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RESEARCH ARTICLE

The impacts of hydraulic clamming in shallow water and the importance of incorporating anthropogenic disturbances into habitat assessments

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Abstract – Hydraulic dredging for shellfish is known to create some of the highest levels of disturbance, affecting the benthic microfaunal community and the physical characteristics of the substrate. Properly conducted benthic habitat assessments are complex and time consuming, resulting in assessments not being conducted increasing the uncertainty in post impact studies. Hydraulic dredging for Atlantic surfclams (*Spisula soldidissima*) took place at Herring Cove, Massachusetts in the winter of 2014–2015 resulting in areas of high impact disturbance of the seafloor. Surveys conducted in the summer of 2015 included hydroacoustics, benthic invertebrate sampling, video, and grain size analysis for the creation of a habitat map of Herring Cove. The four habitats (A–D) identified were a mix of sand, shell, cobble, algae, and eelgrass. Habitat type “D” is a mix of sand, algae and cobble material and occurred at 12 of 18 stations. These 12 stations were distributed across areas of “high” ($n = 4$), “low” ($n = 2$), and “no” ($n = 6$) hydraulic dredge disturbance. Once habitat was accounted for, benthic invertebrate community structure varied significantly (Analysis of similarity; significance level of sample statistic: 0.3%) between areas of “high”, “low” to “no” disturbance. Areas of “low” to “no” dredge track coverage contained high abundances of bivalves, echinoderms, and isopods, whereas highly disturbed areas had highest abundances of polychaetes and oligochaetes. Future mapping efforts, especially surveys with biological components, need to include and quantify the level, type and spatial distribution of anthropogenic alterations. More attention should be given to “reference maps” instead of “baseline maps”. The latter of which omits to acknowledge pre-existing anthropogenic disturbances and has the potential to skew monitoring of restoration and management efforts.

Keywords: Sidescan sonar / benthic communities / habitat / anthropogenic / hydraulic clamming / dredge / disturbance

1 Introduction

Properly conducted benthic habitat assessments are complex and time consuming. Baseline data is often unavailable and post-impact assessments are challenged in differentiating between anthropogenic disturbances and the natural variation in aquatic environments (De Juan et al., 2009). Impacts from benthic fishing are considered one of the most widespread physical disturbances in the world (Hiddink et al., 2017). Specifically, benthic fishing by hydraulic clam dredge, commonly used for commercial fishing of large bivalves, is known to cause some of the highest physical disturbances to the benthic environment (Oberle et al., 2016). Hydraulic dredging utilizes high-pressure jets to fluidize the sediment, allowing a cage to penetrate the substrate up to 0.25 m (Meyer et al., 1981; Smolowitz, 1982) and collect bivalves at depth.

Habitats are an ecological or environmental area inhabited by a particular species or group of species (ICES, 2006) and comprise both abiotic (e.g. grain size, temperature, light, salinity, wave energy) and biotic factors (e.g. food availability, presence of predators). Habitats can be created and altered by ecosystem engineers (e.g. eelgrass, oysters, corals, and tube forming worms) which can change the physical structure of an area, instigate nutrient cycling, and promote productivity (ICES 2006). Adequately describing and defining habitats is challenging as they naturally change over time and are often

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gradients of different descriptors (e.g. sand, mud, eelgrass, algae, shell, and cobble; Legare and Mace, 2017). Thus, data collected within the same area can vary based on timeframe, mapping resolution, and the methodology used to quantify the habitat (Carvalho et al., 2011; Sköld et al., 2018).

Habitat availability influences community composition of benthic invertebrates (Lu et al., 2008; Ellingsen, 2002). In turn, community composition at the base of the food web can affect the faunal community throughout trophic levels (Henderson and Bird, 2010). Colonization patterns of benthic invertebrates, and thus community composition, following anthropogenic disturbances are highly variable on spatial and temporal scales (Ragnarsson et al., 2015; Van der Linden et al., 2016; Mercaldo-Allen and Goldberg, 2011). The fishing/dredging intensity, season, habitat type, and adjacent habitat types all influence the ability of larvae and juvenile invertebrates to recruit to the substrate (Oberle et al., 2016; Sköld et al., 2018). Changes in benthic community measured immediately (hours to days) following a disturbance event are indicative of mortality, scavenging and colonization (Ragnarsson et al., 2015; Gilkinson et al., 2005). Whereas changes in the benthic communities measured in the weeks and months after the disturbance event reflect recruitment, succession, and competition for resources (Gilkinson et al., 2015). It is important to include post-disturbance timelines into surveys to avoid inadvertently creating an unsuitable baseline. For example, high species diversity and abundance may be an indication of colonization by scavengers and predators rather than the natural recovery of the affected species communities (Poirrier et al., 2009; Kennedy and Jacoby, 1999). As the community assemblage changes over time (post-disturbance), disturbed and undisturbed areas may continue to differ due to natural biotic and abiotic factors (Mittermayr et al., 2020a). If the initial stress is intense enough or becomes chronic and the affected community is unable to recover, the ecosystem may permanently shift to a more disturbed state (Sköld et al., 2018).

Hydraulic clamming affects the benthic habitat by physically disturbing the substrate and by removing and/or damaging organisms (both targeted and by-catch), consequently causing short and long-term changes in benthic communities (Johnson, 2002; Mercaldo-Allen and Goldberg, 2011). Stable communities, particularly of sessile long-lived species, are slow to recover, whereas highly opportunistic organisms, mobile species, and scavengers are quick to re-colonize (Hiddink et al., 2017; Collie et al., 2000). System recovery (both biological and physical) after hydraulic clamming is dependent on intensity and frequency of fishing events, and the type of habitat affected (Sköld et al., 2018).

In New England, Atlantic surfclams are a valued fishery with landings, in 2016, of around 9000 metric tons worth 18 million USD, and the majority of which are harvested by hydraulic dredging (https://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index). If hydraulic dredging is conducted in substrates containing sessile, fragile structures like bivalve reefs and beds of submerged aquatic vegetation, a degradation of habitat might follow (Johnson, 2002). Fisheries management plans for Atlantic surfclams (Mid-Atlantic Fisheries Management Council, 2016) state that in sand dominated environments, the effects of dredging on benthic communities are short lived (hours to months), and that, in comparison, disturbances caused by strong currents and storm events would be more severe. These conditions are assumed for the nearshore sandy environment along Cape Cod, yet physical alteration to the substrate in nearshore environment has been shown to last multiple years (Legare et al., 2020a).

Following the significant impacts Hurricane Sandy had on coastal communities and ecosystems in 2012, the National Park Service sought to create a baseline inventory of existing marine habitats in coastal parks for the purpose of using these baseline data to measure future natural and anthropogenic change in these environments (Borrelli et al., 2019). One of these coastal systems included the area called Herring Cove off the coast of Provincetown, Massachusetts, and the baseline inventory was to be accomplished in the summer of 2015. The area off Herring Cove, has had a local moratorium on hydraulic clamming starting in 2007, in which no hydraulic clamming has been documented between 2007 and the winter of 2014–2015. During the winter of 2014–2015, a series of vessels conducted hydraulic clamming within the study site, extensively covering the area (Legare et al., 2020a). The impacts were thought to be ephemeral allowing the baseline inventory to proceed as planned. Upon inspection (Legare et al., 2020a), significant alteration to the substrate was found. This led us to question if the current survey could be considered a baseline.

The present study focuses on understanding benthic invertebrate communities relative to habitat type and disturbance by hydraulic clamming. Analysis of acoustic surveys identified an area of slow (multiple years) physical recovery to the substrate (Legare et al., 2020a) but no information about the biological impacts exists. Using acoustic survey techniques, benthic invertebrate surveys, grain size analysis, and video surveys, this study documents: 1) the habitat types present; and 2) the differences in benthic invertebrate community composition between areas exposed to hydraulic clamming and areas that have not been exposed to hydraulic clamming. This study provides an example of where incorporation of anthropogenic disturbances can be critical in interpreting the benthic invertebrate survey.

2 Materials and methods

2.1 Study site

The Provincetown Hook on Cape Cod, MA, USA from Race Point to Long Point has a narrow shelf and quickly drops from 10 to 45 m (Fig. 1). The area is dominated by submerged sand flats and banks with a median grain size (D50 1–2 mm) of coarse sand (Borrelli et al., 2019). From November 2014 to April 2015, a series of fishing vessels harvested Atlantic surfclams (*Spisula soldidissima)* by hydraulic dredge along the Provincetown Hook, for the first time since a moratorium was imposed in 2007 (Borrelli et al., 2012; Myers, 2015; Legare et al., 2020a). The method of fishing employed involved a hydraulic clam dredge, in which high-pressure jets of water fluidize the substrate allowing a cage to sift clams and other objects larger than bar width into the cage.

2.2 Acoustic survey and dredge tracks

Details of the acoustic survey can be found in Legare et al. (2020a). To summarize, acoustic surveys were performed

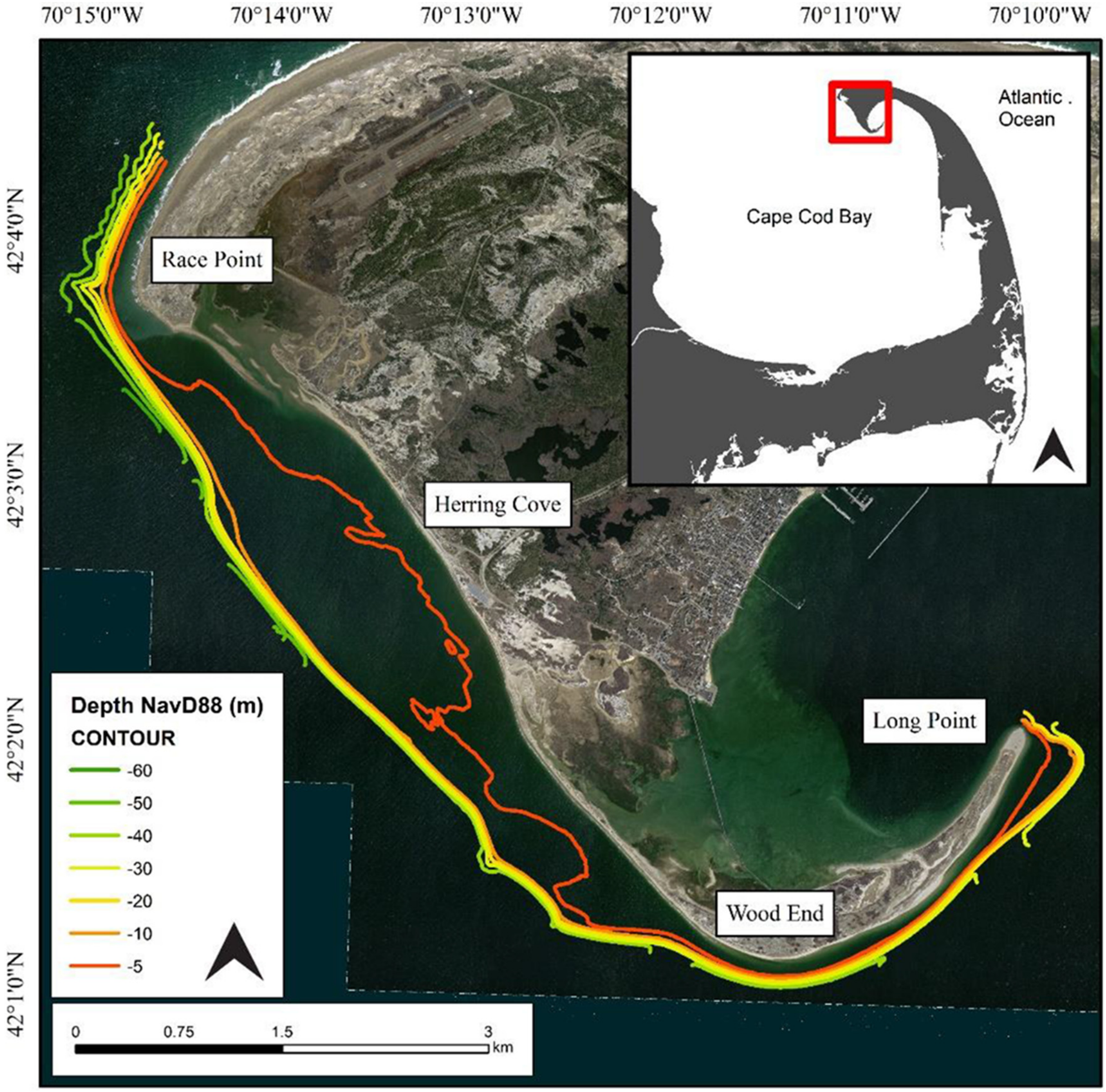

Fig. 1. Study site: Provincetown, Massachusetts (USA) is on the northern end of Cape Cod. The Provincetown Hook forms a barrier beach from Race Point in the North to Long Point in the South with Herring Cove creating a West-South-West facing shore.

using a bow mounted EdgeTech 6205 dual-frequency, phase-measuring sidescan sonar. The EdgeTech 6205 produces both swath bathymetry and duel frequency sidescan imagery, opposed to traditional sidescan sonars that only produce imagery. The EdgeTech 6205 operating frequencies are 550 and 1600 kHz for backscatter imagery and 550 kHz for bathymetry. The sidescan sonar range resolution is 1 cm, and the horizontal beamwidth is 0.5 degrees at 550 kHz. The corresponding quantities at 1600 kHz are 0.6 cm and 0.2 degree (Edgetech, 2014). The bathymetric data has both horizontal and vertical resolutions of up to 1 cm (Edgetech, 2014). A Teledyne TSS DSM-05 Motion reference unit was mounted on the sonar to measure the dynamic motion of the vessel: heave to 5 cm and role and pitch to 0.05 degrees (Teledyne, 2006). Heading was collected using a HemisphereGPS® V110 vector sensor with two differential GPS receivers spaced 2 m apart with a heading accuracy of <0.10 degrees RMS (Hemisphere, 2009). Positional data and tide corrections were collected by a Trimble® GNSS receiver utilizing Real-Time-Kinematic GPS (RTK-GPS). Acquisition was conducted using EdgeTech Discover Bathymetric®, Hypack Survey® and Hypack Hysweep® with raw data outputs as JSF and HSX files. The JSF files were imported into SonarWiz5® where semi-automated processing of bottom tracking, slant range, offsets and gain adjustments were performed. Data were exported as a Geotiff at a resolution of 0.50 m.

Dredge tracks were identified and classified in the digitizer extension in SonarWiz® v5.x from the sidescan imagery. All dredge tracks were hand digitized across each processed survey file. Dredge tracks were exported as a raster (geotiff) at a pixel resolution of 1 m, which is the approximate width the dredge tracks without over estimating their coverage. Exporting as a single band raster allowed for tracks that were digitized multiple times to be combined, thus eliminating duplicate digitization (Fig. 2). For more details see Legare et al. (2020a).

Track density was calculated by converting from raster to point data, each point representing 1 m^2 of seafloor.

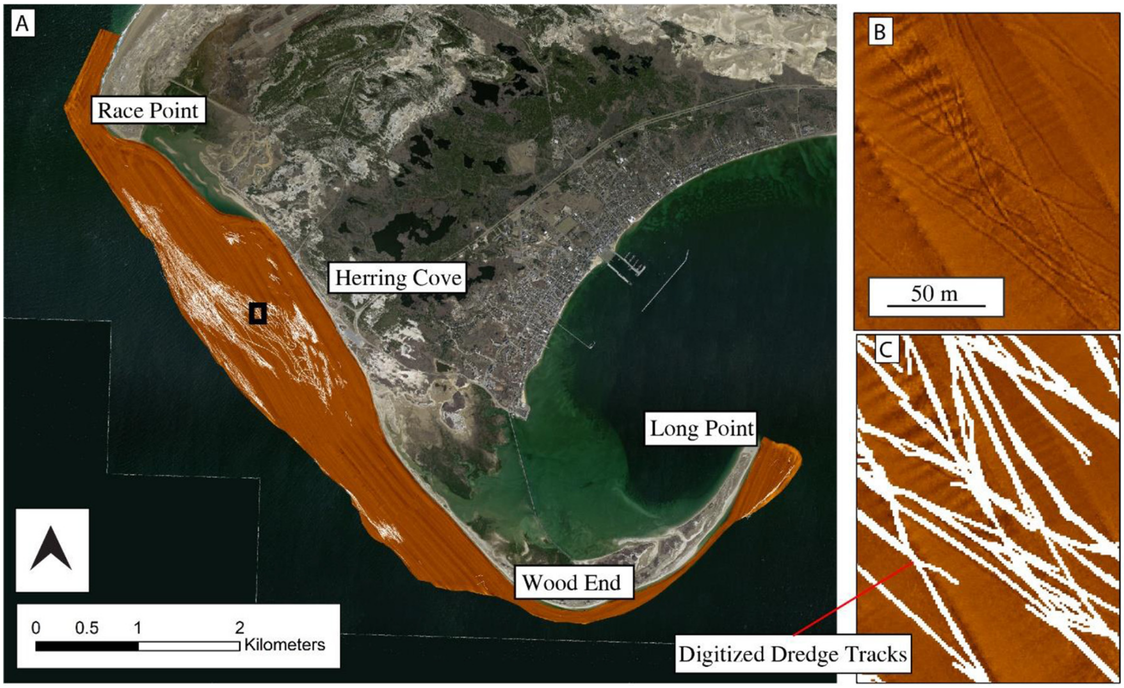

Fig. 2. Overview of sidescan sonar mosaic throughout the study site (a) with dredge tracks (white) digitized. Zoomed in image (b) of the dredge tracks off the shore of Herring Cove and the resulting digitization of the dredge tracks present (c). Dredge tracks digitized in Legare et al. (2020a).

The point-density analyses were conducted with the Spatial Analyst extension in ArcGIS v10. This analysis used 10 × 10 m grid cells and the aggregated count of dredge tracks in each grid cell to display density as a heat map (Bokuniewicz and Jang, 2018). The grid resolution of 10 m (100 m^2) creates the finest resolution heat map while maintaining the original mapping unit (1 m^2) and does not overestimate or underestimate the density of points (Fig. 3). This also produced a density data equivalent to percent cover, e.g. 0.25 dredge tracks per m^2 is equivalent to 25% cover, allowing for the data to translate to other studies.

2.3 Habitat and invertebrate sampling

To address the biological impacts of the dredging activity, field surveys were conducted for benthic invertebrates, sediment (Gravel (%), Sand (%), Silt (%), Clay (%) LOI (%)), and habitat characteristics (Benthic Cover e.g. Algae, Algae Type, Eelgrass, Cobble, Sand) (Tab. 1). As dredge locations and habitat types were unknown at the time of sampling, samples were collected at 18 random stations across the study site. At each station, three benthic invertebrate replicates and one sample for grain size analysis were collected using a Young-Modified Van Veen Grab Sampler (0.04 m^2 surface area, 0.1 m sample depth, 0.004 m^3 sample volume) with an attached GoPro Hero3® to collect habitat data via video (Legare et al., 2020b). The contents of the grab were washed through a 1 mm sieve using a low energy salt water wash. A 1 mm sieve was chosen for cost-benefit reasons (Fox et al., 2009; Hemery et al., 2017). Large bivalves, crabs and vertebrates were returned to the water after they were identified (to the lowest identifiable taxonomic level), counted and measured. The material retained on the sieve was transferred to a fine mesh bag for transport and later preserved in 80% ethanol and Rose Bengal to aid in identification. All replicates (3) at each station were pooled, averaged, and adjusted to individuals per square meter (Tab. 2) for further analysis.

Sediment samples were frozen for subsequent quantification of organic matter, and grain size analysis. Organic content for each sample was obtained by placing 20–30 g of sediment on pre-weighed aluminum trays and wet weight obtained. The samples were dried at 105 °C for 24 hours to record dry weight (Heiri et al., 2001; Borrelli et al., 2019). Samples were placed in a 550 °C muffle furnace for four hours then weighed again (Snyder et al., 2004). The change in mass from pre- and post-ignition is expressed as percent Loss on Ignition (LOI). Grain-size analysis of grains <2 mm in size was conducted using a Beckman-Coulter LS 13 320 Laser Diffraction Particle Size Analyzer at the Woods Hole Oceanographic Institution's Coastal Systems Laboratory. All data were reported using Wentworth grain size thresholds and classes (Folk, 1974).

Underwater video was used to document semi-quantitative habitat information (Legare et al., 2020b). Screen captures of the substrate and surrounding habitat were extracted from the video using Adobe Premiere® for each benthic sample at each station, resulting in 4 screen captures per station, at a minimum. Multiple screen grabs for each sample were examined to refine habitat classifications as necessary, thus more than four images were used at many stations. Types of habitat characteristics were grouped into four categories: bare sand, cobble, algae, and eelgrass on a scale of 0–4 with "0"

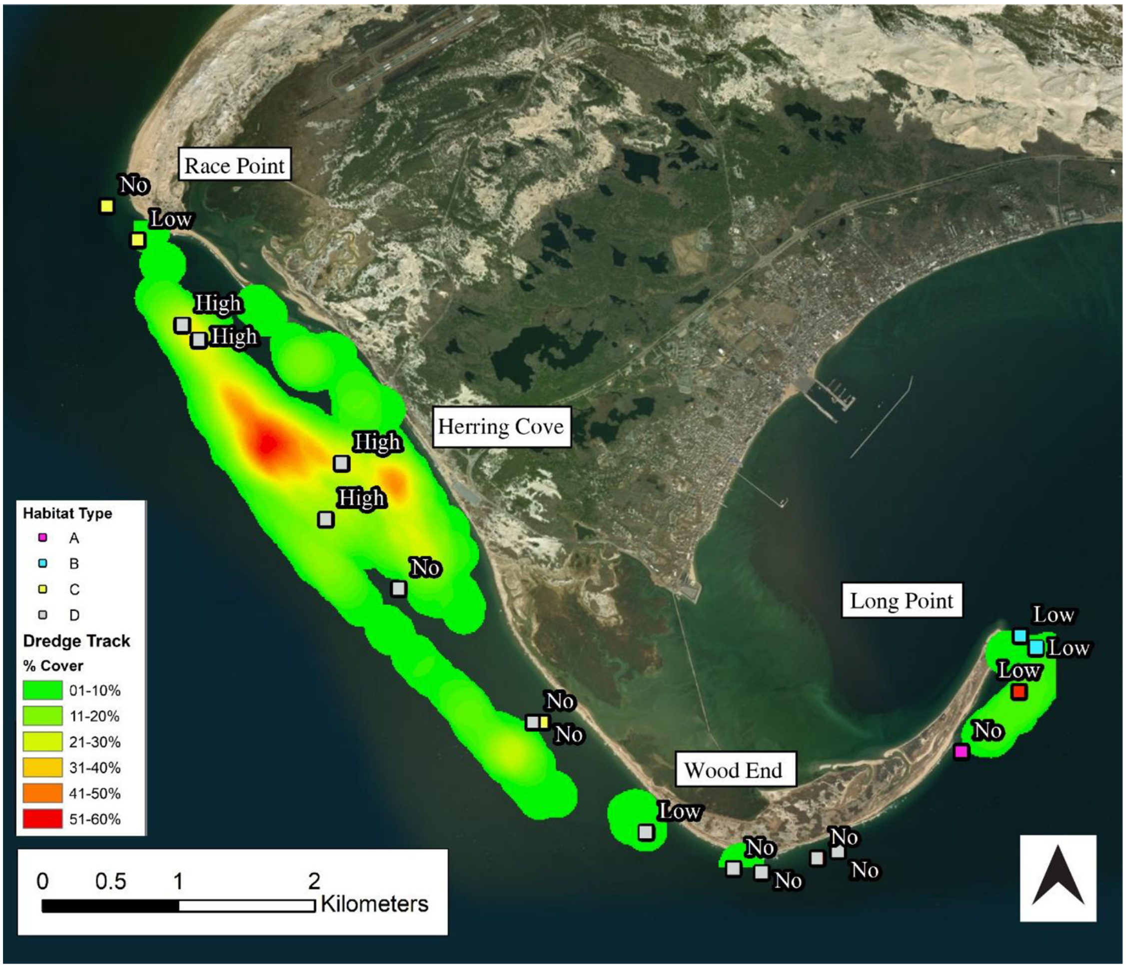

Fig. 3. Dredging extent throughout the study area with invertebrate stations indicated across habitat types A–D and dredge disturbance indicated as "high", "low" or "no" dredge tracks. Habitat type A ($n = 1$) has the highest percent LOI, % gravel, and algae. Habitat type B ($n = 2$) has the highest coverage of eelgrass. Habitat type C ($n = 3$) is identified as bare sand. Habitat type D ($n = 12$) high % sand, presence of algae, and cobbles.

indicating absent, "1" representing >0–25%, "2" has cover of 25–50%, "3" represents cover of 50–75% and "4" indicates 75–100% cover (Roelfsema et al., 2009; Legare et al., 2020b). The dominant type of algae was also indicated as 1–3 with "1" indicating red filamentous, "2" being green filamentous and "3" being a combination of the two.

2.4 Analysis

The outputs of the point density analysis were used to bin each sampling station by dredge track intensity "no" 0%, "low" 1–4%, "high" 14–28% dredge tracks (Fig. 3). No stations were observed in areas between 4 and 14% or above 28% (Tab. 1). In Legare et al. (2020a) an inverse relationship between dredge intensity and physical recovery of the substrate was determined. Areas with dredge density greater than 10% were found to recover slower than areas with a dredge track density 10%. This relationship between dredge intensity and recovery aided in the creation of these categorical bins.

Habitats can be described by gradients of different biotic and abiotic factors. These data sets are often categorical and do not meet the assumption of normality. Additionally, the units of different habitat descriptors are inconsistent or absent, therefore non-parametric statistics are appropriate. In order to create adequate habitat classifications for each station and to capture the suite of attributes (eelgrass, cobble, grain size (<2 mm), LOI, algae presence, algae type), a cluster analysis in Primer7® (PRIMER-E v7, Plymouth) was conducted (Clarke and Warwick, 2001, Clarke and Gorley, 2015). Data was first normalized, and using a resemblance matrix based on Euclidean distance, a Non-metric Multi-Dimensional Scaling (nMDS) plot was created to visualize the distribution, and overlaid trajectories of attributes were used to describe each habitat (Legare et al., 2020b).

Benthic invertebrate species data were analyzed in the statistical software Primer7®. Diversity indICES calculated include number of species (S) and Shannon's diversity Index (H') for each station as a comparison across habitats (A–D) and dredge levels. Species data were standardized across samples and a square-root transformation was applied. A Bray-Curtis resemblance measure was used to create a similarity matrix. Visual inspection of nMDS plots were performed to examine the relationship of assemblages across stations. Dredge

Table 1. Habitat characteristics from grain size analysis (% composition of gravel, sand, silt, clay, and loss on ignition), video survey (cobble, algae, and eelgrass on a scale of 0–4, and algae type) and dredge analysis across stations. Outputs of cluster analysis produces the "habitat type" as a combination of the abiotic sediment characteristics and the biotic habitat types.

Station	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	LOI (%)	Dredge (%)	Dredge category	Algae	Algae type	Eelgrass	Cobble	Habitat type
1	26	74	0	0	0.71	1	Low	2	2	2	2	B
2	7	92	0	0	0.55	1	Low	2	2	3	1	B
3	36	64	1	0	1.12	4	Low	2	2	0	0	D
4	45	55	0	0	2.52	0	No	4	2	1	1	A
5	22	78	0	0	0.28	0	No	3	2	0	2	D
6	12	88	0	0	0.27	0	No	2	2	0	2	D
7	7	93	0	0	0.41	0	No	1	2	0	0	D
8	3	97	0	0	0.4	0	No	1	2	0	0	D
9	10	90	0	0	0.59	1	Low	2	3	1	0	D
10	5	95	0	0	0.33	0	No	0	0	0	0	C
11	18	82	0	0	0.25	0	No	2	3	0	0	D
12	1	99	0	0	0.37	0	No	1	2	0	0	D
13	17	83	0	0	0.31	14	High	2	3	0	1	D
14	18	82	0	0	0.58	26	High	4	2	0	1	D
15	5	95	0	0	0.43	28	High	2	3	0	0	D
16	4	96	0	0	0.39	23	High	2	3	0	0	D
17	19	81	0	0	0.25	1	Low	0	0	0	0	C
18	14	86	0	0	0.21	0	No	0	0	0	0	C

disturbance and habitat characteristics were used as factors displayed on the nMDS. Two-way crossed Analyses of Similarity (ANOSIM) were used to look for effects of dredge level and habitat characteristics (Anderson and Walsh, 2013). Principle components analyses of community assemblages were used to identify which organisms best describe the community assemblages. To more fully examine the relationships between habitat variables, percent dredge and benthic community composition, distance based linear modelling (DistLM) was conducted using the PERMANOVA+ extension on Primer7®. The model analyzes the relationship between multiple variables (eelgrass, cobble, grain size, LOI, algae presence, algae type, and % dredge) and the proportion those variables contribute to the overall benthic community distribution (Mittermayr et al., 2020b; Legare et al., 2020b).

3 Results

3.1 Acoustic survey

Acoustic surveys took place over 10 days between 6 June 2015 and 6 August 2015 and resulted in data along 324 km of survey lines over 5.1 km^2 with an average depth of 7 m. All backscatter imagery was collected at a minimum of 200% overlap (i.e. the seafloor was mapped a minimum of two times). A geotiff raster was created at a pixel resolution of 0.5 m^2 (Fig. 2). Digitized dredge tracks covered a total area of 402 309 m^2 with maximum dredge track density up to 53% (e.g. 53 m^2 of dredge lines per 100 m^2; Fig. 3). Across the 18 benthic sample stations, nine were in areas of no dredging, five in low density areas (1–4% dredged) and four in high density areas (14–28% dredged) (Tab. 1).

3.2 Habitat analysis

Surveys for grain size, video and benthic invertebrates were conducted between 7/11/2015 and 8/6/2015. Grain size analysis for 18 stations showed that all stations are primarily sand (85 ± 12%) mixed with gravel (15 ± 12%; Tab. 1). Loss on Ignition (LOI) varied between 0.21 and 2.52%, indicating low organic content. Algae were present at 15 stations and eelgrass was present at 4 stations. Cobble was present at seven stations (Tab. 1).

Using 6 measured variables, (percent sand, LOI, algae, algae type, eelgrass and cobble) cluster analysis identified four distinct habitat types (Fig. 4). A nMDS plot of the habitat types with overlaying vectors visually describes the habitat composition (Fig. 4). Habitat type A ($n = 1$) has the highest percent LOI, lowest % sand and highest % gravel with higher presence of algae. Habitat type B ($n = 2$) has the highest coverage of eelgrass. Habitat type C ($n = 3$) is identified as bare sand with absence of algae, cobble, or eelgrass. Habitat type D ($n = 12$) is the most abundant habitat type described by a high percentage of sand, presence of algae and cobbles (Tab. 1, Fig. 4). The presences of one habitat characteristic is not mutually exclusive of the others (e.g. algae on cobble in sand).

3.3 Benthic invertebrate analysis

Diversity indices calculated include number of species and Shannon's diversity Index (H') were estimated for each station for comparison across habitats (A–D) and dredge levels. Benthic assemblages at 18 stations contained 152 different species across 93 families and 9 phyla. The only station classified as habitat type A ($n = 1$) contained 48 species.

Table 2. Density of invertebrates (m^{-2}) of each benthic invertebrate group at each station.

Station	Habitat type	Amphipod	Arthropoda	Balanidae	Bivalve	Echinoderm	Hirudinea	Cnidaria	Crab	Gastropod	Halacaridae	Paguridae
1	B	2918	8	0	1093	308	0	0	0	4108	0	0
2	B	6358	0	0	768	743	0	18	33	6193	0	33
3	D	1600	0	0	693	118	0	25	0	5718	18	8
4	A	1750	0	0	16368	1143	0	0	0	1025	18	0
5	D	475	0	0	15433	1183	0	0	8	1600	68	8
6	D	33	0	0	13293	1233	0	0	0	743	8	0
7	D	0	0	0	46718	575	0	0	8	450	8	0
8	D	25	0	0	29900	300	0	0	25	618	8	0
9	D	93	0	0	5800	250	0	75	0	843	0	0
10	C	8	0	8	12420	920	0	95	0	488	63	0
11	D	8	25	0	6143	633	0	0	0	75	68	0
12	D	0	0	0	450	433	33	0	0	0	0	0
13	D	83	0	8	3050	75	0	0	0	1883	8	0
14	D	75	0	0	288	25	0	0	0	400	0	0
15	D	113	0	0	5525	250	0	0	0	925	0	0
16	D	8	0	8	2575	108	0	25	0	625	0	0
17	C	0	0	0	2843	368	0	8	0	18	43	0
18	C	0	0	0	75	388	0	0	0	0	13	0
Station	Habitat type	Hydractiniidae	Isopod	*Microporella ciliata*	Oligochaeta	*Phascolopsis gouldii*	Platyhelminthes	Polychaete	Porifera	Crangonidae	Sipuncula	Tanaid
1	B	0	100	0	1808	0	0	1293	0	0	0	0
2	B	0	8	0	543	0	0	2518	0	8	0	43
3	D	0	18	0	43	0	0	4450	18	0	0	0
4	A	8	58	0	0	0	0	2108	0	8	0	0
5	D	0	443	0	568	0	0	3725	0	18	0	0
6	D	8	258	0	225	0	0	4500	0	8	0	0
7	D	0	358	0	183	0	0	5400	0	0	0	25
8	D	0	875	0	2868	0	0	7058	0	25	0	0
9	D	0	143	0	3233	0	8	3258	0	0	50	0
10	C	0	295	0	10 338	0	8	7688	0	13	8	0
11	D	0	518	0	4650	18	0	3100	0	0	0	0
12	D	0	193	0	158	0	0	750	0	0	8	0
13	D	0	33	8	10 333	0	0	6318	18	0	0	0
14	D	0	25	0	1113	0	0	3563	0	0	0	0
15	D	0	50	0	18 088	0	0	13 263	0	13	0	0
16	D	33	175	0	11 833	8	0	6950	0	0	0	0
17	C	0	93	0	108	0	0	4283	0	18	0	0
18	C	0	0	0	0	0	0	6063	0	0	0	0

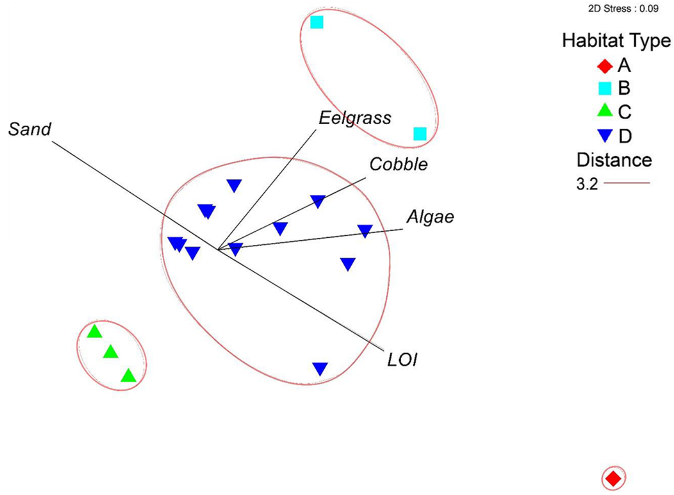

Fig. 4. Non metric multidimensional scaling plot showing habitat classification groups identified as habitat A ($n = 1$): the highest percent LOI and lowest % sand, B ($n = 2$): having the highest coverage of eelgrass, C ($n = 3$): sand with absence of algae, cobble, or eelgrass, and D ($n = 12$): mixed habitat with high percent sand. Classification is based on cluster analysis of all habitat variables.

Stations in habitat type B ($n = 2$) contained 98 different species overall. Stations in habitat type D ($n = 12$) contained a total of 127 different species and ranged from 20 to 68 species per station. Habitat type C stations ($n = 3$) had the lowest species diversity with 44 species overall and individual stations containing 10–37 species (Tab. 2; Fig. 5). Density of individual invertebrates across all stations ranged between 1108 and 31413 m^{-2} (9156 ± 430 m^{-2}; average ± SE). Habitat type A ($n = 1$) contained a density of 2180 m^{-2} invertebrates, where habitat type B ($n = 2$) contained a density of 3118–11632 m^{-2}. Habitat type C ($n = 3$) had a density of 9367 ± 2528 m^{-2} (average ± SE) and habitat type D ($n = 12$) contained 9914 ± 713 m^{-2} (average ± SE). Shannon's diversity across all stations ranged from 0.83 to 2.89 (2.02 ± 0.53; average ± STDEV). Shannon diversity by habitat type A ($n = 1$) was 2.092, B ($n = 2$) contained a density of 2.107–2.694, C ($n = 3$) had a density of 2.177 ± 0.52 m^{-2} (average ± STDEV) and habitat type D ($n = 12$) contained 1.904 ± 0.57 m^{-2} (average ± STDEV).

Benthic invertebrate community assemblages represented in habitat type (D), the most abundant habitat type ($n = 12$), showed a significant difference between the dredge treatment areas (ANOSIM: significance level: 0.1%; $R = 0.616$). Further, the lowest similarity was detected between "high" and "no" dredge stations (Significance level at 0.5%, $R = 0.794$) (Fig. 6). Principle components analyses of community assemblages indicated that oligochaetes and polychaetes are the benthic invertebrate community members most descriptive of areas of "high" dredge, whereas bivalves, echinoderms, and isopods were indicative of areas "no" dredge. Areas with "low" dredging activity contained higher species diversity with species from both "high" and "lno" dredge treatments present (Fig. 7).

More specifically, the polychaetes identified were comprised of 51 different species. Polychaetes in the subclass Errantia were most abundant in "high" dredge areas and least abundant in "low" and "no" dredge areas, whereas polychaetes in the subclass Sedentaria were least abundant in the "high" dredge areas and more abundant in "no" dredge areas (Fig. 7). This distribution is supported as Errantia are characterized as mobile free-swimming worms where polychaetes in the group Sedentaria are known to be sessile tube building worms.

DistLM indicated that the eelgrass and dredge level were the most influential variables in the community distribution. Results of PRIMER's DistLM show that a total of 36.09% species distribution can be explained by the combination of eelgrass ($p = 0.009$) and dredge level ($p = 0.013$). As benthic communities are known to be vastly different in eelgrass compared to other habitat types, eelgrass stations ($n = 2$) were subsequently removed from the analysis as the presence of these stations can overshadow the differences between dredge and non-dredge locations within habitat type (D). When DistLM was performed on stations excluding eelgrass, the variable of percent dredge track explained 46% of the overall distribution ($p = 0.004$).

4 Discussion

This study found clear differences in both biodiversity and community composition across dredge treatments. Dredging effects on the benthic community in sandy, high-energy areas are thought to be short lived and are considered to be overshadowed by the dispersal of sediment by currents and waves (Johnson, 2002; Mercaldo-Allen and Goldberg, 2011). Herring Cove is a shallow (≤10 m) area with a very dynamic shoreline and active sediment transport (Giese et al., 2011). Intuitively, a physical alteration of the seabed by a hydraulic clam dredge would be overshadowed by natural events (e.g. storms), yet the physical disturbance is slow to recover and can last multiple years (Legare et al., 2020a). The recovery rate for benthic communities in this area is unknown, however, the benthic invertebrate community composition differs with proximity to different levels of hydraulic clamming several months after the disturbance occurred.

Baseline inventories are not only a reference level in which to measure future activities against, but are most often associated with a pristine state of an ecosystem (Samhouri et al., 2011). Pristine ecosystems are those systems absent of human activity or exposed to minimal anthropogenic pressures (Halpern et al., 2007; Samhouri et al., 2011). Pristine ecosystems are decreasing globally but contemporary baseline data can be drawn from protected areas such as Marine Protected Areas (MPA; Samhouri et al., 2011). The area off Herring Cove, absent of substantial benthic fishing activity due to federal protection within National Park Waters and local town moratoriums, acted as a de facto MPA and was a suitable site to create a baseline of benthic macroinvertebrate community habitats until the fishing commenced in the winter of 2014/2015. Hydraulic dredging for clams was thought to have a minimal impact on the local benthic substrate in this area, but the use of acoustic surveys provided a complete picture and allowed us to post-stratify our benthic invertebrate samples across various levels of disturbance (Legare et al., 2020a).

The dredging across low-relief unconsolidated sediment found a >90% loss of biogenic structures such as burrowing invertebrate species (Gilkinson et al., 2003). Similar low-relief unconsolidated sediments can be found at Herring Cove with consistent sediment characteristics throughout the study site as shown by grain size and video analysis. The presence of habitat type D across areas of “high”, “low”, and “no”, allowed for an adequate number of stations to be used for references. The study as presented is what is considered a “Natural” experiment compared to a planned “Controlled” experiment, and adequate sampling was determined by post-survey inspection of the data. Stations (each with 3 samples per location) were distributed at a density of 3.5^{-km2} and fishing intensity was quantified a resolution 10 m^2. Compared to other natural experiments which often use proxies for fishing intensity (e.g. vessel monitoring systems) at densities as low as 1 sample per square kilometer (Sköld et al., 2018; Queirós et al., 2006), this study represents one of the finer-scaled studies that did not require scuba divers.

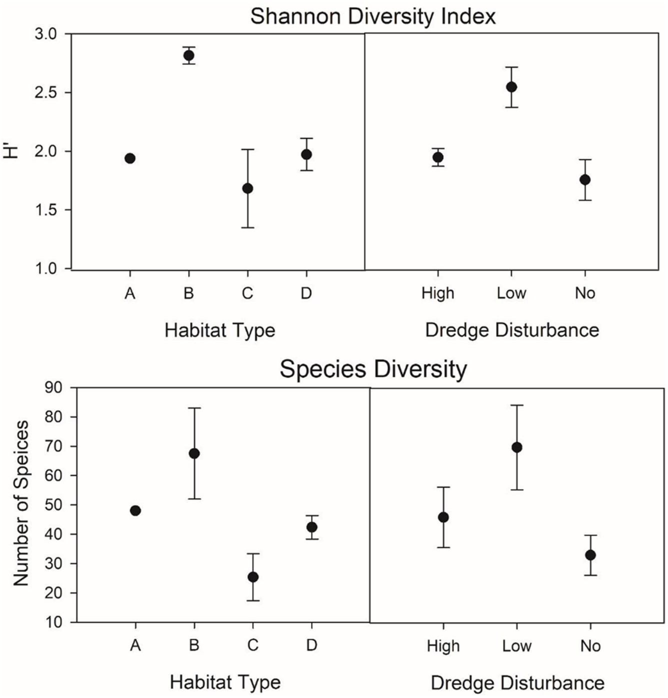

Fig. 5. Shannon diversity index (mean ± SE; top) and species diversity (mean ± SE; bottom) across habitats and across dredge disturbance in habitat type D.

Benthic invertebrate diversity within the most common habitat type D (coarse sand and cobble with algae) shows distinct differences between areas of “high”, “low”, and “no” dredging. Areas of “high” dredging contained greater abundances of oligochaetes and polychaetes and low relative abundances of bivalves, echinoderms and isopods. Bivalves, echinoderms and isopods have lower to no mobility, high site fidelity to benthic structures and/or have not entered into a recruitment period since the disturbance occurred (Medeiros-Bergen et al., 1997; Patricio and Dearborn, 1989; Whitlatch, 1977), thus explaining the low abundances. Polychaete and oligochaete species identified in this study are highly mobile opportunists that can colonize a recently disturbed area following specific succession patterns (Hilbig and Blake, 2000). Interestingly, low levels of disturbance led to a mixture of opportunistic species and site attached, low mobility organisms, consequently resulting in the highest species abundance and diversity that was found. This fits directly with the intermediate disturbance hypothesis, in which varied levels

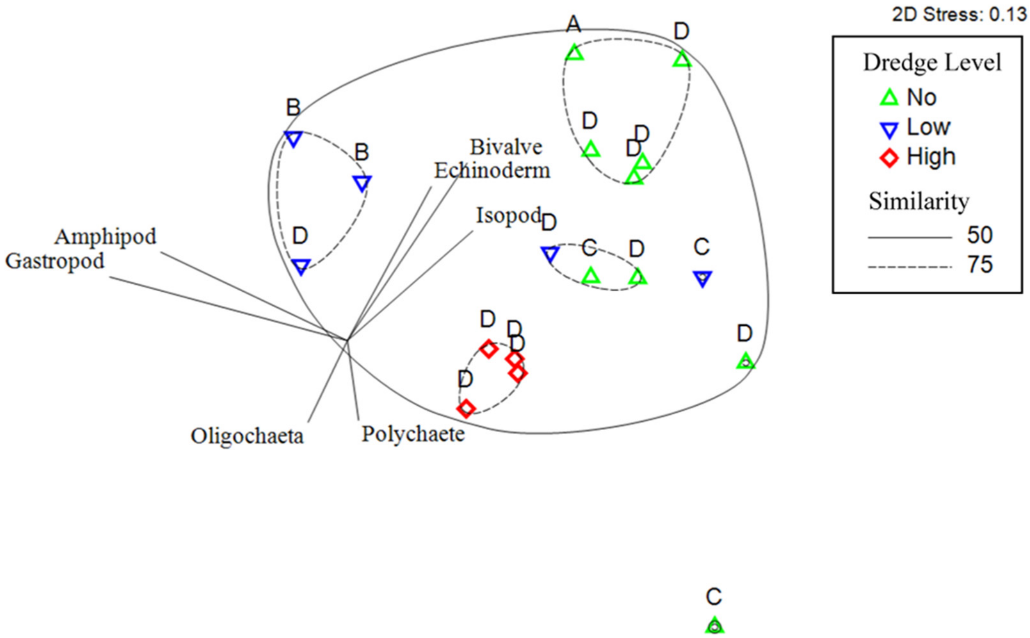

Fig. 6. Non metric multidimensional scaling plot of invertebrate communities across each site with trajectories indicating which species has the largest influence on the overall invertebrate community distribution. Symbols represent level of dredge: “high” (>10%), “low” (<10%), and “no” (0%) and letters (A–D) are representative of habitat types. Principle components analysis was used to determine which species group influenced described the invertebrate communities represented here by the vectors.

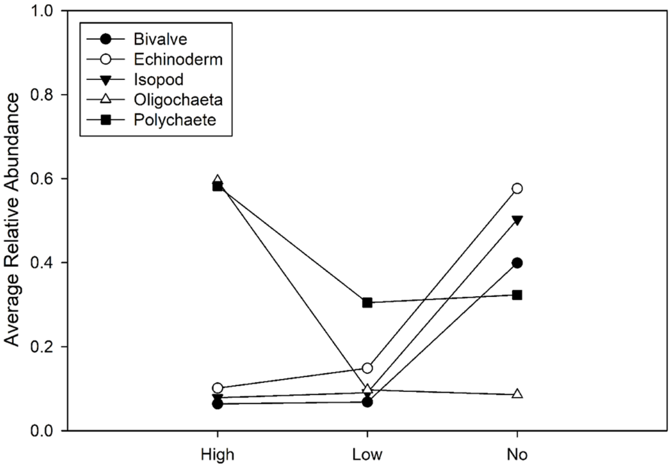

Fig. 7. Relative abundance of different species groups across disturbance level (High-Low) for stations in habitat type (D).

of natural or anthropogenic disturbances result in higher biodiversity and species richness compared to newly formed habitats or mature habitats (Connell, 1978; Lasiak and Field, 1995).

The objective of the acoustic surveys and bottom grab sampling conducted in 2015 was to create a baseline benthic habitat map (Borrelli et al., 2019). Although the investigators were aware of the dredging that had occurred in the area during the previous winter, both the extent and density of dredge tracks was unknown. The combination of high-resolution acoustic mapping and benthic sampling allowed for greater interpretation of the benthic invertebrate communities than grab samples alone. Baseline datasets are useful tools to measure future change and to set ecosystem goals against but the objective during this survey, to create a baseline data set, is offset by the anthropogenic disturbance of hydraulic clamming in the area. Ideally an acoustic survey would be performed prior to surveying allowing for appropriate stratification of stations (Mittermayr et al., 2020a) with the caveat that the further away from the impact event the more difficult it will be to capture the broad ranging effects.

This study was able to successfully stratify stations across dredge treatment after sampling was conducted, but that is not always possible. With an average distance of ~500 m between stations, disturbances (e.g. clam dredging) could be detected and quantified based on benthic invertebrates. Studies of different scales (e.g.10s of kilometers or 10s of meters) may miss the disturbed area due to randomized sampling or may reside entirely within a disturbed area, thus mis-representing local anthropogenic disturbances. Adequate stratification across treatments increases the statistical power of analyses and can be accomplished by post-acoustic survey sampling (Mittermayr et al., 2020a). As the longevity of dredge tracks is site and habitat dependent, there is often little time to spare between dredge events, acoustic survey, and benthic sampling across the disturbance if the extent is unknown.

5 Conclusion

The results of this study show that long-term monitoring is needed in order to create an accurate baseline, allowing the separation of natural versus dredge affected community assemblages. Hydraulic clamming at the intensity found at herring cove changes the benthic invertebrate community yet the long term ecological effects are unknown. Creating habitat maps to assess future anthropogenic disturbances should incorporate local knowledge and a detailed history of past and present anthropogenic disturbances. Here we identify that interpretation of benthic invertebrate community composition were clearer when dredge level was taken into account. Targeted future biological surveys need to document the level, type and spatial distribution of anthropogenic alterations to benthic habitats as well as the long term ecological consequences. When possible, greater emphasis should be put on using habitat maps as points of reference and/or statement of condition of the habitat instead of viewing them as baseline inventories which can mistakenly be referred to as pristine, natural conditions and an ecosystem goal to work towards. Additional monitoring can be used to determine the accuracy of previously established baselines and will track the stabilization of the community composition which can then serve as a reference point to measure future natural and anthropogenic impacts and restorations.

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