




Impact and assessment of hydraulic rake trawling on benthic communities in *Urechis unicinctus* aquaculture zones, Laizhou Bay, China

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ARTICLE INFO

Keywords:

Urechis unicinctus
Hydraulic rake trawling
BACI
Ecological indices

ABSTRACT

Hydraulic rake trawling is essential for *Urechis unicinctus* aquaculture. However, its ecological impacts on benthic ecosystems remain poorly understood, especially in China. This study used a Before-After-Control-Impact design to evaluate the effects of hydraulic rake trawling in Laizhou Bay, China. Sampling was conducted across high, low, and control harvesting intensities before, during, and four months after harvesting. Macrofaunal diversity metrics, such as total abundance, total biomass, species richness, Shannon-Wiener index, Margalef index, and dominance index, were analyzed using a two-way analysis of variance (ANOVA) and the Scheirer-Ray-Hare test (SHR). Community structure was assessed through non-metric multidimensional scaling (NMDS), analysis of similarities, similarity percentage, and permutational multivariate ANOVA (PERMANOVA). Benthic habitat health was assessed using multiple ecological indices, specifically the AZTI Marine Biotic Index (AMBI), Norwegian Sensitivity Index (NSI), Norwegian Quality Index 1 (NQI1), and Dredge Spoil Disposal Index (DSD). Significant temporal changes in macrofaunal diversity were observed. Species richness declined from 14.67 ± 2.08 to 3.00 ± 1.00 species during harvesting, recovering to 12.33 ± 0.58 species after harvesting. The Shannon-Wiener index decreased from 2.57 ± 0.15 to 1.27 ± 0.32 during harvesting, recovering to 2.33 ± 0.12 after harvesting. Dominant species shifted from *Paraprionospio pinnata* (EG IV) and *Umbonium thomasi* (EG II) before harvesting to *Nephtys polybranchia* (EG II) and *Iridona iridescens* (EG I) during harvesting, and *Nephtys polybranchia* (EG II) and Virgulariidae (EG I) after harvesting. The dominant functional groups changed from F/D (Deposit-feeder) and M/C (Crawl) to F/P (Predator) and M/B (Burrow). Ecological indices mirrored these changes, showing declines during harvesting and subsequent recovery. These findings highlight the resilience and vulnerability of benthic ecosystems to hydraulic rake trawling.

1. Introduction

Anthropogenic activities related to marine resource utilization are primarily driven by supporting the food supply and deriving economic benefits. However, excessive exploitation of marine resources poses significant risks to marine environments and ecosystems, often leading to negative impacts on marine life and human well-being (Balmford et al., 2002; Wackernagel et al., 2002).

With advancements in manufacturing and aquaculture technology,

particularly hydraulic technology, mechanized harvesting methods have developed rapidly. Research indicates that mechanized fishing gear is one of the primary anthropogenic activities disturbing benthic habitats and macrofaunal communities in marine ecosystems (Clarke et al., 2018). The novel hydraulic rake trawl, now employed to harvest the fat innkeeper worm *Urechis unicinctus* (Drasche, 1880), should be evaluated for its potential impact on local benthic communities.

The polychaete *U. unicinctus* is a species with high economic value. According to surveys conducted from 2000 to 2010 (Qu, 2011), this

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<https://doi.org/10.1016/j.ocecoaman.2025.107969>

Received 20 March 2025; Received in revised form 8 October 2025; Accepted 9 October 2025

Available online 12 October 2025

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species occupies approximately 70 km² in nearshore zones of Laizhou Bay in southern Bohai Sea. The species has vigorous growth and reproduction and experiences a high fishing intensity, with around 200 medium-sized fishing boats engaged in its harvest in this area. The fishing gear includes bottom rakes equipped with water pumps to impact the seabed. Since 2000, the annual production in the Weifang coastal zone has increased from several thousand tons to tens of thousands of tons, with an economic value of tens of millions of dollars (Qu, 2011).

Mechanized harvesting, in particular, can alter macrofaunal diversity and community structure, with many studies highlighting its potential to cause long-term impacts on biodiversity and ecosystem function. However, based on the published literature, the physical, biological, and chemical effects of shellfish dredging within the inshore coastal zone are generally short-lived, with the rate of recovery varying among studies (Gandara-Martins et al., 2020). However, there is relatively little research on the environmental impact of mechanized harvesting in China. In 2013, the Chinese Ministry of Agriculture implemented a ban on traditional rake nets, leading to the adoption of modified harvesting gear in the region.

In this study, we employed the Before-After-Control-Impact approach to evaluate the impact of hydraulic rake trawling on benthic communities (Clarke and Tully, 2014; Baeta et al., 2021). Measurements and sampling were performed before, during, and after harvesting across three zones with varying harvesting intensities: high intensity

(HHI), low intensity (LHI), and (no harvesting) control zones. The findings will be crucial for understanding the effects of this mechanized harvesting gear on exploited habitats, providing theoretical support for the optimization and standardization of management practices for *U. uncinatus* mechanized harvesting equipment. This study aims to: (1) evaluate the effects of hydraulic rake trawling on benthic communities; (2) quantify the magnitude and extent of these impacts using multiple ecological indices.

2. Materials & methods

2.1. Study area

The study was conducted in Laizhou Bay, southern Bohai Sea, Shandong Province, China (Fig. 1).

This semi-enclosed bay, with an inlet length of approximately 96 km and an average water depth of less than 10 m, serves as a vital marine ecosystem and a key fishing ground. Laizhou Bay receives inputs from 39 rivers, which deliver substantial sediment and nutrients, fostering high primary productivity and supporting diverse benthic communities. Among the species in this region, *U. uncinatus* holds significant economic and ecological value. The bay's environmental conditions provide a complex but favorable habitat for the growth and development of benthic fauna, underscoring its role as a critical fishing ground. We set five stations (L1, L2, L3, C1, C2) to collect the *U. uncinatus* sample

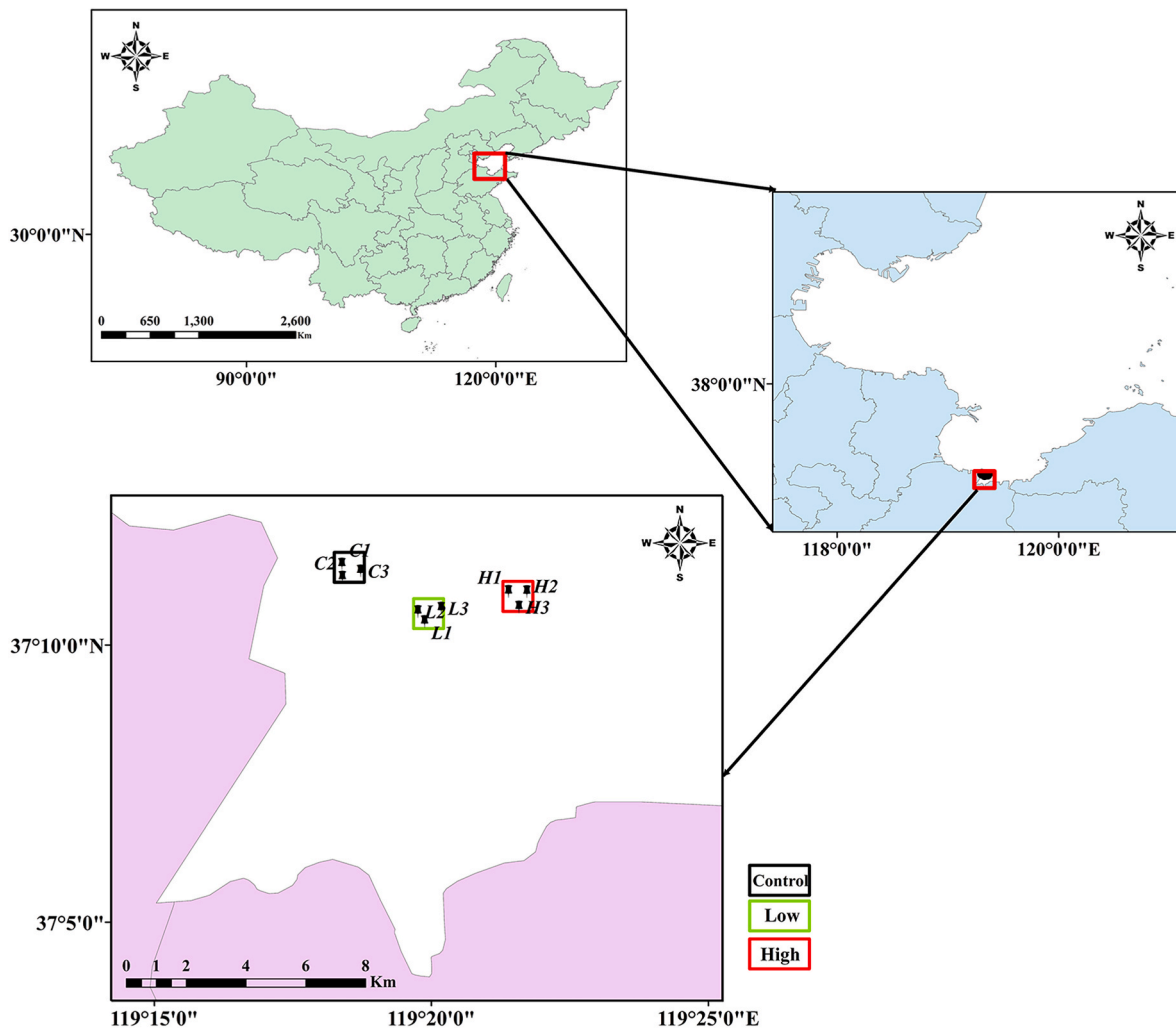


Fig. 1. The location of the study area and sample locations for high- (H1 to H3, HHI) and low-intensity harvesting zones (L1 to L3, LHI) and the control zone (C1 to C3, black).

in August 2023, the average abundance in control area was 5 ± 0.9 ind/ m^2 , the average biomass in control area was 8.93 ± 3.52 g/ m^2 . The average abundance in LLI area was 3.53 ± 1.50 ind/ m^2 , the average biomass in LLI area was 8.35 ± 2.51 g/ m^2 .

The surface water environmental parameters of Laizhou Bay are temporally and spatially variable, reflecting the dynamic nature of this coastal ecosystem. During the survey period, the salinity ranged from 16.7 ± 5.89 to 31.35 ± 0.04 , and the pH levels remained stable at 8.06 ± 0.06 – 8.22 ± 0.06 . Additionally, the dissolved oxygen concentrations fluctuated between 6.68 ± 0.11 and 9.42 ± 0.17 mg/L, the particulate organic matter levels ranged from 5.54 ± 1.21 to 12.13 ± 5.14 mg/L, and the chlorophyll-a concentrations varied from 3.00 ± 0.90 to 14.60 ± 2.15 μ g/L (Appendix A).

2.2. Hydraulic rake trawl for *U. unicinctus*

The hydraulic rake trawl ($2 \times 1.7 \times 0.6$ m) is a mechanized harvesting device (Fig. 2) that is specifically designed to optimize the efficiency of *U. unicinctus* collection while minimizing disturbance to seabed sediments (Laizhou Bay Marine Technology Co, 2021). The gear has a trapezoidal steel frame encased in a mesh net, equipped with an integrated rigid water pipe system and sediment-scraping blades fitted with downward-facing nozzles. Adjustable-height rollers on either side of the frame enable precise control of the rake's penetration depth, which is typically set at 30 cm to align with the habitat layer of *U. unicinctus*. A sediment-leveling blade at the rear of the frame smooths the seabed post-harvest, mitigating turbidity and minimizing habitat disruption. This gear is categorized as a single-vessel towing hydraulic rake trawl for burrowing echinoderms, as described in relevant classifications.

With a harvesting capacity of 15–20 kg per net and a maximum

penetration depth of 60 cm, this gear is employed during the harvesting seasons from February to May and September to November but it is prohibited from May 1 to August 31 during the fishing moratorium. Compared with traditional methods, this gear demonstrates significant improvements in efficiency and environmental sustainability.

During operation, the hydraulic rake trawl is towed by the production vessel at a speed of 0.1 knots (0.1852 km/h). A high-pressure water pump supplies water to vertical nozzles, which dislodge *U. unicinctus* from the sediment, while horizontal nozzles direct the dislodged organisms into the net bag. The rake's penetration depth is precisely controlled by adjustable rollers, which minimize habitat disturbance due to the metal frame. Additionally, a sediment-leveling blade further reduces sediment dispersal, helping to preserve the benthic habitat. Vertical nozzles embedded in the sediment-scraping blades further mitigate damage to the target species.

2.3. Sampling design and measurements

Sampling for this study was conducted at three time points: August 29, 2022 (before harvesting), April 27, 2023 (during harvesting), and August 17, 2023 (after harvesting). Measurements and sampling were performed before, during, and after harvesting across three zones with varying harvesting intensities: high intensity (HHI), low intensity (LHI), and (no harvesting) control zones.

Harvesting activities were tracked using Global Positioning System and vessel speed data from Shipxy (www.shipxy.com) and onboard black-box systems, with valid harvesting periods defined as a minimum of 5 min at speeds of 0.1 knots. This study used a fishing density-based Sweep Area Ratio calculation (SAR) method to more accurately quantify the spatial impact of these harvesting activities. This approach comprehensively accounts for the number of vessels per unit area,

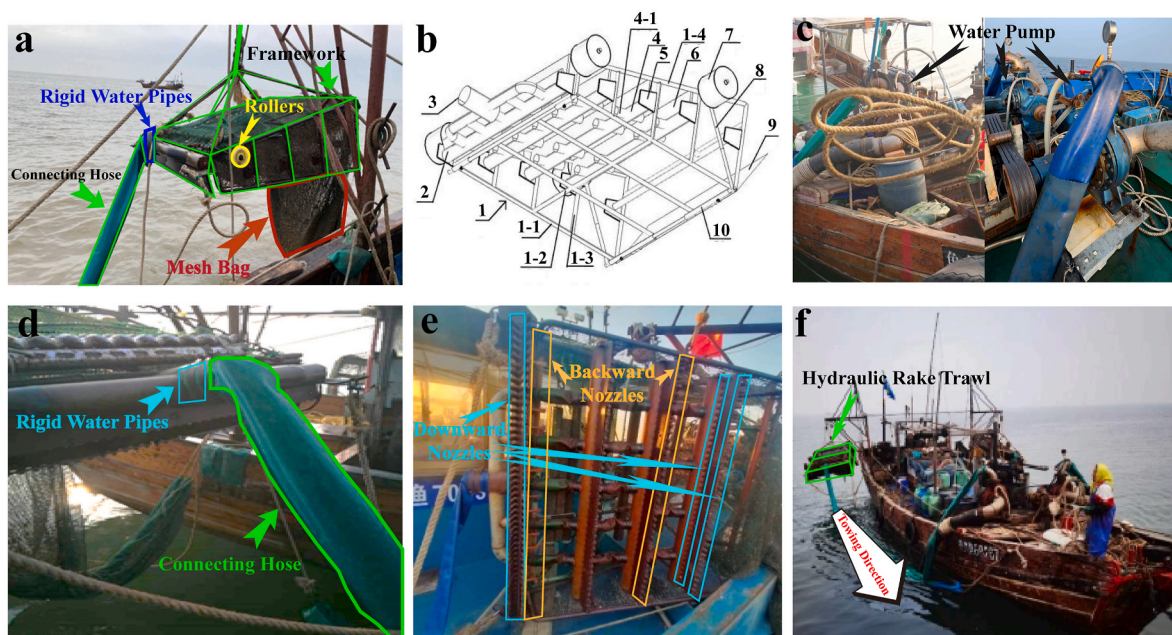


Fig. 2. Structural design and operational setup of the hydraulic rake trawl. **Notes:** a. The hydraulic rake trawl in a suspended state before deployment. Key components include the rigid water pipes (blue), connecting hose (green), mesh bag (red), rollers (yellow), and the trapezoidal framework (green outline), which supports and stabilizes the gear. b. Structural schematic diagram of the trawl gear. The diagram illustrates the trapezoidal frame (1-1 to 1-4), rigid water pipe system (2), connecting hose (3), Secondary Rigid Pipelines (4), Fixing Plate (5), Sediment Scraper (6), rollers (7), Rotating Shaft and Connecting Rod (8), and sediment-leveling blade (9). The framework design ensures efficient operation and minimal seabed disturbance. c. Production vessel equipped with a water pump, providing high-pressure water to the hydraulic rake trawl, which is crucial for driving the sediment-flushing nozzles. d. Close-up view of the rigid water pipes (blue) and connecting hose (green), which direct high-pressure water to the sediment-flushing nozzles embedded in the sediment-scraping blades. e. Detailed view of the sediment-scraping blades, showing downward nozzles (blue) for sediment disturbance and backward nozzles (orange) for directing *U. unicinctus* into the mesh bag. f. Operational setup of the hydraulic rake trawl during harvesting. The trawling gear is towed behind the production vessel, with the framework ensuring stability and efficient collection of *U. unicinctus* from the seabed. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

reflecting the overall intensity of the harvesting activities. The formula is as 'equation (1)':

$$SAR_{density} = \frac{\text{Net width} \times \text{Harvesting duration} \times \text{Net speed} \times D}{\text{Total zone}} \quad (\text{eqn 1})$$

where the net width is 1.7 m, the daily harvesting duration is 0 h/day in the control area, 3 h/day in the LHI area, and 8 h/day in the HHI area, the net speed is 0.1852 km/h, D is the fishing density (number of vessels/km²), and the total zone area is 1 km².

Based on the number of vessels in the different harvesting intensity zones, the values for the fishing density (D) were as follows: Control Zone: 0 vessels/month; LHI Zone: 2–3 vessels/month (average D_{low} = 2.5 vessels/km²/month); HHI Zone: 13–15 vessels/month (average D_{high} = 14 vessels/km²/month). Each zone covered 1 km², with 1.8 km between adjacent zones. Within each intensity level, three sampling stations were established (Fig. 1).

Macrofaunal samples were collected using a 0.05 m² Van Veen grab, with three replicates taken at each station. The collected samples were sieved through 0.5 mm mesh and preserved in 95 % ethanol. In the laboratory, macrofauna were identified to the lowest possible species level, counted, and weighed (to the nearest 0.001 g). Species names were validated using the World Register of Marine Species database (WoRMS Editorial Board, 2023. World Register of Marine Species. Available from <https://marinespecies.org>. Accessed, 2023-08-20).

2.4. Statistics

2.4.1. Parameters for assessing macrofaunal diversity

Macrofaunal diversity was assessed using the total abundance (total number of individuals in each zone), total biomass (sum of the biomass of all the species in each zone), species richness (number of different species present (Silva et al., 2016)), Shannon-Wiener index (H') (diversity index accounting for both richness and evenness of species) (Shannon, 1963), Margalef index (D) (species richness relative to the number of individuals (Margalef, 1958)), and dominance index (Y) (proportion of individuals of the most abundant species) (McNaughton, 1967). The formula are as follows:

$$H' = - \sum_{i=1}^S p_i \log_2 p_i$$

$$D = (S - 1) / \ln N$$

$$Y = f_i \times p_i$$

In the formula, S is the total number of species. N is the total number of individuals. $p_i = n_i/N$, p_i is the proportion of the i species in the total number of individuals, where n_i is the number of individuals of the i species. f_i represents the frequency of the i species at a particular site. Species is considered dominant if $Y \geq 0.02$. These metrics were calculated using the species numbers and abundance at each zone and Primer v6 and PERMANOVA+ (Clarke and Gorley, 2005).

We choose two biological traits (Mobility, Feeding guild) to analyse the change of ecological function (Veríssimo et al., 2012). Based on it, macrobenthic fauna were classified into five groups: Predator (F/P), Herbivorous (F/H), Deposit-feeder (F/D), Filter-feeder(F/F). The mobility was classified into five groups: Sessile (M/SE), Swim (M/SW), Burrow (M/B), Crawl (M/C), Walk (M/W).

2.4.2. Benthic habitat health indices

The benthic habitat health status was assessed using the following biotic indices: AZTI Marine Biotic Index (AMBI), Norwegian Sensitivity Index (NSI), Norwegian Quality Index 1 (NQI1), and Dredge Spoil Disposal (DSD) Index. The AMBI, originally developed for European coastlines, categorizes species based on their tolerance responses to broad-spectrum pollution stressors (Borja et al., 2000; Rygg and Norling,

2013). The NSI, designed specifically for assessing soft-bottom macrofauna ecological quality, defines sensitivity thresholds through empirical calibrations of Norwegian indigenous species (Johansen et al., 2018). The NQI1 includes the AMBI as a measure of sensitivity, with the diversity component described by a factor SN. (Rygg, 2006). The DSD index compares indicator species abundance between impacted and unimpacted sites to evaluate disturbance levels, with higher values indicating greater disturbance (Dauvin et al., 2018).

The formula are as follows:

$$NSI = \sum_i^S \left[\frac{N_i \cdot NSI_i}{N_{NSI}} \right]$$

Where N_i is the number of individuals of species, NSI_i is the sensitivity value of species, N_{NSI} is the total number of individuals assigned a sensitivity value.

$$NQI1 = \left[0.5 \cdot \left(\frac{1 - AMBI}{7} \right) + 0.5 \cdot \left(\frac{\frac{\ln(S)}{\ln(\ln(N))}}{2.7} \right) \cdot \left(\frac{N}{N + 5} \right) \right]$$

Where AMBI is the Marine Biotic Index, S is the number of species, and N is the number of individuals in the sample, $SN = \frac{\ln(S)}{\ln(\ln(N))}$

These indices were calculated using the BBI package in R software (Cordier and Pawlowski, 2018).

2.4.3. Statistical analysis methods

Prior to conducting statistical testing, the data distributions and homogeneity of variance for all the metrics data were assessed using the Shapiro-Wilk (Welham et al., 2014) and Levene's tests (O'Neill and Mathews, 2000). Variables meeting the assumptions of normality and with equal variances were analyzed using a two-way analysis of variance (Shahbaba, 2012) to examine the effects of harvesting time (Before, During, and After harvesting) and the harvesting intensity (Control, Low, and High). For these parametric datasets, significant interactions were further explored using Tukey's honestly significant difference post hoc test. For variables that failed to meet the parametric assumptions, the Scheirer-Ray-Hare test was used, a non-parametric alternative to a two-way ANOVA (Scheirer et al., 1976). Where significant effects were detected under non-parametric conditions, the Kruskal-Wallis test was used followed by Dunn's post hoc tests with Bonferroni correction to identify pairwise differences.

The macrofaunal community composition was examined using square-root transformed species abundance data and performing non-metric multidimensional scaling (nMDS) using Bray-Curtis similarity in Primer v6 (Clarke et al., 2006). Then, a two-way analysis of similarity (ANOSIM) was used to evaluate community differences across harvesting intensities and time periods. Similarity percentage analysis (SIMPER) identified the species contributing the most to community dissimilarities (Clarke et al., 2001). Additionally, permutational multivariate ANOVA (PERMANOVA) was used to test the effects of time (Before, During, and After harvesting) and treatment (Control, Low, and High) on community structure, with both factors treated as fixed effects (Anderson, 2005).

The temporal, treatment, and interaction effects on the DSD, AMBI, NSI, and NQI1 indices were analyzed using the same parametric or non-parametric criteria as described above. Observations missing NSI values (15 % of the dataset) were excluded to ensure consistency, and this exclusion did not significantly impact the robustness of the analyses.

All statistical analyses were performed with a significance level of $p < 0.05$. Univariate and non-parametric tests were conducted in R 4.0.5 (R Core Team, 2020) using the "rcompanion," "stats," "FSA" and "BBI" packages, while the nMDS, SIMPER, ANOSIM, and PERMANOVA analyses were performed in Primer 6 and PERMANOVA+.

3. Results

3.1. Benthic community

3.1.1. Macrofaunal structure and composition

A total of 67 macrofaunal species were identified. Before harvesting, the polychaetes (18 species) and mollusks (13 species) were the dominant species, with *Paraprionospio pinnata* and *Umbonium thomasi* being the most abundant species (Table 1).

During harvesting, the number of species and individuals for polychaetes and mollusks decreased (12 species and 7 species, respectively), with *Nephtys polybranchia* and *Iridona iridescens* as the dominant species. After harvesting, the polychaetes and mollusks recovered, with the number of species and individuals increasing to 20 species and 7 species, respectively, with *Nephtys polybranchia* and as the dominant species.

The macrobenthic assemblages in the control area were consistently dominated by taxa exhibiting a burrowing movement mode (M/B). The representative species, *I. iridescens*, was present throughout all three sampling periods. In terms of feeding guilds, deposit-feeders (F/D) and predators (F/P) predominated. Predators (F/P) was represented by *N. polybranchia*, whose average abundance increased to 33.33 ind/m² after harvesting, establishing it as the most dominant species.

In the LHI area, the dominant mobility changed from crawl (M/C)—exemplified by *U. thomasi*, which reached an abundance of 984.44 ind/m² and a biomass of 114.47 g/m² before harvesting—to burrow (M/B). The feeding guild also changed from deposit-feeding (F/D) dominance to predatory (F/P) dominance, with the abundance of *N. polybranchia* increasing to 31.11 ind/m² after harvesting.

In the HHI area, the mobility remained co-dominated by both burrow (M/B) and sessile (M/SE) taxa. Among these, the sessile family Virgulariidae sp. was predominant, with average abundance changing from 37.78 ind/m² before harvesting to 40.00 ind/m² after harvesting, again becoming the most dominant taxon. The feeding guild groups changed from three types (F/D, F/F, F/P) before harvesting to being dominated by two types (F/D and F/F).

Multivariate Analysis (Table 2) revealed that the sampling time factor (Ti) significantly influenced the community structure (Pseudo-F

= 4.6422, *p* = 0.001). The harvesting intensity (Tr) also had a significant effect (Pseudo-F = 1.5743, *p* = 0.021). However, the interaction between time and treatment (Ti × Tr) did not have a significant impact (Pseudo-F = 1.2342, *p* = 0.107). Among the factors, time accounted for the largest proportion of observed variation (30 %), almost twice the contribution of differences in treatment (11.91 %).

The nMDS ordination revealed significant temporal and spatial differences in macrofaunal assemblages, particularly during the harvesting period. Typically, stations in the LHI and HHI zones exhibited higher dispersion compared with the control zone (Fig. 3).

The SIMPER analysis showed an average dissimilarity of 88.2 % between the communities before and after harvesting, 84.7 % between during and after harvesting, and 64.6 % between before and during harvesting (Appendix B-D). The main contributing species before, during, and after harvesting were Virgulariidae (66.9 %), *Iridona iridescens* (56.6 %), and *Nephtys polybranchia* (52.4 %). The average dissimilarity between samples from the control and HHI zones was higher than that between the control zone and LHI zone, with the main contributing species being *Iridona iridescens* (119.9 %), *Nephtys polybranchia* (91.2 %), *Scoloplos* (86.1 %), *Umbonium thomasi* (76.1 %), and *Nitidotellina lischkei* M. Huber, Langlet & Kreipl, 2015 (56.4 %).

3.1.2. Abundance and biomass

The total abundance significantly decreased temporally (*H* = 18.17, *p* = 0.0001) during harvesting, which was followed by partial recovery after harvesting. In the HHI zone, the total abundance declined from 271.11 ± 44.44 ind/m² to 26.67 ± 10.18 ind/m² during harvesting and then recovered to 200 ± 13.88 ind/m² after harvesting. There were no significant effects of time (*H* = 1.55, *p* = 0.46), treatment (*H* = 4.58, *p* = 0.10), or their interaction (*H* = 5.88, *p* = 0.21) on the total biomass based on the Scheirer-Ray-Hare test (Fig. 4).

3.1.3. Macrofaunal diversity

Significant temporal changes were observed in the macrofaunal diversity, with limited treatment effects (harvesting intensity) (Fig. 4). The species richness significantly decreased temporally (*p* = 0.0003) during harvesting, which was followed by a partial recovery afterward. In the

Table 1
Dominant species in each harvesting intensity zone during the research period.

		Dominant species	Y	Ecological group	Mobility	Feeding guild	Average abundance (ind/m ²)	Average biomass (g/m ²)
Control	Before	<i>Paraprionospio pinnata</i>	0.15	EG IV	M/B	F/D	46.67	0.18
		<i>Iridona iridescens</i>	0.12	EGI	M/B	F/D	37.78	1.87
		<i>Glycinde gurbanovae</i>	0.09	EG II	M/B	F/P	28.89	0.20
	During	<i>Nephtys polybranchia</i>	0.15	EG II	M/B	F/P	15.56	1.07
		<i>Iridona iridescens</i>	0.13	EGI	M/B	F/D	13.33	2.13
		<i>Scoloplos</i> sp.	0.11	EG II	M/B	F/D	11.11	0.60
	After	<i>Nephtys polybranchia</i>	0.22	EG II	M/B	F/P	33.33	1.67
		<i>Iridona iridescens</i>	0.08	EG I	M/B	F/D	17.78	1.67
		Virgulariidae sp.	0.07	EG I	M/SE	F/F	11.11	0.20
LHI	Before	<i>Umbonium thomasi</i>	0.27	EG II	M/C	F/D	984.44	114.47
		<i>Iridona iridescens</i>	0.02	EGI	M/B	F/D	31.11	4.87
	During	<i>Balanoglossus misakiensis</i>	0.17	EGI	M/B	F/D	106.67	8.33
		<i>Protankyra bidentata</i>	0.05	EG II	M/B	F/D	6.67	16.27
		<i>Nephtys polybranchia</i>	0.04	EG II	M/B	F/P	17.78	1.07
	After	<i>Umbonium thomasi</i>	0.11	EG II	M/C	F/D	33.33	4.73
		<i>Nephtys polybranchia</i>	0.10	EG II	M/B	F/P	31.11	0.53
		<i>Sigambra bassi</i>	0.07	EG IV	M/B	F/P	20.00	0.07
	HHI	Before	<i>Iridona iridescens</i>	0.16	EG I	M/B	F/D	44.44
Virgulariidae			0.14	EG I	M/SE	F/F	37.78	0.80
Nemertea			0.09	EG I	M/C	F/P	17.78	0.20
During		<i>Anthopleura</i> sp.	0.11	EG II	M/SE	F/P	4.44	3.33
		<i>Nephtys polybranchia</i>	0.08	EG II	M/B	F/P	6.67	0.87
		<i>Thyasira tokunagai</i>	0.06	EG II	M/B	F/D	4.44	0.27
After		Virgulariidae	0.13	EG I	M/SE	PI	40.00	0.60
		<i>Nephtys polybranchia</i>	0.1	EG II	M/B	F/F	20.00	0.47
		<i>Iridona iridescens</i>	0.09	EGI	M/B	F/D	17.78	0.73

Note: F/P – Predator, F/H – Herbivorous, F/D - Deposit-feeder, F/F - Filter-feeder, M/SE - Sessile, M/SW - Swim, M/B - Burrow, M/C - Crawl, M/W - Walk.

Table 2

Permutational multivariate analysis of variance based on Bray Curtis similarities comparing macrofaunal abundances (square root transformed) sampled across time (fixed; time-harvesting period) and treatment (fixed; treatment-harvesting intensity).

Source of variation	Df	MS	Pseudo-F	<i>p</i>	Unique permutations	Effect size (%)
Time	2	10325	4.6422	0.001	997	30.00
Treatment	2	3501.6	1.5743	0.021	999	11.91
Time × Treatment	4	2745	1.2342	0.107	999	13.18
Residual	18	2224.2				47.17
Total	26					

Note: Effect size was calculated as a proportion of the estimated contribution of the components of variation for each factor. Df – Degrees of Freedom, MS – Mean Square. The *p* values in bold denote significance at *p* < 0.05.

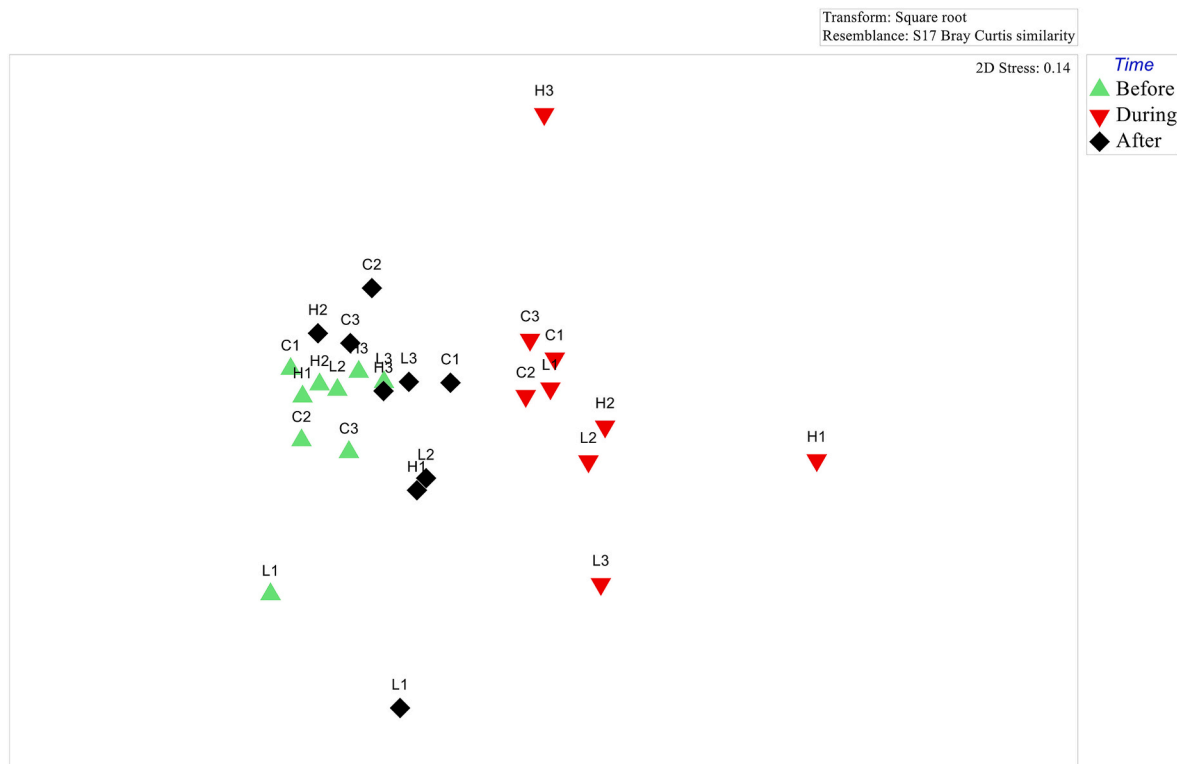


Fig. 3. Non-metric multidimensional scaling ordination treatments based on Bray-Curtis similarities in August 2022 (Before Harvesting), April 2023 (During Harvesting), and August 2023 (After Harvesting). Notes: C - Control; L - Low-intensity zone; H - High-intensity zone.

LHI zone, the species richness declined from 12.67 ± 3.06 before harvesting to 6.33 ± 2.52 during harvesting, with a complete recovery to 13.00 ± 6.08 after harvesting. In the HHI zone, the values decreased from 14.67 ± 2.08 to 3.00 ± 1.00 during harvesting and then recovered to 12.33 ± 0.58 after harvesting.

The Shannon-Wiener diversity index exhibited significant temporal changes ($H = 10.13, p = 0.006$), with a decline observed during harvesting, followed by subsequent recovery. In the LHI zone, the index increased from 1.77 ± 1.36 before harvesting to 1.90 ± 0.44 during harvesting, and it increased to 2.30 ± 0.53 after harvesting. In the HHI zone, the index dropped from 2.57 ± 0.15 to 1.27 ± 0.32 during harvesting and recovered to 2.33 ± 0.12 after harvesting.

The Margalef Index significantly decreased temporally ($F = 8.92, p = 0.002$) during harvesting, which was followed by recovery after harvesting. In the LHI zone, the Margalef Index decreased from 2.90 ± 1.22 before harvesting to 2.30 ± 0.66 during harvesting, followed by an increase to 3.57 ± 1.19 after harvesting. In the HHI zone, the index declined from 4.00 ± 0.79 to 1.57 ± 0.21 during harvesting and then recovered to 3.33 ± 0.21 after harvesting.

3.2. Benthic habitat health assessment

The results of multiple benthic habitat health indices demonstrated distinct temporal and treatment-related patterns (Fig. 5). The AMBI index demonstrated that a higher harvesting intensity ($F = 5.69, p = 0.012$) significantly impacted benthic habitat health. From pre-harvesting to the harvesting period, the value increased in the LHI (from 1.11 ± 0.35 to 1.48 ± 0.89) and HHI (from 0.60 ± 0.28 to 0.75 ± 0.75) areas, while it decreased in the Control area (from 2.28 ± 0.31 to 1.03 ± 0.18). After harvesting, AMBI values increased further in all groups, reaching 1.59 ± 0.57 (Control), 1.99 ± 0.17 (LHI), and 1.32 ± 0.50 (HHI).

The DSD index primarily reflected temporal changes ($H = 25.68, p = 0.0003$), with significant fluctuations over time but minimal influence due to the harvesting intensity. Before harvesting, the DSD values were stable across the regions at 4.60 ± 0.00 (Control), 4.63 ± 0.06 (LHI), and 4.60 ± 0.00 (HHI). During harvesting, the index increased uniformly to 4.80 across all zones, which was followed by a significant decline to 3.20 after harvesting.

The NSI index highlighted that the harvesting intensity ($H = 9.06, p = 0.011$) significantly affected the benthic habitat health, particularly between the control and HHI areas. For the NSI index, treatment

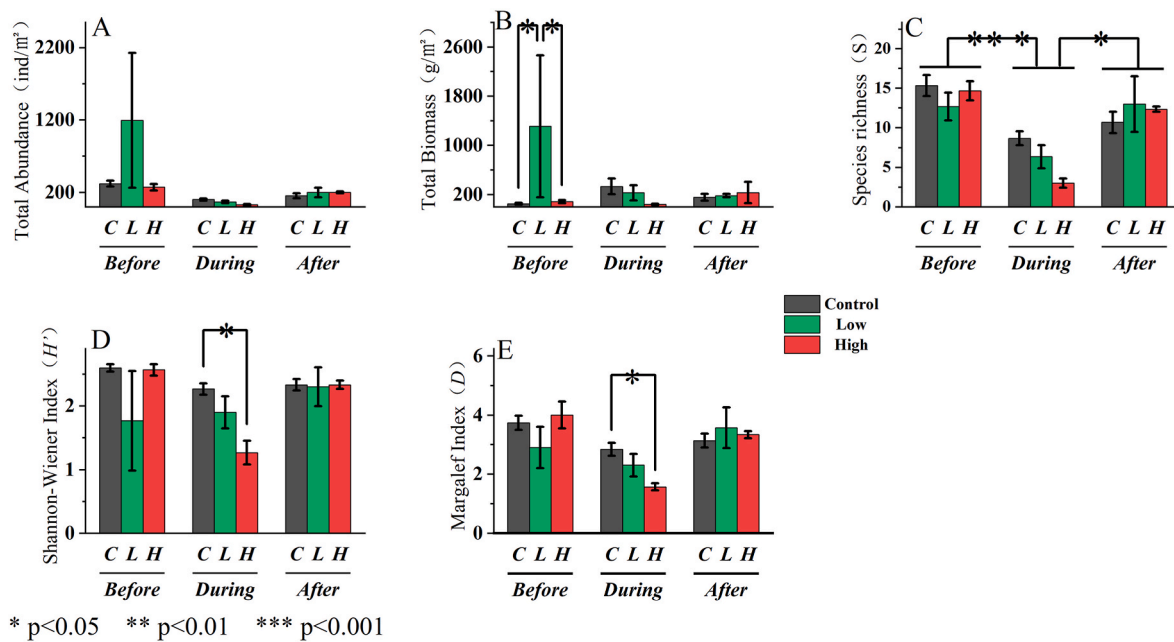


Fig. 4. Mean values (± standard error) of the total abundance, total biomass, and the biodiversity indices across the treatments and time periods. Note: (A) Total abundance, (B) Total biomass, (C) Species richness, (D) Shannon-Wiener index, and (E) Margalef index.

intensity had a significant effect, while there were no significant temporal effects ($H = 2.43, p = 0.297$) or interactions ($H = 2.07, p = 0.724$). Before harvesting, the NSI values were 20.63 ± 0.71 (Control), 21.67 ± 0.66 (LHI), and 19.96 ± 0.88 (HHI). During harvesting, the NSI increased to 23.82 ± 0.00 for the Control and LHI but decreased to 19.63 ± 0.00 for the HHI. After harvesting, the NSI partially recovered to 21.81 ± 0.96 (Control), 22.31 ± 2.09 (LHI), and 20.03 ± 0.94 (HHI). Significant differences were observed between the Control and HHI ($p = 0.036$) and between the LHI and HHI ($p = 0.015$).

The NQI1 index showed no significant effects of either harvesting intensity ($H = 2.35, p = 0.309$) or temporal factors ($H = 2.83, p = 0.242$) on benthic habitat health. However, the interaction between time and treatment approached significance ($H = 9.37, p = 0.052$). Before harvesting, the NQI1 values were stable at 0.60 ± 0.02 (Control), 0.64 ± 0.08 (LHI), and 0.71 ± 0.02 (HHI). During harvesting, the index declined across all the zones to 0.66 ± 0.03 (Control), 0.57 ± 0.06 (LHI), and 0.57 ± 0.06 (HHI). A slight recovery was observed after harvesting, with values rising to 0.60 ± 0.08 (Control), 0.61 ± 0.03 (LHI), and 0.65 ± 0.03 (HHI).

The indices collectively revealed that the benthic habitat health declined during harvesting, with partial recovery observed after harvesting. The AMBI and NSI values highlighted treatment-related differences, while the DSD demonstrated significant temporal variation. Before harvesting, the benthic habitat health was predominantly classified as “good” or “very good”. However, during harvesting, conditions deteriorated to “moderate” or “poor” in the LHI and HHI zones. After harvesting, the habitat conditions improved across all the zones to “good”.

4. Discussion

4.1. Impact of hydraulic rake trawling on macrofaunal diversity

The low abundance of benthic macrofauna (1079 individuals from 81 grabs) in our study aligns with known ecological constraints of sandy substrates (Zhao et al., 2021). This study demonstrated significant temporal shifts in the total abundance, species richness, Margalef Index, and Shannon-Wiener diversity index, and the harvesting intensity did not significantly affect these metrics. Notably, the macrofaunal richness

markedly declined during harvesting, with partial recovery observed afterward. These results align with previous studies demonstrating that bottom trawling causes short-term declines in benthic species (Hall and Harding, 1997; Ferns et al., 2001), but partial recovery often occurs following the disturbance (Collie et al., 2000; Hiddink et al., 2006).

The decline in the total abundance during harvesting, particularly in both the LHI and HHI zones, reflects disturbance-induced mortality patterns in benthic communities (Kaiser et al., 2006). In the LHI zone, where the disturbance was less severe, the abundance dropped significantly (from 1193 to 66 ind/m²), which was followed by partial recovery after harvesting. The high before-harvesting abundance in the LHI zone was largely driven by the dominance of *Umbonium thomasi*, a species that thrives under environmental conditions that are influenced by seasonal factors, such as freshwater influx from the Yellow River (Frey et al., 1987; Gao et al., 2020; Wang et al., 2022). This unusually high abundance may not fully represent typical conditions, as it is influenced by transient factors (e.g., freshwater input, temperature, and food availability) rather than long-term ecological drivers (Gibson et al., 2002). While *Umbonium thomasi* likely contributed to the relatively high abundance before harvesting, the observed decline in abundance during harvesting, followed by recovery, cannot be solely attributed to this species. Other factors, such as changes in community structure and environmental conditions, likely played a role in these shifts (Pilskaln et al., 1998).

In contrast, the total abundance in the HHI zone declined sharply from 271.11 ± 44.44 ind/m² before harvesting to 26.67 ± 10.18 ind/m² during harvesting, before recovering to 200.00 ± 13.88 ind/m² after harvesting. These recovery patterns align with those reported by Pusceddu et al. (2014), who showed that the species richness can recover after only a few months of reduced fishing pressure, although the species composition may shift in response to disturbance (Pusceddu et al., 2014). The differing recovery trajectories between the LHI and HHI zones underscore the complex interaction between the disturbance intensity and local environmental conditions. The high recovery in the LHI zone may also be attributed to the restoration of favorable environmental conditions (e.g., salinity and oxygen levels) after harvesting, which likely facilitated the re-colonization by opportunistic species (Tarnowski, 2006).

Additionally, the Shannon-Wiener index demonstrated temporal

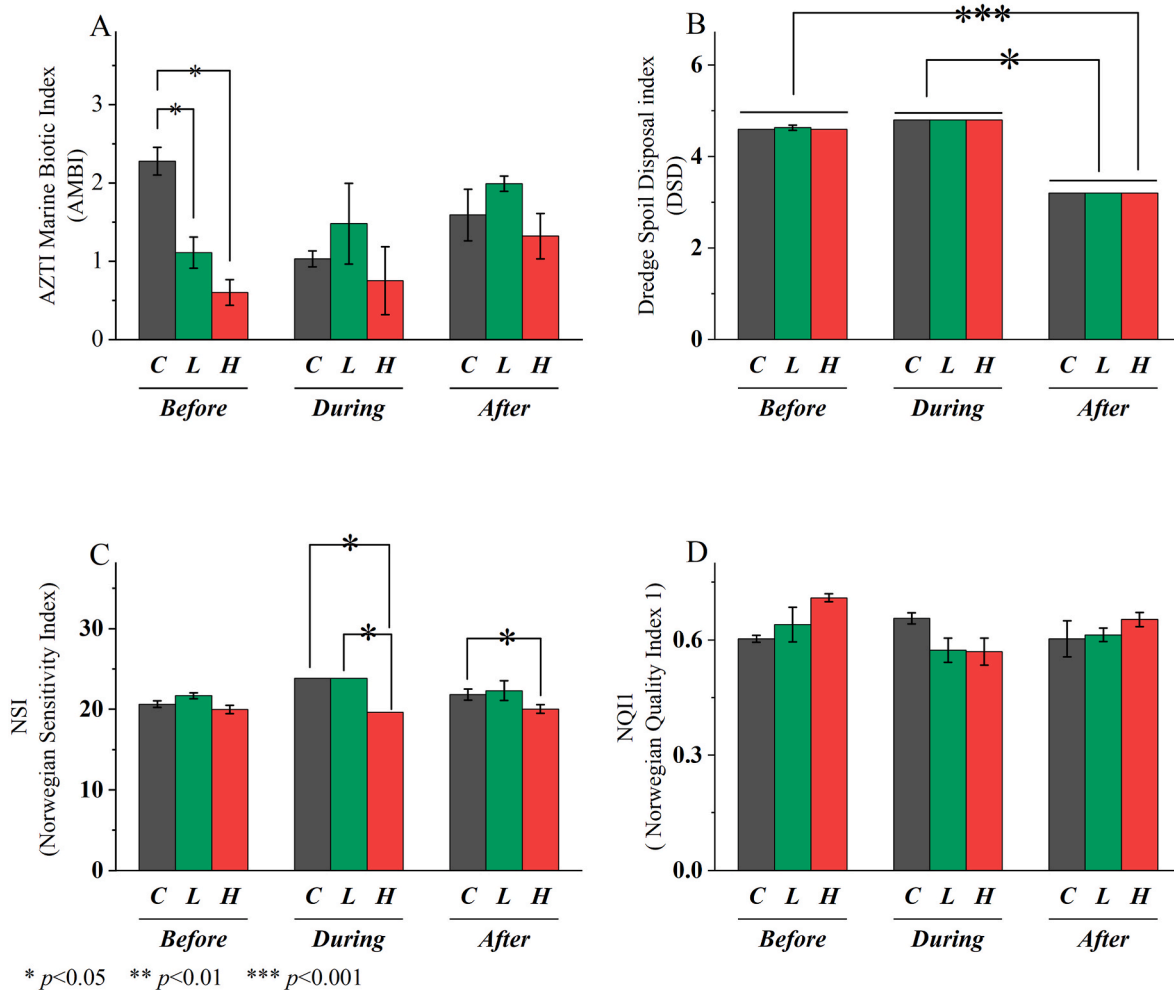


Fig. 5. Temporal and treatment-related variation in the benthic habitat health indices. Notes: (A) AZTI Marine Biotic Index (AMBI), (B) Dredge Spoil Disposal Index (DSD), (C) Norwegian Sensitivity Index (NSI), and (D) Norwegian Quality Index 1 (NQI1) across three treatments (Control [C], Low Intensity [L], and High Intensity [H]) during three periods (Before, During, and After harvesting). Significant differences are indicated by asterisks (* $p < 0.05$, ** $p < 0.01$, and *** $p < 0.001$).

variability, recovering in both zones after harvesting. However, while the species richness may recover relatively quickly, community evenness can remain altered, as certain opportunistic species (e.g., *Nephtys polybranchia* and *Iridona iridescens*) tend to dominate disturbed zones. This pattern reflects the broader ecological shifts typical of trawled ecosystems, where disturbance-tolerant species prevail at the expense of more sensitive and competitive species (Collie et al., 2000). Despite their numerical dominance, these opportunistic species contribute minimally to total biomass. Due to their typical exhibition of high fecundity, short life cycles, and rapid dispersal capacity, enabling swift colonization of vacant niches. However, their small body size and low individual biomass result in limited contribution to total biomass despite substantial abundance fluctuations (Levinton, 1970).

4.2. Impact of hydraulic rake trawling on the macrofaunal community

The PERMANOVA analysis confirmed that while temporal effects accounted for the largest variation in the community composition, treatment (harvesting intensity) also influenced the community structure, though to a lesser extent. These findings suggest that community shifts were primarily driven by the temporal dynamics of disturbance, with recovery trajectories largely determined by the post-harvesting period. The significant temporal effects on the community structure further support the notion that benthic communities undergo significant restructuring over time and that trawling disturbances can drive shifts in

community composition, favoring disturbance-tolerant species such as *Nephtys polybranchia* and *Iridona iridescens* (Hall and Harding, 1997; Sivadas et al., 2010).

The nMDS ordination clearly illustrated temporal shifts in community structure, with higher dispersion observed in the LHI and HHI zones during harvesting compared with the control zone. This increased variability likely reflects the heterogeneous impact of trawling on macrofaunal assemblages, where disturbance intensity, substrate type, and species resilience all contribute to community variation (Kaiser et al., 2006). Consequently, the control zone remained more stable, while the harvested zones exhibited increased variability, suggesting that the effects of trawling are not uniform and can lead to substantial changes in species composition even after trawling ceases.

The SIMPER analysis further emphasized that species, such as Virgulariidae, *Iridona iridescens*, and *Nephtys polybranchia* contributed significantly to community dissimilarity before and after harvesting. These species, characterized by their ability to exploit disturbed habitats (Thrush et al., 2002), underscore the functional shifts within the community. While Infaunal polychaetes, which are more generally resilient to physical disturbances, exhibited faster recovery in both the LHI and HHI zones, mollusks, such as *Umbonium thomasi*, were slower to recover. This illustrates the differential resilience of benthic species to trawling impacts (Hiddink et al., 2006; Constantino et al., 2009). The post-trawling dominance of opportunistic species like *Nephtys polybranchia* (Thrush et al., 2002) supports the idea that

disturbance-tolerant species often dominate after a disturbance event, altering the functional characteristics of the ecosystem (Veale et al., 2000; Thrush et al., 2005). These changes can have cascading effects on ecosystem functions, including nutrient cycling and sediment biogeochemistry (Badino et al., 2004), as the functional roles of key species shift following disturbance (Kaiser et al., 2006; Lucchetti et al., 2021).

The change in the functional groups of microbenthic fauna in the LHI area from “crawl (M/C)” to “burrow (M/B)” demonstrated that “disturbance-succession” theory (Pearson, 1978). During harvesting, the disappearance of *U. thomasi* (M/C) is highly consistent with previous studies indicated that slow-moving epifaunal species are highly sensitive to physical disturbances (e.g., trawling, dredging) (Thrush et al., 1998). In contrast, infaunal burrowers such as *N. polybranchia* and *B. misakiensis*, which inhabit deeper sediment layers, are not only better able to avoid the physical stress caused by disturbance but can also act as pioneer opportunistic species that rapidly colonize disturbed habitats (Kaiser et al., 2000). After harvesting, the change in feeding functional groups from deposit-feeder to predator suggested that hydraulic rake trawling may have reduced the quality of surface sediments as a food source by altering sediment structure, organic matter content, or microbial communities (Yan et al., 2023). Additionally, the increase of predators (F/P) such as *N. polybranchia* and *S. bassii* may be attributed to their ability to prey on abundantly available small opportunistic species (e.g., polychaetes). This further indicated that hydraulic rake trawling not only directly affects species but also reshapes the energy pathways within the entire community (Thrush et al., 2002).

In the HHI area, although community abundance and biomass decreased significantly, the dominant species composition and functional group structure (coexistence of burrow and semi-sessile organisms, multiple feeding modes) quickly returned to initial conditions. The persistence and rapid recovery of key species such as *I. iridescens* and *Virgulariidae* sp. were critical. *Virgulariidae* as semi-sessile filter feeders, typically indicate a relatively stable and less disturbed sedimentary environment (Ross et al., 2021). Their ability to rapidly regain dominance after harvesting may be due to the fact that the fundamental sedimentary conditions (e.g., sediment type, hydrodynamic forces) remained largely unchanged, allowing timely recovery (Hiddink et al., 2006).

4.3. Benthic habitat health assessment

4.3.1. Impact of hydraulic rake trawling on benthic habitat health

The health of benthic habitats, assessed using four ecological indices (AMBI, DSD, NSI, and NQI1), revealed distinct temporal and treatment-related variations. These differences underscore the complexity of benthic ecosystem responses to disturbance and highlight the importance of selecting appropriate indices to capture the nuances of habitat degradation and recovery. The findings illustrated varying sensitivities among these indices, with some being more responsive to specific ecological components or stress gradients, depending on the nature of the disturbance and the community structure.

The AMBI index, developed to assess the ecological health of benthic environments, demonstrated significant effects of treatment intensity ($F = 5.69$, $p = 0.012$) and a notable interaction between time and disturbance ($F = 2.96$, $p = 0.048$). During the harvesting period, the AMBI values decreased markedly, with the lowest values observed in the HHI zone (0.60 ± 0.28) compared with the control (2.28 ± 0.31) and LHI zones (1.11 ± 0.35). This decline aligns with previous studies indicating that disturbances such as trawling or dredging reduce the abundance of sensitive species while favoring the proliferation of more tolerant or opportunistic species (Borja et al., 2000; Forchino et al., 2011). After harvesting, the AMBI values exhibited partial recovery, but they did not return to pre-harvest levels, particularly in the HHI zone. This incomplete recovery suggests that prolonged disturbances can cause lasting shifts in community structure. These findings are consistent with observations by Hiddink et al. (2017), who reported that ecosystems

subjected to trawling disturbances often experience slow and incomplete recovery after the disturbance ceases (Hiddink et al., 2017). The statistically significant differences observed between the control and LHI and HHI zones ($p < 0.05$) support the utility of the AMBI as an effective tool for identifying disturbance impacts and monitoring recovery.

Despite its widespread application for monitoring habitat health, the AMBI has certain limitations in detecting subtle shifts in species composition due to its broad classification system, which groups species into five sensitivity categories. Recent research has highlighted the potential for overestimations in AMBI assessments, particularly when species sensitivities within the same species group differ markedly (Rygg and Norling, 2013). These findings underscore the value of more refined indices, such as the NSI and NQI1, which are optimized to capture finer ecological nuances.

In contrast, the DSD index, which quantifies sediment disruption from dredging and other anthropogenic activities, exhibited significant temporal effects ($H = 25.68$, $p < 0.0001$), with values increasing during harvesting (from 4.6 to 4.8) and subsequently declining after harvesting (3.2), indicating habitat recovery. This variation is likely related to changes in the species abundance ratios resulting from disturbance. The increase during harvesting may be attributed to the rapid proliferation of opportunistic polychaetes, such as *Nephtys polybranchia*, which exploit newly available resources and altered environmental conditions resulting from the harvesting activities.

The post-harvest decline in the DSD index likely reflects a reduction in the abundance of opportunistic species, resulting in a more balanced community structure (Grassle and Sanders, 1973; Grassle and Grassle, 1974; Clarke et al., 2018; Dauvin et al., 2018). However, no significant response to treatment intensity was observed ($H = 0.0264$, $p = 0.9869$), suggesting that the DSD index is primarily sensitive to the physical disruption of the sediment rather than variations in the fishing intensity. This makes the DSD index more useful for detecting immediate impacts on sedimentary habitats. These findings align with a previous study by Ellis et al. (2000), which indicated that sediment disturbance often temporarily increases following dredging or trawling, but improves once the disturbance ceases (Ellis et al., 2000). However, due to its focus on macrofaunal abundance, the DSD index may be less effective in capturing broader ecological changes, such as alterations in species composition or community structure (Dauvin et al., 2018).

The NSI index, which categorizes species based on their sensitivity to environmental changes, revealed significant treatment intensity effects ($H = 9.06$, $p = 0.011$). Specifically, during harvesting, the control and LHI zones exhibited higher sensitivity (high value), characterized by a higher relative proportion of sensitive species (EG-I), while the HHI zones were dominated by disturbance-tolerant species (EG-III and EG-IV). The significant increase in the NSI values during harvesting, followed by partial recovery post-harvest, highlights the utility of the NSI as a tool for assessing the resilience of benthic communities to varying disturbance levels. By identifying shifts in community structure, the NSI provides valuable insights into responses to disturbance (Rygg and Norling, 2013). However, the absence of significant temporal effects ($H = 2.43$, $p = 0.297$) suggests that the NSI may be less sensitive to longer-term recovery processes, particularly when the shifts in species composition occur gradually, as observed in the HHI zones.

The NQI1 index, a composite measure incorporating species diversity (Shannon-Wiener index) and the AMBI, showed no significant temporal or treatment-related effects ($H = 2.83$, $p = 0.242$; $H = 2.35$, $p = 0.309$). However, the interaction effect approached significance ($H = 9.37$, $p = 0.052$). Specifically, values before harvesting were relatively stable across all the zones, with slight decreases observed during harvesting, which was followed by minor recovery after harvesting. These results suggest that the NQI1, as a broader habitat quality index, may be less responsive to short-term disturbances compared with more specialized indices such as the AMBI and NSI. Nevertheless, the NQI1 remains valuable for assessing long-term trends in habitat quality, especially in

environments where biodiversity and overall species composition are of primary concern (Cordier and Pawlowski, 2018). The lack of significant differences across the treatment zones in this study indicates that the NQI1 may not fully capture the finer ecological nuances during short-term disturbances or in the early stages of recovery.

4.3.2. Comparative analysis of indices

Each index employed in this study offers distinct advantages and limitations in assessing benthic habitat health. The AMBI is particularly sensitive to shifts in species composition and is effective in detecting changes in habitat quality due to disturbances. However, its broader classification system can obscure finer-scale changes in community structure, and it may overestimate disturbance severity due to variances in species sensitivities. In contrast, the NSI provides a more nuanced view of species sensitivity and community structure, making it especially useful for assessing the resilience of benthic communities to disturbances. As demonstrated in this study, the NSI better reflects disturbance gradients compared with the AMBI (Rygg and Norling, 2013), suggesting that it is a more precise index for assessing ecological conditions, particularly under varying disturbance intensities.

The DSD is invaluable for measuring the immediate physical impacts of sediment disturbance, such as those caused by dredging or trawling, but it is less effective in capturing broader ecological changes. Therefore, it is best used in conjunction with other indices to gain a comprehensive understanding of ecological impacts, from sediment disruption to shifts in species composition. Finally, the NQI1 provides a broad overview of habitat quality, which makes it ideal for long-term ecological monitoring. However, its lack of sensitivity to short-term disturbances or recovery processes may reduce its applicability in environments where disturbance-induced changes are more pronounced or occur over shorter timescales. Although we used multiple ecological indices to evaluate the impact of hydraulic rake trawling, the results varied due to the different parameters targeted by each index. Additionally, while we conducted a year-long, cross-seasonal sampling survey, the monitoring was not continuous. Future studies should involve longer, continuous, and systematic monitoring.

4.3.3. Management strategies

The impact of mechanized fishing on benthic habitats is closely related to fishing gear, and frequency, thus requiring differentiated management (Hiddink et al., 2003). (1) In vulnerable habitats (e.g., biogenic reefs), fishing activities should be avoided or limited in scale to minimize impacts on species and ecological recovery, while in nearshore areas with strong natural disturbances, mechanized fishing can be conducted moderately, ensuring quick recovery. (2) It is recommended to establish dedicated enhancement, seed replenishment, and restricted fishing zones based on existing aquaculture area divisions, and to choose appropriate fishing tools to reduce negative environmental impacts. (3) A rotational fishing management system should be adopted, with fishing areas rotated annually, allowing sufficient recovery time for benthic habitats. (4) Future assessments of benthic habitat health should use multiple indices for comparative analysis to construct a comprehensive evaluation system that addresses the impacts of hydraulic rake trawling and other human activities.

5. Conclusions

This study demonstrated that hydraulic rake trawling directly impacted benthic habitats. The findings were as follows: (1) macrofaunal diversity and community composition shift dramatically during harvesting, but recovery is incomplete within a four-month period. (2) The ecological indices revealed a decline in benthic habitat during harvesting, with partial recovery observed after harvesting. (3) The dominant functional groups of macrobenthic fauna changed from F/D and M/C to F/P and M/B. (4) The AMBI and NSI values highlighted treatment-related differences, while the DSD underscored significant

temporal variation.

This study not only enhances our understanding of disturbance and recovery processes in benthic communities but also provides critical insights for ecosystem-based management strategies, offering theoretical support for the scientific management policies of mechanized fishing in China. Future studies should refine the habitat assessment tools, explore the cumulative effects of multiple human disturbances, and examine diverse marine environments. This will be crucial for developing adaptive management practices that promote the long-term sustainability of marine ecosystems in the face of increasing anthropogenic pressures.

CRedit authorship contribution statement

Yuze Tang: Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Conceptualization. **Erwann Legrand:** Writing – review & editing. **Jiaqi Li:** Writing – review & editing, Conceptualization. **Haiqing Wang:** Writing – review & editing. **Ang Li:** Investigation. **Lulei Liu:** Investigation. **Zirong Liu:** Investigation. **Suyan Xue:** Writing – review & editing. **Shoutuan Yu:** Investigation. **Yuze Mao:** Writing – review & editing, Writing – original draft, Validation, Supervision, Project administration, Methodology, Funding acquisition, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was supported by National Key R&D Program of China (2024YFD2400404), Central Public-interest Scientific Institution Basal Research Fund, CAFS (2023TD54), Shandong Province Science and Technology based Small and Medium sized Enterprises Innovation Capability Enhancement Project (2023TSGC0768).

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ocecoaman.2025.107969>.

Data availability

Data will be made available on request.

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