous research in the Gulf of Maine and elsewhere in the region suggest the same.

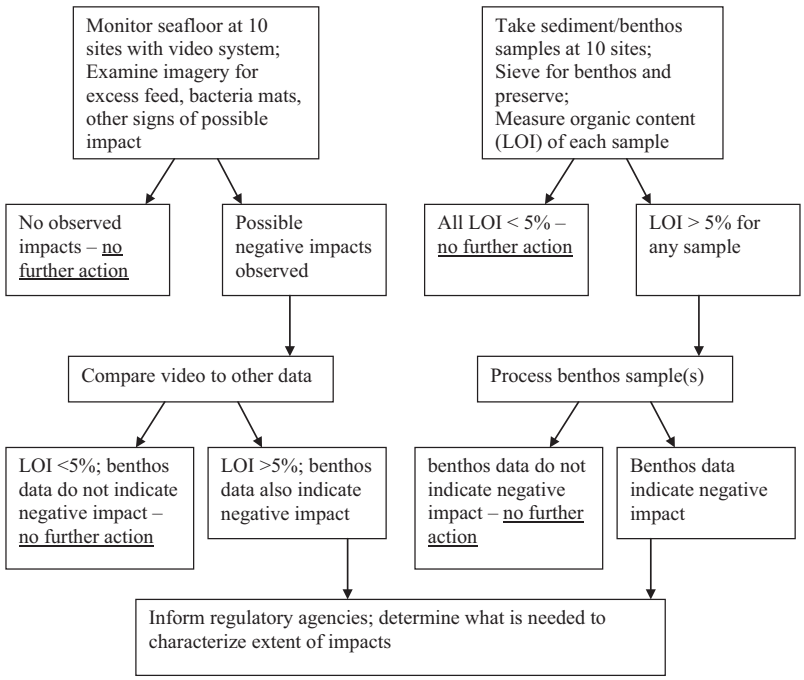

Fig. 5. Flow chart of the chronology of events involved in the overall adaptive monitoring protocol used at the study site during 2008 (see text for details).

Macroinfaunal communities at the study site were typical of those in soft sediments in this region of the Gulf of Maine (Maciolek and Grassle, 1987; Larsen and Doggett, 1991; Theroux and Wigley, 1998; Hilbig and Blake, 2000). Direct quantitative comparisons among the datasets are difficult because of the variety of methods (e.g., different samplers and sieve mesh sizes) used. Nonetheless, some comparisons can be made. For example, in the present study two polychaete families (Spionidae and Paraonidae) made up ~50% of the total abundances from all samples collected, with only minor differences between impact and reference areas (Table 4). Maciolek and Grassle (1987) reported the same two families as density dominants in soft sediments in their historical studies on Georges Bank, an area far offshore and not likely affected by any sources of organic enrichment. A spatially extensive study (Grizzle et al., 2009) using methods similar to the present study (e.g., Wildco box corer, 0.5 mm mesh sieve), but conducted further offshore in areas not likely to be affected by organic wastes, found very similar ranges in soft sediments (mud to sand) for total biomass (<5 to 12 g/0.1 m^2) and taxonomic richness (14–23 taxa/sample), but somewhat lower densities (<1000 to 1800 individuals/0.1 m^2), compared to the present study (compare to our Fig. 3).

It should be noted, however, that for the years 2006 and 2007 when fish biomass was highest in the cages, there was significantly lower benthic community taxonomic richness at the impact sites compared to the reference sites. Benthic density and biomass, however, did not differ. Nor were there significant differences in relative distributions of the taxonomic composition of the benthic communities from the impact and reference areas. Depressed taxonomic richness could have been a response to increased organic loadings, but changes in density, biomass, and taxonomic composition of the benthic communities would also have been expected (Pearson and Rosenberg, 1978; Grizzle and Penniman, 1991; Kennish, 1998). Thus, while there may have been a "signal" (decreased taxonomic richness) of increased organic loadings during the years of highest fish biomass, the data are not conclusive.

Sediment organic content was measured in the present study as loss on ignition (LOI; also expressed as total volatile solids [TVS]), but other metrics (e.g., total organic carbon [TOC]) are also commonly measured in pollution impact studies. Regardless of the metric, sediment organic content is affected by factors other than organic pollutants (e.g., grain size; see discussion below) and the relationship between organic content and pollution loading is not straightforward. Even so, LOI values <5%, as was consistently found in the present study (Table 3) are substantially lower than that reported from "enriched" or "polluted" sediments in other areas (e.g., Grizzle and Penniman, 1991; Holmer and Kristensen, 1992; Staniszewska et al., 2011; Wilding et al., 2012a,b). It should be noted that 5% LOI was chosen in our protocol (Fig. 5) as a "trigger" for further sampling based on our work in muddy continental shelf waters in New England and enriched muddy estuarine sediments, and may not be suitable for other areas. Because sediment organic content typically varies with grain size, "trigger" levels might also vary based on grain size distribution (WAC, 1995; Heinig, 2001).

In sum, our benthic and sediment data indicate no or minimal negative impacts to the seafloor. Why might this be the case? We suggest that wastes (fish feed and feces) were sufficiently dispersed most years by hydrodynamical conditions at the farm site so that no detectable impacts to the seafloor occurred. This is one of the major reasons for the movement of fish farms further offshore where there is generally greater dispersal and dilution of wastes compared to shallower near-shore areas (Holmer, 2010; Price and Morris, 2013), and it seems reasonable to conclude that our monitoring efforts demonstrated this situation for our study site. A particularly relevant factor supporting this conclusion is the magnitude of winter storms that typically occur in the study area. Fredriksson et al. (2005) described a storm with measured wave heights >8 m with a dominant period of 10.24 s. Such longer period waves have orbital velocities that can penetrate the entire depth of the water column (~50 m) with the possibility of inducing resuspension of fish farm wastes. Although such storms are likely typical of a 25-yr event, annual maximum wave events, however, have wave heights of ~6.5 m with a dominant period of 10 s (Fredriksson, 2001). The possibility that organic matter initially deposited in the impact zone could have been transported outside the modeled impact zone seems quite likely. Additionally, fish

survival rates were relatively low most years, ranging from 0% to ⩽60% for 5 of the 8 trials (Table 1). Thus, total fish biomass in the cages and potential deposition of organic matter to the seafloor was low compared to typical cage culture operations. Although our study was not designed to fully quantify the relationship between farm wastes and environmental conditions, this might be the next step. Existing models (e.g., Rensel et al., 2006; Kiefer et al., 2011) could be further parameterized and refined using our data.

There is an enormous amount of global literature on the effects of organic wastes on the seafloor in coastal waters (e.g., Pearson and Rosenberg, 1978; Maurer et al., 1993; Diaz and Rosenberg, 1995; Kennish, 1998; Nilsson and Rosenberg, 2000), including the particular case of aquaculture (e.g., Findlay et al., 1995; GESAMP, 1996; Findlay and Watling, 1997; Black, 2001; Hargrave, 2010; Wang et al., 2012; Wilding et al., 2012a,b; Price and Morris, 2013). And there is a substantial literature on the environmental permitting process (DeVoe, 2000; Cincin-Sain et al., 2005). However, very little published literature deals with permitting and environmental monitoring as inter-related processes. Our adaptive protocol (Fig. 5) was patterned after protocols described by Heinig (2001) and it is similar to existing monitoring and permitting requirements in other areas (e.g., Washington State, USA; WAC, 1995). Such an approach has two major advantages compared to traditional intensive seafloor sampling approaches: lower cost and better environmental protection. The first tier of sampling (towed video and box corer) and analysis (visual inspection of video and LOI analysis) are designed to quickly show if excessive organic wastes have reached the seafloor; this is essentially a low-cost "early warning system." Additional sampling or sample processing occurs only if this initial assessment reveals potentially negative impacts. Thus, the first advantage is a substantial cost savings over traditional methods. The second advantage is also related to the early warning system as well as the cost savings. An adaptive protocol results in less cost per unit sample because most samples are not fully processed, so more samples can potentially be taken for a given overall program budget. This potentially allows more frequent sampling and thus detection of negative impacts before they become excessive or widespread. In sum, these two advantages—lower cost and better environmental protection—represent a significant improvement in traditional environmental monitoring approaches.

In their extensive review of the global literature on marine finfish aquaculture, Price and Morris (2013) note that adaptive monitoring protocols also provide useful information for farm management. For example, early detection of negative impacts could allow managers to make appropriate changes in fish stocking rates, feeding schedules or other protocols so that further impacts are avoided. Finally, adaptive protocols may be combined with mathematical models of waste discharge, assimilation and accumulation to refine the models so they are more sensitive to impact detection (Dudley et al., 2000; Cromey et al., 2002; Kiefer et al., 2011). In her recent review of offshore aquaculture, Holmer (2010) concluded that there is an urgent need for global consensus on monitoring protocols in order to better ensure long-term sustainability of offshore fish farming. We suggest that adaptive protocols might play an important role as we move towards this goal.

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