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An Investigation of the Cumulative Impacts of Shrimp Trawling on Mud Bottom Fishing Grounds in the Gulf of Maine: Effects on Habitat and Macrofaunal Community Structure

Anne W. Simpson

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**AN INVESTIGATION OF THE CUMULATIVE IMPACTS OF SHRIMP
TRAWLING ON MUD BOTTOM FISHING GROUNDS IN THE GULF
OF MAINE: EFFECTS ON HABITAT AND MACROFAUNAL
COMMUNITY STRUCTURE**

By

Anne W. Simpson

B.S. Mary Washington College, 1997

A THESIS

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

(in Oceanography)

The Graduate School

The University of Maine

December, 2003

Advisory Committee:

Leslie Watling, Professor of Oceanography, Advisor

Joseph Kelley, Professor of Marine Geology

Sara Lindsay, Assistant Research Professor of Marine Sciences

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Thesis Advisor: Dr. Leslie Watling

An Abstract of the Thesis Presented
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Despite the widespread occurrence of trawl fisheries on mud-bottoms, there is limited knowledge concerning the effects of trawling induced disturbance on these habitats and their resident macrofaunal communities. I investigated the cumulative impacts of seasonal commercial shrimp trawling on infaunal habitat and macrofaunal community structure on two mud-bottom fishing grounds in the Gulf of Maine from June 2000 – December 2001. One fishing ground located near the Outer Pumpkin Ledges (Pumpkin) experienced trawling activity during the 2000-2001 fishing season. In contrast, the other fishing ground near Monhegan Island was not trawled during the same period because shrimp abundances were low. Consequently, impacts of trawling reported for the Pumpkin fishing ground are representative of both past and more recent short-term effects of trawling while results reported for the Monhegan are indicative of longer-term, cumulative effects only. To further examine short-term effects on trawling on mud

bottom habitat structure, a Before-After Control-Impact (BACI) experiment was carried out at a different location (Thrumcap).

Images of infaunal habitat structure obtained by sediment x-radiography showed no evidence of changes in overall structure (as measured by relative sediment density) related to commercial or experimental (BACI) shrimp trawling disturbance; however, excess ^{210}Pb activity profiles suggest that trawling may affect sediment mixing regimes.

Macrofaunal communities on the two fishing grounds exhibited different responses to shrimp trawling disturbance which I attribute to disparities in levels of fishing activity during the 2000-2001 shrimp season. Multivariate community analysis showed that the Pumpkin fishing ground displayed significant differences in macrofaunal community structure compared to adjacent untrawled areas. Abundances of opportunistic polychaete families were higher in the trawled areas while disturbance-sensitive taxa, such as bivalves, were more abundant in the untrawled area. Similar patterns in taxa abundance were not observed at Monhegan. Results from mud bottom fishing grounds suggest that seasonal shrimp trawling disturbance produced at least short-term changes in infaunal community structure, but did not appear to result in long-term cumulative changes. Resilience to trawling disturbance may be due in part to high levels of biological sediment disturbance from high densities of large surface-dwelling megafauna such as lobsters, fishes, and brittle stars. These animals rework sediments to a depth of 16-17 cm by burrowing, pit-digging and possibly foraging. Sediment reworking by these benthic megafauna creates disturbance that appears to maintain macrofaunal communities in a perpetually low successional state, thereby potentially minimizing trawling impacts.

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exposure and development. Dr. Bob Steneck was kind enough to provide me with his lobster abundance data from Monhegan Island. Dr. Dan Belknap and Allen Gontz deserve a special word of thanks for collecting and processing side-scan sonar data. C. McClellan and M. McClellan and the F/V Adventurer made it possible for me to conduct ROV operations. J. Higgins and R. Downs deftly captained the R/V Ira C. during video quadrat and ROV operations.

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INTRODUCTION

Bottom trawling is a pervasive agent of anthropogenic disturbance in many of the world's most productive marine environments (Dayton et al. 1995, Kaiser 1998, McConnaughey et al. 2000, Messiah et al. 1991, Rijnsdorp et al. 1998, Sainsbury 1987, Thrush 1998, Watling and Norse 1998). Among the immediate and direct effects of trawling are the removal of target and non-target species (Hall 1999, Jennings and Kaiser 1998, Kaiser and de Groot 2000), and physical disturbance of biogenic and abiotic habitat structures (Auster et al. 1996, Collie et al. 1997, Engel and Kvitek 1998, Freese et al. 1999). The ecological consequences of trawling disturbance vary depending on both the substrate type and the degree of natural disturbance present in the environment. A growing body of scientific literature supports the idea of a continuum of trawling impacts in which habitats with high-relief biogenic structures, such as sponges, corals, and seagrass beds, as well as areas with few large-scale natural disturbances, are more severely impacted by trawling than shallow, low-relief areas of the seabed that experience frequent large-scale natural disturbances (Reviews in Auster and Langton 1999, Collie 1998, Hall 1999, Jennings and Kaiser 1998, Collie et al. 2000, National Research Council 2002).

Trawling reduces sea floor structural complexity, (Auster et al. 1996, Auster and Langton 1999, Freese et al. 1999) a habitat feature which is positively associated with biodiversity (Thrush et al. 2001). The effects of trawling on habitat structure are most apparent in relatively high relief areas of the sea bed such as boulder, gravel, and even rippled sand bottoms. Mud bottoms characterized by overall low topographic structure,

but a high degree of small-scale physical, chemical, and biological habitat complexity, may also be severely impacted. On mud bottoms, much of the biological community resides below the sediment surface, thus, most of the habitat structure is not visible from above the seabed.

Trawling causes resuspension of the upper layers of sediment and may also mix or bury remaining substrate (Caddy 1973, Churchill 1989, Mayer et al. 1991). This intense disturbance can alter physical and chemical sediment profiles (Mayer et al. 1991, Pilskaln 1998, Smith, C.J. et al. 2000, Watling et al. 2001) and surficial habitat complexity by creating deep gouges or ruts on the sediment surface (Caddy 1973, Friedlander et al. 1999, Roberts et al. 2000); however, the impact of chronic trawling disturbance on infaunal habitat structure has not been well studied. Trawling may secondarily reduce the degree of mud bottom habitat structure by removing sediment-dwelling organisms that increase physical and chemical heterogeneity in soft sediments through burrowing, feeding, and related activities (Aller 1982, Aller 1988, Wheatcroft 1990).

Comparatively few studies on trawling impacts have been conducted on mud (silt/clay) bottoms relative to sand, gravel and mixed sediment types, despite the existence of commercial trawl fisheries for groundfish and shellfish on fine sediment habitats. Most studies conducted on mud bottoms have employed manipulative field experiments to examine short term effects of known levels of trawling effort (Hansson et al. 2000, Lindegarth et al. 2000, Mayer et al. 1991, Riemann and Hoffman 1991, Sanchez et al. 2000, Sparks-McConkey and Watling 2001, Tuck et al. 2000). The results of short-term experiments provide insight into specific mechanisms of trawling disturbance;

however, these experiments cannot approximate either the spatial extent or chronic levels of disturbance present in commercial trawl fisheries (Jennings et al. 2002; Thrush 1998)

The only study to date to examine the effects of a commercial trawl fishery on a mud bottom habitat was conducted by Smith et al. (2000) in the eastern Mediterranean Sea. Results from this study show strongly negative impacts of trawling on benthic macrofauna and megafauna, as well as significant differences in sedimentary organic carbon, chlorophyll, and phaeopigments between lightly and heavily fished areas (Smith, C.J. et al. 2000). Other studies on fishing grounds with mixed sand and mud (silt/clay) substrates have shown that chronic levels of trawling disturbance reduce the biomass of infauna and epifauna (Jennings et al. 2001), decrease production of large infauna (Jennings et al. 2002), and alter the size structure of benthic infaunal communities (Duplisea et al. 2002).

One of the most important trawl fisheries occurring on mud bottom habitats in the Gulf of Maine is an annual winter fishery for the Northern or Pink shrimp (*Pandalus borealis*). In an area which extends from approximately the southwestern portion of the Gulf of Maine through western Penobscot Bay to Cape Cod, this species is harvested by trawls and traps when ovigerous female shrimp migrate from deeper waters to shallower, near-shore areas where egg release occurs. The shrimp trawling fleet is made-up mainly of vessels <25 m long that tow otter trawls. Nets generally have 70 ft sweeps, 2 inch mesh, and are equipped with rock-hopper gear. The fishing season is characterized by intense fishing activity on mainly mud, muddy sandy, and the edge of hard bottom. The length of the fishing season is highly variable from year to year (0-180 days) due to dramatic fluctuations in regional shrimp populations.

I examined the cumulative impacts of the seasonal shrimp trawling fishery on the infaunal habitat and macrofaunal community structure of relatively deep (>80 m), near-shore mud-bottom fishing grounds in the mid-coast region of Maine. Based on the assumption that these mud bottom habitats possessed a high degree of small-scale structural complexity and were subject to few large-scale natural disturbances, I anticipated that habitat and macrofaunal community structure would be significantly affected by trawling disturbance.

This study consisted of an 18-month field sampling program conducted on two shrimp fishing grounds and adjacent unfished areas. The purpose was to determine the cumulative effects of commercial levels of shrimp trawling on: (1) the physical structure of sedimentary habitats and; (2) the biological structure of the resident benthic macrofaunal community. Common difficulties associated with locating appropriate untrawled areas for control sampling sites (Hall 1999) were dealt with by situating control treatment sampling stations in areas that are the current and historical territory of fixed gear fishermen, and thus, not subject to trawling. In addition to the presence of fixed gear as an obstacle to trawling, one of the two control areas was located in a lobster conservation zone, while the other control site was located in an area that is physically challenging to fish with mobile gear.

To compare the immediate effects of a known level of shrimp trawling effort on infaunal habitat structure with chronic impacts on fishing grounds, a short-term, manipulative, Before-After Control-Impact experiment (Greene 1979) was conducted. A second objective of this experiment was to attempt to elucidate specific mechanisms of sediment disturbance created by the shrimp trawling gear.

METHODS

Measurative Field Experiment on Long-time Shrimp Fishing Grounds

Study Site Location and Description

To assess the cumulative effects of historic and recent shrimp trawling on infaunal habitat and community structure, a measurative field experiment was conducted on shrimp fishing grounds in the mid-coastal region of Maine. Based on information on the location of past and present shrimp trawling effort provided by local fishermen and fisheries scientists, two study areas with similar habitat features and fishing history were selected. Each study area encompassed both traditional fishing grounds and an adjacent area of similar habitat type where shrimp trawling is not known to occur. The first study area was located approximately 2 km north-east and due east of Monhegan Island (N 43° 45° W 69° 18°) (Fig. 1). The other study area was approximately 13 km due south of Boothbay Harbor, south-west of the Damariscove Island (N 43° 44° W 69° 34°), near the Outer Pumpkin Ledges (Fig. 1). Water depths in the study areas ranged from 86-102 m and 84-91 m at Monhegan and Pumpkin, respectively. Bottom water temperature and salinity were similar at both sites (Fig. 2). The substrate in both study areas was predominately fine, relatively high-water content mud with grain surface areas characteristic of silt-sized particles (Mayer and Rossi 1982) (Table 1), and occasional large boulders, most likely glacial relics were present at Monhegan. Visual inspection of the sediment surface at both study sites using a remotely operated vehicle (Phantom 300, Deep Ocean Engineering) showed accumulation of fine flocculent material and no evidence of physical disturbance from storm wave or tidal current scouring.

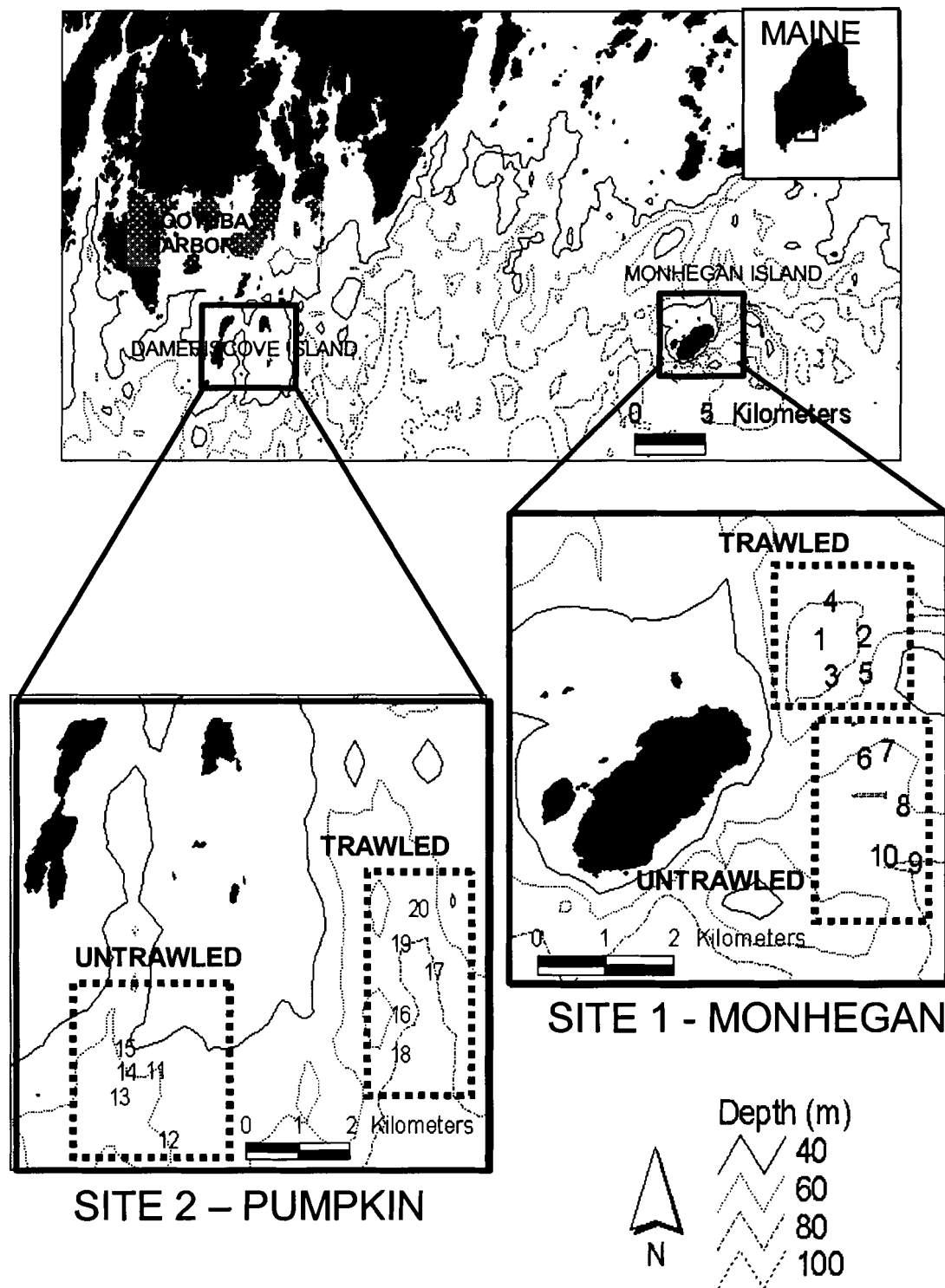


Figure 1. Location of study areas on mud bottom shrimp fishing grounds in the mid-coastal region of Maine. Sampling stations indicated by numerical labels.

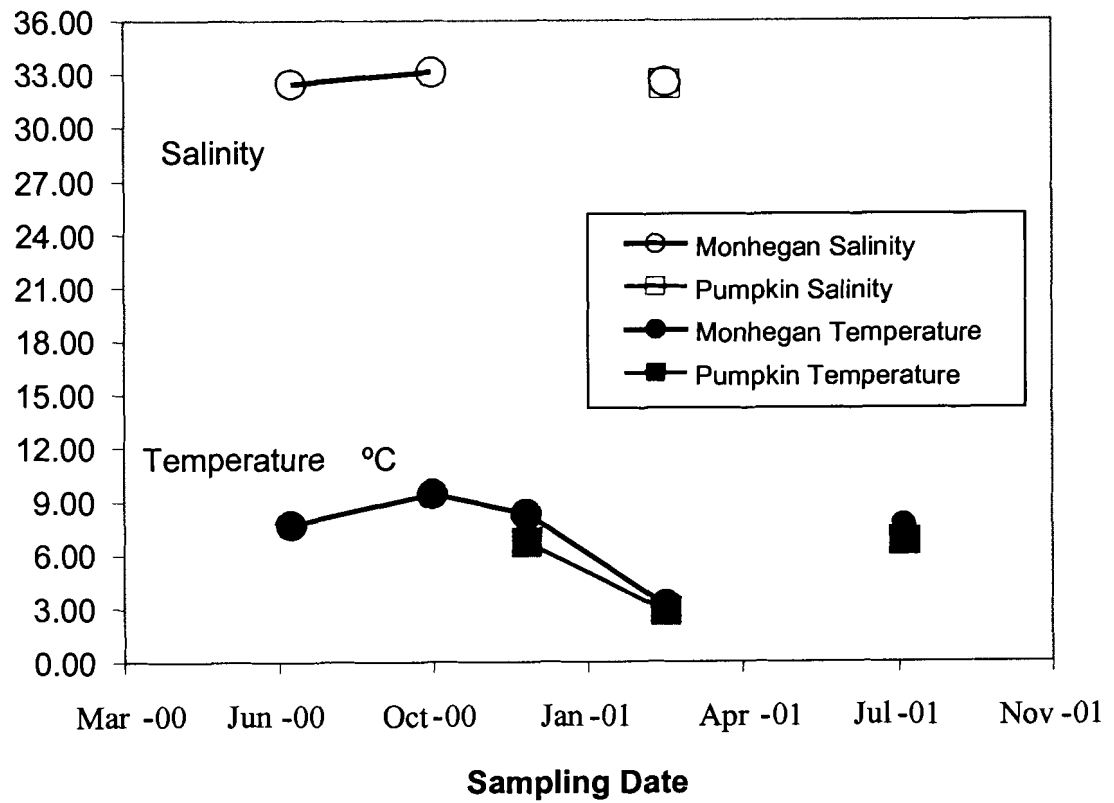


Figure 2. Bottom water temperature (°C) and salinity at fishing ground study areas.

Table 1. Mean grain surface areas (GSA) at Monhegan (stations 1-10) and Pumpkin (stations 11-20) study areas. Only values for 0-1 cm, 4-5 cm, and 9-10 cm sediment depth layers are shown. GSA values $>8 \text{ m}^2/\text{g}$ are characteristic of sediments with grain size medians in the silt size range.

| Sampling Station | Depth in Sediment (cm) | m^2/g | std dev |
|------------------|------------------------|-----------------------|---------|
| 1 | 0-1 | 27.64 | 0.136 |
| | 4-5 | 25.00 | 0.128 |
| | 9-10 | 22.94 | 0.068 |
| 2 | 0-1 | 21.83 | 0.000 |
| | 4-5 | 22.57 | 0.071 |
| | 9-10 | 20.05 | 0.168 |
| 3 | 0-1 | 24.16 | 0.064 |
| | 4-5 | 23.73 | 0.128 |
| | 9-10 | 23.05 | 0.055 |
| 4 | 0-1 | 22.45 | 0.201 |
| | 4-5 | 20.57 | 0.117 |
| | 9-10 | 20.49 | 0.000 |
| 5 | 0-1 | 23.45 | 0.089 |
| | 4-5 | 21.69 | 0.000 |
| | 9-10 | 22.33 | 0.072 |
| 6 | 0-1 | 24.42 | 0.066 |
| | 4-5 | 21.12 | 0.000 |
| | 9-10 | 19.22 | 0.140 |
| 7 | 0-1 | 23.70 | 0.069 |
| | 4-5 | 25.74 | 0.134 |
| | 9-10 | 27.20 | 0.078 |
| 8 | 0-1 | 24.31 | 0.000 |
| | 4-5 | 26.00 | 0.057 |
| | 9-10 | 21.22 | 0.166 |
| 9 | 0-1 | 21.89 | 0.260 |
| | 4-5 | 25.34 | 0.079 |
| | 9-10 | 20.42 | 0.124 |
| 10 | 0-1 | 22.63 | 0.077 |
| | 4-5 | 22.59 | 0.083 |
| | 9-10 | 20.82 | 0.199 |

| Sampling Station | Depth in Sediment (cm) | m^2/g | std dev |
|------------------|------------------------|-----------------------|---------|
| 11 | 0-1 | ns | ns |
| | 4-5 | 30.00 | 0.166 |
| | 9-10 | 28.16 | 0.092 |
| 12 | 0-1 | 30.82 | 0.051 |
| | 4-5 | 24.48 | 0.081 |
| | 9-10 | 20.80 | 0.173 |
| 13 | 0-1 | 29.35 | 0.163 |
| | 4-5 | 26.61 | 0.083 |
| | 9-10 | 26.96 | 0.000 |
| 14 | 0-1 | 30.67 | 0.285 |
| | 4-5 | 31.17 | 0.097 |
| | 9-10 | 30.09 | 0.109 |
| 15 | 0-1 | 26.74 | 0.102 |
| | 4-5 | 28.18 | 0.000 |
| | 9-10 | 25.09 | 0.176 |
| 16 | 0-1 | 28.12 | 0.000 |
| | 4-5 | 27.18 | 0.070 |
| | 9-10 | 26.25 | 0.141 |
| 17 | 0-1 | 26.88 | 0.096 |
| | 4-5 | 29.19 | 0.157 |
| | 9-10 | 27.22 | 0.225 |
| 18 | 0-1 | ns | ns |
| | 4-5 | 27.58 | 0.000 |
| | 9-10 | 27.34 | 0.063 |
| 19 | 0-1 | 26.00 | 0.236 |
| | 4-5 | 27.73 | 0.061 |
| | 9-10 | 26.71 | 0.094 |
| 20 | 0-1 | 27.73 | 0.218 |
| | 4-5 | 27.77 | 0.183 |
| | 9-10 | 26.19 | 0.428 |

No sample = ns.

Structures created by biological activity, including tubes, burrows, and pits, were plentiful at both sites. Commonly observed surface-dwelling animals were the brittlestar (*Ophiura sarsi*), lobster (*Homarus americanus*), northern shrimp (*Pandalus borealis*), and longfin hake (*Phycis chesteri*).

Study Site 1 – Vicinity of Monhegan Island

The Monhegan study area was located in one of the state of Maine's two Lobster Conservation Zones. Lobstermen in this zone have the exclusive right to fish the waters in a 3 mile radius around Monhegan Island with fixed gear during a 6 month fishing season, Dec. 1st – May 1st. Several Monhegan Island lobstermen stated that no shrimp trawling had been observed, and no fixed gear had been lost due to trawling in several deep mud basins off the east side of the island (S. Stanley and M. Thompson, pers.comm.). This area appears to be sheltered from shrimp trawling activity by both historic lobster fishing territory boundaries and relatively high densities of fixed gear during the winter shrimp season. Based on the above information, the untrawled area at Monhegan was located in one large, and a second adjacent smaller, muddy basin (N 43° 45° W 69° 17°) roughly 2 km east of Monhegan Island. The trawled area at Monhegan (N 43° 46° W 69° 17°) was situated approximately 1 km northeast of the island on a relatively wide expanse of mud bottom. This area has been historically trawled for shrimp by boats from the neighboring Port Clyde fishing fleet. Both Monhegan Island lobstermen and fishermen from the near-by fishing town of South Bristol reported observing Port Clyde fishing vessels trawling for shrimp in this location in recent years (E. Gastaldo and M. Thompson pers. comm.).

Study Site 2 – Vicinity of Outer Pumpkin Ledges

The untrawled area at the Outer Pumpkin Ledges study area was located in a narrow and short, north-south running channel (N 43° 43' W 69° 35'), which makes it a difficult and undesirable area to tow. Both mobile and fixed gear fishermen noted that this area is dominated by fixed gear (lobster and shrimp traps) during the winter shrimp fishing season (pers. comm. B. McLean, E. Gastaldo, W. Audio). Local fishermen confirm, and fisheries monitoring data support, the notion that shrimp trawling has not occurred at this location in recent or past decades (State of Maine Department of Marine Resources [DMR] unpublished data). The trawled area at Pumpkin was situated in a wide, easily towed, north-south running channel (N 43° 44' W 69° 33'). Information supplied by fishermen and independent fisheries monitoring data document historic and recent shrimp trawling in this location (E. Gastaldo, B. McLean, M. McLellan, and W. Audio per comm.; DMR unpublished data).

Study Design

This study was a mesurative, (as opposed to a manipulative), field study carried out using a block design where the treatment variable, the presence of shrimp trawling, was assigned with *a priori* knowledge of fishing activity. To avoid difficulties associated with pseudoreplication, data were collected at two separate study sites, each consisting of an area of trawled and adjacent untrawled bottom. At each study site, five replicate box cores were collected at initially haphazardly selected stations in each treatment area. Bad weather and equipment malfunction created sampling difficulties resulting in fewer than

the desired number of samples being collected during the first two sampling periods, yielding an unbalanced design in June 2000 and October 2000. During the first sampling period (June 2000), five replicate box cores were collected in the Monhegan untrawled area; however, only three replicate box cores were collected in the Monhegan trawled area, and Pumpkin trawled and untrawled areas. Box cores collected during the second sampling period (October 2000) were identical in number and location to those in the first sampling period with the exception of one less box core sample collected in the Monhegan untrawled region. Sampling stations were re-located using a Global Positioning System (GPS) and cores were collected within approximately 25 m of the designated station location.

Sample Collection

From June 2000 through December 2001, sediment samples were collected approximately every 60-120 days at both study sites using a 0.625 m² GOMEX style box core (Boland and Rowe 1991). Cores were sub-sampled for sediment x-radiography, porosity, and macrofauna. In March 2001, one box core collected in the Monhegan trawled area and a second box core collected in the untrawled study area were sub-sampled for excess ²¹⁰Pb activity along vertical profiles. In August 2001, all porosity cores were sub-sampled for grain surface area measurements. Video quadrat surveys were conducted in April 2002 and June 2002 to determine surface burrow densities in both study areas. Qualitative observations of surficial sediment structures and surface-dwelling macro- and mega-fauna were carried out in October 2001 and 2002 using a Phantom 300 remotely operated vehicle (ROV). Side-scan sonar mapping was conducted

in July 2002 at Monhegan in order to resolve substrate type and to detect trawl door marks. Bottom temperature and salinity measurements were made throughout the study using a CTD (Sea-bird Electronics, Inc.) and temperature data logger (HOBOTemp, Onset Computer Corporation).

Sampling Procedures and Data Analysis

Sediment X-Radiography

To determine whether shrimp trawling causes changes in habitat structure by altering particle distribution, and consequently sediment water content, x-radiographs of intact box core “slices” were obtained; image gray level values corresponding to relative sediment density were analyzed as a proxy for water content. A hollow 24 x 55 cm rectangular core was used to extract a 2.5 cm or 4.0 cm thick cross section from near the center of each box core. A protective bottom piece was fitted over the sediment filled end of the rectangular core to prevent loss of material while the sample was prepared for x-radiography. The exterior surface of the core was carefully cleaned to remove any adhering sediment prior to x-ray exposure. The core was placed upright at one end of a 50 x 150 x 60 cm lead-lined box and secured directly in front of a 38 x 46 cm x-ray cassette (with L-Plus enhancement screens) housing 35 x 43 cm Kodak X-OMAT AR Scientific Imaging film (Fig. 4). Exposures were made at 50 kV for 0.24 seconds. (for cores 2.5 cm thickness) and 0.48 seconds. (for cores 4 cm thickness) at a target distance of 1 m using a HF 80+ 15 mA Mini-XRay, portable veterinary x-ray unit. X-radiographs produced a 1/1 image of fine scale sediment density including biogenic structure (tubes, burrows, and feeding voids), as well as images of calcified sediment-dwelling animals

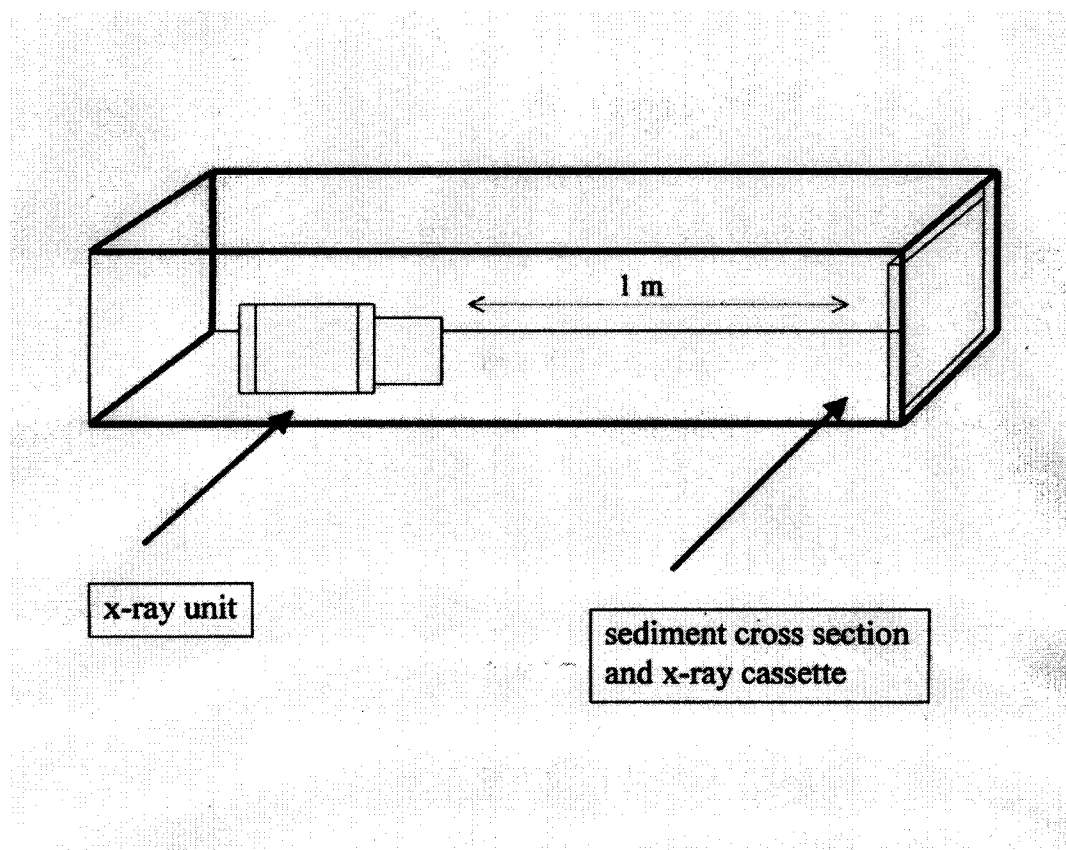


Figure 3. Configuration of sediment core x-raying equipment used on sampling vessel.

such as brittlestars, bivalves, and gastropods (Fig. 12). Exposed x-ray film was developed within 24 hours of exposure at Miles Memorial Hospital, Damariscotta, Maine, using an automated x-ray film developing system. The use of a calibrated, automated developing system minimized variation in x-radiography development (gray level) due to temperature processing differences, variable developer strength, and length of time in developer that might have occurred had the film been developed in the field.

Sediment x-radiographs were digitized at 1000 dpi using a UMAX Mirage II x-ray film scanner. Using image analysis software (Image Pro Plus 4.1, Media Cybernetics, Inc.), relative sediment density was evaluated by measuring the average gray level value (gray level scale 0-255) from digitized images in successive 1 cm depth layers from the sediment surface to the bottom of the x-radiograph (Fig. 13). Mean \pm 1 standard deviation of gray level values were recorded for all sediment depth layers in each image. Biogenic structures and sediment-dwelling animals captured in the x-radiographs were noted.

X-ray image gray levels from replicate box cores collected in trawled and untrawled areas were first analyzed to detect differences in sediment density structure with depth using a Kruskal-Wallis non-parametric rank test (Zar 1996). Image gray level data was then analyzed by individual 1 cm depth layers using a Mann-Whitney rank sum test (Zar 1996), to examine differences in sediment density between like depth layers in cores from trawled and untrawled areas. Because depth of the sediment sample obtained varied between box cores, and thus x-radiographs, only the upper 10 cm of the core were included in the analysis. Gray level data from each sampling period was analyzed

separately because x-ray core sample thickness and exposure time were varied throughout the course of the study.

Video Quadrat Surveys

Surface penetrating burrow densities were measured using video quadrat surveys to determine if shrimp trawling disturbance had an impact on surficial habitat complexity provided by this type of biogenic structure. A 1.0 m² aluminum frame quadrat with an attached drop camera was lowered to the bottom to obtain a visual record of burrows in a 1.0 m² field of view. Digital video tape from the quadrat survey was analyzed by obtaining a single frame immediately before the quadrat came to rest on the bottom. At each study site a 100 quadrats total were obtained, 50 quadrats from each treatment area. From each image, the number and size (diameter) of all visible burrows was systematically recorded. Burrows were grouped into the following seven categories based on size (measured as diameter): (a) >1 cm; (b) 1 cm; (c) 2 cm; (d) 3 cm; (e) 4 cm; (f) 5 cm; (g) <5 cm.

Burrow densities for each size category were examined with a Mann-Whitney rank test. Data from Monhegan and Pumpkin were analyzed separately based on the assumption that different sampling blocks were not equivalent (Krebs 1999.)

Porosity

Sediment porosity was measured to: (1) examine the effect of shrimp trawling on this physical structural property and; (2) provide a means of ground-truthing sediment density measurements made from x-radiographs. Porosity samples were collected from

each box core using a clear acrylic tube with a 10 cm interior diameter (i.d.). Each tube core was vertically extruded and sectioned at 1 cm intervals to a depth of 10 cm. Sections were placed into individual air tight containers for transport to shore. In the laboratory, each sample was homogenized and an approximately 2.0 g (wet weight) sub-sample was transferred to an aluminum drying pan. Sediment wet-weight was measured using a Mettler AE 100 calibrated electronic balance. All samples were dried to a constant weight in an oven at 60°C for approximately 24 hours. Dry samples were allowed to cool in a moisture-free environment and then were re-weighed to obtain total dry weight. Sediment porosity was calculated using the following formula:

$$\text{Porosity} = (\text{wet weight} - \text{dry weight}) / (\text{wet weight} - \text{dry weight}) + (\text{dry weight} / 2.65)$$

Differences in sediment porosity between trawled and untrawled areas were analyzed by individual 1 cm depth increments from 0-10 cm using a three-way ANOVA with sampling date (DATE), study site (SITE), and the presence/absence of trawling activity (TREATMENT) as the main factors. Data were tested for normality and homoscedasticity and met these criteria in all but a few cases.

Excess ²¹⁰Pb Activity

Physical re-working of upper sediment layers was examined using ²¹⁰Pb, a naturally occurring particle-reactive radioisotope. Cores for excess ²¹⁰Pb activity profiles were collected from both a trawled and untrawled station at Monhegan using a 10 cm i.d. clear, acrylic tube. Sediment from each tube core was vertically extruded and sectioned

at 1 cm intervals for the complete length of the core. The outer-most layer (approximately 5 mm) of sediment was carefully scraped away and discarded to avoid sample contamination from down-core smearing during core extrusion. In the laboratory, each sediment sample was homogenized and a 2 gram sub-sample was removed for porosity measurement. The remaining portion of each sample was lyophilized to remove water and then disaggregated using a mortar and pestle. Five grams of dried sediment were transferred to pre-weighed plastic vials where samples were packed to produce uniform geometry among samples. Uranium-series activities, including that of ^{210}Pb , were measured by non-destructive gamma spectrometry using a single, closed-end, coaxial well germanium detector (Canberra model GCW3523/S). ^{210}Pb activity was measured at the 46.5 keV energy peak. Supported levels of ^{210}Pb were obtained by measuring the activity of Pb-214 at the 352 energy peak. Detector counting efficiencies were determined by counting a U.S. Department of Energy certified “pitchblende ore” uranium standard (CRM-103A) of known activity, with all daughter isotopes in secular equilibrium. Sources of error associated with counting, background radioactivity and detector counting efficiency were calculated and corrected for in final ^{210}Pb activity measurements. Both error and activity measurements were corrected for radioactive decay which occurred between sample collection and counting time.

Excess ^{210}Pb activity values were plotted by depth. Total excess ^{210}Pb inventories were obtained by summing activity profiles of both the trawled and untrawled core(s).

Grain Surface Area

Sediment grain size characteristics were determined for untrawled and trawled treatment areas at both study sites using grain surface area (GSA) measurements, which are a function of sediment grain size (Mayer et al and Rossi 1982). In August 2001, subsamples from porosity cores were obtained from all box cores for GSA measurements. Only the 0-1, 4-5, and 9-10 cm sediment depth fractions were analyzed for GSA using the one point BET-method described in Mayer 1994. Briefly, samples were washed in an acetone/water solution to remove salt and freeze-dried to remove all moisture. Organic matter was removed by placing sediment in 550⁰C oven for 12 hours. Samples were then dried in a vacuum overnight and stored in a desiccator the following day. Dried samples were placed in a Quantachrome Monosorb gas analyzer (model MS-6, Quantachrome Corp.) where samples were cooled by immersion in liquid nitrogen while N₂ gas was passed through the sample, causing a monolayer of N₂ gas to adsorb to the surface of each sediment grain. Subsequent warming of the sample resulted in desorption of the monolayer of N₂ molecules coating the surface of each sediment particle. The total amount of N₂ gas released from the sample was measured and the total solid surface area, was calculated.

Differences in mean grain surface area between trawled and untrawled areas were analyzed using a Mann-Whitney rank sum. Monhegan and Pumpkin GSA values were analyzed separately based on experimental design assumptions that separate sampling blocks were not equivalent (Krebs 1999).

Side-scan Sonar

High resolution side scan sonar bottom mapping was performed at Monhegan using an Edgetech digital side scan sonar to check for the presence of marks in the substrate from recent shrimp trawling activity. This effort was prompted by suspicions that the trawled treatment area at Monhegan may not have been trawled during the course of this study due to the near record low levels of fishing effort resulting from low shrimp abundance during the 2000-2001 season (DMR unpublished data). Fisheries port sampling data show no records of fishing northeast of Monhegan Island and local fishermen reported that the area was perhaps only very lightly fished (DMR unpublished data, E. Gastaldo, pers.comm.)

Sediment Macrofauna

Effects of shrimp trawling disturbance on community structure and composition were determined by examining patterns in macrofaunal abundance. A sample for quantitative community analysis was collected from each box core using a clear acrylic tube with a 6.5 cm i.d. Each tube core was vertically extruded, sectioned into 0-2cm, 2-4cm, 4-6cm, 6-10 cm sediment depth increments and placed in separate containers for transport to the lab where all samples were sieved using a 0.5 mm screen mesh and preserved with a 10% buffered formalin solution <12 hours from the time of sample collection. To obtain larger and deeper (>10cm) burrowing macrofauna for qualitative community description, sediment remaining in the box core after the removal of sub-cores was sieved over a 2.0 mm mesh. Material remaining on the sieve was placed in a sample container with 10% buffered formalin.

In the laboratory, macrofauna samples were sorted from non-biological material using an Olympus (SZ40) dissecting microscope. As a quality control, 20% of samples were re-examined to insure all macrofauna had been removed. All taxa were identified to the family level (Somerfield and Clarke 1995), or next lowest possible level above family, and counted for community analysis.

Macrofaunal abundance data from both tube cores and sediment remaining in the box core was combined to determine the relative proportions of phyletic groups in study area communities. Temporal patterns of total abundance, average taxa abundance (richness), diversity (Shannon-Wiener), and evenness (Pielou's), were examined using data from the macrofaunal tube cores. Data for diversity and evenness measures were square-root transformed prior to analysis. Due to unequal sample sizes, data from the first two sampling periods, June 2000 and October 2000 were not included when examining temporal trends.

Non-metric multi-dimensional scaling (MDS) ordination was used to identify patterns in assemblage composition. This analysis was performed with the MDS routine in PRIMER 5.0 (Clarke and Warwick 1994) using Bray-Curtis similarity measures applied to square-root transformed taxa abundance data. Results of this analysis are displayed in two-dimensional ordination diagrams where the positions of sample units (depicted as points in ordination space) are indicative of relative degree of similarity to other sample units. Stress values for MDS ordination diagrams are reported; this value is a measure of the accuracy with which the 2-dimensional ordination diagram represents the relative sample unit position in multi-dimensional space.

To test for statistically significant differences between treatments (untrawled and trawled areas) in the multivariate taxa abundance data set depicted graphically in MDS ordination diagrams, a one-way Analysis of Similarities (ANOSIM) (Clarke and Green 1988) was performed on the square root transformed taxa abundance matrix for each study site using PRIMER 5.0. Taxonomic groups which contributed the most proportionally to average Bray-Curtis similarity or dissimilarity between sample groups were identified using the similarity of percentages (SIMPER) (Clarke 1993) routine in PRIMER 5.0.

Relationships between the abundance of individual taxa and environmental variables, including sediment depth and the presence of shrimp trawling activity, were examined using Redundancy Analysis (RDA) (CANOCO 4.5). This ordination technique is a constrained variation of principle components analysis in which axes are linear combinations of explanatory variables; however, RDA assumes a linear model of taxa response to environmental gradients (McCune and Grace 2002). Only taxa with greater than 1% total abundance at either study site were included in this analysis. The rationale for exclusion of rare taxa was based on the need to: (1) reduce the potential for rare taxa to exert a disproportionate influence of community analysis; (2) reduce the overall number of null abundance values because it has been frequently argued that Euclidean distance based RDA is an inappropriate analytical method for data sets with large numbers of null abundance values (e.g. Legendre and Legendre 1998). Taxa abundance data were square-root transformed prior to RDA ordination. Statistical significance of each environmental (explanatory) variable axis was tested with a Monte Carlo permutation test (499 unrestricted permutations). Forward selection was used to rank the

relative importance of each environmental variable based on the amount of variance it captured in the macrofaunal abundance data set. Values for percent variance explained were reported for environmental variable axes with statistically significant relationships to the data set. Reported values are based on marginal effects only (i.e. each variable is treated as the sole predictor variable). The contribution of each variable in conjunction with covariables (i.e. conditional effects) was not considered because all environmental variables included in this analysis were categorical and therefore, not suited to variable/covariable analysis.

Shrimp Trawling Before-After Control-Impact Experiment (BACI Experiment)

Study Site Location and Description

A manipulative field experiment was conducted in July 2001 for the joint purpose of determining immediate impacts of a known level of shrimp trawling disturbance on habitat structure and to elucidate mechanisms of trawling-induced sediment disturbance. This small-scale experiment was carried out in the mid-coast area of Maine near the mouth of the Damariscotta River, southeast of Thrumcap and Little Thrumcap Islands (N 43° 49' W 69° 33') (Fig. 4). The experimental site was situated in an area of relatively open mud bottom at roughly 60 m depth. Information from local fishermen and fisheries port sampling records (E. Gastaldo pers. comm.; DMR unpublished data) indicate that this area was not targeted by shrimp trawlers.

Experimental Design

A before-after control-impact (BACI) design (Green 1979) was used for this experiment. Six sampling stations were initially haphazardly selected in the study area. Box cores were collected in each of these locations for initial sediment structure analysis. Two weeks after the first sample collection, three of the six stations were repeatedly trawled (5 passes) using a standard shrimp trawling net with rock-hopper gear. All stations were re-sampled with a box core two weeks after trawling to examine gear effects on sediment structural properties.

Sample Collection

All box cores were sub-sampled for porosity and x-radiography. An additional sub-sample was collected for excess ^{210}Pb activity analysis from one station in each treatment area after trawling. Following the initial sampling, stations were relocated using GPS. An ROV was used to verify that stations in the trawled treatment showed visible evidence of trawling disturbance during post-trawl sampling.

Sampling Procedures and Data Analysis

BACI experiment samples for x-radiography, porosity, and ^{210}Pb were collected and analyzed according to the procedures outlined in the previous section.

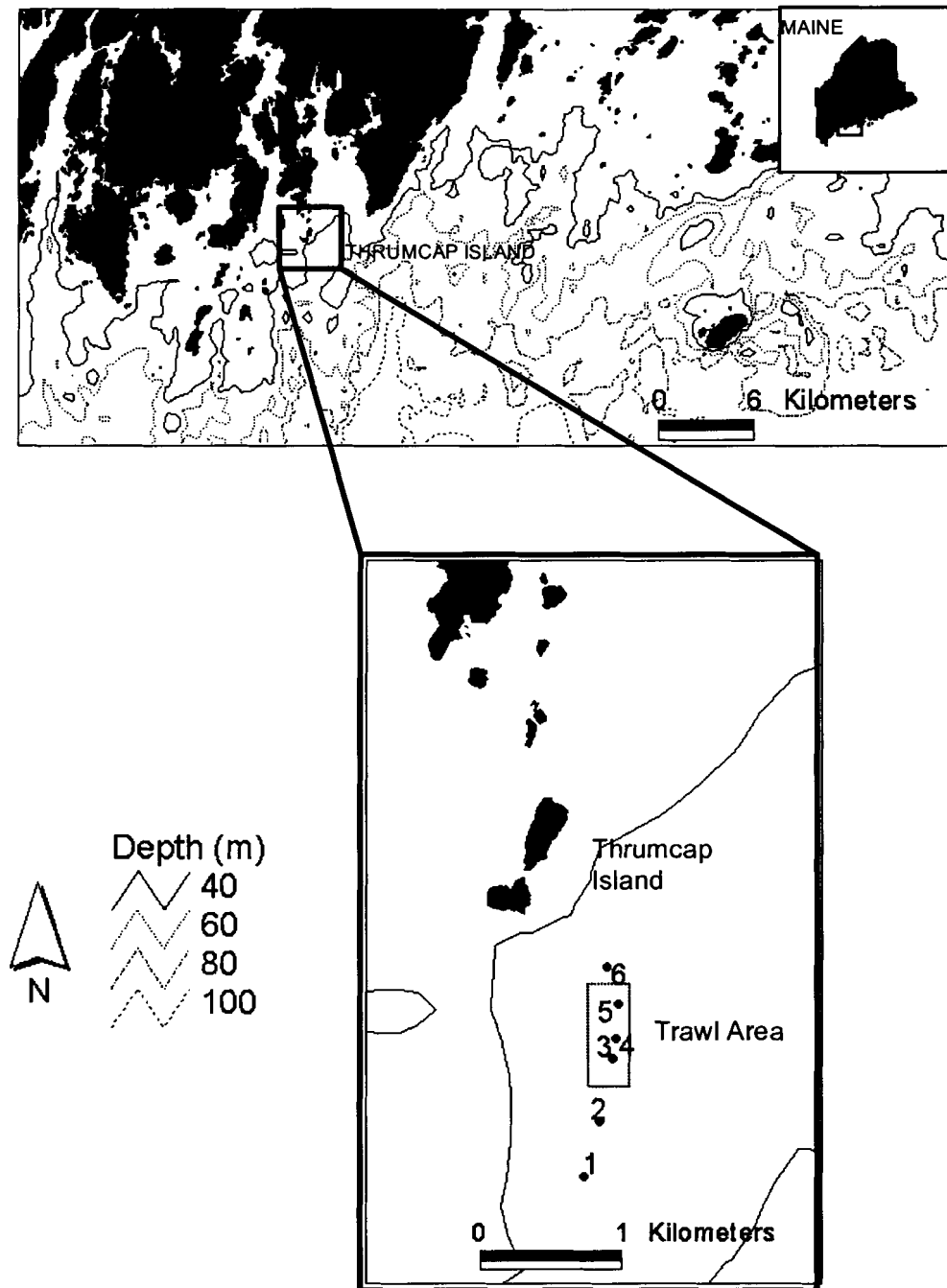


Figure 4. Location of BACI shrimp trawling experimental site near Thrumcap Island. Sampling stations labeled numerically. Boxed area delineates treatment area where experimental shrimp trawling was conducted.

RESULTS

Cumulative Impacts of Shrimp Trawling on Mud Bottom Fishing Grounds

Shrimp Trawling Impacts on Habitat Structure

Sediment Density Structure

Although sediment density structure varied significantly with depth (p value) in all sampling periods, no clear patterns between sediment density and the presence of shrimp trawling were apparent (Table 2). Differences in the range of gray-level values between the first three sampling periods (June 2000 – Fig. 5, October 2000 – Fig. 6, and December 2000 – Fig. 7) and the remaining four periods (Figs. 5-11) were due to an increase in the thickness of x-ray cores from 2.5 cm to 4 cm. Exposure time was increased from 0.24 seconds to 0.48 seconds to optimize x-ray image quality of the thicker core sections. Core thickness was increased during the course of this study in an effort to reduce the degree of sediment compression in x-ray cores by decreasing the surface area to volume ratio. The use of wider x-ray corers did appear to slightly reduce the degree of sediment compression inside the x-ray core; however, it was later determined that the apparent compression in x-ray cores (the difference the height of the sediment surface inside and outside the x-ray core) was largely the result of sediment displacement outside the x-ray corer when it was pushed down into the sediment-filled box core.

During the first three sampling periods (June 2000, October 2000, and December 2000), overall sediment density structure was similar in untrawled and trawled areas at

Table 2. Mann-Whitney rank sum test results comparing x-ray image gray level in trawled and untrawled areas. Data were analyzed by 1 cm depth increments. P-values <0.05 denote significant differences at $\alpha = 0.05$ confidence levels.

| MONHEGAN | | | | | | | |
|----------|--------|--------|--------|--------|--------|--------|--------|
| Depth | Jun-00 | Oct-00 | Dec-00 | Mar-01 | Jun-01 | Aug-01 | Dec-01 |
| 0-1 | 0.655 | 0.480 | 0.117 | p<0.05 | p<0.05 | 0.917 | p<0.05 |
| 1-2 | 0.881 | 0.724 | 0.117 | p<0.05 | 0.175 | 0.754 | p<0.05 |
| 2-3 | 0.881 | 0.724 | 0.251 | p<0.05 | 0.175 | 0.754 | p<0.05 |
| 3-4 | 0.655 | 0.724 | 0.117 | p<0.05 | 0.602 | 0.754 | p<0.05 |
| 4-5 | 0.456 | 0.480 | 0.175 | p<0.05 | 0.754 | 0.602 | p<0.05 |
| 5-6 | 0.297 | 0.480 | 0.251 | p<0.05 | 0.602 | 0.465 | p<0.05 |
| 6-7 | 0.297 | 0.724 | 0.251 | p<0.05 | 0.251 | 0.602 | p<0.05 |
| 7-8 | 0.180 | 0.724 | 0.251 | p<0.05 | 0.347 | 0.602 | p<0.05 |
| 8-9 | 0.180 | 0.724 | 0.251 | p<0.05 | 0.347 | 0.465 | p<0.05 |
| 9-10 | 0.180 | 0.724 | 0.251 | p<0.05 | 1.000 | 0.754 | p<0.05 |

| PUMPKIN | | | | | | | |
|---------|--------|--------|--------|--------|--------|--------|--------|
| Depth | Jun-00 | Oct-00 | Dec-00 | Mar-01 | Jun-01 | Aug-01 | Dec-01 |
| 0-1 | 0.513 | 0.275 | 0.175 | 0.076 | 0.175 | 0.076 | 0.754 |
| 1-2 | 0.275 | 0.513 | 0.175 | 0.347 | 0.251 | 0.076 | 0.754 |
| 2-3 | 0.275 | 0.275 | 0.175 | 0.465 | 0.117 | 0.076 | 0.465 |
| 3-4 | 0.127 | 0.513 | 0.175 | 0.175 | p<0.05 | 0.076 | 0.347 |
| 4-5 | 0.275 | 0.513 | 0.117 | 0.251 | p<0.05 | 0.047 | 0.465 |
| 5-6 | 0.275 | 0.827 | 0.221 | 0.251 | p<0.05 | 0.076 | 0.917 |
| 6-7 | 0.513 | 0.827 | 0.327 | 0.347 | p<0.05 | 0.117 | 0.917 |
| 7-8 | 0.513 | 0.827 | 0.327 | 0.465 | 0.117 | 0.076 | 0.754 |
| 8-9 | 0.827 | 0.827 | 0.327 | 0.347 | 0.117 | 0.076 | 0.602 |
| 9-10 | 0.275 | 0.827 | 0.221 | 0.347 | 0.251 | 0.076 | 0.465 |

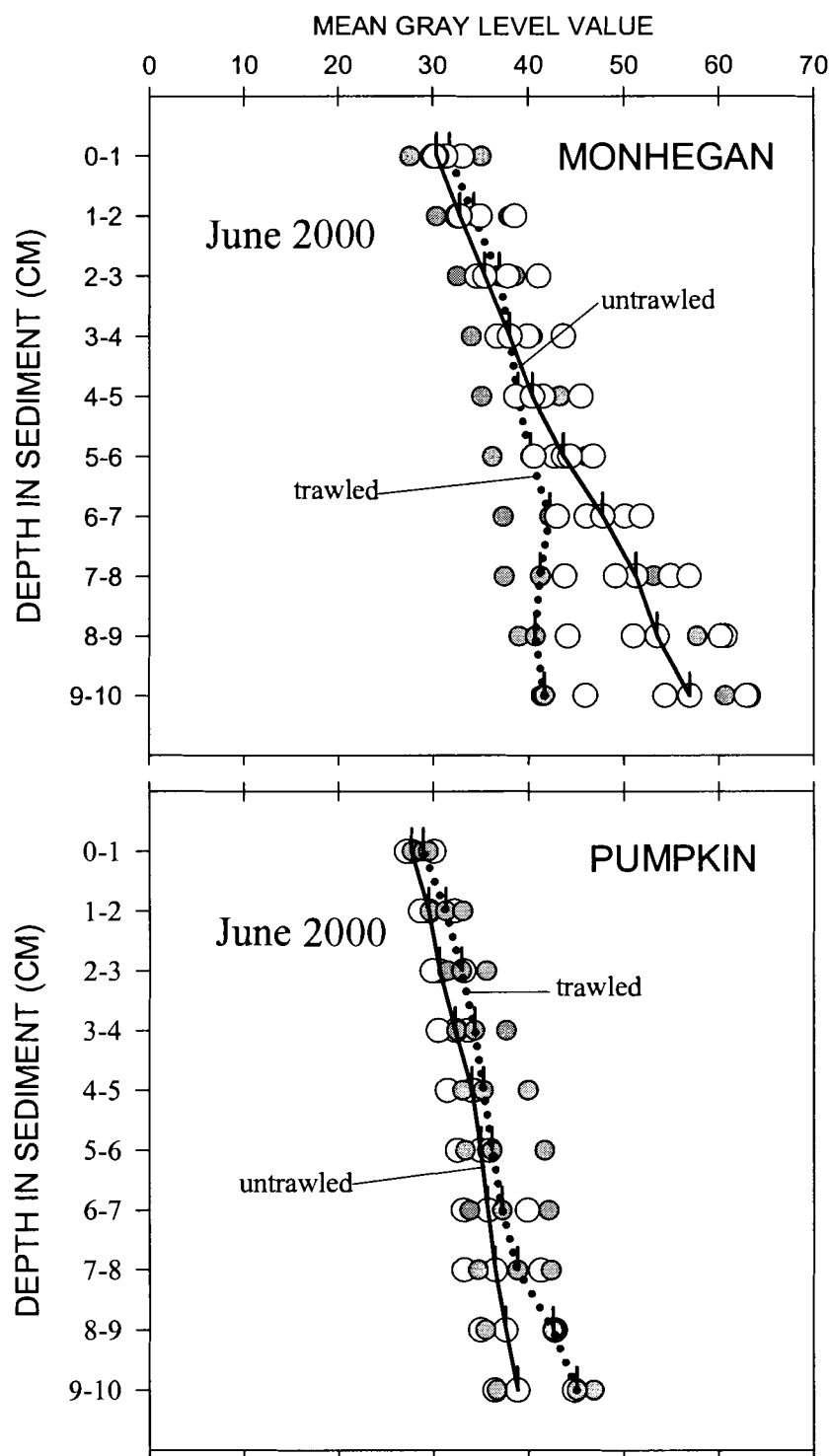


Figure 5. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in June 2000. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

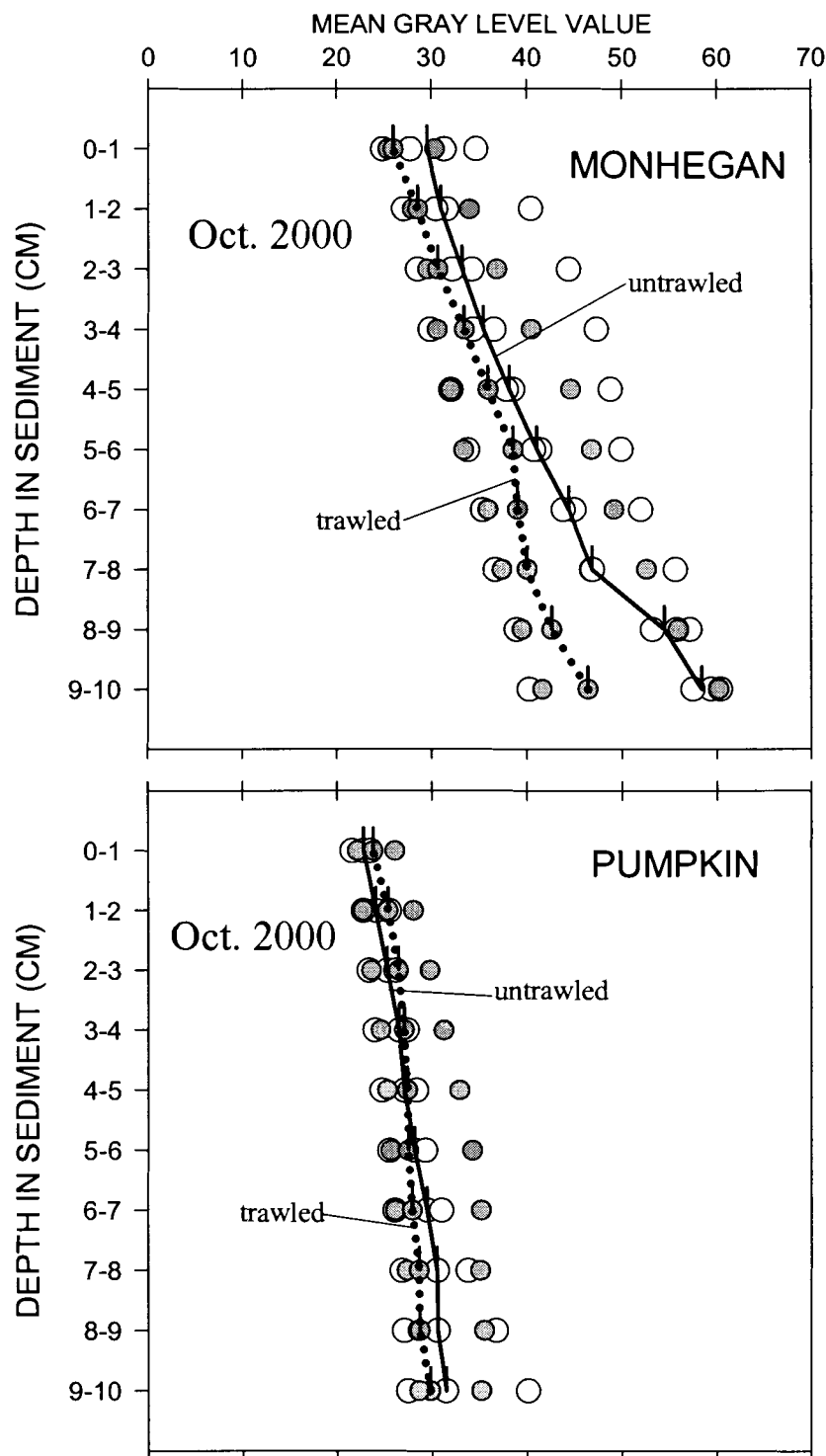


Figure 6. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in October 2000. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

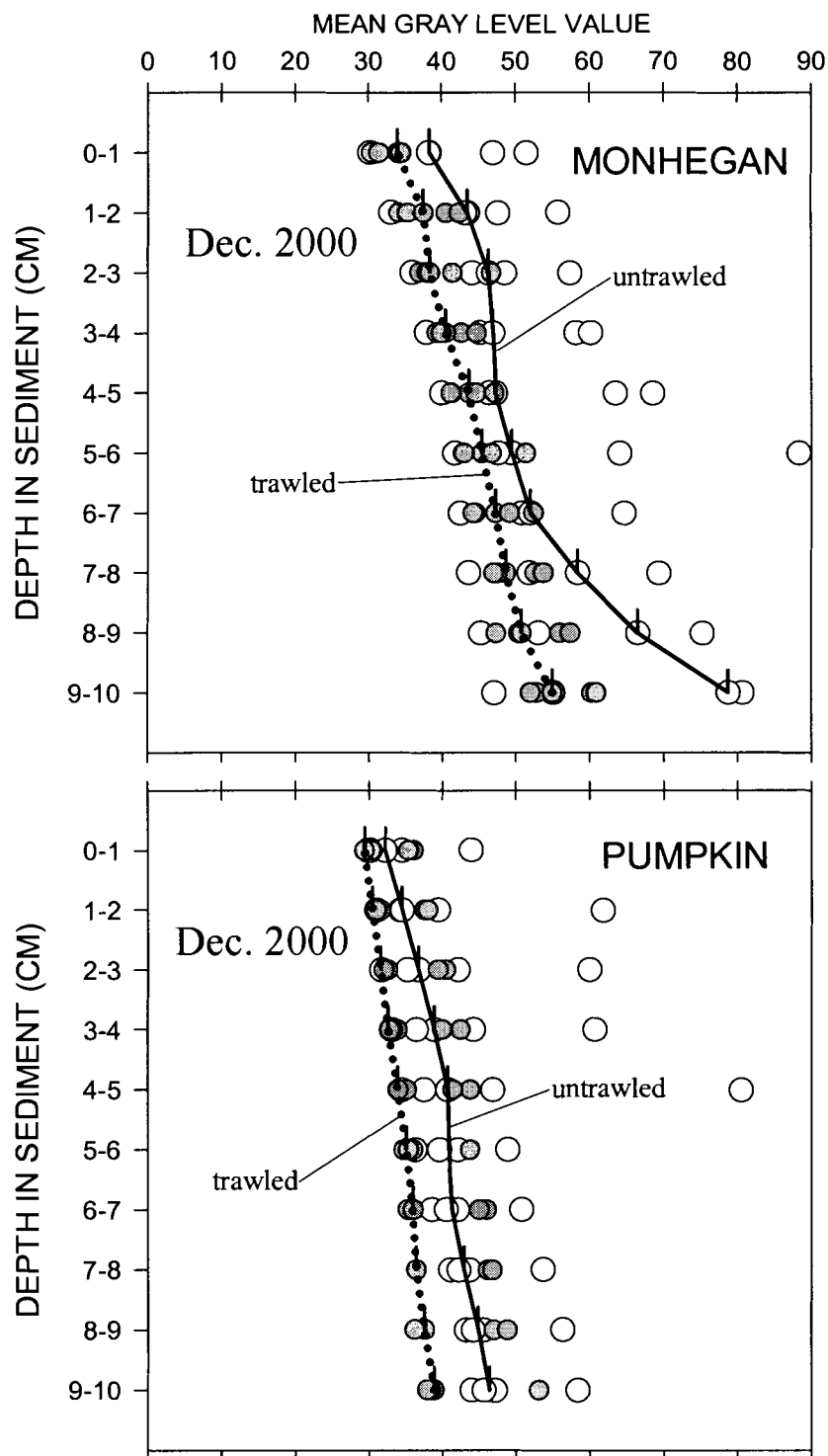


Figure 7. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in December 2000. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

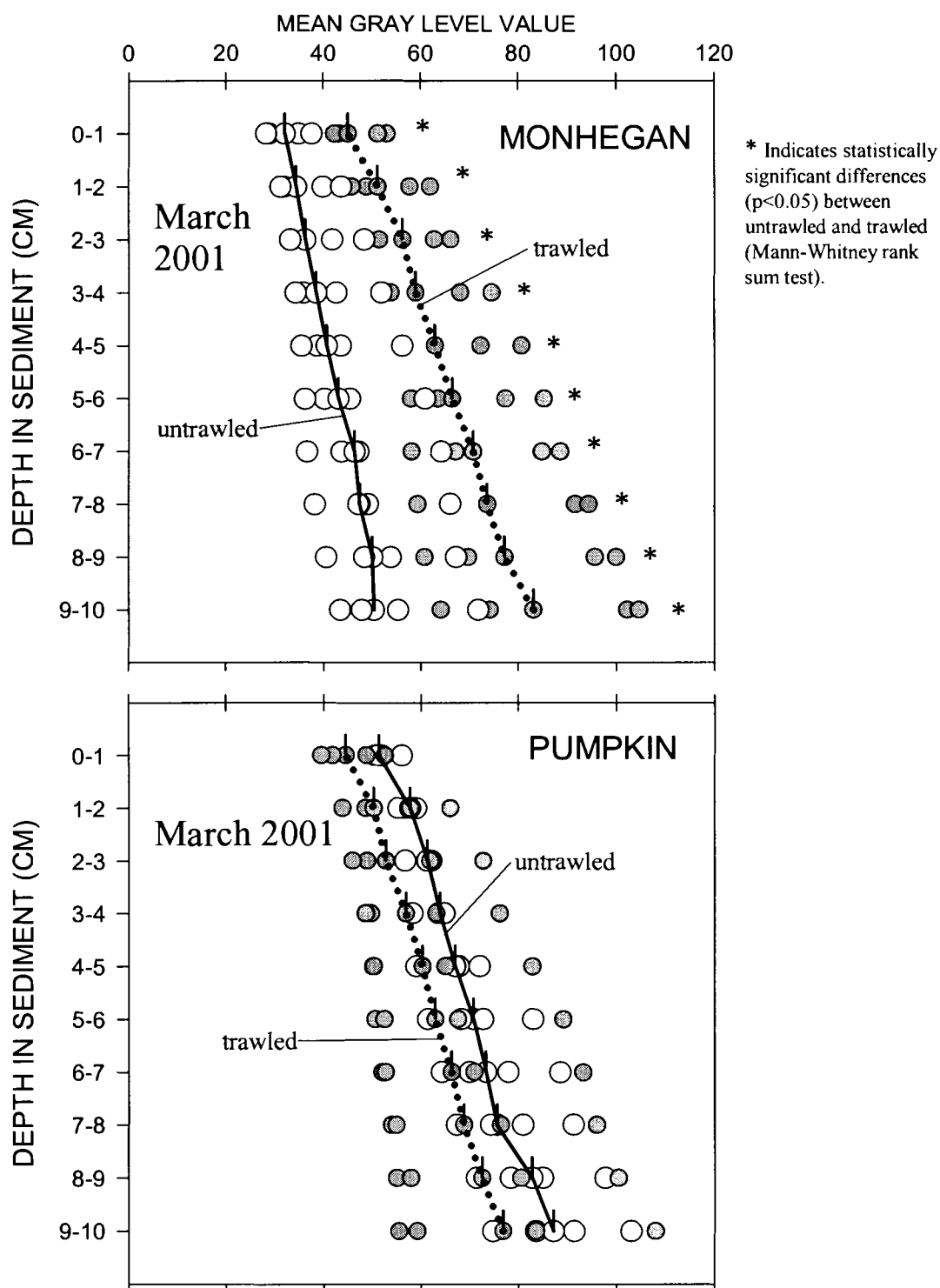


Figure 8. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in March 2001. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

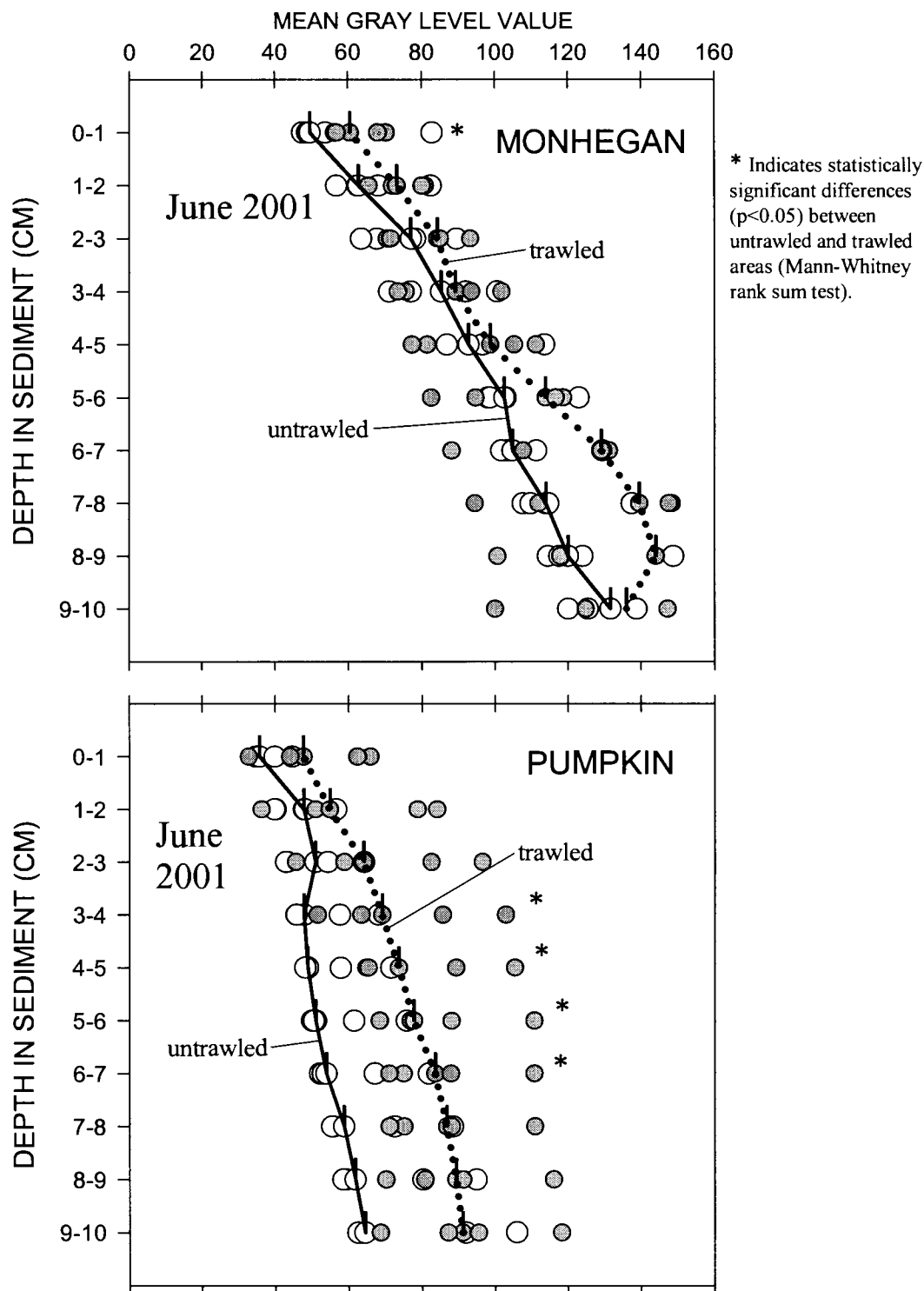


Figure 9. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in June 2001. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

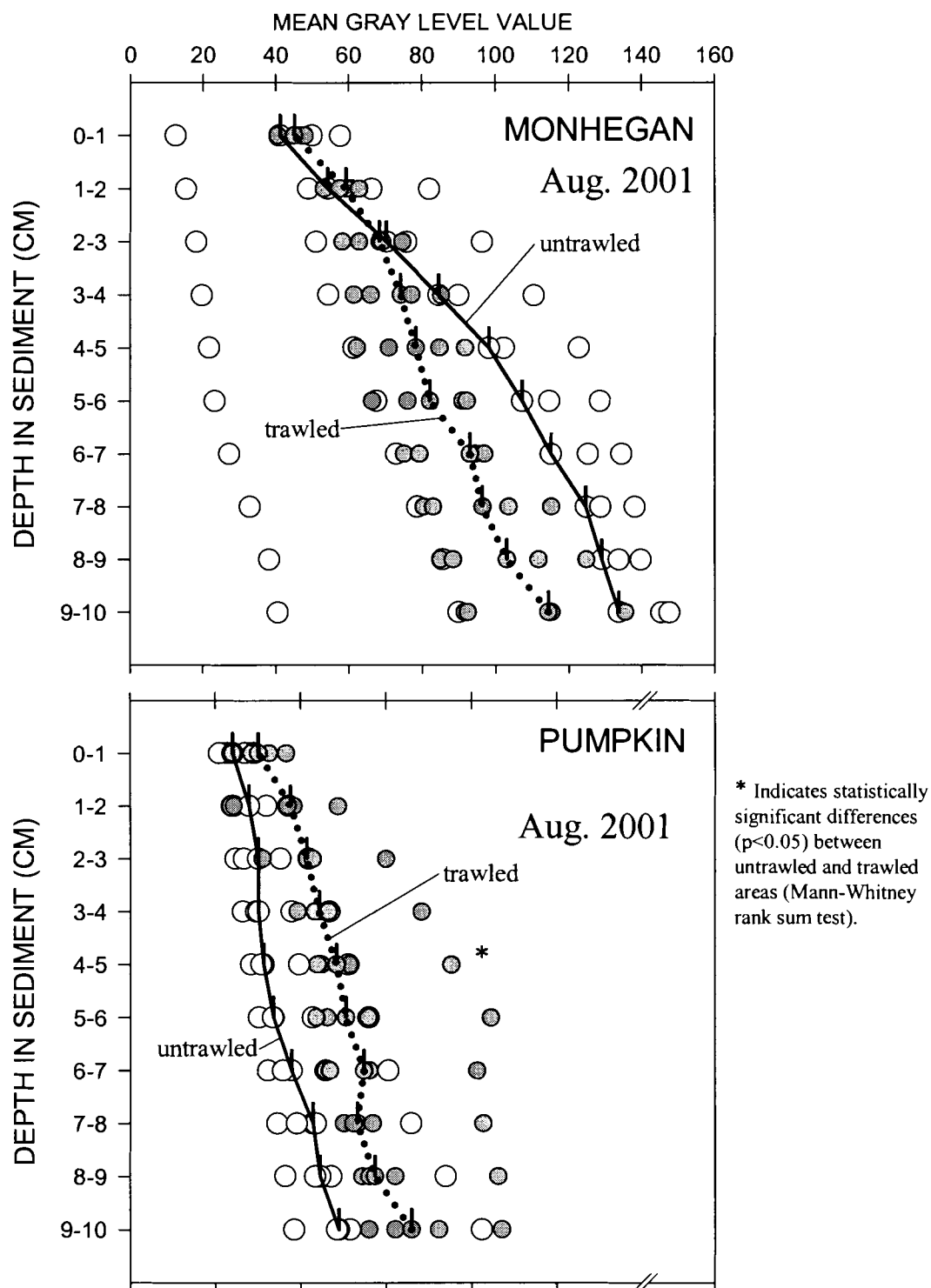


Figure 10. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in August 2001. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

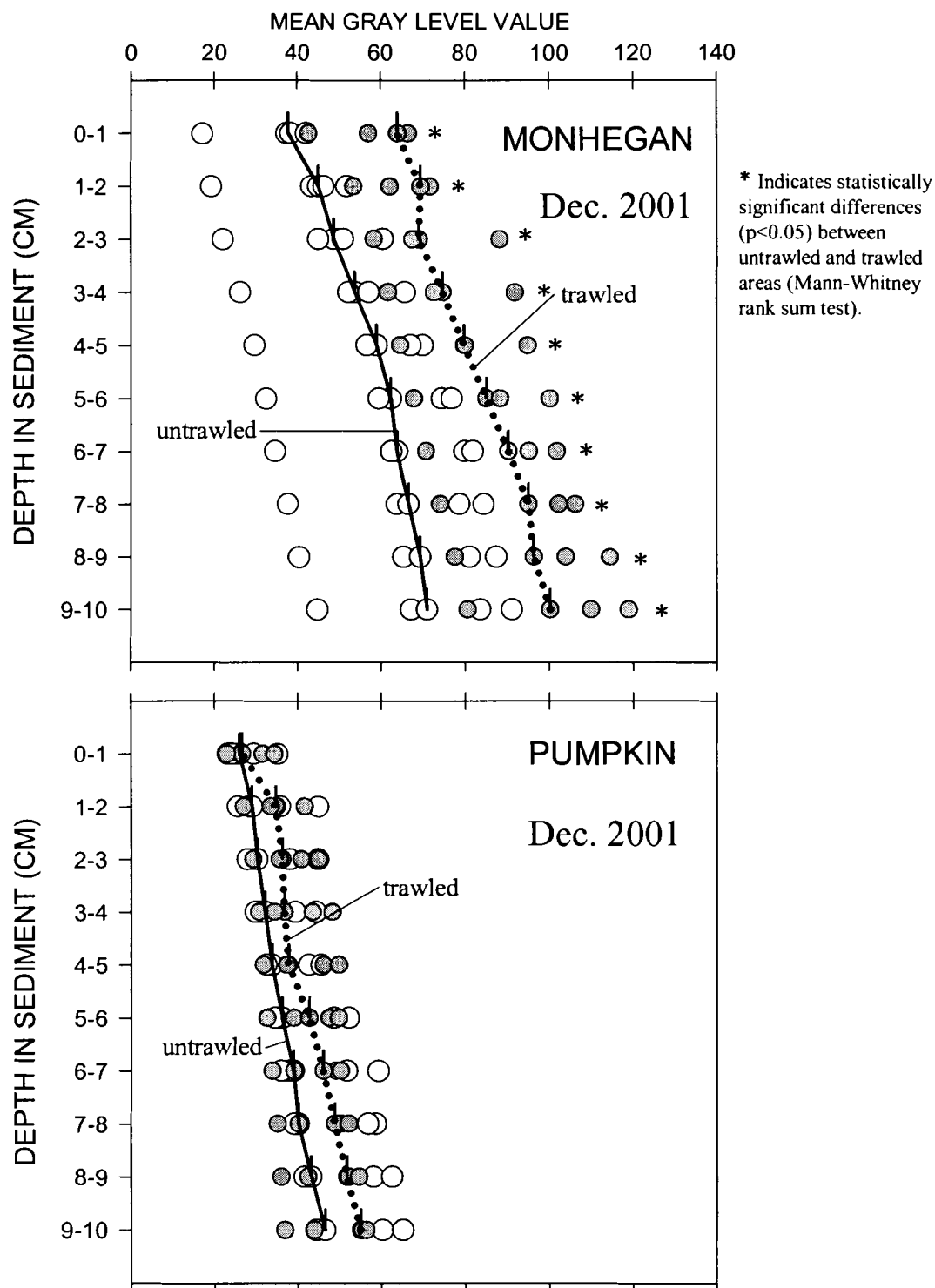


Figure 11. Sediment x-radiography mean gray level values from trawled (shaded circles) and untrawled (open circles) areas at Monhegan and Pumpkin in December 2001. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in trawled (dotted line) and untrawled (solid line) areas.

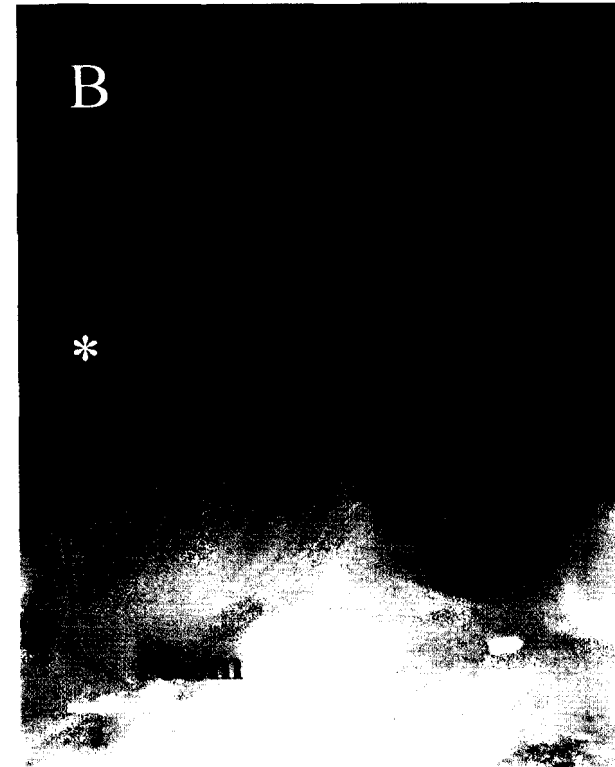
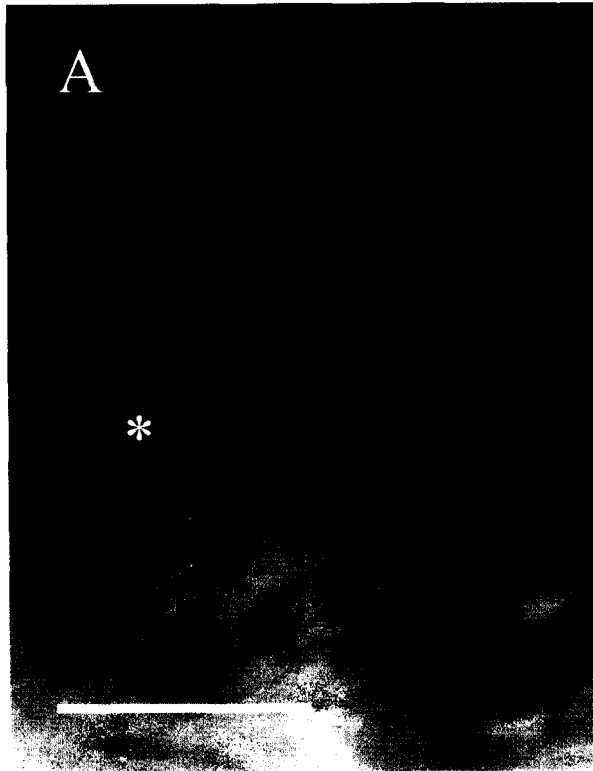


Figure 13. Sediment x-radiographic images from Monhegan trawled (A) and untrawled (B) areas in June 2001. Note the presence of the brittlestar (*Ophiura sarsi*) indicated by * in image A and a parchment worm tube also indicated by * in image B. Darker areas in x-radiograph correspond to less dense areas of the core while lighter colored areas correspond to more dense material

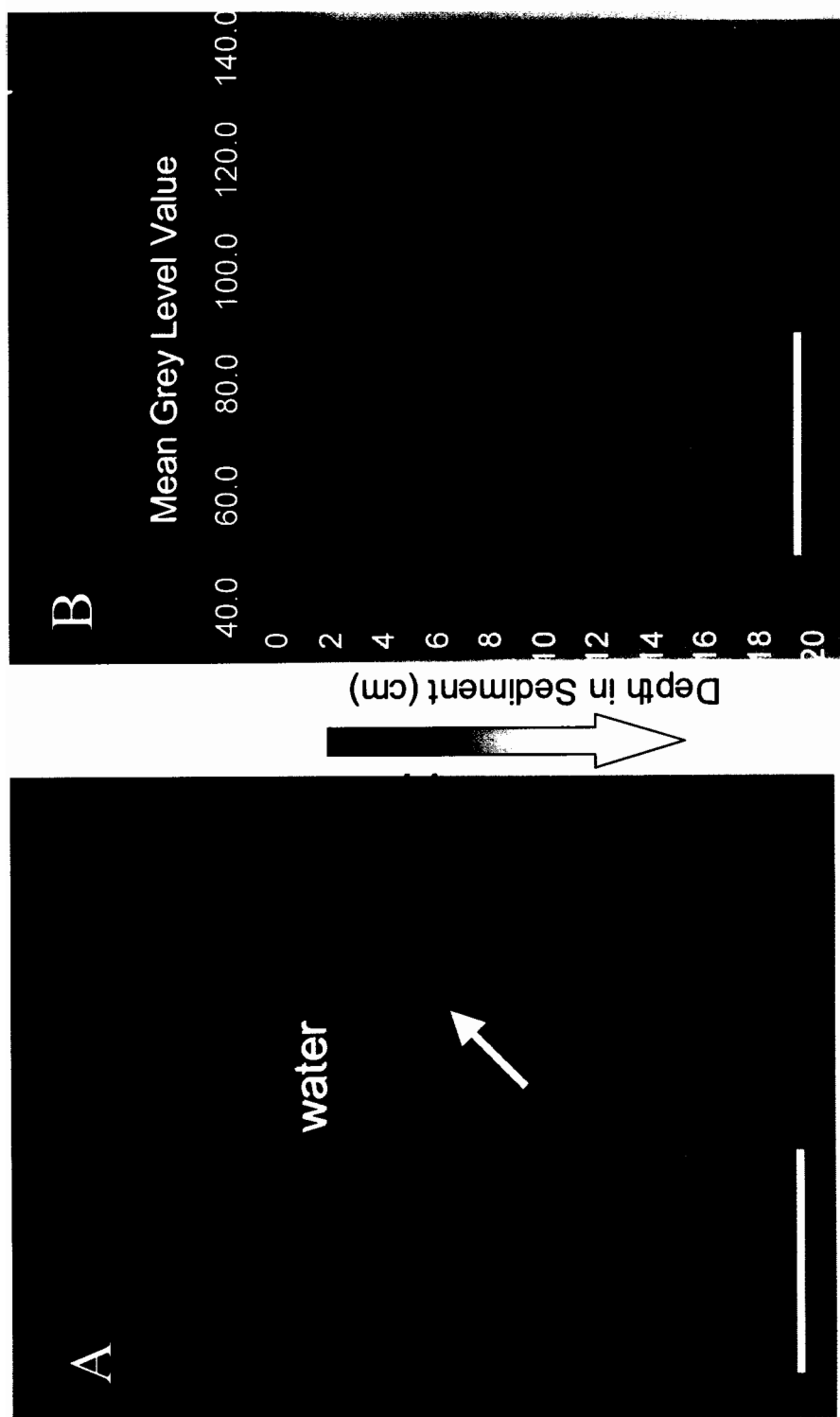


Figure 14. Image analysis of sediment x-radiographs. Measurement of average gray level and standard deviation values in successive 1 cm layers (A). Downcore profiles of relative sediment density were generated by plotting mean \pm 1 standard deviation gray level value (B). Higher (i.e. lighter) gray level values correspond to increased sediment density.

both study sites, with median gray level values ranging from 26.0 - 78.7, and 22.8 - 46.4, at Monhegan and Pumpkin, respectively (Figs. 5, 6, & 7). Results of a Mann-Whitney rank sum tests show no significant differences in sediment density between like depth layers (from 0-10 cm) in untrawled and trawled areas during these first three sampling periods (see Table 2).

Sediment density was significantly different ($p < 0.05$, Mann-Whitney rank sum test) between untrawled and trawled areas in all depth layers at Monhegan for the sampling period coincident with shrimp fishing season (March 2001); however, this pattern was not present at Pumpkin where no significant differences were detected. Median gray level values for the trawled area at Monhegan were higher than those observed in the untrawled area and in both areas at Pumpkin (Fig. 8).

Three months after the end of the shrimp fishing season (June 2001), only the surface layer (0-1 cm) of sediment at Monhegan showed a significant difference ($p < 0.05$, Mann-Whitney rank sum test) in sediment density between untrawled and trawled areas (Table 2). At Pumpkin, sediment density was similar between treatment areas; however, 2 of the 5 cores examined from the trawled area displayed high gray level values from 3-7 cm depth in two of the cores resulting in overall statistically significant differences between trawled and untrawled areas ($p < 0.05$, Mann-Whitney rank sum test) (Fig. 9).

The median and overall range of gray level values differed between Monhegan and Pumpkin during the June 2001, August 2001, and December 2001 sampling periods. Median gray level values were always higher and the range of values was always greater at Monhegan (Figs. 9, 10, and 11). In August 2001, the untrawled area at Monhegan displayed the widest range of gray level values (sediment density) recorded during the

course of this study (12.3 – 147.5 gray level values). Sediment density structure was less variable at Pumpkin during the same sampling period. No significant differences between sediment density in untrawled and trawled areas were detected during the August 2001 sampling period with a single exception: in the 4-5 cm depth layer at Pumpkin there was a significant difference ($p < 0.05$, Mann-Whitney rank sum test) between treatment areas (Table 2).

In December 2001, Monhegan and Pumpkin displayed different sediment density patterns in untrawled and trawled areas. At Monhegan, there was a significant difference ($p < 0.05$, Mann-Whitney rank sum test) in sediment density in all depth layers between treatment areas (Table 2). Median gray level values were higher always in the trawled area (Fig. 11). No significant differences in sediment density structure were detected between treatment areas at Pumpkin, during the same sampling period. Median gray level values were only slightly higher in the trawled compared to the untrawled area (Fig. 11).

Surface Burrow Densities

Analysis of surface penetrating burrow densities showed no clear persistent effect of shrimp trawling on this biogenic habitat feature. Burrows were significantly more numerous in the trawled area ($p < 0.05$, Mann-Whitney rank sum test) for five of the seven burrow diameter classes examined at Monhegan (Fig. 14). This was not the case for burrows of any diameter in the trawled area at Pumpkin. Rather, burrows were significantly more numerous ($p < 0.05$, Mann-Whitney rank sum test) in the size 3 cm and 4 cm diameter classes in the untrawled area at Pumpkin.

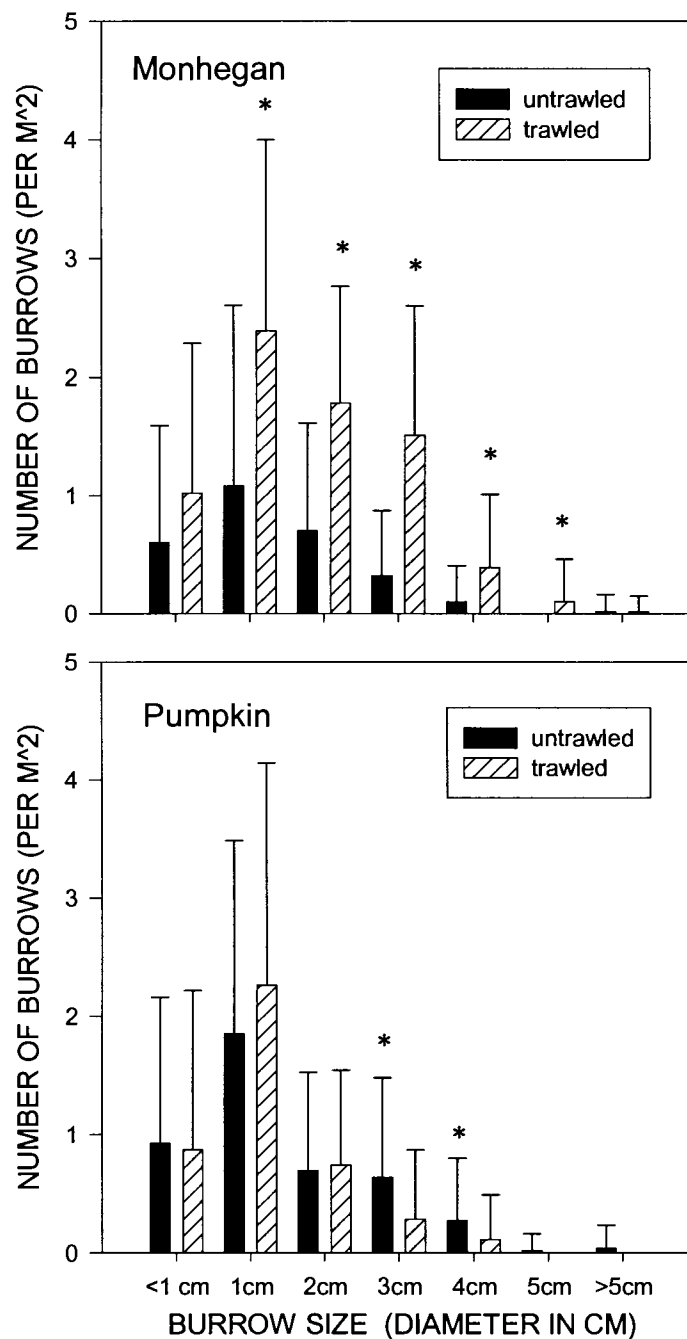


Figure 14. Mean surface burrow densities in trawled (diagonally striped bars) and untrawled (solid bars) and areas at Monhegan and Pumpkin. Burrows grouped by diameter in 1 cm size classes. Error bars represent ± 1 standard deviation from the mean. Asterisks (*) indicate significant differences ($p < 0.05$) between trawled and untrawled areas (Mann-Whitney test).

Porosity

Samples for sediment porosity measurements were collected on all sampling dates; however, due to a sampling processing error which reduced data quality, only data for the sampling periods, August 2001 and December 2001, whose quality could be assured, are presented here.

Porosity decreased significantly with depth in sediment ($p < 0.05$) at both study sites (Table 3; Figs. 15 & 16). Analysis of individual 1 cm depth layers from 0-10 cm, showed significant differences in porosity (three-factor ANOVA) between Monhegan and Pumpkin at all depths; however, no significant differences between untrawled and trawled areas at either site were detected (Table 3).

Excess ^{210}Pb Activity

Excess ^{210}Pb activity profiles from untrawled and trawled areas at Monhegan revealed a relatively deep mixed layer (~16-17 cm) in both areas (Fig. 17). Sediment inventory of excess ^{210}Pb activity was higher (3.311 Bq/g) in the trawled area compared with the untrawled area (2.909 Bq/g). The elevated inventory in the trawled area was due in part to consistently higher excess ^{210}Pb activity, especially in a 6 cm thick layer from 11-17 cm depth in the sediment. Surface activity of this isotope was slightly higher in the untrawled area (0.233 Bq/g), but decreased with depth in sediment to activity values similar to those observed in the trawled area. There was a single sub-surface peak in ^{210}Pb activity in the 5-6 cm depth layer in the untrawled area profile; however, all other activity values in the upper 20 cm of the profile were lower than those in the trawled area.

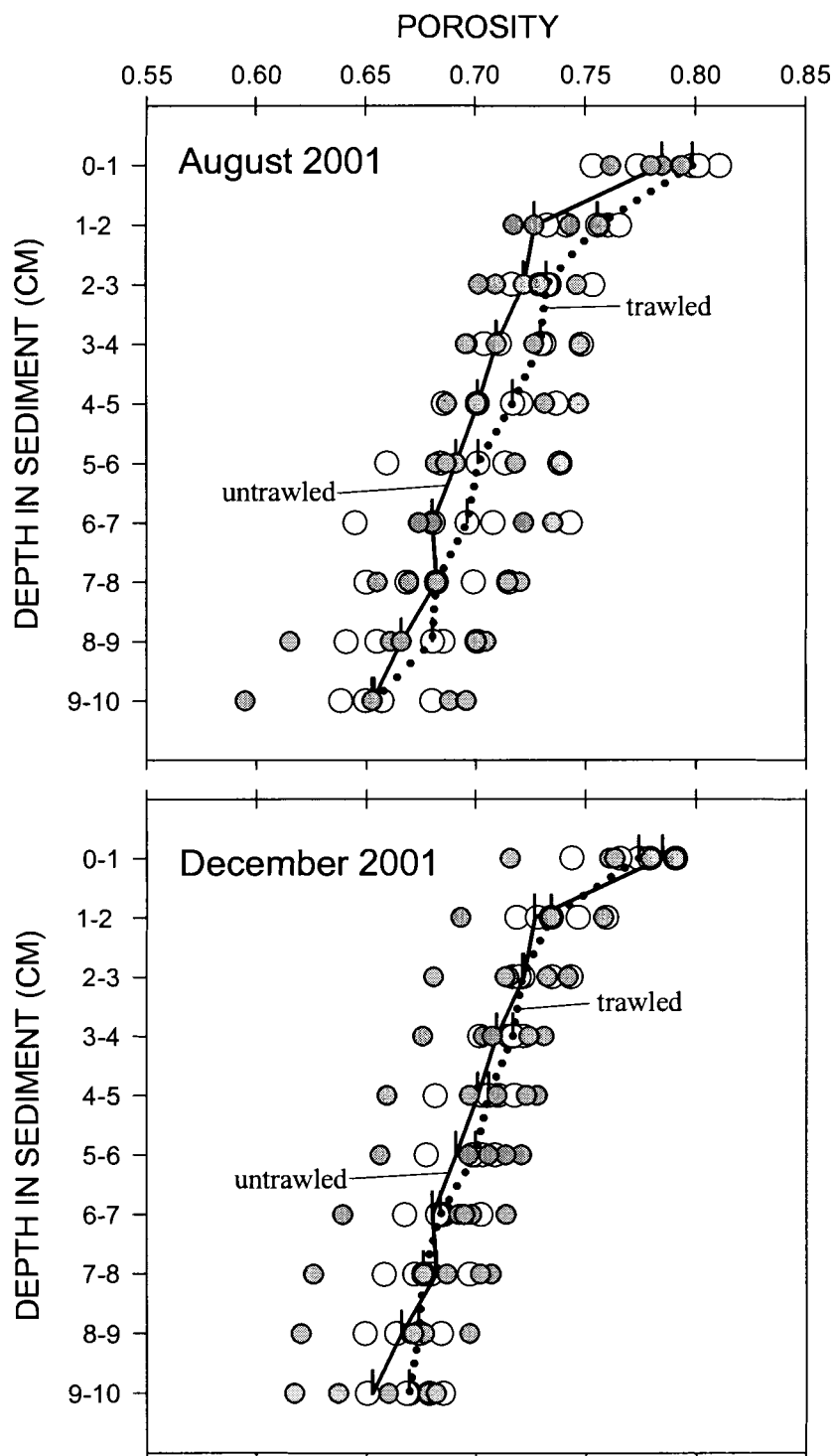


Figure 15. Monhegan porosity profiles from trawled (shaded circles) and untrawled (open circles) in August and December 2001. Data displayed as circles are replicates from separate box cores. Vertical bars represent median porosity values at each depth in trawled (dotted line) and untrawled areas (solid line).

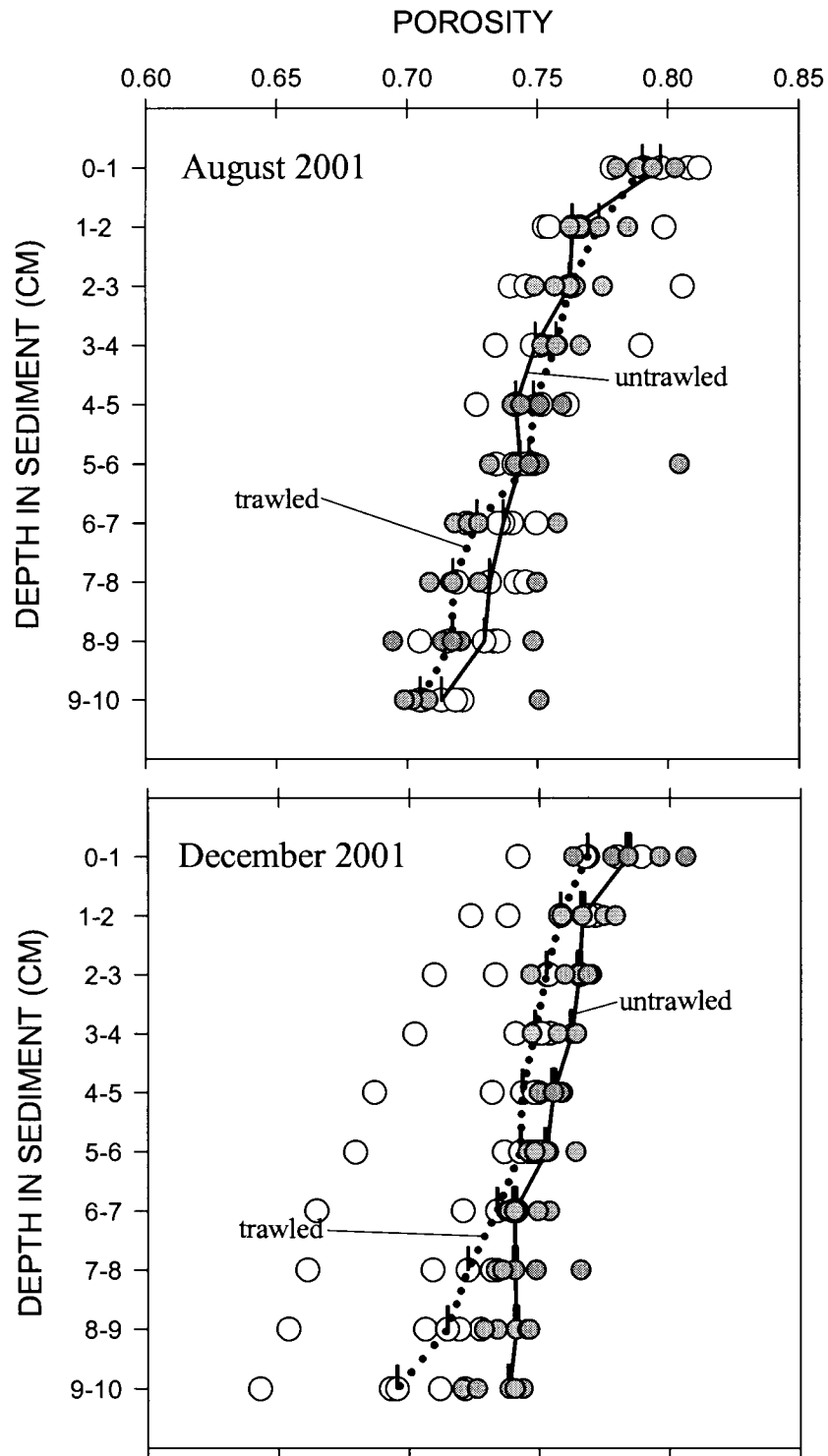


Figure 16. Pumpkin porosity profiles from trawled (shaded circles) and untrawled (open circles) areas in August and December 2001. Data displayed as circles are replicates from separate box cores. Vertical bars represent median porosity values at each depth in trawled (dotted line) and untrawled (solid line) areas.

Table 3. ANOVA results comparing sediment porosity in untrawled and trawled areas in August and December 2001. Three factors, sampling date (DATE), study site (SITE), and presence/absence of shrimp trawling (TREATMENT) were analyzed for each 1 cm depth increments from 0-10 cm. Bold p* values indicate significance at $\alpha = 0.05$.

| DEPTH IN SEDIMENT (CM) | SOURCE | SS | DF | MS | F | p |
|---------------------------|---------------------|-------|----|-------|--------|---------------|
| 0-1 | DATE | 0.003 | 1 | 0.003 | 9.506 | 0.004* |
| | SITE | 0.001 | 1 | 0.001 | 3.164 | 0.085 |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.143 | 0.708 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.024 | 0.878 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 0.079 | 0.781 |
| | SITE*TREATMENT | 0.001 | 1 | 0.001 | 2.378 | 0.133 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.349 | 0.559 |
| | ERROR | 0.011 | 32 | 0.000 | | |
| 1-2 | DATE | 0.001 | 1 | 0.001 | 2.016 | 0.165 |
| | SITE | 0.007 | 1 | 0.007 | 22.870 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.096 | 0.759 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.098 | 0.759 |
| | DATE*TREATMENT | 0.001 | 1 | 0.001 | 3.085 | 0.089 |
| | SITE*TREATMENT | 0.001 | 1 | 0.001 | 2.081 | 0.159 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.105 | 0.748 |
| | ERROR | 0.009 | 32 | 0.000 | | |
| 2-3 | DATE | 0.001 | 1 | 0.001 | 1.875 | 0.180 |
| | SITE | 0.011 | 1 | 0.011 | 34.664 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.002 | 0.965 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.177 | 0.676 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 0.686 | 0.424 |
| | SITE*TREATMENT | 0.001 | 1 | 0.001 | 3.677 | 0.064 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.558 | 0.461 |
| | ERROR | 0.010 | 32 | 0.000 | | |
| 3-4 | DATE | 0.001 | 1 | 0.001 | 2.093 | 0.158 |
| | SITE | 0.014 | 1 | 0.014 | 47.373 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.000 | 0.989 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.015 | 0.904 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 1.518 | 0.227 |
| | SITE*TREATMENT | 0.001 | 1 | 0.001 | 2.292 | 0.140 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.845 | 0.365 |
| | ERROR | 0.009 | 32 | 0.000 | | |
| 4-5 | DATE | 0.000 | 1 | 0.000 | 1.064 | 0.310 |
| | SITE | 0.013 | 1 | 0.013 | 37.712 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.689 | 0.413 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.243 | 0.625 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 1.135 | 0.295 |
| | SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.560 | 0.460 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 1.396 | 0.246 |
| | ERROR | 0.011 | 32 | 0.000 | | |

Table 3. (continued).

| DEPTH IN SEDIMENT (CM) | SOURCE | SS | DF | MS | F | p |
|---------------------------|---------------------|-------|----|-------|--------|---------------|
| 5-6 | DATE | 0.000 | 1 | 0.000 | 0.487 | 0.490 |
| | SITE | 0.021 | 1 | 0.021 | 41.445 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.263 | 0.612 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.068 | 0.796 |
| | DATE*TREATMENT | 0.001 | 1 | 0.001 | 1.248 | 0.272 |
| | SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.024 | 0.877 |
| | DATE*SITE*TREATMENT | 0.001 | 1 | 0.001 | 1.709 | 0.200 |
| | ERROR | 0.016 | 32 | | | |
| 6-7 | DATE | 0.000 | 1 | 0.000 | 0.581 | 0.451 |
| | SITE | 0.018 | 1 | 0.018 | 31.042 | 0.000* |
| | TREATMENT | 0.001 | 1 | 0.001 | 1.424 | 0.242 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.285 | 0.597 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 0.379 | 0.543 |
| | SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.643 | 0.428 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.305 | 0.585 |
| | ERROR | 0.018 | 32 | | | |
| 7-8 | DATE | 0.000 | 1 | 0.000 | 0.581 | 0.451 |
| | SITE | 0.018 | 1 | 0.018 | 31.042 | 0.000* |
| | TREATMENT | 0.001 | 1 | 0.001 | 1.424 | 0.242 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.285 | 0.597 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 0.379 | 0.543 |
| | SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.643 | 0.428 |
| | DATE*SITE*TREATMENT | 0.000 | 1 | 0.000 | 0.305 | 0.585 |
| | ERROR | 0.018 | 32 | 0.000 | | |
| 8-9 | DATE | 0.000 | 1 | 0.000 | 0.024 | 0.879 |
| | SITE | 0.027 | 1 | 0.027 | 49.732 | 0.000* |
| | TREATMENT | 0.001 | 1 | 0.001 | 1.392 | 0.247 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.048 | 0.829 |
| | DATE*TREATMENT | 0.001 | 1 | 0.001 | 1.108 | 0.300 |
| | SITE*TREATMENT | 0.001 | 1 | 0.001 | 2.300 | 0.139 |
| | DATE*SITE*TREATMENT | 0.001 | 1 | 0.001 | 0.978 | 0.330 |
| | ERROR | 0.017 | 32 | 0.001 | | |
| 9-10 | DATE | 0.000 | 1 | 0.000 | 0.142 | 0.709 |
| | SITE | 0.027 | 1 | 0.027 | 48.094 | 0.000* |
| | TREATMENT | 0.000 | 1 | 0.000 | 0.585 | 0.450 |
| | DATE*SITE | 0.000 | 1 | 0.000 | 0.173 | 0.681 |
| | DATE*TREATMENT | 0.000 | 1 | 0.000 | 0.769 | 0.388 |
| | SITE*TREATMENT | 0.002 | 1 | 0.002 | 0.299 | 0.094 |
| | DATE*SITE*TREATMENT | 0.002 | 1 | 0.002 | 3.642 | 0.066 |
| | ERROR | 0.017 | 30 | 0.001 | 0.001 | |

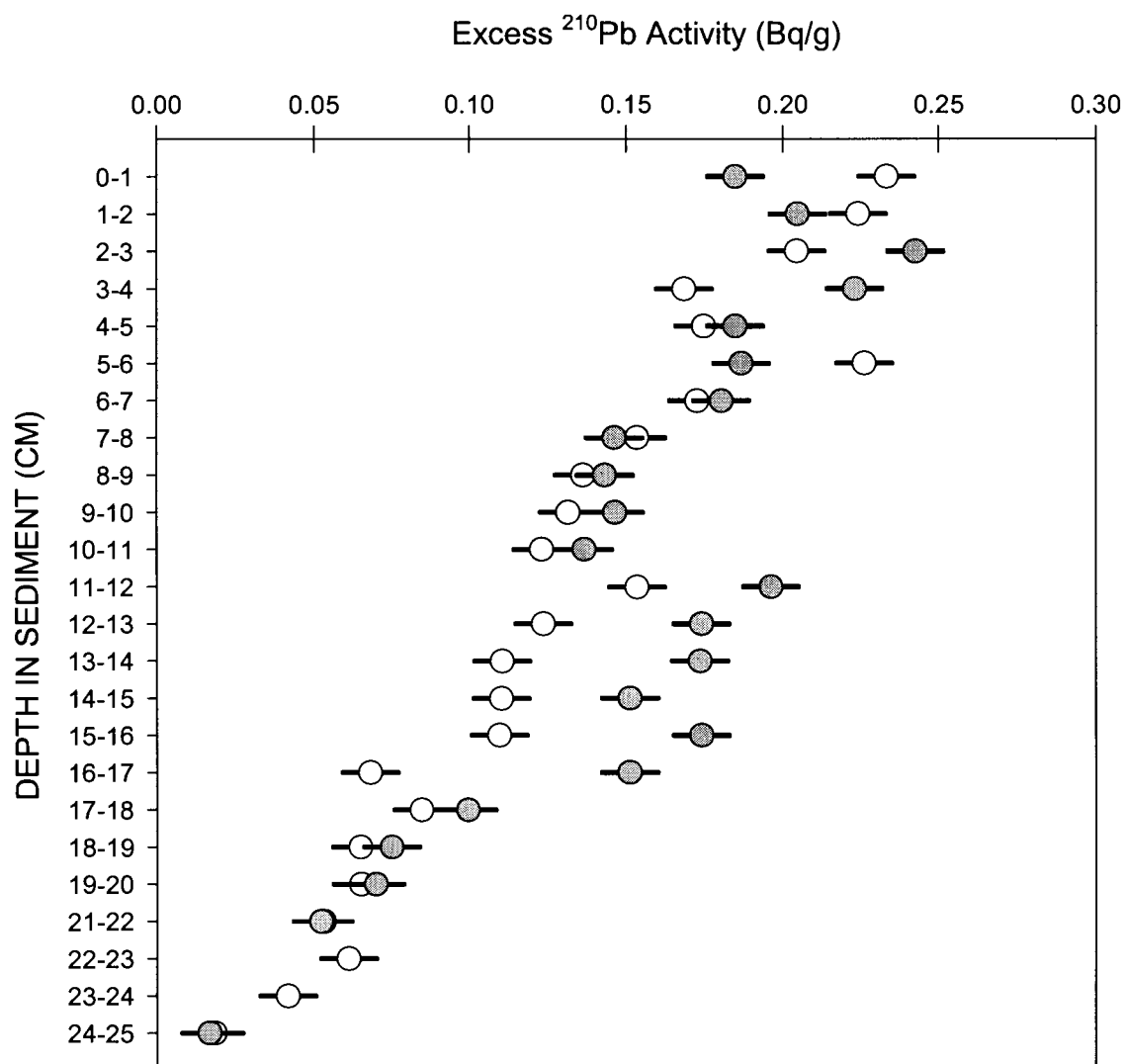


Figure 17. Excess ^{210}Pb activity profiles from trawled (shaded circles) and untrawled (open circles) areas at Monhegan. Error bars show total analytical error (± 1 SD).

Grain Surface Area

Using the surface area to grain size distribution relationship developed by Mayer and Rossi (1982), sediments in both study areas were classified as silt ($2\ \mu\text{m} - 62.5\ \mu\text{m}$). Sediment surface areas at Monhegan ranged from $27.6\text{--}21.8\ \text{m}^2/\text{g}$, $25.9\text{--}21.1\ \text{m}^2/\text{g}$, and $27.1\text{--}19.2\ \text{m}^2/\text{g}$ for the 0-1, 4-5, and 9-10 depth layers, respectively (Table 1). At Pumpkin, values ranged from $30.8\text{--}25.9\ \text{m}^2/\text{g}$, $31.1\text{--}24.4\ \text{m}^2/\text{g}$, and $28.1\text{--}20.7\ \text{m}^2/\text{g}$ for the 0-1, 4-5, and 9-10 depth layers, respectively. Grain surface area decreased slightly with depth in sediment, indicating a subtle coarsening with increased distance from the sediment surface. Grain surface areas, and therefore sediment grain size distributions were similar in trawled and untrawled areas with no significant differences (Mann-Whitney rank sum test) detected by depth layers at either study site.

Shrimp Trawling Impacts on Macrofaunal Community Structure

During the course of this study 14,224 individuals in 52 families and 8 higher taxonomic levels were collected. Annelids, represented by 24 polychaete families, were the most numerous and diverse group, making up from 61% - 66% of the community assemblage at Monhegan, and 75% - 81% of the community at Pumpkin (Fig. 18). The second most numerous phylum was molluscs, which was represented by five families and accounted for 27% - 29% of the community at Monhegan and 8% - 19% at Pumpkin. Echinoderms made up less than 8% of the community assemblage at all sites and were represented by only two families, Caudinidae and Ophiuridae. Arthropoda and Nemertea accounted for only 1% - 3% of the community assemblage at all sites.

The highest abundances of macrofauna residing in the upper 10 cm of sedimentary habitat were found in the 0-2 cm depth range (Fig. 19). Similarly, at Pumpkin 46% - 66% of the total abundance of macrofaunal was recorded in the upper 2 cm of sediment (Fig. 19).

Temporal patterns in total macrofaunal abundance in untrawled and trawled areas at each study site showed a seasonal peak in recruitment during the summer months (Fig. 20). Abundance of macrofauna was highest in at the untrawled site at Monhegan. Seasonal abundance peaks were lower at Pumpkin, but roughly equal in both untrawled and trawled areas at that site.

The average taxa richness was significantly higher ($p < 0.05$ Mann-Whitney rank sum test) at Monhegan compared to Pumpkin in all sampling periods from December 2000 – December 2001. No significant differences in the average taxa richness between untrawled and trawled areas were detected at either study site (Fig. 21).

Univariate measures of taxa diversity (Shannon-Wiener) and evenness (Pielou's) in untrawled and trawled areas were not significantly different at either site during the course of this study. Taxa diversity at Monhegan was significantly higher ($p < 0.05$, Mann-Whitney rank sum test) than at Pumpkin. Evenness did not differ significantly between sites (Fig. 23).

MDS ordination of pooled Monhegan and Pumpkin macrofaunal abundance data shows apparent differences in assemblage structure between the study sites (Fig. 23). Differences between sites were tested using a one-way ANOSIM and were found to be statistically significant ($p = 0.01$; Global $R = 0.52$). MDS ordination of the same data set shows no clear differences between benthic assemblages in untrawled and trawled areas

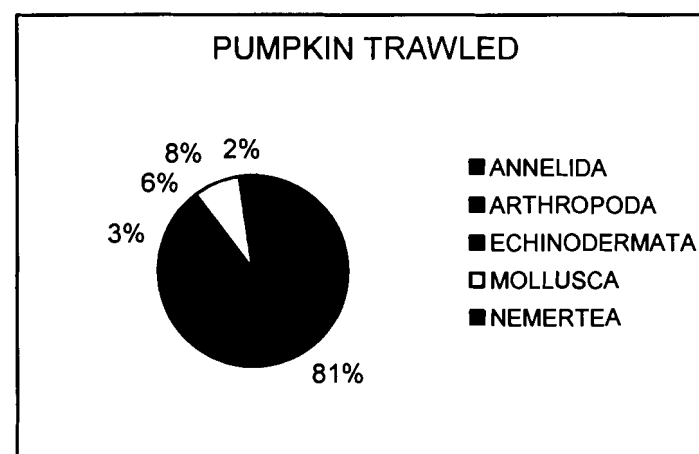
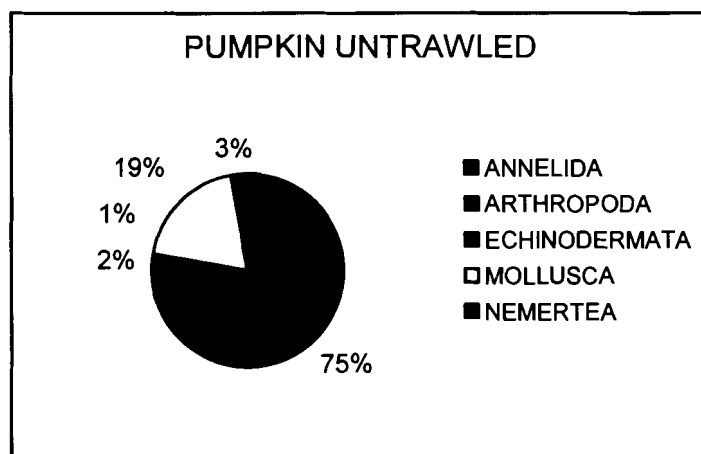
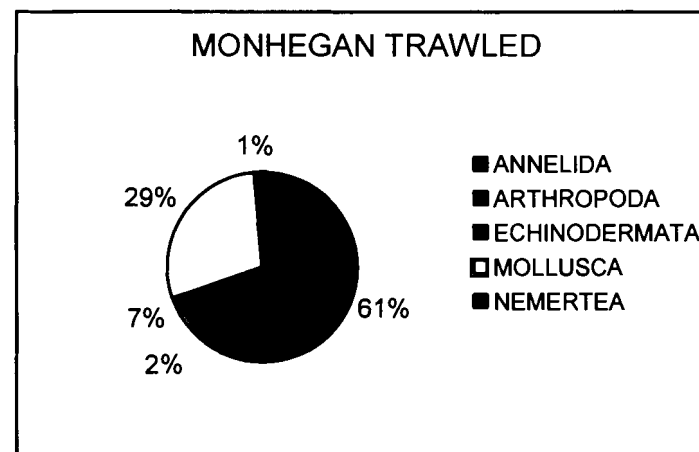
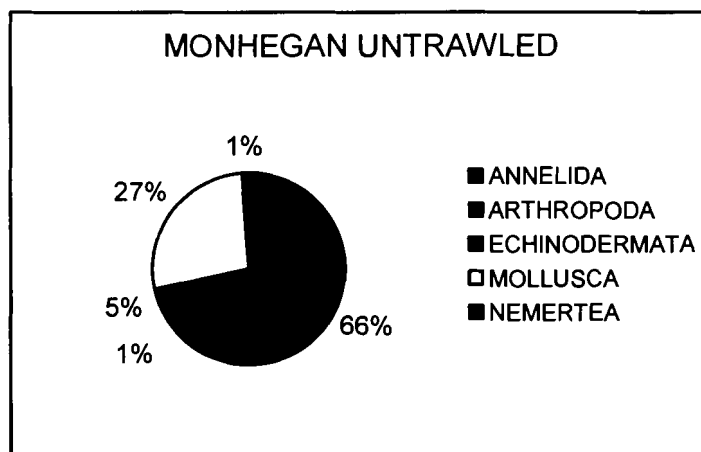


Figure 19. Percent composition of macrofaunal community at the level of phylum.

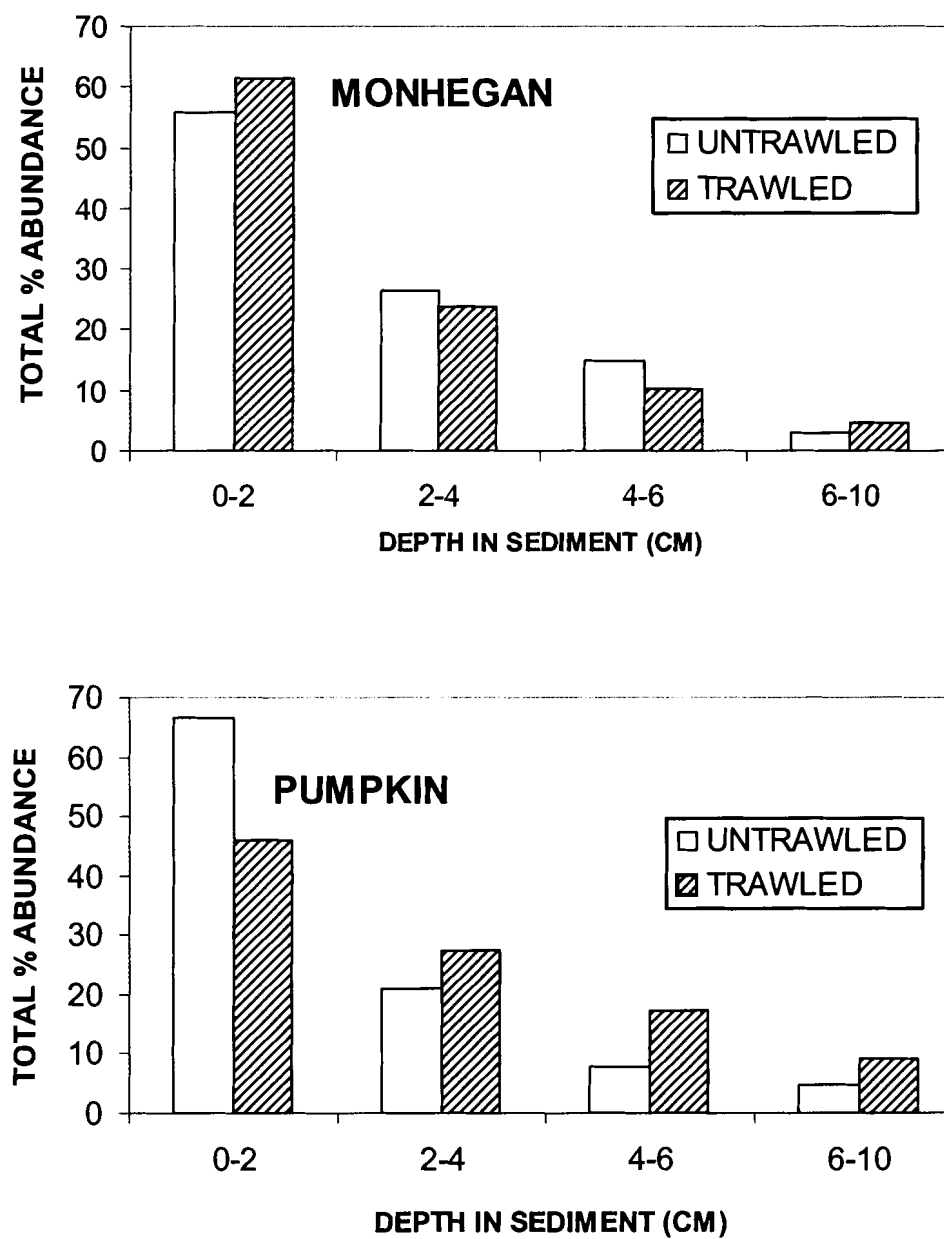


Figure 19. Percent abundance of macrofauna in the upper 10 cm of sediment in trawled and untrawled areas at Monhegan and Pumpkin.

(Fig. 24). Results of one-way ANOSIM supported this conclusion, with no significant differences detected overall between untrawled and trawled areas ($p = 0.32$; Global $R = 0.013$) when abundance data from both study sites was combined.

Macrofaunal assemblages show different responses to shrimp trawling in MDS ordination diagrams of each study areas. At Monhegan, there were no clear differences in community assemblage character between untrawled and trawled areas (Fig. 25). In contrast, there was a clear separation of community assemblages in untrawled and trawled areas at Pumpkin (Fig. 25). Statistically significant separation of treatments at Pumpkin was confirmed by the results of ANOSIM ($p=0.012$; Global $R=0.355$). No significant difference was detected between trawled and untrawled areas at Monhegan (ANOSIM $p=0.579$; Global $R= -0.032$).

SIMPER analysis was used to determine contributions of macrofaunal taxon abundance to the average Bray-Curtis dissimilarity between untrawled and trawled sites at Monhegan and Pumpkin depicted in the MDS ordination diagrams (Tables 4 & 5). The average dissimilarity between areas at Monhegan was lower (average dissimilarity = 29.70) then at Pumpkin (average dissimilarity = 35.00). Families contributing cumulatively to 50% of the average dissimilarity (Cossuridae, Paraonidae, Sabellidae, Nucleidae, Spionidae, Ampharetidae, Capitellidae) were always on average more numerous in the untrawled area. Conversely, only two (Spionidae and Nucleidae) of the eight families contributing cumulatively to >50% of the dissimilarity between areas at Pumpkin were on average more abundant in the untrawled area; the remaining six families (Paraonidae, Cossuridae, Ampharetidae, Sabellidae, Phoxocephalidae, Cirratulidae) were all on average more numerous in the trawled area. Relationships

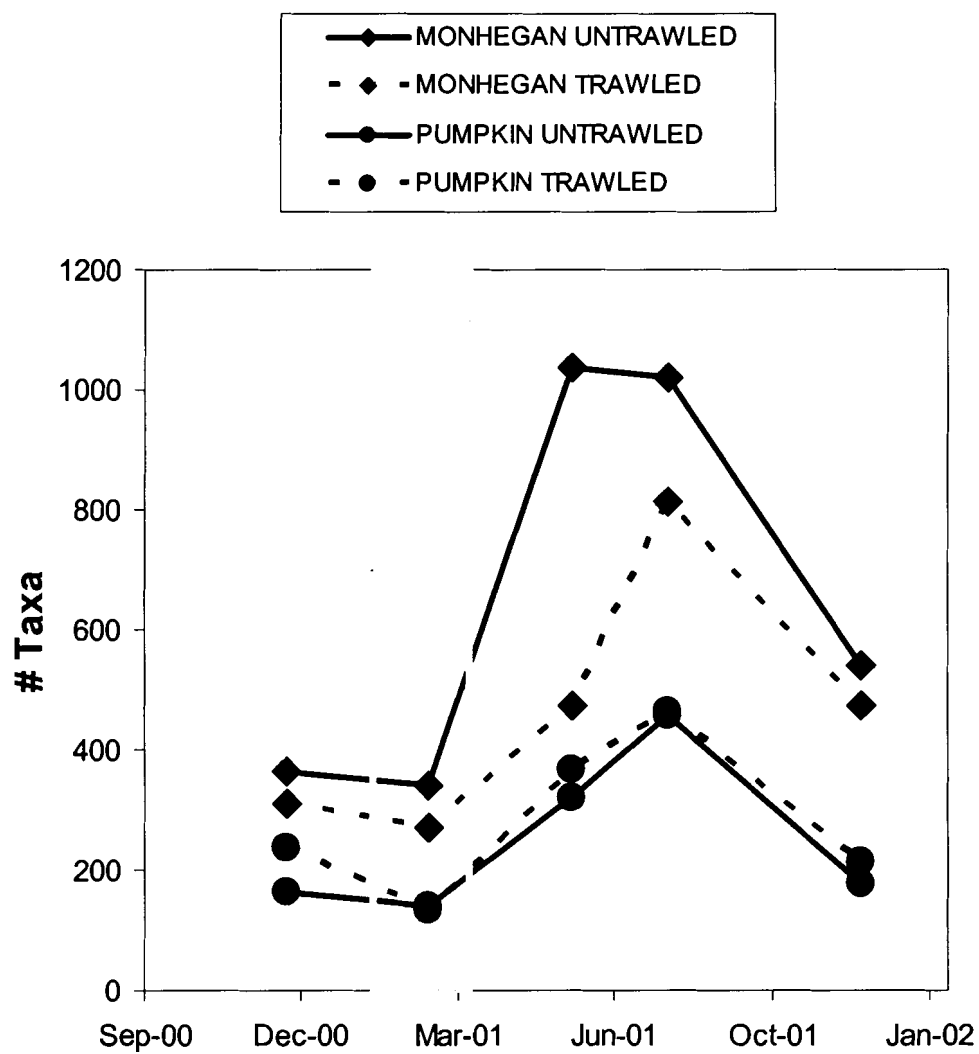


Figure 20. Temporal trends in macrofaunal abundance in trawled and untrawled areas at Monhegan and Pumpkin.

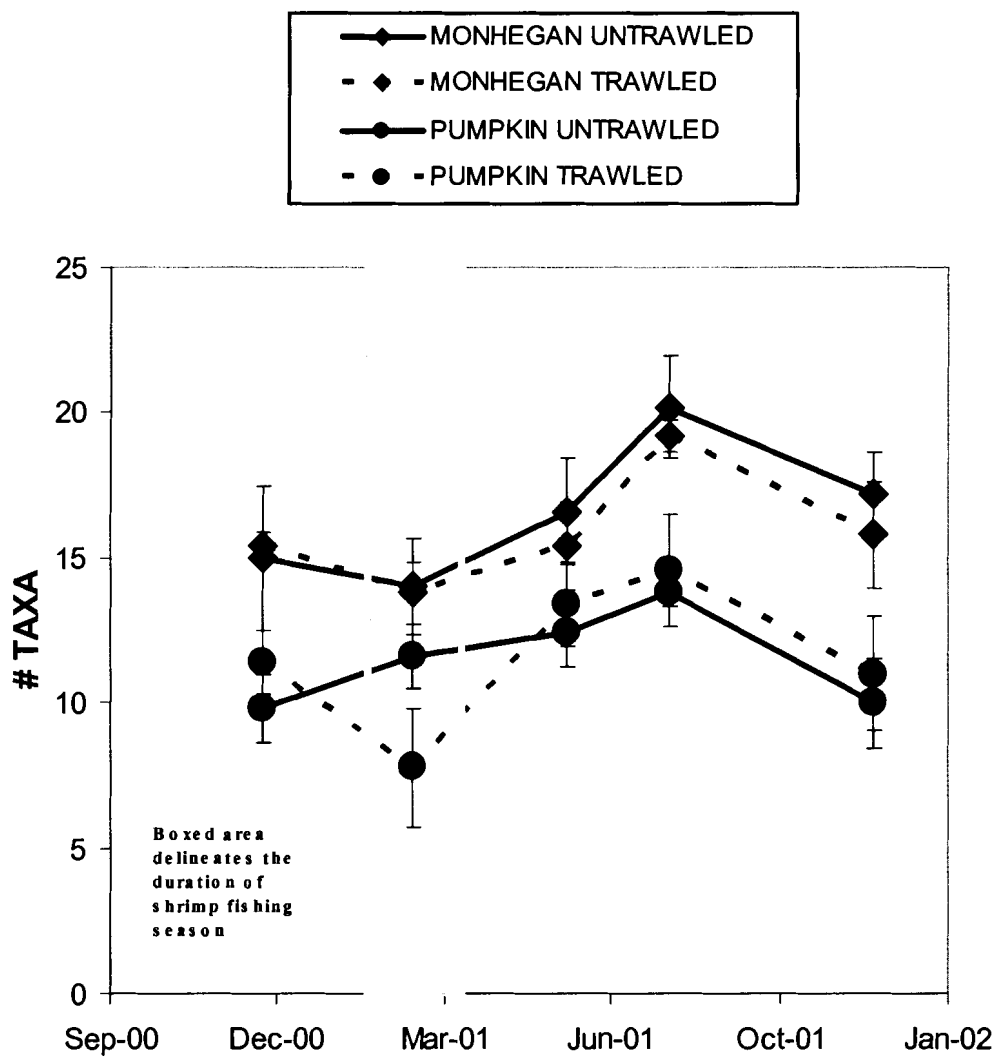


Figure 21. Temporal trends in the average number of taxa in trawled and untrawled areas at Monhegan and Pumpkin.

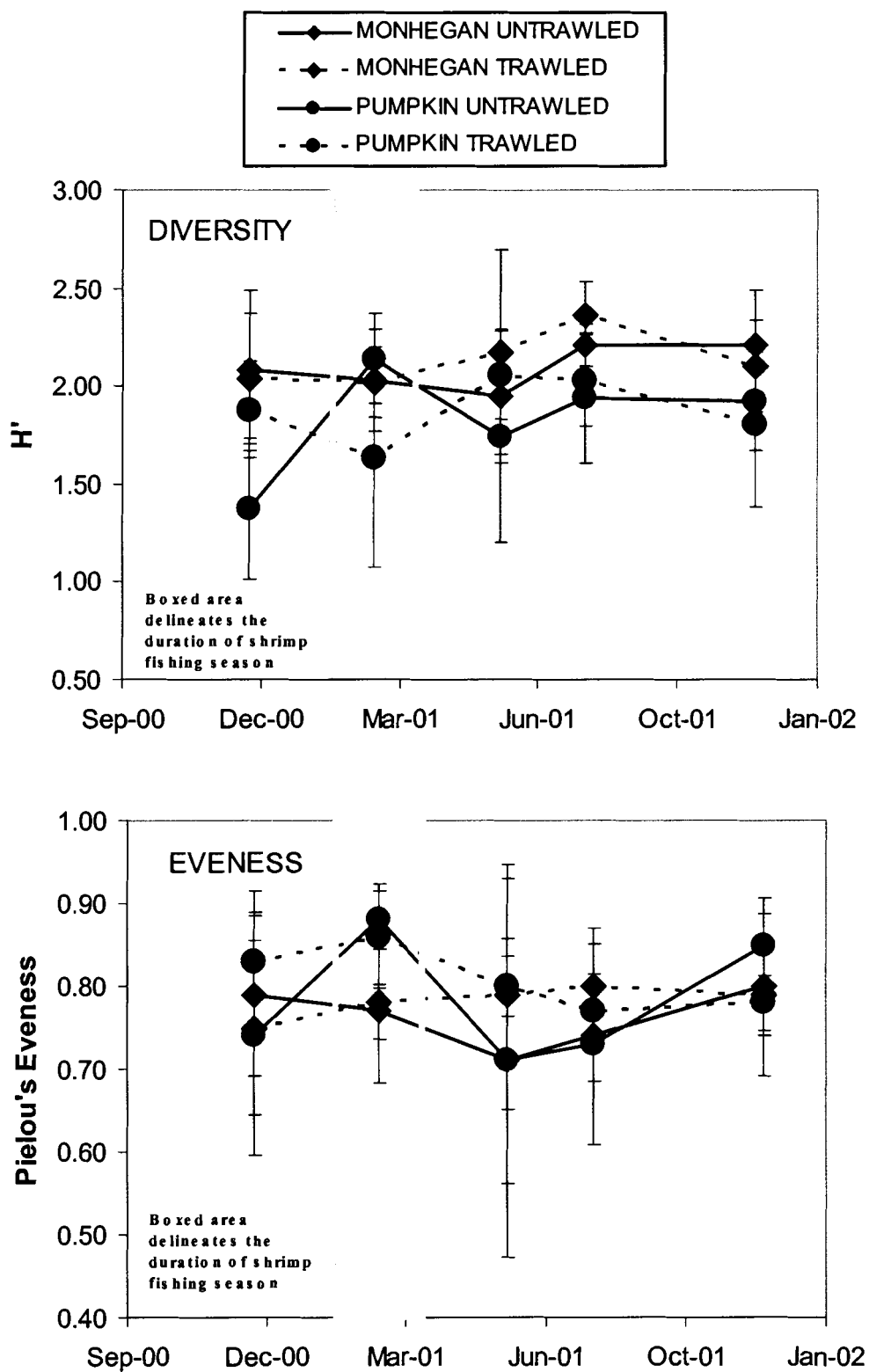


Figure 22. Temporal trends in taxa diversity and evenness in trawled and untrawled areas at Monhegan and Pumpkin. Errors bars show ± 1 SD.

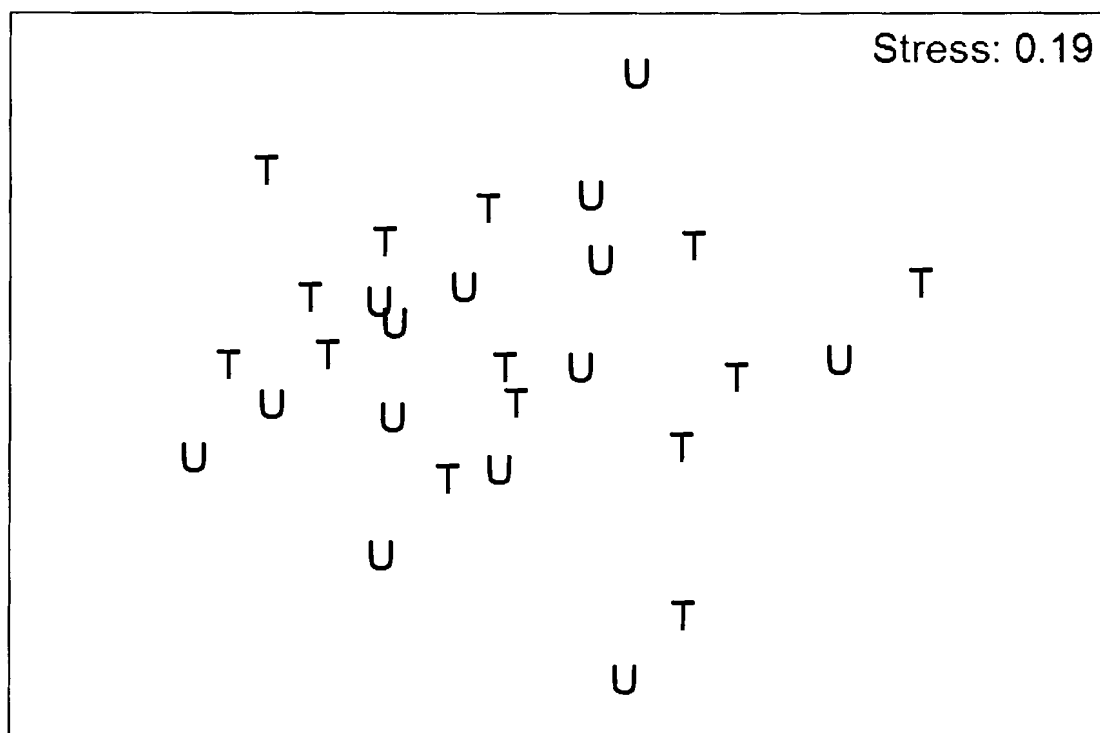


Figure 24. Two-dimensional MDS ordination plot of pooled macrofaunal abundance data from trawled (T) and untrawled (U) areas.

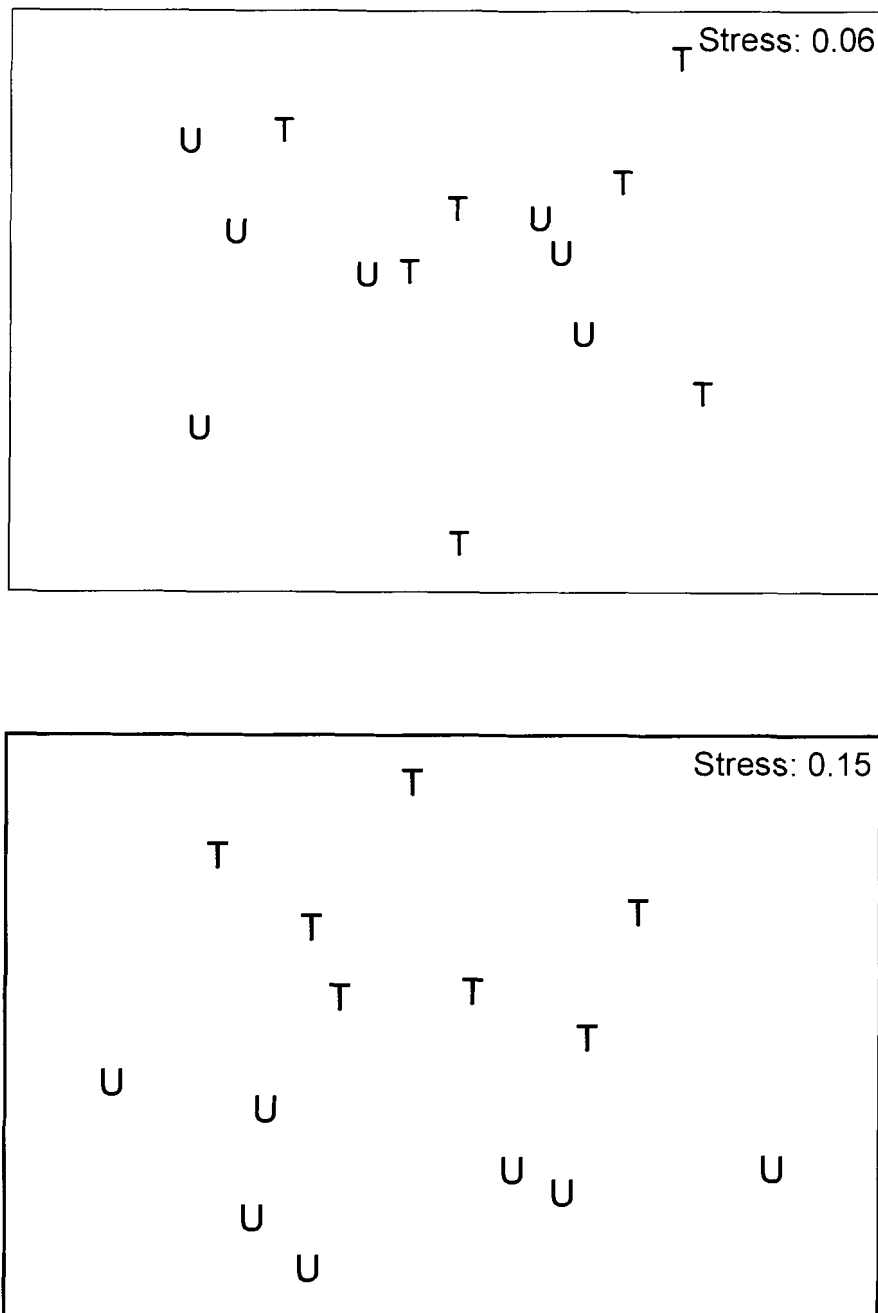


Figure 25. Two-dimensional MDS ordination plots of macrofaunal abundance data trawled (T) and untrawled (U) areas in each fishing ground study area (Monhegan and Pumpkin). Results of ANOSIM between community assemblages in trawled and untrawled areas show no significant differences (global $R = -0.032$ and $p = 0.579$) in assemblages at Monhegan; and a significant difference between assemblages at Pumpkin (global $R = 0.355$ and $p = 0.012$.)

between macrofaunal assemblage variability and environmental factors (explanatory variables, CANOCO 4.5, ter Braak 1990) including depth in sediment (0-2 cm, 2-4 cm, 4-6 cm, 6-10 cm) and the presence/absence of trawling activity, were explored using RDA analysis. Forward selection with Monte Carlo permutation tests (499 permutations) identified environmental factors in order of significance according to the amount of variance explained. At Monhegan the order of environmental variables was: 0-2 cm, 6-10 cm, 4-6 cm, UNTRAWL, TRAWL, and 2-4 cm. Together these 6 variables accounted for 54.3% of the total variability present in the macrofaunal abundance data set at this study site (Table 6). Only 5 out of the 6 environmental factors showed a significant relationship to macrofaunal abundance. The percent variance explained by each of these significant environmental variables when examined individually (marginal effects) was greatest for 0-2 cm depth range (40.4%), followed by 6-10 cm depth range (16.1%), 4-6 cm depth range (7.7%). The absence of trawling activity (3.7%) and presence of trawling activity (3.7%) accounted for only a minimal amount of observed variability in the macrofaunal community assemblage. At Pumpkin, (site 2), the order of environmental variables was: 0-2, 6-10, 4-6, UNTRAWL, TRAWL, and 2-4. Together these 6 variables accounted for 50.1% of the total variability present in the macrofaunal abundance data at this study site (Table 6). A Monte Carlo permutation test showed that there was a statistically significant relationship between only the first 4 of the 6 environmental factors. The percent variance explained by each of these significant environmental variables when examined individually (marginal effects) was greatest for 0-2 cm depth range (36.2%), followed by 6-10 cm depth range (14.8%), 4-6 cm depth range (8.2%). The absence of trawling activity (4.8%) and the presence of trawling activity (4.8%)

Table 4. Results of SIMPER analysis of macrofaunal abundance data for Monhegan. Average dissimilarity = 29.70 between trawled and untrawled areas.

| Taxa | Group T Av.Abund | Group U Av.Abund | Av.Diss | Diss/SD | Contrib% | Cum.% |
|----------------------|---------------------|---------------------|---------|---------|----------|--------|
| Cossuridae | 38.86 | 127.29 | 3.07 | 1.28 | 10.34 | 10.34 |
| Paraonidae | 24.86 | 64.29 | 2.25 | 1.39 | 7.56 | 17.90 |
| Sabellidae | 24.57 | 40.29 | 1.94 | 1.45 | 6.54 | 24.44 |
| Nuculidae | 98.71 | 119.57 | 1.89 | 1.19 | 6.36 | 30.80 |
| Spionidae | 17.29 | 49.00 | 1.78 | 1.26 | 5.99 | 36.79 |
| Ampharetidae | 56.71 | 83.86 | 1.74 | 1.31 | 5.86 | 42.66 |
| Capitellidae | 16.00 | 25.00 | 1.43 | 1.40 | 4.81 | 47.46 |
| Cirratulidae | 39.43 | 46.00 | 0.95 | 1.59 | 3.19 | 50.66 |
| Maldanidae | 11.29 | 4.14 | 0.87 | 1.27 | 2.94 | 53.60 |
| Nuculanidae | 8.43 | 7.00 | 0.81 | 1.34 | 2.73 | 56.33 |
| Ophiuridae | 5.00 | 4.57 | 0.76 | 1.31 | 2.54 | 58.87 |
| Scalibregmidae | 1.14 | 2.57 | 0.61 | 1.05 | 2.04 | 60.91 |
| Trichnobranchidae | 3.71 | 7.00 | 0.58 | 1.29 | 1.95 | 62.87 |
| Class Scaphopoda | 2.57 | 2.29 | 0.57 | 1.38 | 1.92 | 64.79 |
| Astartidae | 0.71 | 3.29 | 0.55 | 1.00 | 1.84 | 66.62 |
| Periplomatidae | 5.00 | 4.86 | 0.54 | 1.11 | 1.80 | 68.43 |
| Rissoidae | 2.00 | 0.57 | 0.53 | 1.27 | 1.80 | 70.22 |
| Lumbrineridae | 10.86 | 7.00 | 0.51 | 1.59 | 1.72 | 71.94 |
| Phylum Nemertea | 5.29 | 6.29 | 0.49 | 1.29 | 1.65 | 73.58 |
| Nephtyidae | 6.57 | 5.71 | 0.47 | 1.24 | 1.58 | 75.17 |
| Sternaspidae | 1.43 | 1.57 | 0.46 | 1.09 | 1.57 | 76.73 |
| Syllidae | 0.57 | 1.00 | 0.45 | 1.15 | 1.50 | 78.24 |
| Class Copepoda | 1.00 | 1.43 | 0.44 | 1.21 | 1.48 | 79.71 |
| Cardiidae | 1.00 | 1.00 | 0.41 | 1.26 | 1.39 | 81.10 |
| Leuconidae | 1.14 | 1.71 | 0.40 | 1.17 | 1.34 | 82.44 |
| Flabelligeridae | 0.86 | 0.29 | 0.40 | 1.14 | 1.34 | 83.78 |
| Goniadidae | 2.57 | 2.00 | 0.39 | 1.15 | 1.30 | 85.08 |
| Class Ostracoda | 1.00 | 0.43 | 0.38 | 1.16 | 1.29 | 86.37 |
| Diastylidae | 0.29 | 0.57 | 0.36 | 1.16 | 1.21 | 87.58 |
| Polynoidae | 1.00 | 1.43 | 0.35 | 1.11 | 1.19 | 88.77 |
| Chaetopteridae | 0.71 | 0.43 | 0.35 | 0.96 | 1.18 | 89.95 |
| Cheirocratidae | 1.29 | 0.14 | 0.31 | 0.65 | 1.06 | 91.01 |
| Class Aplacophora | 0.14 | 0.71 | 0.31 | 0.90 | 1.04 | 92.05 |
| Phoxocephalidae | 0.00 | 0.71 | 0.30 | 0.81 | 1.01 | 93.06 |
| Nannastacidae | 0.43 | 0.43 | 0.30 | 0.71 | 1.01 | 94.06 |
| Orbiniidae | 0.43 | 0.29 | 0.27 | 0.72 | 0.92 | 94.98 |
| Oedicerotidae | 0.71 | 0.14 | 0.27 | 0.54 | 0.89 | 95.87 |
| Opheliidae | 0.14 | 0.43 | 0.21 | 0.72 | 0.71 | 96.58 |
| Order Isopoda | 0.00 | 0.43 | 0.17 | 0.60 | 0.58 | 97.16 |
| Class Cephalocaridae | 0.14 | 0.14 | 0.17 | 0.56 | 0.56 | 97.72 |
| Phyllodocidae | 0.29 | 0.00 | 0.13 | 0.40 | 0.43 | 98.14 |
| Pandalidae | 0.29 | 0.00 | 0.11 | 0.40 | 0.37 | 98.52 |
| Sigalionidae | 0.14 | 0.00 | 0.08 | 0.40 | 0.26 | 98.78 |
| Corophiidae | 0.14 | 0.00 | 0.08 | 0.40 | 0.26 | 99.05 |
| Crangonidae | 0.14 | 0.00 | 0.08 | 0.40 | 0.26 | 99.31 |
| Pleustidae | 0.14 | 0.00 | 0.07 | 0.40 | 0.23 | 99.54 |
| Photidae | 0.00 | 0.14 | 0.07 | 0.40 | 0.23 | 99.77 |
| Class Arachnida | 0.00 | 0.14 | 0.07 | 0.40 | 0.23 | 100.00 |

Table 5. Results of SIMPER analysis of macrofaunal abundance data for Pumpkin. Average dissimilarity = 35.00 between trawled and untrawled areas.

| Species | Group T Av.Abund | Group U Av.Abund | Av.Diss | Diss/SD | Contrib% | Cum.% |
|-------------------|---------------------|---------------------|---------|---------|----------|--------|
| Spionidae | 50.00 | 93.86 | 3.57 | 1.28 | 10.20 | 10.20 |
| Paraonidae | 48.14 | 38.57 | 2.65 | 1.45 | 7.58 | 17.78 |
| Cossuridae | 30.86 | 8.29 | 2.54 | 1.62 | 7.26 | 25.04 |
| Ampharetidae | 39.71 | 19.57 | 2.27 | 1.82 | 6.47 | 31.51 |
| Sabellidae | 12.43 | 9.71 | 2.08 | 1.40 | 5.93 | 37.44 |
| Nuculidae | 8.14 | 20.71 | 1.86 | 1.54 | 5.30 | 42.74 |
| Phoxocephalidae | 4.43 | 0.86 | 1.29 | 1.75 | 3.69 | 46.43 |
| Cirratulidae | 29.86 | 19.57 | 1.16 | 1.68 | 3.32 | 49.75 |
| Trichnobranchidae | 3.86 | 1.57 | 1.12 | 1.29 | 3.21 | 52.96 |
| Nuculanidae | 3.71 | 4.71 | 1.10 | 1.45 | 3.15 | 56.11 |
| Orbiniidae | 1.29 | 3.00 | 1.03 | 1.27 | 2.93 | 59.04 |
| Capitellidae | 2.14 | 2.14 | 1.00 | 1.19 | 2.86 | 61.90 |
| Maldanidae | 2.14 | 5.43 | 0.98 | 1.41 | 2.80 | 64.69 |
| Nephtyidae | 2.86 | 1.57 | 0.86 | 1.27 | 2.46 | 67.16 |
| Ophiuridae | 1.57 | 0.29 | 0.84 | 1.37 | 2.40 | 69.56 |
| Phylum Nemertea | 4.43 | 6.71 | 0.80 | 0.91 | 2.29 | 71.85 |
| Class Ostracoda | 0.00 | 0.86 | 0.76 | 2.25 | 2.16 | 74.01 |
| Goniadidae | 1.86 | 1.71 | 0.74 | 1.24 | 2.10 | 76.11 |
| Polynoidae | 0.29 | 1.29 | 0.73 | 1.04 | 2.09 | 78.20 |
| Syllidae | 0.57 | 1.43 | 0.71 | 1.44 | 2.02 | 80.22 |
| Periplomatidae | 0.43 | 0.71 | 0.68 | 1.57 | 1.95 | 82.17 |
| Leuconidae | 1.00 | 1.14 | 0.65 | 1.22 | 1.86 | 84.03 |
| Lumbrineridae | 4.00 | 6.71 | 0.61 | 1.35 | 1.75 | 85.78 |
| Cardiidae | 0.71 | 0.71 | 0.59 | 1.11 | 1.67 | 87.45 |
| Class Copepoda | 1.14 | 0.71 | 0.54 | 1.07 | 1.55 | 89.00 |
| Sternaspidae | 0.43 | 0.57 | 0.53 | 0.95 | 1.50 | 90.51 |
| Rissoidae | 0.14 | 0.57 | 0.49 | 1.07 | 1.39 | 91.89 |
| Class Scaphopoda | 0.71 | 0.29 | 0.47 | 0.83 | 1.35 | 93.24 |
| Trochochaetidae | 0.86 | 0.00 | 0.32 | 0.40 | 0.92 | 94.16 |
| Ampeliscidae | 0.00 | 0.57 | 0.31 | 0.61 | 0.90 | 95.06 |
| Flabelligeridae | 0.29 | 0.14 | 0.31 | 0.70 | 0.89 | 95.94 |
| Opheliidae | 0.14 | 0.14 | 0.24 | 0.56 | 0.68 | 96.62 |
| Scalibregmidae | 0.14 | 0.29 | 0.23 | 0.57 | 0.65 | 97.27 |
| Class Aplacophora | 0.14 | 0.14 | 0.21 | 0.55 | 0.60 | 97.87 |
| Order Acarina | 0.00 | 0.14 | 0.14 | 0.40 | 0.40 | 98.27 |
| Order Isopoda | 0.14 | 0.00 | 0.14 | 0.40 | 0.39 | 98.66 |
| Diastylidae | 0.00 | 0.14 | 0.13 | 0.40 | 0.38 | 99.04 |
| Phyllodocidae | 0.00 | 0.14 | 0.13 | 0.40 | 0.36 | 99.41 |
| Oedicerotidae | 0.00 | 0.14 | 0.10 | 0.40 | 0.30 | 99.70 |
| Pandalidae | 0.14 | 0.00 | 0.10 | 0.40 | 0.30 | 100.00 |

accounted again for a minimal amount of the explained variability in the macrofaunal data set.

Patterns in macrofaunal assemblage variability revealed similar responses to measured environmental variables at both study sites (Figs. 26 & 27). Macrofaunal abundance for nearly all taxa (Ampharetidae, Capitellidae, Maldanidae, Nucleidae, Nucleanidae, Paraonidae, Phylum Nemertea, Sabellidae, Spionidae, and Trichobranchidae) was positively related to 0-2 cm sediment depth layer at Monhegan. Two families, Cirratulidae and Lumbrineridae, were positively correlated with an absence of trawling activity while the family Cossuridae exhibited a strong positive relationship with 2-4 cm depth range (Fig. 25). An RDA ordination diagram of Pumpkin macrofaunal abundance data (Fig. 26) displays patterns similar to those observed at Monhegan, with strong positive correlations between most taxa and the 0-2 cm depth range and Lumbrineridae showing a weak but positive relationship with the absence of trawling activity; however a few taxa exhibit responses that differ from those observed in the previous analysis. Annelid polychaetes in the families Cirratulidae and Cossuridae appear to be more positively correlated with slightly deeper sediment layers at Pumpkin compared to Monhegan. Certain families including Nucleidae, Nucleanidae, and Spionidae, appear to be more strongly correlated with 0-2 depth range at Pumpkin than at Monhegan.

Before-After Control-Impact Experiment (BACI Experiment)

Sediment Density Structure

Sediment density, as measured by x-radiograph gray level values, increased

Table 6. Results of RDA ordination analysis of macrofaunal abundance data.

| Axes | MONHEGAN | | | | | PUMPKIN | | | | |
|----------------------------------|----------|--------|--------|---------|----------------|---------|--------|--------|---------|----------------|
| | 0-2 | 2-4 | 4-6 | 6-10 | Total variance | 0-2 | 2-4 | 4-6 | 6-10 | Total variance |
| Eigenvalues | 0.487 | 0.063 | 0.013 | 0.001 | 1.000 | 0.431 | 0.054 | 0.015 | 0.001 | 1.000 |
| Species-environment correlations | 0.906 | 0.600 | 0.577 | 0.196 | | 0.847 | 0.710 | 0.576 | 0.291 | |
| Cumulative percentage variance | | | | | | | | | | |
| of species data | 48.700 | 55.000 | 56.300 | 56.400 | | 43.100 | 48.500 | 50.000 | 50.100 | |
| of species-environment relation | 86.400 | 97.500 | 99.900 | 100.000 | | 86.000 | 96.800 | 99.700 | 100.000 | |
| Sum of all eigenvalues | | | | | 1.000 | | | | | 1.000 |
| Sum of all canonical eigenvalues | | | | | 0.564 | | | | | 0.501 |

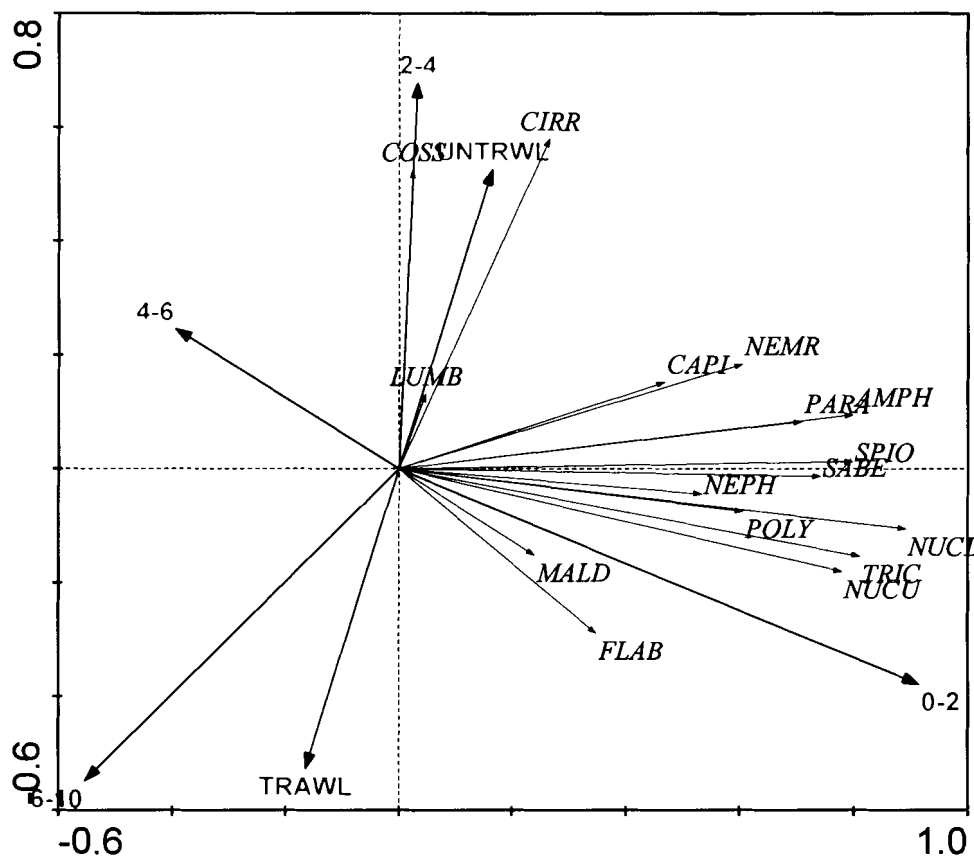


Figure 26. Redundancy analysis (RDA) ordination diagram for taxa abundance data at Site 1 - Monhegan. Only taxa that account for >1% total abundance were included in this analysis. For results on RDA analysis see Table X. Individual taxa and environmental variables (depth in sediment and presence of trawling activity) are shown as vectors. Length of vector arrows represent relative importance of variable in representing variation in community assemblage data set. Direction of vector arrow indicates relative correlation of variable to ordination axes and other vectors. Taxa are identified by first four letters of family name with the following exceptions: (1) members of the phylum Nemertea are identified as NEMR and (2) members of the family Nuculidae are identified as NUCL. The following families were included in this analysis: Ampharetidae, Capitellidae, Cirratulidae, Cossuridae, Flabelligeridae, Lumberineridae, Maldanidae, Nephtyidae, Nuculanidae, Nuculidae, Polynoidae, Sabellidae, Spionidae, Trichnbranchidae.

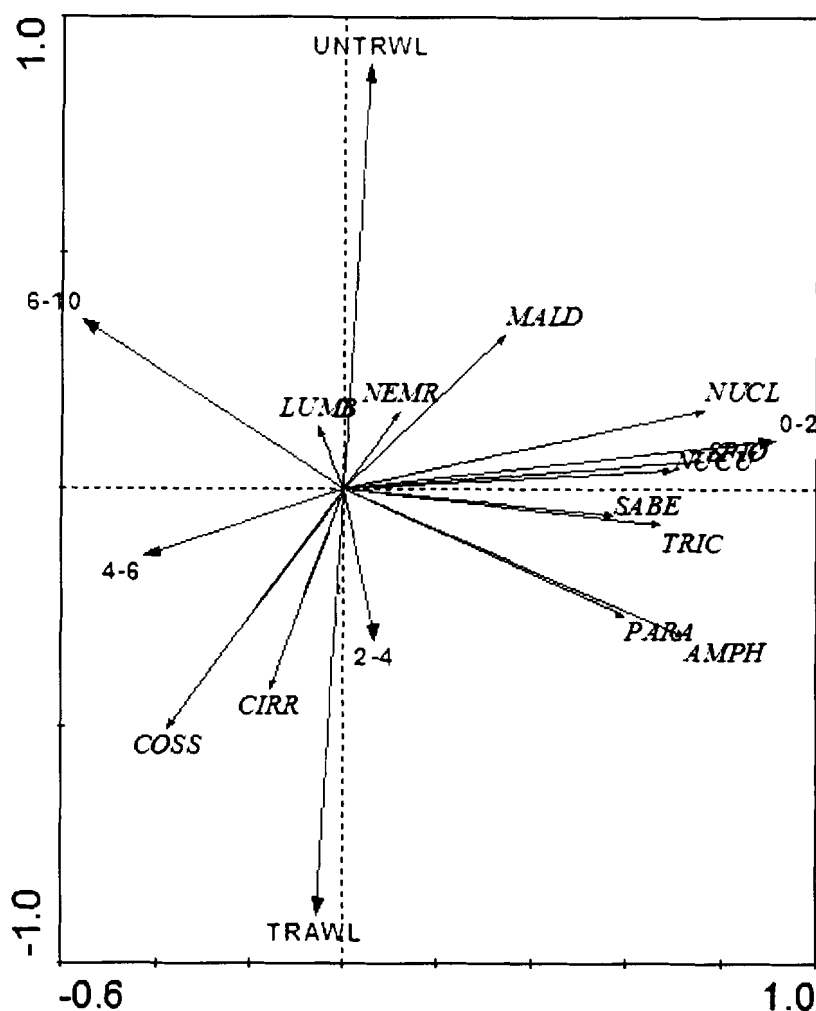


Figure 27. Redundancy analysis (RDA) ordination diagram for taxa abundance data at Pumpkin. Only taxa that account for >1% total abundance were included in this analysis. For results on RDA analysis see Table X. Individual taxa and environmental variables (depth in sediment and presence of trawling activity) are shown as vectors. Length of vector arrows represent relative importance of variable in representing variation in community assemblage data set. Direction of vector arrow indicates relative correlation of variable to ordination axes and other vectors. Taxa are identified by first four letters of family name with the following exceptions: (1) members of the phylum Nemertea are identified as NEMR; (2) members of the family Nuculidae are identified as NUCL. The following families were included in this analysis: Ampharetidae, Cirratulidae, Cossuridae, Lumberineridae, Maldanidae, Nuculanidae, Nuculidae, Paraonidae, Sabellidae, Spionidae, Trichnbranchidae.

significantly ($p < 0.05$, Mann-Whitney) with depth layer (cm) from 0-10 cm before and after experimental shrimp trawling. No significant differences in gray level values were detected between like depth layers in control or treatment areas before and after trawling (Table 7). Median gray level values were similar in control and treatment areas before experimental shrimp trawling occurred in the latter (Fig. 28). Following the disturbance, median gray level values in the control area concomitantly decreased and exhibited a wider range relative to the treatment area (Fig. 28); however, comparisons between treatments are limited because data from only two cores is available from the control area due to an exposure error in the x-radiograph from the third core. One of the two cores measured from the control area after trawling exhibited notably lower overall sediment density structure relative to cores collected before experimental shrimp trawling. The relatively low gray level values captured in this core likely result from natural heterogeneity in sediment density structure possibly from localized physical or biological structuring processes.

Porosity

Porosity decreased significantly with depth in sediment (cm) from 0-10 cm ($p < 0.05$, Mann-Whitney) in both treatment areas before and after experimental shrimp trawling (Fig. 29). Results of a Mann-Whitney rank sum test showed no significant differences in sediment porosity related to trawling disturbance between like depth layers; however, there were significant changes in porosity at the control stations before and after trawling (Table 8).

Table 7. Mann-Whitney rank sum test results from BACI experiment comparing x-ray image gray level values before and after shrimp trawling. P-values <0.05 denote significant differences at the $\alpha = 0.05$ confidence levels.

| Sampling Date | Depth in Sediment | U Test Statistic | Probability |
|---------------|-------------------|------------------|-------------|
| Control | 0-1 | 3.00 | 1.000 |
| | 1-2 | 3.00 | 1.000 |
| | 2-3 | 3.00 | 1.000 |
| | 3-4 | 3.00 | 1.000 |
| | 4-5 | 3.00 | 1.000 |
| | 5-6 | 3.00 | 1.000 |
| | 6-7 | 3.00 | 1.000 |
| | 7-8 | 3.00 | 1.000 |
| | 8-9 | 3.00 | 1.000 |
| | 9-10 | 3.00 | 1.000 |
| Trawled | 0-1 | 0.00 | <0.05 |
| | 1-2 | 3.00 | 0.513 |
| | 2-3 | 3.00 | 0.513 |
| | 3-4 | 2.00 | 0.275 |
| | 4-5 | 2.00 | 0.275 |
| | 5-6 | 0.00 | <0.05 |
| | 6-7 | 1.00 | 0.127 |
| | 7-8 | 3.00 | 0.513 |
| | 8-9 | 1.00 | 0.127 |
| | 9-10 | 1.00 | 0.127 |

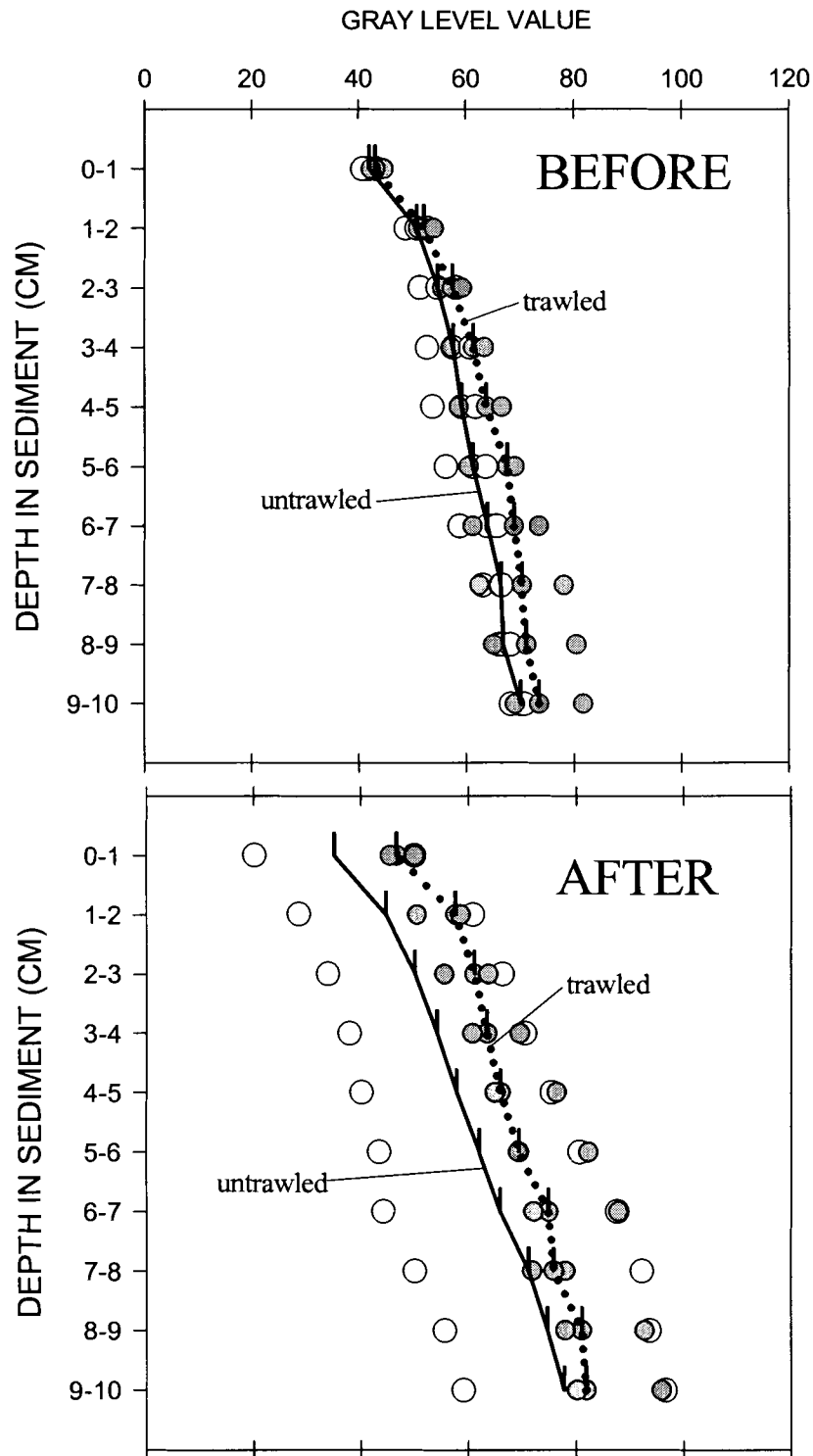


Figure 28. Sediment x-radiography mean gray level values from control and (open circles) and trawl treatment (shaded circles) areas at Thrumcap (BACI experiment) before and after shrimp trawling. Gray level values are a measurement of relative sediment density. Data displayed as circles are replicates from separate box cores. Vertical bars represent median gray level values at each depth in control (solid line) and trawl (dotted line) areas.

Excess ^{210}Pb Activity

Profiles of excess ^{210}Pb activity from trawled and untrawled treatment areas following experimental shrimp trawling show a deeply mixed layer extending from the surface to roughly ~18-20 cm depth in sediment (Fig. 30). Surface excess ^{210}Pb activity in the control area is anomalously low (0.134 Bq/g) in comparison with the trawled area (0.190 Bq/g). It is difficult to explain the occurrence of low surface activity value in the untrawled area by physical or biological processes that do not appear to similarly impact the rest of the core.

Comparison of excess ^{210}Pb between like depth layers shows a relatively subtle pattern of consistently higher activity in the untrawled core. This pattern is suggestive of a loss of 1-2 cm of surface material resulting in a downward shift in the overall ^{210}Pb profile after trawling. Furthermore, the sediment inventory of excess ^{210}Pb activity was lower in the trawled treatment area (3.37 Bq/g) compared to the control (3.51 Bq/g).

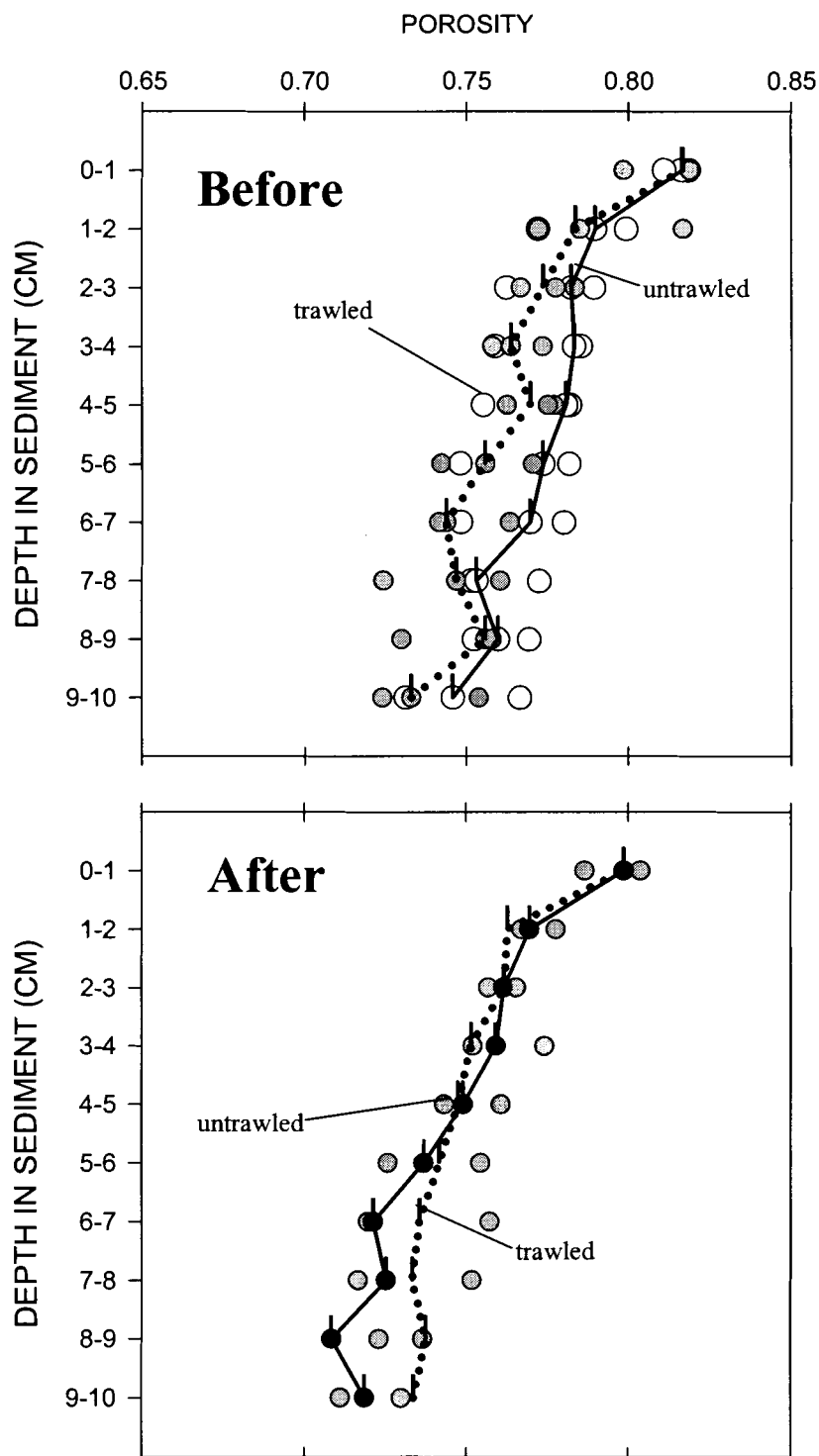


Figure 29. Porosity profiles from control treatment (open circles) and trawl treatment (shaded circles) at Thrumcap (BACI experiment) before and after shrimp trawling. Data displayed as circles are replicates from separate box cores. Vertical bars represent median porosity values at each depth in control (solid line) and trawl treatment (dotted line) areas.

Table 8. Mann-Whitney rank sum test results comparing sediment porosity before and after experimental shrimp trawling in control and treatment areas.

| DEPTH | 0-1 | 1-2 | 2-3 | 3-4 | 4-5 | 5-6 | 6-7 | 7-8 | 8-9 | 9-10 |
|----------------------|--------|------|-------|-------|-------|-------|-------|-------|--------|--------|
| BEFORE/AFTER CONTROL | p<0.05 | .126 | 0.126 | 0.275 | 0.126 | 0.126 | 0.126 | 0.126 | p<0.05 | p<0.05 |
| BEFORE/AFTER TRAWLED | 0.275 | .126 | 0.126 | 0.275 | 0.827 | 0.512 | 0.275 | N/D* | 0.275 | 0.827 |

*No data.

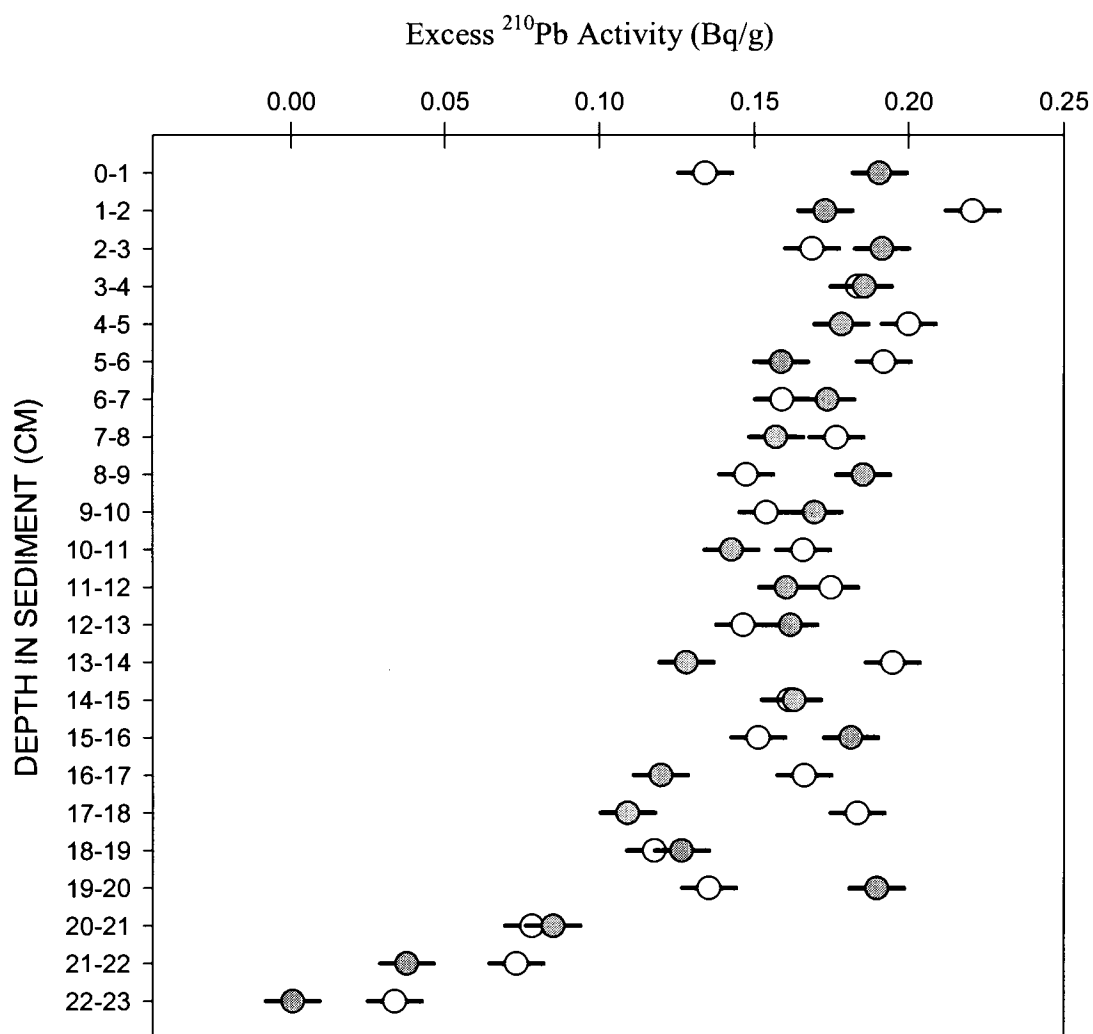


Figure 30. Excess ^{210}Pb activity profiles collected before (open circles) and after (shaded circles) shrimp trawling at Thrumcap (BACI experiment). Error bars show total analytical error (± 1 SD).

DISCUSSION

General Impact of Shrimp Trawling in the Gulf of Maine

Habitat and community structure in marine mud bottom environments are dictated by both physical and biological processes that act as agents of disturbance (Sousa 1984; Thistle 1983; Aller 1982; Widdicombe and Auster 1999). The results of this study indicate that the cumulative impacts of shrimp trawling are not a significant disturbance factor affecting the long-term physical and biological structure of mud bottom fishing grounds in the Gulf of Maine; however, shrimp trawling did appear to alter sediment mixing regimes and produce short-term changes in macrofaunal community structure.

The impact of mobile fishing gear on the seabed is related to both the intensity and frequency of fishing (Watling and Norse 1998). The intensity of disturbance created by otter trawls, like those used in the Gulf of Maine shrimp fishery, is considered to be lower than for other types of mobile gear, such as beam trawls and dredges (Collie et al. 2000, Kaiser et al. 1996). The frequency of benthic disturbance in the shrimp trawling fishery is highly variable throughout the course of the year, and between years. During the shrimp fishing season, from late winter through early spring, the frequency of disturbance in certain areas may be very high because the same well-known areas of unobstructed bottom are repeatedly towed; however, no shrimp trawling activity is present in these areas during the rest of the year (off-season). Interannual variability in the frequency of shrimp trawling is due to large fluctuations in the population of shrimp (*Pandalus borealis*) in the Gulf of Maine. During the course of this study, the near record low numbers of shrimp (Assessment Report for Gulf of Maine Northern Shrimp

2000) resulted in extremely limited fishing activity, from a high of 180 days in 1994 year to 51 days in 2000 and only 25 days in 2001 (M. Hunter pers. comm.).

Consequent with the reduction in fishing effort, shrimp trawling occurred at only at one of the two experimental study sites during the 2000-2001 shrimp season. Fishery port sampling records (DMR unpublished data) and observations by fisherman both noted the presence of trawling activity only on the Pumpkin fishing grounds. Side-scan sonar images of the Monhegan fishing grounds also did not show any evidence of trawl door marks that would have been indicative of recent fishing gear disturbance. In spite differences in the length of recovery period between Monhegan and Pumpkin, my ability to assess the overall cumulative impacts of decades of shrimp trawling effort on these historic fishing grounds was not comprised. It is important to note however, that differences in trawling effects between Monhegan and Pumpkin may be due to short-term impacts of trawling based on disparities in recent levels of fishing disturbance.

Impacts of Shrimp Trawling on Habitat Structure

Previous studies on the effects of trawling on habitat structure have examined impacts on surficial habitat features only. One objective of this study was to determine the effects of shrimp trawling on habitat features in muddy areas of the seabed, where a large proportion of the three-dimensional structure is located below the sediment surface. The most common approach to measuring the impact of fishing gear disturbance on sea floor habitat structure is to record the variety and abundance of biotic and sometimes abiotic habitat features from visual or acoustic images of the substrate surface (Engel and Kvitek 1998; Schwinghamer et al. 1998, Thrush et al. 2001). In this study, I examined

the impact of trawling on infaunal sub-surface habitat structure. Images of infaunal habitats were obtained using sediment x-radiography, a method commonly used to study sedimentological features such as depositional events (Bouma 1979, Kuehl et al. 1995), depth of sediment mixing, and biogenic sediment structures such as burrows (Bouma 1979, Smith, C.R. et al. 2000).

Initially I attempted to assess habitat structure from sediment x-radiographs by measuring: (1) the variety and abundance of discrete biogenic features (burrows, tubes, and feeding voids) and (2) sediment density, a bulk physical property that is a function of the mass of particulate material relative to its volume. My rationale for this approach was that individual biogenic features create microenvironments that enhance habitat structural complexity. Additionally, sediment density is a biologically relevant physical property that provides a measure of overall sediment fabric structure. Upon examining sediment x-radiographs I determined that in a total of 135 images, only a few had biogenic structures that were large enough to be easily discerned. The extremely limited number of biogenic structures captured in x-radiographs precluded the use of these features as measures of habitat structure, thus I examined sediment density structure as the primary measure of habitat structure.

Relative sediment density was determined from gray shade values captured in sediment x-radiographs. Gray shade level was related to the relative degree to which x-rays were able to penetrate through the sediment matrix and expose the film. Areas of the sediment fabric that contained densely packed sediment grains and relatively fewer interstitial spaces filled with water reduced x-ray penetration yielding a lighter gray shade (and higher numerical gray level values) whereas areas with less tightly packed grains

and greater amount of interstitial water allowed greater penetration resulting in darker gray shades (and lower numerical gray level values). In addition to mineral grains and water, other components of the sediment fabric, such as the amount of organic matter present, influenced density structure based on the materials' ability to absorb and scatter x-rays. One potential source of error in measuring relative sediment density from x-radiographic images may have resulted from variable amounts of electron dense elements, such as iron contained in pyrite deposits. The presence of such elements could have disproportionately reduced the penetration of x-rays relative to actual sediment density; however, no unusual structures with high density were noted in any x-radiographs.

Because the mud bottom habitats in this study were characterized by relatively fine, silicious, high water content sediments, I assumed that density structure captured in x-ray images primarily reflected the density distribution of sediment particles. Since large-scale physical disturbances from storm waves were most likely infrequent or absent in the study areas, the distribution of particles, and therefore habitat density structure, was most likely primarily influenced by bioturbation in areas not impacted by shrimp trawling. Conversely, on fishing grounds, I predicted that sediment resuspension from shrimp trawling, and possibly related changes in infaunal community structure, would alter particle distributions and therefore, habitat density structure. Previous work on x-radiographs of California shelf sediments showed that trawling on mixed sand/silt sediments appeared to create lag deposits with a layer of overlying fines (L. Watling pers. comm.). These apparent trawling induced changes in sediment density structure were

presumably due to resuspension and subsequent sorting when sediments were re-deposited at different rates based on size-specific settling velocities.

The results of both the mesurative and BACI experiment showed that, contrary to the prediction that shrimp trawling disturbance would alter mud bottom habitat structure in impacted areas, no significant differences in sediment density consistent with trawling impacts were detected (Tables 2 & 7). In March 2001 and December 2001, sediment density was significantly greater from 0-10 cm in the trawled area at Monhegan (Figs. 8 & 11). It is unlikely that these differences in habitat structure resulted from even short-term effects of trawling disturbance, because although the sampling period in March 2001 was coincident with the shrimp fishing season, no trawling occurred at this study area during either sampling period. Furthermore, no similar pattern was observed at Pumpkin during these same periods. It is also unlikely that a large-scale physical disturbance, such as that generated by storm waves, was responsible for the differences in habitat structure seen during the winter and early spring sampling periods, because the trawled and untrawled study areas were located in close proximity and in the same water depths and would presumably have exhibited similar impacts resulting from a large-scale natural disturbance.

Median sediment density profiles were higher in the trawled area at Monhegan during both sampling periods where significant differences between treatment areas were present (Figs. 8 & 11). Because sediment density always increased with depth in sediment such that gray level values were statistically significant with depth in sediment due to sediment compaction (Friedman and Sanders 1978), removal of the less dense, upper sediment layers would yield an overall higher density profile. Intensive lobster

trap fishing that occurs annually from December-May in Monhegan waters creates very localized disturbance that removes or compacts upper sediment layers (personal observation). Lobster traps can create depressions in the mud and leave ruts and gouges in the substrate when they are dragged along the bottom during retrieval (personal observation). Although lobstering occurs in both the trawled and untrawled area at Monhegan, gear distribution may be patchy. It is possible, although purely conjectural, that at least two sampling stations in March 2001 and three stations in December 2001 that exhibited notably higher density profiles, may have been impacted by lobster gear. Similar patterns of sediment density profiles were not recorded at Pumpkin; however, this area is not part of a seasonal lobster zone and trap densities were lower at this site during the same sampling periods (personal observations).

No changes in sediment porosity related to trawling activity were detected on mud bottom fishing grounds (Table 3). Although porosity profile data was analyzed for only the last 2 out of the 7 sampling periods, these results support the conclusion drawn from analysis of sediment x-radiographs, that shrimp trawling disturbance was not associated with changes in habitat density structure. Porosity profiles at Monhegan and Pumpkin were significantly different ($p < 0.05$, ANOVA) from one another and thus, although the study sites shared many common environmental features, they were not equivalent. Significant decreases in sediment density with depth seen in x-radiographs were mirrored in porosity profiles. Surprisingly, differences between treatments areas observed in the x-radiograph analysis from Monhegan in December 2001 were not detected in porosity profiles. It is possible that quantitative image analysis of sediment x-radiographs may have provided a more sensitive means of measuring changes in bulk sediment density

properties. Although, the sensitivity of this technique was not determined for the purposes of this study, the use sediment x-radiography to measure sediment density and related physical properties, such as porosity, may warrant further investigation.

Sparks-McConkey and Watling (2001) found that experimental shrimp trawling on high-water content silt/clay substrate in Penobscot Bay, Maine, significantly lowered surface porosities presumably due to removal of sediment surface layers. Based on these results, I expected to detect a significant short term effect of trawling on sediment porosity in the BACI experiment however, results of a Mann-Whitney rank sum test showed no differences in porosity associated with trawling effects (Table 8). There were significant differences ($p < 0.05$ Mann-Whitney) in selected depth layers (i.e. 0-1 cm, 8-9 cm, 9-10 cm) in the control treatment before and after trawling.

Changes in sediment porosity with sampling time that were not related to trawling effects must have been produced by some other mechanism of sediment disturbance. Video images of the experimental study area showed high biological activity in the form of numerous burrows and pits, some of which appeared to be quite large (>5 cm). In addition to evidence of biological activity on the sediment surface, numerous epibenthic, megafaunal predators including shrimp (*Pandalus borealis*, name & date), lobster, (*Homarus americanus*, name & date) and a variety of demersal fishes were observed. Studies by Rowden et al. (1998), Botto and Iribarne (2000), Hall et al. (1991) and others have shown that burrowing and pit-digging by large benthic invertebrates such as crabs, shrimp, and brittle stars can locally alter sediment physical properties such as water content, permeability, and median grain size. Based on the apparent high level of biological activity from large, resident megafauna, I suggest that short-term variations in

porosity and perhaps overall sediment density structure may have been due to high levels of biological sediment disturbance.

Experimental shrimp trawling may have produced a short-term change (<2 weeks) in sediment porosity that was not detected because post-trawling sample collection took place 14 days after experimental trawling occurred; however, even if short-lived effects were present, the results of both the manipulative and BACI experiments suggest that the overall impact of shrimp trawling on both sediment porosity and mud bottom habitat structure was minimal.

Both fishing technique and the physical and biological characteristics of the mud bottom study areas may help to explain the minimal impact of trawling on this type of habitat. The target species, Northern or Pink shrimp (*Pandalus borealis*), tend to sit on or swim above the sediment surface (personal observation). Fishing efficiency is maximized when the net is towed very close to or just touches the bottom, capturing shrimp but not unwanted benthic organisms such as brittle stars or lobsters (E. Gastaldo pers. comm.). It is not advantageous for the gear to “dig” into the substrate and consequently, the overall intensity of shrimp trawling disturbance on bottom sediments is likely quite low compared to other types of otter trawling and beam trawling.

The mud bottom fishing grounds examined in this study possess two notable habitat features that may limit the impact of trawling in these areas. First, sediments in the study areas were fine and well-sorted to at least a depth of 10 cm. Results of grain surface area analysis presented in this study showed that shrimp trawling disturbance had no effect on grain size distribution, a biologically important habitat feature that influences community structure (Gray 1974). If trawling disturbance had caused sorting and

subsequent changes in grain size, a greater impact on sediment physical properties and macrofaunal community composition would have been expected. This idea is supported by results reported by Hall et al. (1990) who showed that the use of a hydraulic dredge on mixed sandy sediment resulted in changes in sediment grain size distribution and persistent changes in community composition. Second, the resilience of resident infaunal communities to cumulative trawling impacts may have been an additional factor that contributed to the minimal effect of this disturbance on mud bottom habitat structure. Even in the absence of abiotic physical disturbance, soft sediments are dynamic habitats due to constant re-working by the resident macrofaunal community (Rhoads 1974, Rhoads and Boyer 1982). Small infauna and organisms living below the depth of trawling disturbance are not likely to be crushed, injured, or otherwise impacted by the gear (Duplisea et al. 2002). If biological sediment mixing is primarily carried out by infauna and a significant portion of this community escapes harm, then biological sediment mixing would be expected to resume shortly after trawling disturbance, thereby rapidly restoring habitat structure. The resilience of mud bottom infaunal communities to experimental trawling has been documented in several studies (Drabsch et al. 2001, Lindegarth et al. 2000; Sparks-McConkey and Watling 2001) suggesting that many organisms may be able to withstand at least a limited amount of trawling disturbance.

Similar to effects of trawling on infaunal habitat structure, surficial habitat structure showed no clear impacts due to trawling disturbance. At Monhegan, burrows from 1 - 5 cm in diameter were more abundant in the trawled area; however no similar pattern was observed at Pumpkin where burrow densities were never significantly higher in the trawled area and 3 – 4 cm diameter burrows showed significantly higher densities

in the untrawled area (Fig. 14). Undoubtedly, numerous factors contributed to variation in surficial habitat structure; however, based on qualitative observations of the seabed in both study areas, I speculate that variability in burrow densities result from natural spatial variation in biological activity.

Profiles of excess ^{210}Pb activity from trawled and untrawled areas revealed overall differences in sediment inventory which are suggestive of potential differences in sediment mixing regimes. Both profiles showed relatively deep mixing to a depth of ~16-17 cm; however, the inventory of ^{210}Pb was higher in the trawled area (Fig. 17). Aller et al. (1980) proposed that elevated rates of physical or biological re-working of surficial sediments may produce higher inventories of ^{234}Th . Although ^{210}Pb has a much longer half-life (22.3 years) than ^{234}Th (24.5 days), both radioisotopes are particle reactive and are expected to be subject to similar mixing processes even though mixing rates may differ (Smith et al. 1993). Thus, elevated sediment excess ^{210}Pb inventory may have resulted from higher levels of physical mixing on mud bottom fishing grounds; however, it is not possible to discount higher rates of biological mixing in the trawled area as a factor responsible for elevated radiotracer inventory although biological data showed similar assemblages, and therefore presumably similar mixing regimes, in trawled and untrawled areas at Monhegan.

Results of experimental shrimp trawling on excess ^{210}Pb profiles showed lower sediment ^{210}Pb inventory in the trawled area compared with the control, suggesting possible net loss of material due to fishing disturbance (Fig. 30). This idea is supported by a subtle, but relatively consistent pattern of lower ^{210}Pb activity which is consistent with the loss of upper sediment layers resulting in an upward shift of the profile in the

area subject to trawling. The net difference in sediment inventory between trawled and control areas (0.1494 Bq/g) was equivalent to a loss of less than 1 cm of surficial sediment material; however, this estimate included an anomalously low surface activity measurement in the control area. If surface layer excess ^{210}Pb activity in the control area was similar to or higher than in the trawled area, the estimated net loss of sediment in the trawled area would increase to at least 1-2 cm of surface activity material.

Profiles from the non-manipulative and BACI experiments seem to suggest different mechanisms of sediment re-working from shrimp trawling disturbance. A deeply mixed layer is a feature common to all profiles; however, experimental shrimp trawling disturbance seemed to cause net loss of at least a small amount of surficial sediment while commercial shrimp trawling on mud bottom fishing ground appeared to possibly mix sediments at a higher than natural rate. From the results of this study, it is not possible to determine conclusively how or even if these mechanisms are related.

Impacts of Shrimp Trawling on Mud Bottom Macrofaunal Community Structure

The macrofaunal community in trawled and untrawled areas exhibited features characteristic of a disturbed faunal assemblage (sensu Rhoads and Boyer 1982). Small, and mainly short-lived polychaetes were the most abundant group at both study sites. In the Pumpkin trawled area, the fishing ground known to have experienced trawling activity during the course of this study, polychaetes displayed the overall highest proportional abundance (Fig. 18), suggesting that shrimp trawling disturbance may have had at least short-term impacts on the structure of even an apparently perturbed community by creating conditions favorable to small, disturbance-tolerant polychaetes.

The second most abundant group observed in this study was mollusks, represented almost completely by small bivalves in the families, Nuculidae and Nuculanidae. Bivalves are known to be sensitive to trawling (Ball et al. 1999, Tuck et al. 1998) and may be useful as indicator species of fishing gear disturbance (Witbaard and Klein 1994). The proportional abundance of mollusks was similar in the Monhegan untrawled and trawled area, the latter of which had not been subject to trawling disturbance during the course of this study. In contrast, the untrawled area at Pumpkin exhibited more than double the relative proportion of mollusks compared to the trawled area. These results are important because they suggest that the effects of shrimp trawling on community structure may be related to relatively recent trawling impacts and are therefore not necessarily cumulative in nature.

Temporal trends in average taxa richness provide further support the idea that shrimp trawling disturbance has a relatively short-term impact on the macrofaunal community. A decline in taxa richness in only the Pumpkin trawled area concurrent with the 2000-2001 shrimp fishing season was followed by a rebound to values similar to the untrawled areas 3 months later (Fig. 21). Although the decline at Pumpkin was not statistically significant, it is suggestive of a short-term trawling impact on the macrofaunal community.

Shrimp trawling disturbance did not appear to cause changes in macrofaunal community diversity or evenness (Fig. 23). These results differ from those reported in a commercial otter trawling study conducted by Smith, C.J. et al. 2000 in Mediterranean Sea where diversity was significantly lower and evenness higher in fished areas. Similar reductions in diversity and decreased evenness were reported in experimental trawling

impact studies on mud bottoms (Tuck et al. 1998; Sparks-McConkey and Watling 2001); however, Sanchez et al. (2000) found that trawling had no effect on community diversity. The variability in diversity and evenness measurements in response to trawling disturbance was most likely related to initial level of individual community diversity and evenness as well as the ability of constituent species to tolerate the disturbance. Because the communities examined in this study exhibited overall low diversity and were primarily comprised of small, disturbance-tolerant fauna, shrimp trawling did not appear to alter the heterogeneity of the macrofaunal community.

The total abundance of organisms varied seasonally with a peak, probably due to annual recruitment to the benthos, during the summer months (Fig. 20). Overall abundance was lowest during the winter sampling periods when shrimp trawling occurred. Small-scale chemical and hydrodynamic regime changes associated with sediment disturbance can affect settling and survival rates of larval and juvenile benthic organisms (Butman 1987, Woodin 1991). Thus, the timing of trawling disturbance to the benthos during the period of lowest macrofaunal abundance might have been a factor that decreased the impact of this disturbance on benthic community dynamics.

Temporal patterns in abundance were consistent in both study areas; the trawled area at Monhegan had lower total abundances of organisms compared with the untrawled area, especially during the first summer sampling period (Fig. 20, June 2001). It is difficult to determine the factors that may have contributed to reduced macrofaunal abundance in trawled area; however, trawling disturbance is not believed to be a factor because there was no similar decrease in abundance in the Pumpkin trawled area. Patchy disturbance created by seasonal lobster fishing in the waters around Monhegan may have

contributed to lowered total macrofaunal abundance. The greatest difference in abundance between trawled and untrawled areas occurred approximately one month after the close of the 6-month lobster fishing season. Possibly, recruitment or emigration had not yet fully re-populated the community in the month following the cessation of fishing. Later in the summer after several months without lobster trap disturbance, the total abundance of macrofauna at Monhegan in the trawled area increased to densities closer to those observed in the untrawled area. Total abundance trends were similar in trawled and untrawled areas at Pumpkin, but displayed lower overall abundances compared to Monhegan.

Results of MDS ordination analysis show that, although the Monhegan and Pumpkin study areas have similar environmental features (including depth, salinity, temperature ranges and sediment grain sizes), community structure differed slightly between the two study areas (Fig. 23). These results were not surprising given the differences in the proportional abundance of community constituent groups between the two study areas; however, despite proportional differences in abundance, communities were nearly identical in their faunistic composition at the family level. Therefore, similar community responses to cumulative trawling effects were expected.

Substantial differences in levels of shrimp trawling activity at the two sites during this study had not been anticipated. Because the impacts of trawling disturbance may be highly temporally dependent (Watling and Norse 1998, Kaiser et al. 2002), it is problematic to compare the responses of a recently impacted community (Pumpkin) to a similar community that may show only persistent effects from long-term impacts (Monhegan). Results of MDS ordination analysis comparing communities in trawled and

untrawled areas at both study sites show no apparent differences between treatment areas (Fig. 24); however, when community similarity between trawled and untrawled areas was analyzed separately for each study site it was clear that recent shrimp trawling disturbance at Pumpkin produced changes in macrofaunal community structure, whereas the effects of past trawling activity at Monhegan did not result in persistent differences in the macrofaunal community (Fig. 25). These results provide further evidence that shrimp trawling disturbance caused changes in macrofaunal community structure on fishing grounds; however, the impact of this disturbance did not appear to alter community structure on a cumulative or long-term basis.

Results of a SIMPER analysis of macrofaunal abundance data also provide support for the idea that shrimp trawling disturbance altered community structure on recently impacted fishing grounds. Communities in trawled and untrawled areas at Pumpkin were more dissimilar (average dissimilarity = 35.00) than at Monhegan (average dissimilarity = 29.70), the less recently impacted site. Dissimilarity between trawled and untrawled areas from recent trawling impacts at Pumpkin was due in part to increased abundance of burrowing and/or disturbance-tolerant taxa such as Cirratulidae, Cossuridae, and Paraonidae on fishing grounds. Trawling sensitive taxa, such as the bivalve families Nuculidae and Nuculanidae, were more abundant in the untrawled area. Similar patterns in common taxa abundance were not present at Monhegan where nearly all of the aforementioned taxa (except Nuculanidae) were more abundant in the untrawled area.

Mortality rates of infaunal organisms due to trawling disturbance are related to their body size and depth distribution in the sediment (Duplisea et al. 2002). Nearly all of

the animals collected in this study were very small (<1.0 cm) and were surface or near-surface dwelling. The depth distribution of sediment-dwelling macrofauna showed that 50 - 65% of the total abundance of organisms was found in the upper 2 cm of sediment except in the Pumpkin trawled area where only ~45% of the total macrofauna inhabited this upper sediment layer (Fig. 19). Macrofaunal depth distributions suggest that recent shrimp trawling disturbance may have caused mortality at Pumpkin that reduced the abundance of organisms in the upper 2 cm of sediment. Results of ^{210}Pb activity profiles presented here, which indicated a possible loss of the upper 1-2 cm layers of sediment, provide a mechanism to support this conclusion. It is interesting to note that while the percent abundance of organisms in the 0-2 cm sediment layer in the trawled area at Pumpkin was (~20%) lower than the untrawled area, the percent abundance of animals in deeper sediment layers that presumably were not directly impacted by the gear (2-4, 4-6, 6-10 cm) was 5-10% greater in the trawled area.

Sediment depth layer and the presence/absence of shrimp trawling activity, accounted for 53.4% and 50.1% of the variability in community structure at Monhegan and Pumpkin study areas, respectively (Table 6). Most of the variability in community structure was related to the sediment depth layer organisms were associated with. RDA ordination plots show that the abundance of most taxa was positively correlated with the 0-2 cm depth layer (Figs. 26 & 27). Unlike other taxa, polychaetes in the families Cossuridae and Cirratulidae were positively correlated with deeper sediment depth layers. Interestingly, there appears to have been a downward shift in the depth layer these families were associated with from Monhegan to Pumpkin. Cossurids and Cirratulids at Monhegan were positively correlated with the 2-4 cm depth layer (Fig. 26) while these

families were more positively correlated with the 4-6 cm depth layer at Pumpkin (Fig. 27). By creating disturbance and possible net loss of upper sediment layers, recent shrimp trawling activity at Pumpkin may have either (1) caused these taxa to actively seek refuge in deeper undisturbed sediment layers or; (2) resulted in increased mortality of animals inhabiting the 2-4 cm depth layer relative to deeper sediment layers.

Commercial shrimp trawling activity was not a disturbance that appeared to have an ecologically significant impact on macrofaunal community structure. Results of RDA ordination analysis showed that the presence of shrimp trawling activity accounted for only 3.7% - 4.8% of variability in community structure and was not consistently a statistically significant environmental factor related to variability in the abundance of macrofauna (Table 6).

Biological Sediment Disturbance by Large, Predatory Benthic Megafauna:

Possible Effects on Habitat and Macrofaunal Community Structure

Contrary to initial predictions, shrimp trawling did not measurably affect habitat structure and produced only relatively short-term changes in macrofaunal community structure on mud bottom fishing grounds examined in this study. Low intensity and frequency of fishing effort, especially over the course of the study, may have been factors that contributed to the minimal impact of fishing activity; however, I suggest that there is evidence that high levels of biological sediment disturbance, probably caused by large, predatory megafauna, maintained macrofaunal communities in disturbance-tolerant, low successional state that may have minimized the impact of shrimp trawling disturbance on both habitat and community structure.

Several forms of evidence suggest that biological sediment disturbance on mud bottom fishing grounds was high and primarily associated with the activities of resident megafauna. First, ^{210}Pb activity profiles show a ubiquitous, deeply mixed layer (~16-17 cm) (Fig. 17). Due to the water depth (~100 m) and the lack of visual evidence of storm wave, tidal current, or other abiotic disturbance, it is unlikely that sediment mixing was the result of large-scale physical disturbance. Moreover, the small body size and predominately near-surface depth distribution of the macrofaunal community suggests that it is unlikely that these organisms were capable of producing the deeply mixed layer. Second, observations of the sediment surface revealed many small and larger burrows and pits probably created by burrowing and digging crustaceans (lobsters and shrimp) as well as demersal fishes. These animals were commonly observed, along with dense patches on brittlestars, creating localized sediment disturbance while simply moving over and/or apparently foraging on the mud bottom fishing grounds examined in this study.

Based on the evidence given above, I speculate that sediment disturbance by large, predatory megafauna including fishes and especially lobsters, created the ~16-17 cm mixed layer and favored the persistence of lower successional state, disturbance-tolerant communities observed on mud bottom habitats in this study. Crustaceans such as ghost and mud shrimp are capable of intense sediment mixing which subsequently lower infaunal abundance and diversity (Posey 1986, and Posey et al. 1991) while sediment disturbance by pit-digging rays and the crab, *Cancer pagurus*, has been shown to have similar localized effects on community structure (Van Blaircom 1982 and Hall et al. 1991). Predatory control of infaunal populations has been reported for the blue crab, *Callinectes sapidus*, and the bottom feeding fish *Leiostomus xanthurus* (Virnstein 1977).

Hall et al. (1990) noted that the effects of sediment disturbance and predation by large megafauna are difficult to separate because predation is often associated with sediment disruption. Both physical sediment disturbance and predation by resident megafauna may affect macrofaunal community structure on mud bottom fishing grounds however the relative importance of each process is unknown.

Although several species of benthic fish and invertebrates may locally alter habitat and macrofaunal community structure, I speculate that the American Lobster (*Homarus americanus*) due to its abundance and behavior, may exert a disproportionately large effect on sediment mixing and infaunal communities. The abundance of lobsters in the Gulf of Maine has increased over the past few decades in conjunction with the declining predator populations such as cod and other groundfish (Factor 1995). While lobsters are known to inhabit hard substrates there is evidence that their populations may be increasing on soft sediments (DMR unpublished data). Population surveys on deep mud bottoms around Monhegan Island showed lobster densities approached 2-5 individuals/100m² (R. Steneck unpublished data). Lobsters are known to both burrow and dig pits for shelter in soft sediments (Cooper and Uzmann 1980). Lobsters are generalist predators as well as scavengers. They are known to forage in soft sediment for prey items. Gut content analysis shows that lobsters routinely consume polychaetes (Factor 1995).

Based on their ability to re-work sediments while burrowing/digging and foraging, as well as their relatively high abundance on mud bottom fishing grounds, lobsters along with other megafaunal predators, could maintain a natural disturbance regime that controls habitat and macrofaunal community structure to a greater degree

than periodic shrimp trawling disturbance on mud bottom fishing grounds in the Gulf of Maine.

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Before attending college, she studied for one academic year at Lycee Aristide Briand in Saint Nazaire, France. She attended The University of California at Santa Cruz and Mary Washington College, and graduated from the latter with a Bachelor of Science in Biology in 1997.

After receiving her degree, Anne plans to enter the doctorate program in Marine Biology at the University of Maine to begin research examining the reproductive biology of cold water octocorals. Anne is a candidate for the Master of Science degree in Oceanography from The University of Maine in December 2003.