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Assessing the Ecological Impact of Jersey's First No Take Zone: Insights from Potting Trials and Diurnal/Nocturnal Surveys

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Abstract

There has been a global push to increase the protection of the world's oceans with a particular interest in increasing the coverage of Marine Protected Areas (MPAs). MPAs vary in level of protection, with No Take Zones (NTZ) serving as the highest level of protection currently imposed. In Jersey, Channel Islands, its First NTZ was implemented in 2022 within Portelet Bay NTZ (PTLNTZ), to assess its impact on biodiversity and potential for benefitting local fisheries. This study marks the 5th year of monitoring within Portelet using Baited Remote Underwater Video (BRUV) data from 2021-2025 to measure change in overall abundance, species richness and species assemblages, furthermore, Nocturnal BRUV deployments were used for the first time in 2025. Secondly, annual potting data from 2022-2025 provided insights into a shift in size and abundance of Spider crab (*M. brachydactlya*) and European Lobster (*H. gammarus*) in the NTZ. Results saw increased overall abundance within PTLNTZ, with species richness remaining unchanged. Species assemblages shifted but did not differ from the control site. Nocturnal BRUVs recorded unique species compared to diurnal deployments, yet overall patterns remained similar between PTLNTZ and the control. Within PTLNTZ, spider crab showed increases in both size and CPUE, while lobster size increased and CPUE initially rose but subsequently declined. Importantly though, lobster individuals by 2025 were significantly larger than those at all comparable unprotected sites. This study has provided evidence that NTZ protection can increase the abundance of individuals, even within a small area. Furthermore, this study provides the first evidence that NTZs can contribute to the management of commercially important crustacean species in Channel Island waters.

Acknowledgements

Thanks, are firstly extended to my advisor Dr Sam Blampied, who not only guided me throughout the planning and completion of this project but also gave up many hours of her time outside of work hours to deploy and recover BRUVs from early mornings to late nights. I would also like to thank Tom Vallois for giving up his time to help with the BRUVs both on the water and in the shed helping with the setup process. Further thanks extend to the rest of the Jersey Marine Resources team for welcoming me aboard the Norman Le Brocq during the 2025 potting trials.

This project would not have been possible without funding from the Ecology trust who thanks to their funds, allowed the creation of the newly designed BRUVs and equipment that can be used for future Portelet NTZ research. Also, Jersey Marine Conservation which helped during the early phases of BRUV design. Further thanks extend to friends; Curtis England, Myles Wilcox, my partner Filipa Olivença and my father Simon Swain for helping during on the water testing and design of various iterations of the BRUV unit prior to use during this project. Further thanks are given to undergraduate students Matt and PJ for their assistance during the BRUV video review process.

Finally, my thanks must be extended to JICAS, specifically Sean Dettman, who initially put me in contact with the Jersey Marine Resources team, leading to this collaboration and subsequent report.

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1. Introduction

Commercial fishing succeeds all other anthropogenic pressures in catalysing ecological extinction (Jackson et al., 2001), by not only depleting fish populations, but also degrading benthic habitats, ultimately negatively impacting ecosystem function. Improvements in fishing technology have allowed for a greater intensity of exploitation, reducing the number of undisturbed habitats to virtually zero (Roberts, 2012). Effective Marine Protected Areas (MPAs) are now recognised as a crucial step in the conservation of biodiversity (Mazaris et al., 2019) and the effective management of fisheries (Rassweiler et al., 2012; Kriegl et al., 2021), allowing the reduction of anthropogenic pressures facing marine systems.

MPAs vary in their level of protection with No Take Zones (NTZs), representing the highest level currently granted to a marine system, whereby extraction of any marine organisms is prohibited. Often referred to as fully protected areas, NTZs currently cover 2.9% of the world's oceans (Marine Protection Atlas, 2025) and have been attributed to increases in abundance of individuals and species richness, both within NTZs (Hoskin et al., 2011; Bergström et al., 2022) and in adjacent fished waters (Ashworth et al., 2005; Lippi et al., 2022). The top-down protection provided by NTZs allows for the recovery to pre-intensive fishing community structure, reinstating complex ecological structures and improving overall ecosystem resilience, an essential factor considering increasing stresses placed on the marine environment by climate change (Oliver et al., 2018; Worm and Lotze, 2021). NTZs are not always successful due to weak compliance or poor decision-making (Campbell et al., 2012). Additionally, populations often exhibit unique responses to NTZ designation, with some negative effects a possibility (Davies et al., 2014). Considering these complexities, ongoing research is crucial for refining NTZ management strategies and enhancing their effectiveness. This is particularly important in regions with high commercial fishing pressures, where the potential for spillover and larval dispersal aiding species recovery is of great interest to fisheries management and conservation practitioners.

Jersey, located in the northeastern English Channel, presents an ideal case study due to its long-standing dependence on coastal fisheries and recent implementation of its first NTZ. The island, with its strong historical and economic ties to marine resource use, relies heavily on its coastal fisheries for both local livelihoods (MEP, 2022) and cultural identity (Le Maistre, 2011). During the 19th century, Jersey was one of the few parts of Britain with any remaining fisheries

laws, though these were outdated and failed to address harmful new practices (Chambers, 2024). Detailed in a presentation by local naturalist James Hornell in 1897, substantial declines in lobster, crab and flatfish stocks were presented, highlighting the vulnerability of the local marine ecosystems to overfishing (Chambers, 2024). As of 2023, 113 licensed Jersey vessels and 97 licensed French vessels operate in these waters (Marine Resources Annual Report, 2024), primarily targeting shellfish. However, wet fish species remain an important component of the local fishery, typically caught on a smaller scale by inshore boats. With many local families both in Jersey and France relying on the continued strength of stocks, the importance of effective marine conservation measures cannot be understated.

In 2022, Jersey designated its first NTZ in Portelet Bay as part of wider efforts to improve marine conservation and support the recovery of commercially targeted species (Gov.je, 2022). The creation of Portelet NTZ (PTLNTZ) has provided a natural laboratory for investigations into the local effectiveness of fully protected areas in the Channel Islands. A condition of PTLNTZ designation was that ecological monitoring must take place for the first five years. Monitoring began in 2021, making this year the fifth and final year of the initial assessment period.

The importance of the Portelet NTZ has grown following Jersey's 2023 Marine Spatial Plan, which sets out clear priorities for expanding NTZ coverage in line with the global '30 by 30' conservation target (Jersey Marine Spatial Plan, 2024). Under this plan, PTLNTZ is to be retained and monitored (NB1a), and a second NTZ is planned at Les Sauvages reef (NB1b). If both sites show positive ecological outcomes, further NTZs will be considered to meet both biodiversity and social objectives (NB1c). In this context, data from PTLNTZ will play a pivotal role in informing future designations. Demonstrating tangible ecological benefits, particularly in commercially valuable species, it will both strengthen the government's decision-making process and improve local perception of NTZs.

This study has set out to answer a key research question: Has the designation of PTLNTZ in 2022 led to measurable ecological change, with particular interest in commercially important species? To answer this, the research objectives of this project are as follows:

- Annual potting data collected inside and beyond the NTZ since 2022 will be used to evaluate changes in the abundance and size of commercially important crustaceans (*Homarus gammarus*, *Cancer pagurus*, *Maja brachydactyla*).

- Diurnal and Nocturnal Baited Remote Underwater Video (BRUV) will be used to investigate changes across the recording period (2021-2025) in overall abundance, species richness and species assemblage, with a particular interest in commercially important species.

Following these objectives, this study sets out to explore the hypotheses outlined below:

1. The number and Size of Crustaceans increase inside PTLNTZ.

2. Overall biodiversity (abundance, species richness, and species assemblage) will increase more strongly within the PTLNTZ over time relative to the control.

3. The individual abundance and species richness of commercially important species will increase more strongly within the PTLNTZ over time relative to both the control and overall biodiversity trends in PTLNTZ.

4. In 2025, nocturnal communities (abundance, richness, and assemblage composition) within the PTLNTZ will differ from both (a) the control and (b) diurnal surveys.

Although the Portelet NTZ is small (0.26km2), evidence from similar UK sites suggests that even small, well-enforced NTZs can deliver measurable ecological benefits within a short timeframe (Howarth, 2012). This study aims to determine whether such outcomes are emerging at PTLNTZ.

2. Methods

2.1 Study Site

Located in the Gulf of St Malo, the Channel Islands comprise of five inhabited islands, with Jersey, the largest serving as the location of this study (Figure 1). The Bailiwick of Jersey is predominantly made up of marine territory, with only a small portion comprising of land. While the island itself covers just 120 km², it is surrounded by a much larger territorial sea area of approximately 2,455 km².

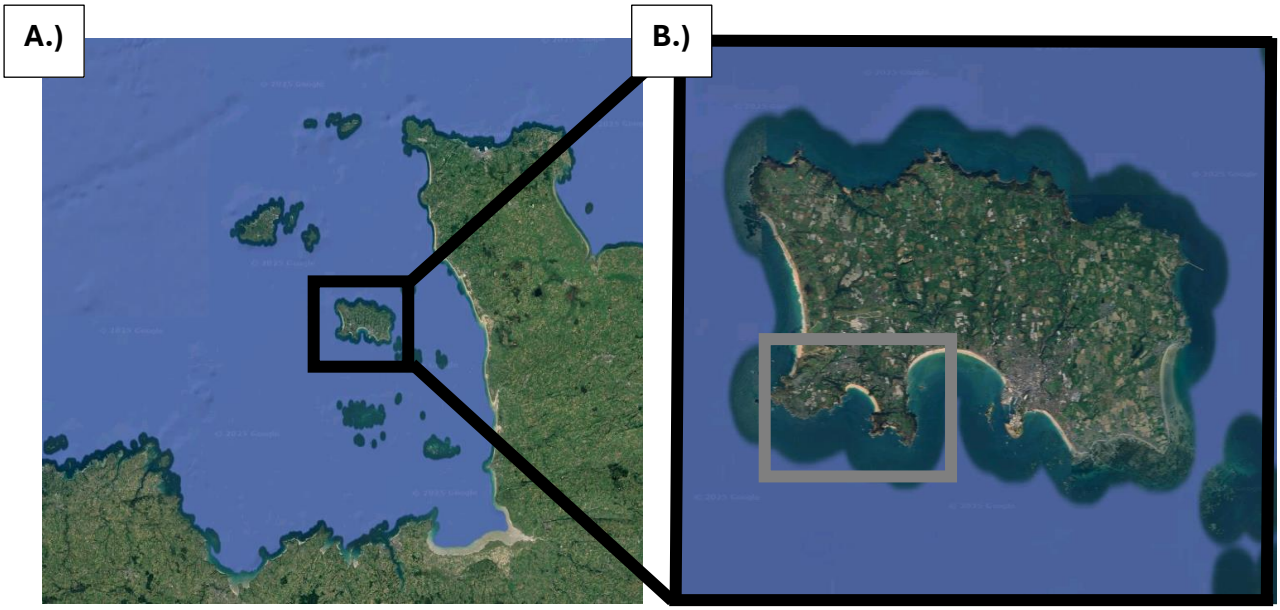


Figure 1, Satellite images denoting A.) Location of Jersey within the gulf of St Malo and B.) Location within which all data collection was undertaken.

This study focuses on two ecologically comparable bays in Jersey: Portelet Bay and Beauport Bay (Figure 2). Portelet Bay, designated as a No-Take Zone (NTZ) in 2022, is currently the only NTZ in Jersey, serving as this study's primary site. Beauport Bay functions as the control site due to its similar ecological characteristics, including a south-facing orientation, comparable gradient, and a mix of sandy substrate and kelp-covered rocky habitats.

2.2 Data Collection

2.2.1 Lobster Pot Trials

In 2025, the annual spring potting trials were conducted with Jersey Marine Resources, continuing the annual monitoring established in 2004. Thirty pots were deployed at each of six locations (Paternosters, Rigdon Bank, Passage Rock, Ouaisné, Outside PTLNTZ (OPTLNTZ) and PTLNTZ) across May and June, selected specifically to best represent the island's lobster fishery (Figure 2).

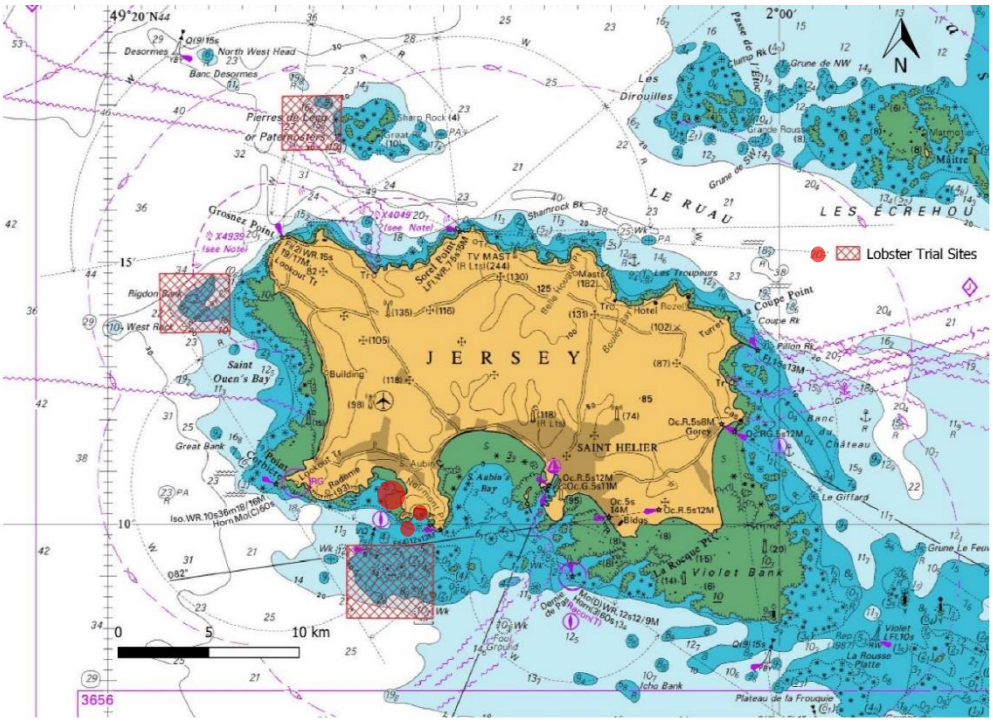
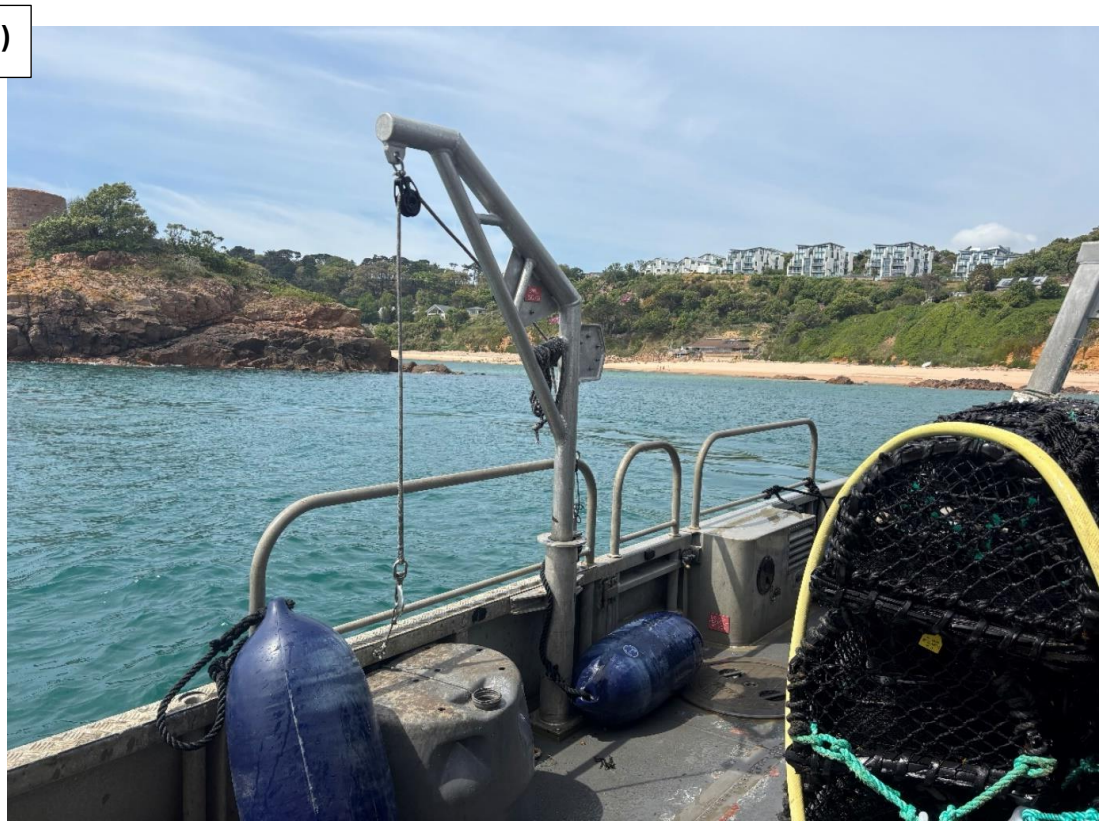


Figure 2, Locations of deployment sites for annual spring potting trials taken from Binney et al., (2024) and modified to include the three new sites adopted in 2022 and used for analysis.

Parlour pots in strings of 10 with escape gaps blocked were baited with two gurnard and set for a 48-hour soak. On retrieval, pots are recovered, and all crustaceans are identified and sexed. For each European lobster (*Homarus gammarus*), carapace length, telson width, and crusher claw width, length, and depth were measured. For other crustacean species, including spider crab (*Maja brachydactyla*), brown crab (*Cancer pagurus*), and lady crab (*Necora puber*), carapace length, width, and depth were recorded. Specimens were also assessed for physical damage, and the presence and colour of eggs were noted for females. Additionally, all bycatch species captured in each string were identified and recorded (Figure 3a, b and c).

A.)



B.)

C.)

Figure 3, A.) Onboard the Norman Le Brocq, prior to deployment of pots inside PTLNTZ, B.) The contents of a pot hauled inside PTLNTZ and C.) Measurements taken of a female lobster using a set of vernier callipers.

2.2.2 Baited Remote Underwater Video (BRUV)

BRUV Design

Each homemade BRUV unit (Figure 4) consisted of a single dive weight to anchor the rig on the seabed, and two small buoys to maintain vertical orientation in the water column. A GoPro Hero 13 camera was mounted to record continuous video footage. Two Xtar D30 1600 dive torches were attached to each rig, with batteries removed during diurnal deployments. Each dive weight was connected via ~5 metres of rope to a secondary ~10 kg lead block. This anchor block was then connected to a surface buoy that was labelled “Marine Resources” to assist in locating and recovering the unit.

Figure 4, The complete homemade BRUV rig including a surface bobber, ~25m of mainline, a ~10kg lead block, 10m of secondary rope and BRUV unit consisting of 2 torches, 1 GoPro 13 and 2 smaller bobbers to maintain vertical position.

BRUV Deployment (Day)

Three homemade Baited Remote Underwater Video (BRUV) units were deployed during daylight hours at two sites: PTLNTZ and the control site, Beauport Bay (figure 5a). Deployments were carried out on the 11th, 17th, 18th, 22nd, and 23rd of July 2025, with three BRUVs deployed in three distinct locations (Figure 5b and Figure 5c) at each site per day, located at least 200m apart. In total, there were 30 deployments, all of which were conducted during neap tides to minimise current strength and maximise stability and visibility on the seafloor.

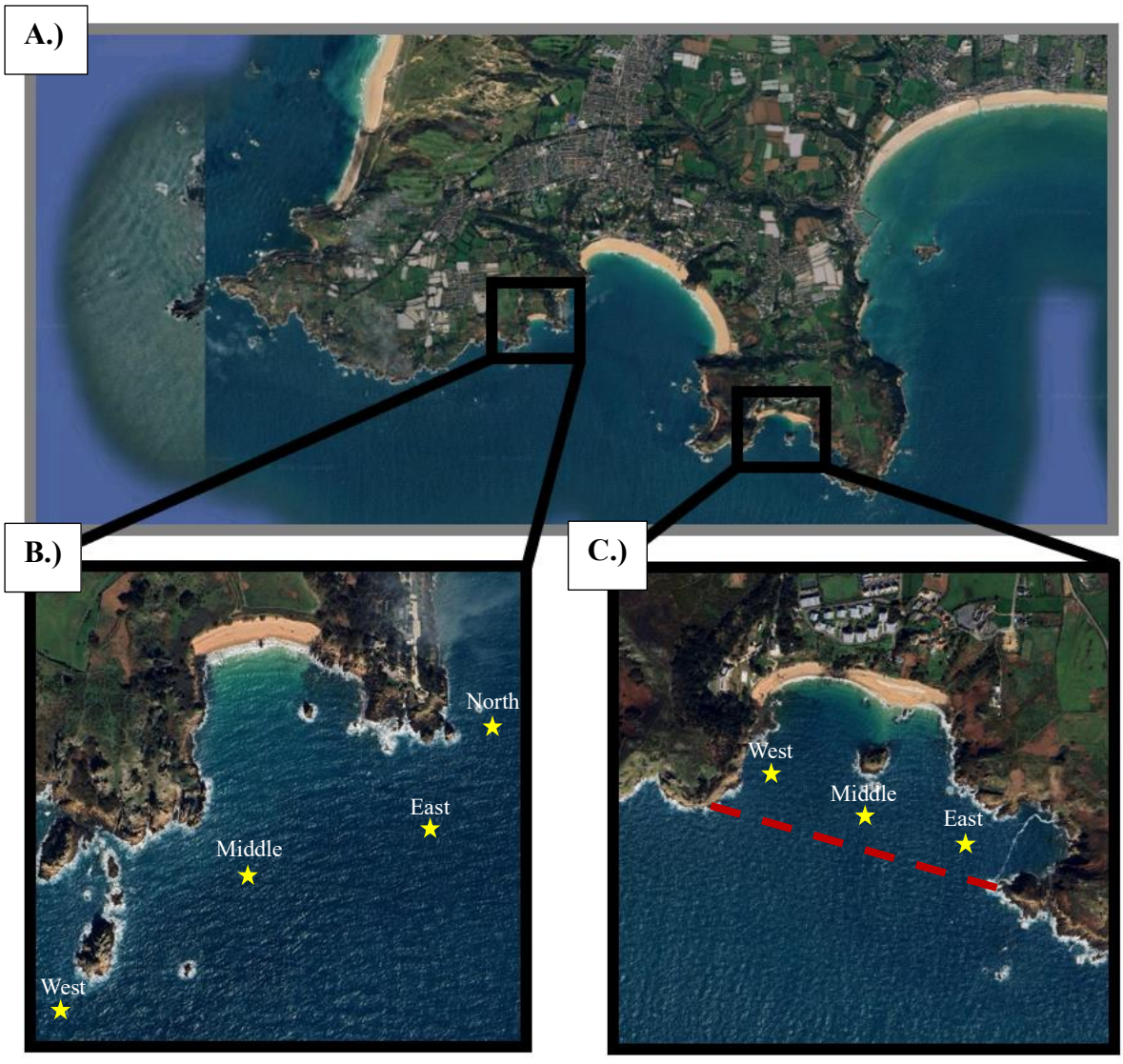


Figure 5 , Satellite images depicting A.) Both sites identified within figure 1b, B.) Beauport Bay including locations of BRUV deployments; North, West, East and Middle and C.) Portelet Bay including locations of BRUV deployments; West, East and Middle and red line denoting border of NTZ.

Each BRUV unit was baited with three defrosted chunks of mackerel placed in a bait cage positioned within the camera's field of view. The Cameras were configured to record at 1080p resolution using the "Linear" field of view setting, selected to optimise battery life and reduce the risk of overheating. Prior to deployment, recording was initiated, and the camera angle was adjusted to ensure the bait cage was fully visible in the frame. Date and site location were communicated to the camera through both hand signage and verbal narration before the rig was gently lowered into the water to avoid disturbing camera alignment. Deployment time was recorded, and the rig was not retrieved unit at least 1 hour had passed to ensure 40 minutes of usable footage (Figure 6a and 6b).

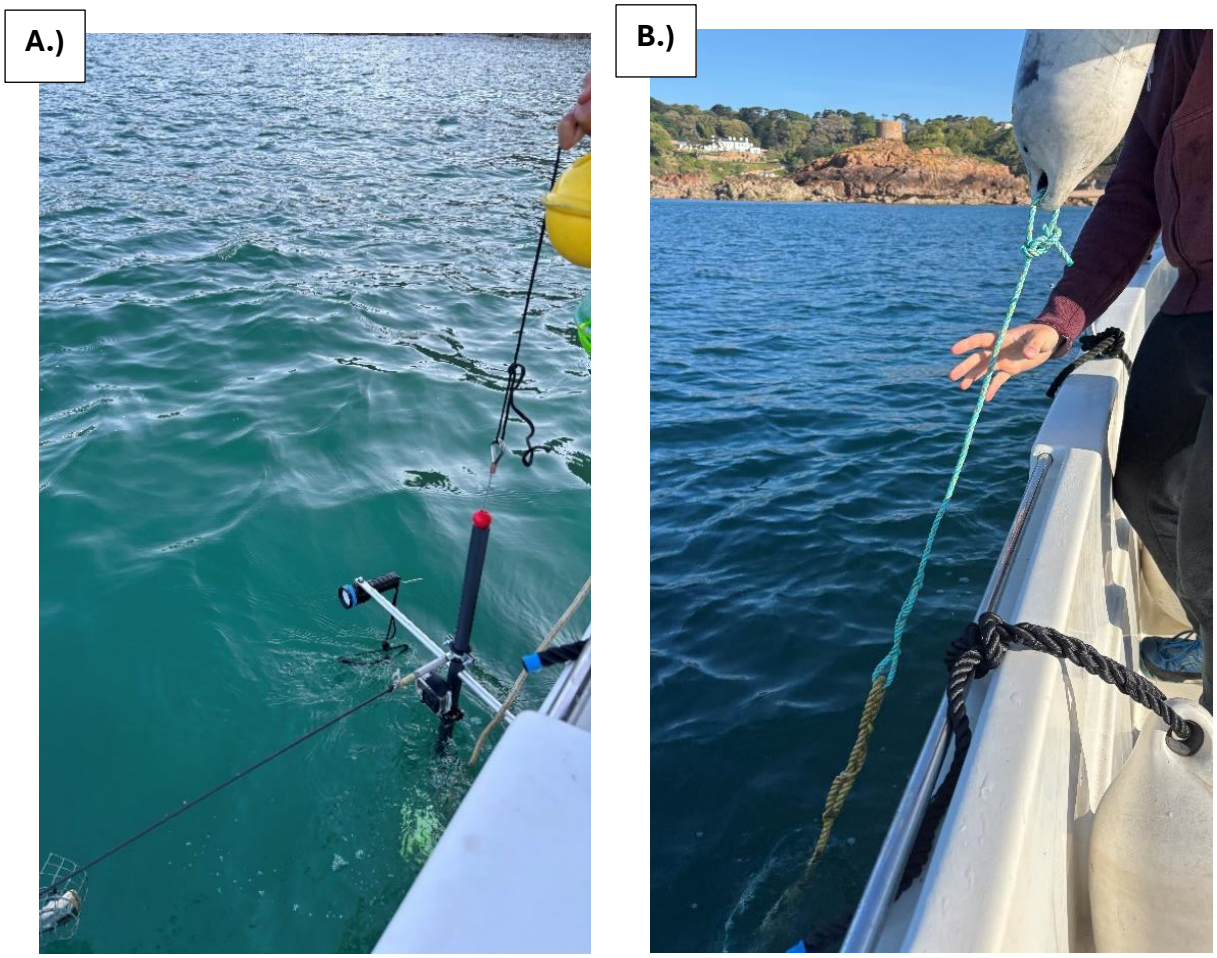


Figure 6, Photos taken during testing of the new homemade BRUV onboard private vessel depicting A.) Deployment of the unit and B.) Retrieval of the unit one hour later

BRUV Deployment (Night)

Three homemade Baited Remote Underwater Video (BRUV) units were deployed during dusk at two sites: PTLNTZ and the control site, Beauport Bay. Deployments were carried out on the 16th, 17th, 21st and 22nd of July 2025, with an extra supplementary drop on the 8th of August 2025. Three BRUVs were deployed at each site per night, totalling 30 deployments. Deployments occurred ~30 minutes after sunset and during neap low tides, but due to specific dusk timing, ideal tide conditions were not always achieved.

For night-time deployments, the bait and camera setup remained consistent with daytime methods; however, the Xtar D30 1600 dive torches were activated by inserting batteries, selecting white light, and setting brightness to 800 lumens (Figure 7a). White light was chosen over red light to maximise visibility and ensure accurate species identification (pers. obs., 2025), despite potential for behavioural disturbance (Harvey et al., 2012). After all rigs were deployed Jersey Coastguard was informed of locations and units were left in place overnight and recovered in the morning (Figure 7b). On the 17th and 22nd of July, camera batteries were replaced after recovery, and rigs were re-deployed for daytime sampling.

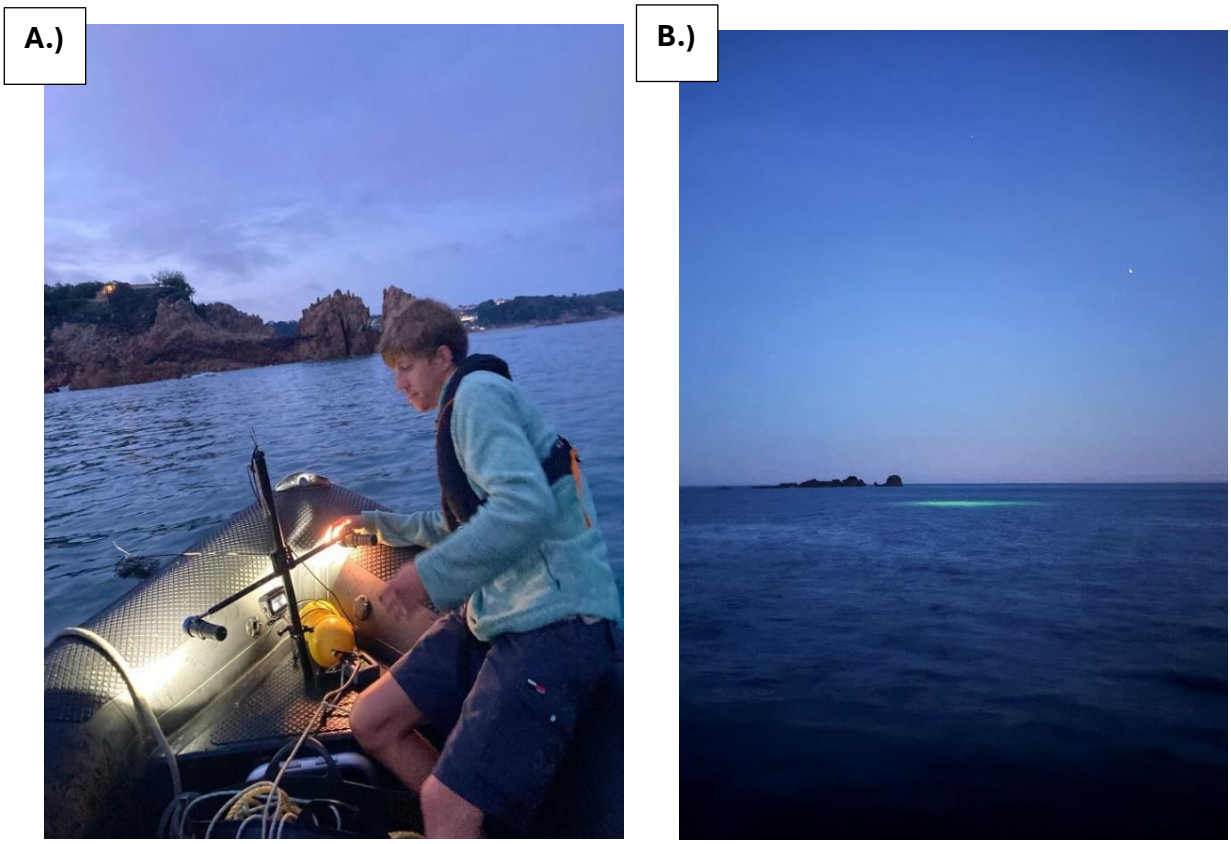


Figure 7, Photos taken during deployment of homemade BRUV onboard Marine resources vessel "Ecrehou" A.) Turning on of torches prior to deployment and B.) Light produced from the unit on the seabed

BRUV Footage Review

Following retrieval of the BRUV system, each video recording was reviewed to confirm a minimum of 40 minutes of usable footage for diurnal deployments and 2 hours of footage for nocturnal deployments, during which the bait remained present within the cage and the field of view was unobstructed (Figure 8a and b).

BRUV footage was first uploaded from the GoPro camera to a computer for analysis. Each video was reviewed from the point at which the rig had visibly settled on the seabed, ranging from 2-5 minutes into the recording. From this point, a 40-minute segment of footage was analysed for diurnal deployments and 2 hours for nocturnal deployments. For each minute of the selected timeframe, the Maximum Number of individuals of each observed species visible within a single frame of the footage was recorded (MaxN). This process was repeated for all identifiable species throughout the designated time.

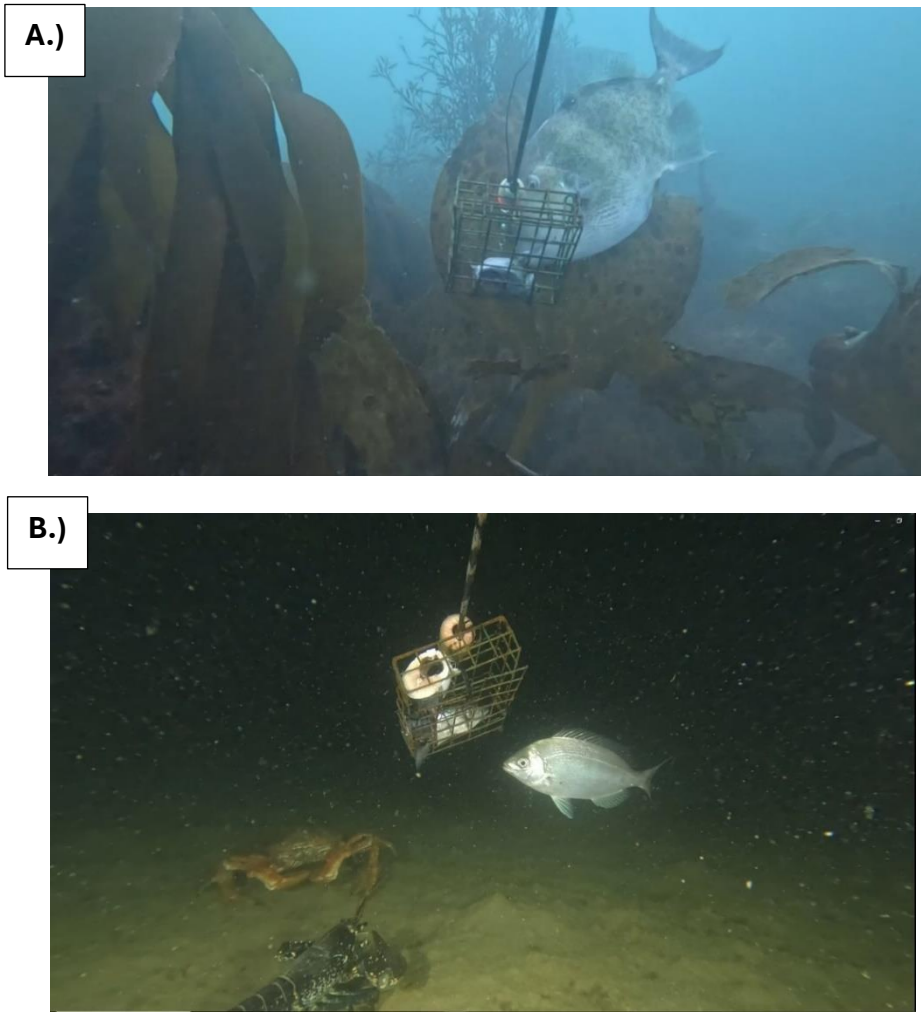


Figure 8, Underwater view of A.) Diurnal BRUV with one grey trigger fish feeding on the bait cage and B.) The nocturnal view with a black bream, European lobster and spider crab present

2.3 Data Analysis

All statistical analysis was performed in RStudio (R Core Team, 2024) Version (4.4.1) using the dplyr package (Wickham et al., 2023) for data filtering and manipulation, statistics for generalized linear models and model selection. The lme4 package (Bates et al., 2015) was utilised for generalized linear mixed models. The emmeans package (Lenth, 2024) allowed for pairwise comparisons of model estimates and the vegan package (Oksanen et al., 2024) provided community ecology analyses, including PERMANOVA and SIMPER. Finally, the ggplot2 package (Wickham, 2016) was responsible for the visualization of raw data and model results.

2.3.1 Lobster pot trials

Crustacean Size

H. gammarus and *M. brachydactyla* were analysed for changes in size following the implementation of PTLNTZ. The long-term dataset was filtered to include all the years the PTLNTZ was included in potting trials (2022-2025) and sites to compare were restricted to two sites; OPTLNTZ and Ouaisné, which share shallower water deployments with PTLNTZ. Carapace length was then analysed to assess differences between locations and over time using a General Linearized Model. An interactive GLM fitted using the Gaussian model, including location and year, was produced. Following this, pairwise comparisons were performed to identify significant differences between locations across the years. Trends in *H. Gammarus* and *M. brachydactyla* size over time were visualized using line plots of raw data.

Crustacean Biomass

H. Gammarus and *M. brachydactyla* catches were analysed to assess both temporal and spatial patterns in relation to PTLNTZ. Firstly, overall Catch Per Unit Effort (CPUE) was calculated as the mean number of each species per set (10 pots per set) for all the shallow water sites; OPTLNTZ, PTLNTZ and Ouaisné. To test for differences in *H. gammarus* and *M. brachydactyla* abundance across the years, CPUE and location were included in an interactive GLM fitted using Poisson model. Post GLM pairwise comparisons were produced for each location within each year. Line plots of raw data were used to depict changes in *H. gammarus* and *M. brachydactyla* size over time.

2.3.2 BRUVs

Diurnal Overall Abundance and Diversity

Combined GLMM's using Poisson distribution were fitted to assess temporal changes in the overall abundance of individuals and species in PTLNTZ and Beauport Bay. For overall abundance, Netted dog Whelk *(Tritia reticulata)* counts were removed to reduce overdispersion of the models. Total counts were modelled as a function of bay, year and their interaction with habitat type as a random effect following best fit from the Akaike Information Criterion (AIC).

Site-specific GLMM's were performed by filtering the dataset to include only one bay, allowing for assessment of temporal overall abundance and overall diversity change in each bay. Differences between locations across the five years were assessed using pairwise comparisons. Bar and line plots of raw data were produced to complement GLMM results and furthermore, two tables were formed identifying both species that saw the greatest increases and decreases in abundance at PTLNTZ.

Diurnal Commercial Species Filter

To test if commercially targeted species were disproportionately benefiting from the NTZ data was filtered to contain only species with an annual landing value greater than £1,000 (Table 1), *M. brachydactyla* was excluded from this filtered dataset, as it was the only commercially relevant species of the class *Malacostraca* recorded on the Diurnal BRUVs and due to its specific analysis already within potting trial data. The commercial dataset was then analysed looking at overall abundance and diversity following same methodology as the unfiltered data.

Diurnal Species Assemblage

To examine changes in diurnal species composition in PTLNTZ and Beauport between 2021 and 2025 PERMANOVAs were conducted using Bray–Curtis dissimilarities with 999 permutations, found testing for significant differences between years. To identify species contributions towards dissimilarity, Similarity Percentage (SIMPER) was performed. The top ten contributors were then plotted to visualize changes in species abundance.

*Table 1, All Wet fish species recorded in Diurnal BRUV's with > £1,000 landed annual value in Jersey. * S. cantharus and S. officianis also have high reported landings in 2023 by French vessels in Jersey waters; 18,,000kg and 17,000kg respectively, but annual landing value not detailed. Data taken from the Marine Resources Annual Report (2024).*

COMMERCIAL SPECIES	NET WEIGHT (KG)	ANNUAL LANDING VALUE (£)
European seabass (*Dicentrachus labrax*)	4,611	55,000
Atlantic Mackerel (*Scomber scombrus*)	5,142	20,000
Black seabream (*Spondyliosoma cantharus*) *	3,493	15,000
Mullet spp. (*Chelon spp.)*	1,119	14,000
Common cuttlefish (Sepia officianis) *	9,952	10,000
Pollack (*Pollachius pollachius*)	1,997	8,000
Ballan wrasse (*Labridae bergylta*)	759	4,000
Small-Spotted catshark (*Scyliohinus canicula*)	DD	3,000
Common sole (*Solea solea)*	133	2,000
Bull huss (*Scyliorhinus stellaris*)	887	1,000

Nocturnal BRUVs

Abundance and species data was analysed using GLM's with Poisson distribution. The model, including both Bay and Habitat as fixed effects, provided the best fit, selected based on AIC. Like the diurnal data boxplots of abundance and species data were produced. Species assemblage was analysed following the Diurnal BRUV data methods. Finally overall species proportions in 2025 between diurnal and nocturnal were plotted displaying difference in assemblage between diurnal and nocturnal data collection

3. Results

All raw outputs of GLMs, GLMMs, Pairwise Comparisons, PERMANOVAs and SIMPER Tables can be found in appendix.

3.1 Lobster Pot Trials

Between 2022 and 2025, three sets of ten pots were deployed annually at each of the three sites: PTLNTZ, OPTLNTZ and Ouaisné. In total, 36 strings were deployed over the study period, with M. brachydactlya being the most frequently caught species, followed by *H. gammarus* and *C. pagarus* (Table 2). *C. pagarus* data was not analysed due to low individual counts.

Table 2, Total captures of the 3 main commercial species caught in Jerseys commercial potting industry split by site (PTLNTZ, OPTLNTZ and Ouaisné) and Year (2022-2025)

SPECIES	SITE	2022	2023	2024	2025	TOTAL
H. gammarus	PTLNTZ	29	85	56	39	209
	OPTLNTZ	28	35	26	24	113
	Ouaisné	16	25	12	25	78
SUBTOTAL		**73**	**145**	**94**	**88**	**400**
M. brachydactyla	PTLNTZ	17	29	26	56	128
	OPTLNTZ	88	96	61	77	322
	Ouaisné	52	92	80	127	351
SUBTOTAL		**157**	**217**	**167**	**260**	**801**
C. pagarus	PTLNTZ	0	3	0	2	5
	OPTLNTZ	5	12	1	1	14
	Ouaisné	0	3	0	2	5
SUBTOTAL		**5**	**18**	**1**	**5**	**24**

3.1.1 Crustacean Size

Hommarus Gammarus

Since recording started in 2022, *H. gammarus* has always been larger in the Portelet than the two comparable locations, OPTLNTZ and Ouaisné, with PTLNTZ 2025 achieving the highest mean carapace length recorded to date (85.3 ± 1.44, Figure 9). Despite this, the interactive GLMM model showed there was no significant effect on carapace length, only a marginal significant difference between Ouaisné and PTLNTZ in 2025 ($p = 0.0790$) (see Table S1a).

Post GLM pairwise comparisons saw no difference in mean carapace length between 2022-2024, however, in 2025 PLTNTZ *H. gammarus* were found to be significantly larger (85.3 ± 1.44) than both OPTLNTZ (76.9 ± 1.83, $p = 0.0019$) and Ouaisné (75.6 ± 1.79, $p = 0.0002$). No significant differences were observed between OPTLNTZ and Ouaisné in 2025 (see Table S2a).

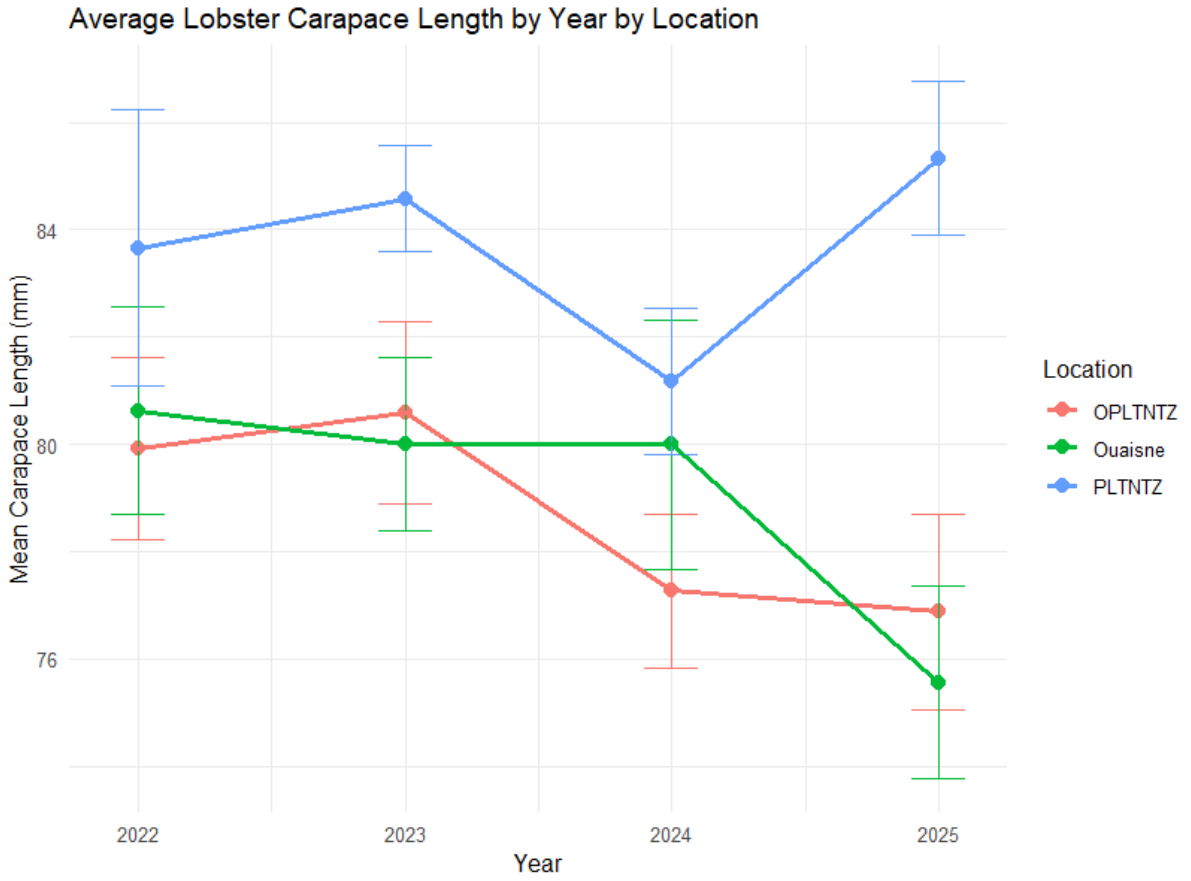


Figure 9, Average carapace length of European lobster between sites (PTLNTZ, OPTLNTZ and Ouaisné) with error bars denoting Standard Error (SE)

Maja brachydactyla

Average carapace length of *M. brachydactyla* across sites increased from the baseline in 2023 ($p = 0.035$), 2024 ($p = 0.002$) and 2025 ($p < 0.001$), driven by increases in PTLNTZ (see Figure 10). In contrast, OPLTNTZ and Ouaisné did not follow this increasing trajectory, instead showing relatively stable or slightly decreasing mean sizes. These divergent temporal patterns were reflected in significant interaction effects, with both OPLTNTZ and Ouaisné differing from Portelet in 2024 ($p \leq 0.002$ for both) and 2025 ($p \leq 0.006$ for both) (see Table S1b).

Pairwise comparisons of estimated marginal means indicated that differences in *M. brachydactyla* carapace length between positions varied across years. In 2022, crabs from PLTNTZ (110.1 ± 5.36) were significantly smaller than those from both OPLTNTZ (127.1 ± 1.55) and Ouaisné (127 ± 1.80) ($p < 0.01$ for both). In 2023 and 2024, no significant differences among positions were detected. However, by 2025 PLTNTZ (128.5 ± 2.34) and OPLTNTZ (130.27 ± 1.99) carapace length was significantly larger than Ouaisné (121.7 ± 1.88) ($p < 0.05$ for both) (see Table S2b).

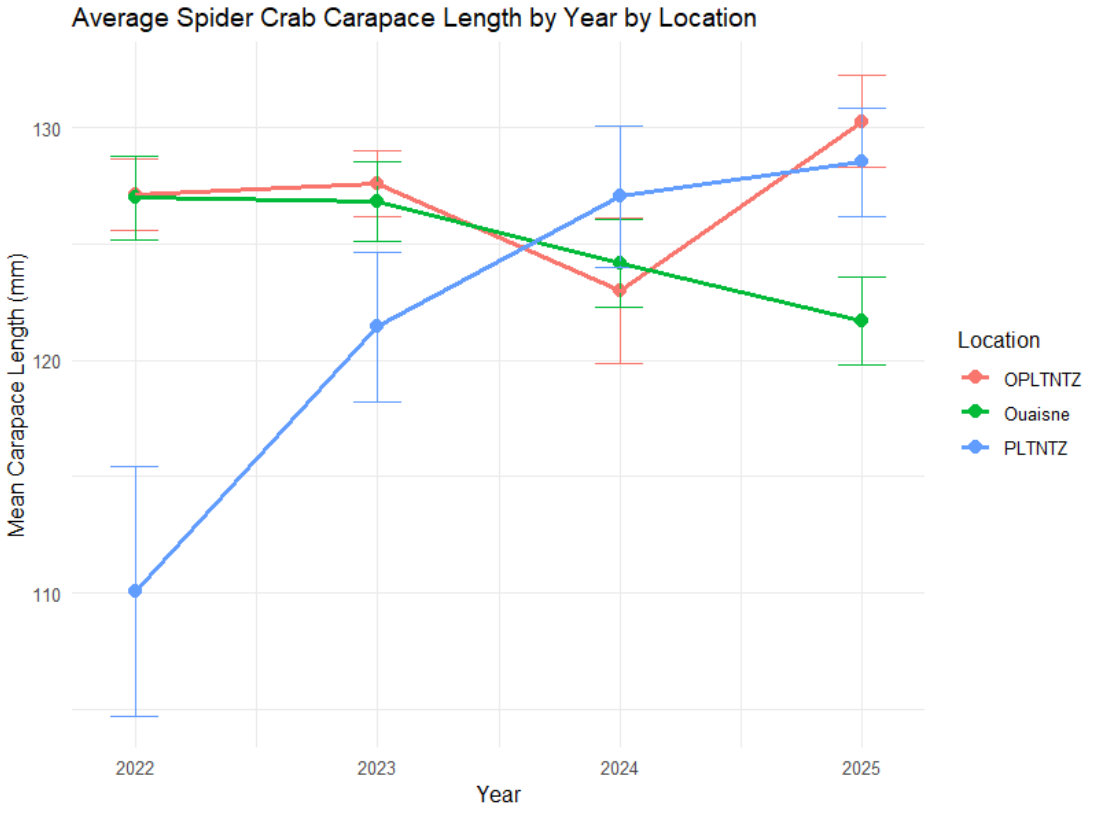


Figure 10 , Average carapace length of Spider Crab between sites (PTLNTZ, OPTLNTZ and Ouaisné) with error bars denoting Standard Error (SE).

3.1.2 Crustacean Biomass

Hommarus Gammarus

The GLM examining *H. gammarus* catch counts across sites and years indicated a significant increase in CPUE in 2023, driven by the significant increase at PTLNTZ from 2022 (1.45 ± 0.15) to 2023 CPUE (2.83 ± 0.13, p = 0.0018), whereas 2024 and 2025 did not significantly differ from the baseline CPUE. Neither OPLTNTZ or Ouaisné had significantly higher baseline catches in 2022 compared to PLTNTZ; however, both sites diverged significantly from PTLNTZ's trajectory in 2023 (OPTLNTZ; p = 0.010 and Ouaisné; p = 0.103) and 2024 (OPTLNTZ; p = 0.040 and Ouaisné; p = 0.034), reflecting an increase in CPUE in PTLNTZ (see Table S3a). By 2025, site-specific trends no longer differed significantly (Figure 11).

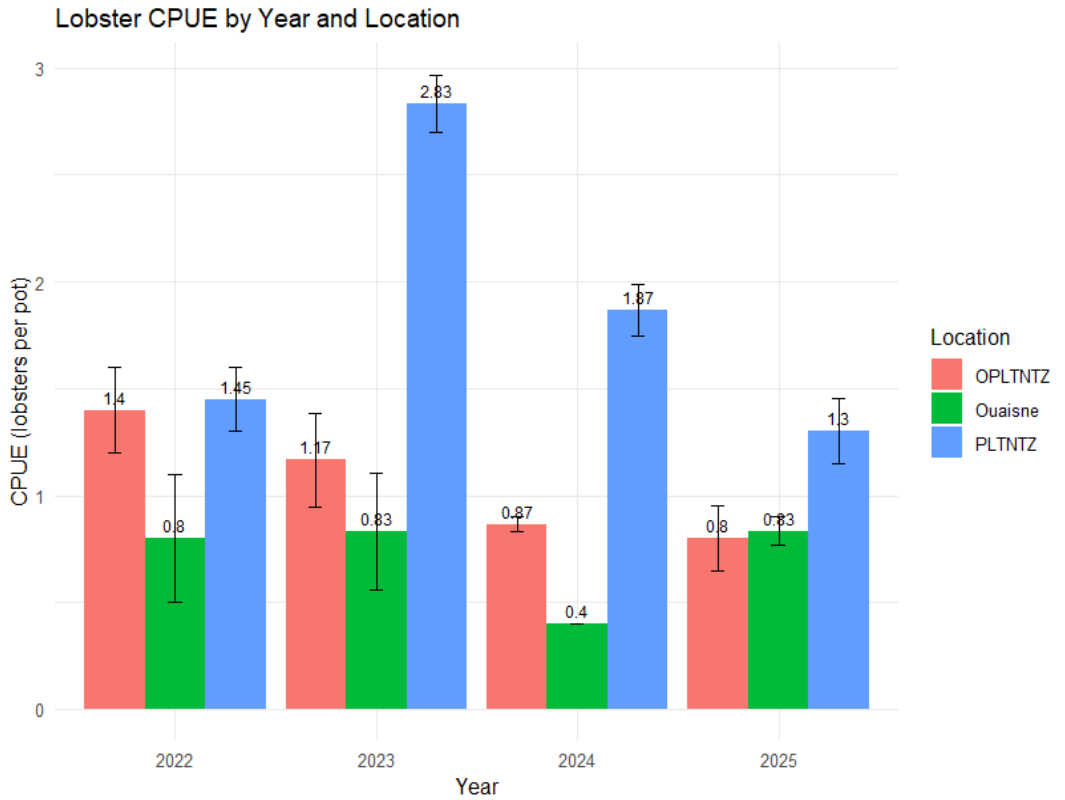


Figure 11, Barplot of average CPUE of European lobster at each site (PTLNTZ, OPTLNTZ and Ouaisné) in each year (2022-2025) with error bars denoting SE.

Pairwise comparisons revealed significantly lower CPUE in 2023 ($p < 0.0001$) and 2024 ($p < 0.01$) within OPTLNTZ (2023; 1.2 ± 0.22, 2024; 0.9 ± 0.03) and Ouaisné (2023; 0.8 ± 0.27, 2024; 0.4 ± 0) compared to PTLNTZ (2023; 2.8 ± 0.13, 2024; 1.9 ± 0.12). There was no significant differences between OPTLNTZ and Ouaisné across the study (see Table S4a).

Maja brachydactyla

Across the study period, CPUE in PTLNTZ increased year on year. The interactive GLM examining CPUE across sites and years indicated that by 2025, CPUE had a significant increase relative to the 2022 baseline level ($p = 0.0045$). This was driven by increases in both PTLNTZ and Ouaisné. Interaction terms revealed that change in PLTNTZ and OPLTNTZ significantly differed in 2024 ($p = 0.025$) and 2025 ($p = 0.0038$), reflecting PLTNTZ's steady increases in comparison to OPTLNTZ's stable CPUE (Figure 12, see Table S3b). Ouaisné did not differ significantly from PTLNTZ.

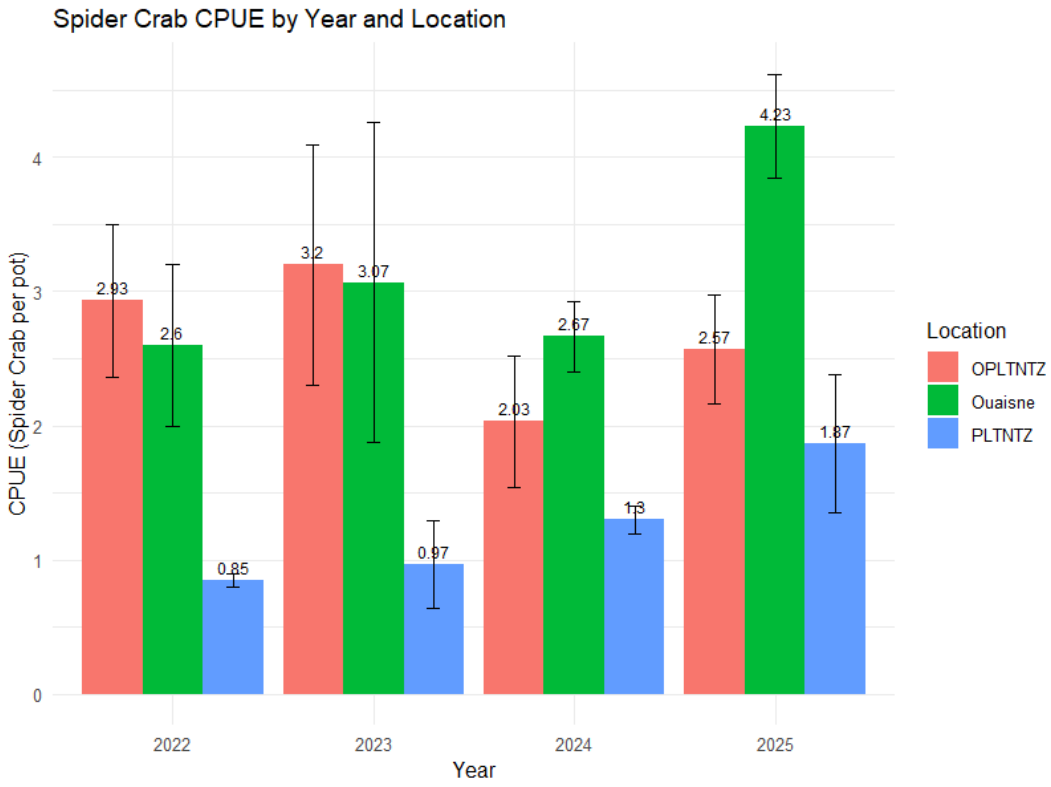


Figure 12, Barplot of average CPUE of Spider Crab at each site (PTLNTZ, OPTLNTZ and Ouaisné) in each year (2022-2025) with error bars denoting SE.

Pairwise comparisons showed that in 2022, CPUE at OPLTNTZ (2.93 ± 0.57) and Ouaisné (2.6 ± 0.60) were significantly higher than Portelet (0.85 ± 0.05, both $p \leq 0.002$). By 2023 and 2024, no significant size differences were detected among sites. However, in 2025, Ouaisné had significantly greater CPUE (4.23 ± 0.38) than both Portelet (1.9 ± 0.52, $p = 0.043$) and OPLTNTZ (2.57 ± 0.41, $p = 0.002$), while Portelet and OPLTNTZ did not differ (see Table S4b).

3.2 Diurnal BRUVs

Between 2021-2025 in total, 94 successful Diurnal BRUVS were deployed, capturing a total of 63 hours of footage. Throughout this time, 33 species were recorded, of which 19 were *Actinopterygii*, 7 were *Elasmobranchii*, 4 were *Malacostraca*, 1 was *Gastropoda,* and 1 was *Cephalopoda* (Table 3)

Table 3, Total counts of each species recorded across 2021 – 2025 Diurnal BRUV surveys in brackets, grouped by class

Actinopterygii	***Elasmobrancii***
S. cantharus (306)	*S. canicula (42)*
T. tracharus (101)	*S. stellaris (3)*
P. pollachius (86)	*R. undulata (2)*
T. luscus (78)	*R. microocellata (2)*
Chelon spp (60)	*Raja spp. (2)*
M. surmulletus (51)	*R. brachyura (1)*
L. bergylta (46)	*T. marmorata (1)*
C.lyra (37)	
D. labrax (30)	***Malacostraca***
S. melops (15)	
S. scombrus (15)	*M. brachydactyla (136)*
A. tobianus (13)	*Pagarus spp. (89)*
G. flavescens (11)	*Liocarcinuss pp. (79)*
H. lanceolatus (10)	*M. squinado (13)*
C. conger (10)	*Macropodia spp. (4)*
L. mixtus (8)	
C. rupestris (5)	***Cepholapoda***
B. capriscus (4)	
C. lucerne (3)	*S. officianis (3)*
	Gastropoda
	T. reticulata (1354)

3.2.1 Overall Abundance

Following the GLMM, it was found that Beauport had higher overall abundance across the study period than Portelet ($p < 0.001$). Nevertheless, interaction with year showed the Portelet overall abundance shifted significantly differently from expectations in 2022 ($p = 0.003$), 2024 ($p < 0.001$), and 2025 ($p < 0.001$), Beauport saw no significant shifts in overall abundance with year (Figure 13). Variance was < 0.01 and therefore, the random factor habitat was deemed not a contributing factor to overall abundance (see Table S5a). Species with highest change in abundances driving change were compiled (Table 4 and 5).

Pairwise comparisons confirmed that overall abundance in Beauport (13.3 ± 2) was significantly higher than Portelet (7.7 ± 1.23) in 2021 ($p < 0.0001$); it was no longer significantly different in 2022, 2024 and 2025 (see Table S6a).

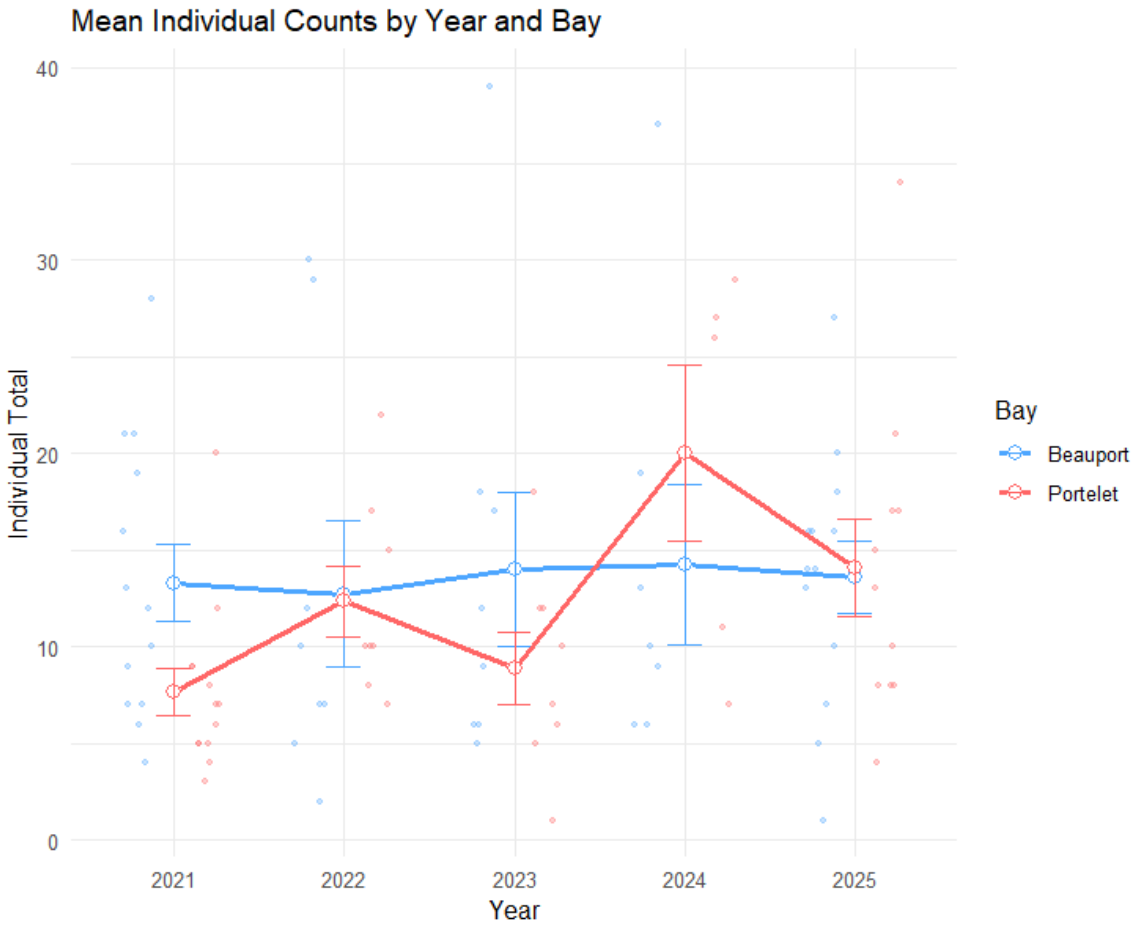


Figure 13, Raw overall abundance counts at Beauport and PTLNTZ, across the study period (2022-2025). Line plot showing means with standard error bars, with jittered points representing individual BRUV deployments

Table 4, Species with > 0.2 mean abundance increase in PTLNTZ, with abundance change in Beauport comparison. Species with greater increase within PTLNTZ compared to Beauport highlighted in red

SPECIES	ABSOLUTE MEAN ABUNDANCE INCREASE (PORTELET 21-25)	ABSOLUTE MEAN ABUNDANCE INCREASE (BEAUPORT 21-25)
Black seabream (*Spondyliosoma cantharus*)	5.21	5.54
Mullet spp. (*Chelon spp.)*	2.10	0.92
European seabass (*Dicentrachus labrax*)	0.73	0.31
Red mullet *(Mullus surmulletus)*	0.545	0.077
Ballan wrasse (*Labridae bergylta*)	0.433	0.538
Grey triggerfish *(Balistes capriscus)*	0.27	0
Lesser sand eel *(Ammodytes tobianus)*	0.27	0.23
Conger eel *(Conger conger)*	0.27	0.303

Table 5, Species with <-0.2 mean abundance decreases in PTLNTZ, with comparable abundance change in Beauport

SPECIES	ABSOLUTE MEAN ABUNDANCE DECREASE (PORTELET 21-25)	ABSOLUTE MEAN ABUNDANCE DECREASE (BEAUPORT 21-25)
Hermit Crab spp. (*Pagarus spp.)*	-1.462	-2.846
Small-spotted catshark (*scyliohinus canicula*)	-0.944	-1.000
Pollack (*pollachius pollachius*)	-0.734	-0.154
Common spider crab (*Maja brachydactyla*)	-0.510	-3.384

3.2.2 Commercial Abundance

A similar pattern was observed for commercially targeted species, but individual total saw a greater increase in PTLNTZ from 2021 (5.1 ± 1.43) to 2025 (11.5 ± 2.42) (Figure 14). Portelet saw significant increases relative to Beauport in 2024 (p = 0.019) and 2025 (p = 0.012) (see Table S5b). Pairwise results only saw significant differences between sites in 2021 (p = 0.0039) (see Table S6b).

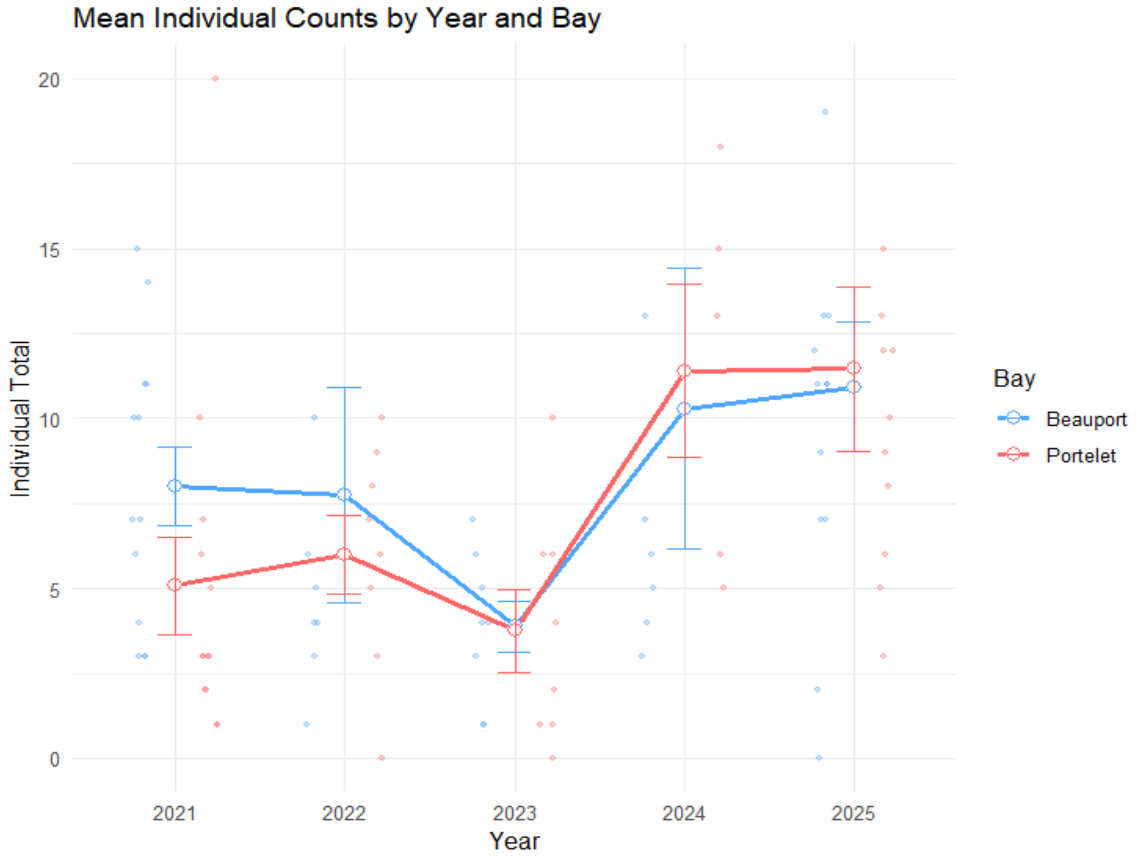


Figure 14, Raw overall commercial species abundance counts at Beauport and PTLNTZ across the study period (2022-2025). Boxplots showing median and interquartile ranges, with jittered points representing individual BRUV deployments.

3.2.3 Species Richness

The GLMM examining species richness indicated that counts at Portelet increased significantly in 2024 relative to baseline (p = 0.003). No other years showed significant changes at Portelet (Figure 15) There was no evidence of differing temporal trajectories between bays. Variance was < 0.001 indicating habitat was not contributing to differences in species richness (see Table S7a).

Post GLMM pairwise comparisons confirmed no differences in species richness for each year, although Portelet 2024 saw a marginal significant difference to Beauport ($p = 0.0545$), but 2024 data must be taken with caution due to low replicate counts (see Table S8a).

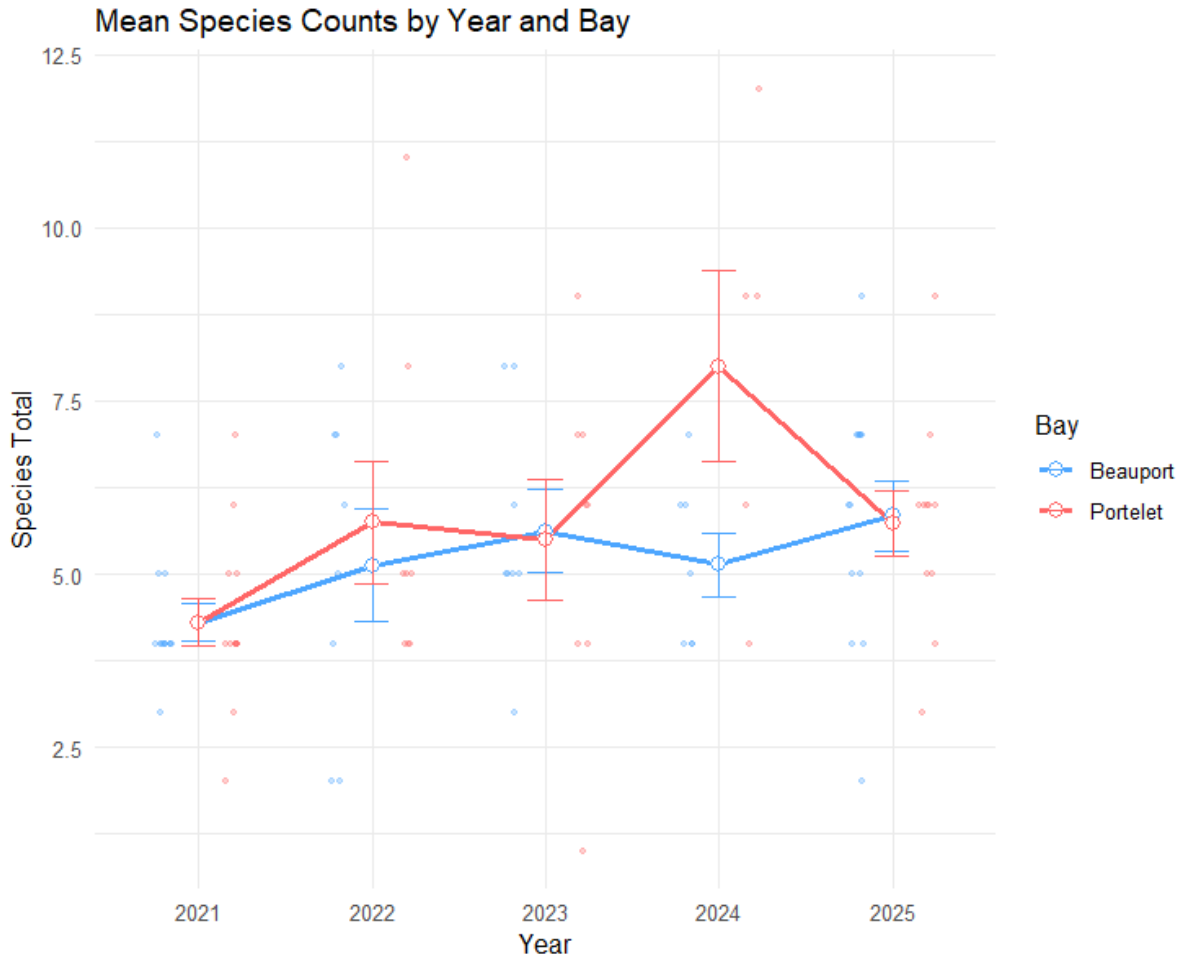


Figure 15, Raw species counts at Beauport and PTLNTZ across the study period (2022-2025). Line plot showing means with SE error bars, with jittered points representing individual BRUV deployments

3.2.4 Commercial Species Richness

When restricted to commercial species only, the GLMM produced the same outcome, no significant temporal changes were detected at Beauport, just the increase at Portelet in 2024 was no longer of marginal significance (see Table S7b).

3.2.5 Species Assemblage

Portelet and Beauport

PERMANOVA's revealed that year had a significant effect at both Beauport (R^2 = 0.2577, p = 0.0001) and PTLNTZ (R^2 = 0.1940, p = 0.0003) (see Table S9a and S10a). SIMPER analysis of PTLNTZ (Figure 16a) saw *S. cantharus* as the highest contributor to dissimilarity (0.24730, p = 0.002). Other contributors included *Chelon spp.* (0.06589, p = 0.003), *D. labrax* (0.03441, p = 0.002) and *M. surmulletus* (0.02294, p = 0.004). Several additional taxa contributed < 2% towards dissimilarity. A further number of taxa contributed to dissimilarity but were not significantly different between sites (see Table S9b).

SIMPER analysis of Beauport (Figure 16b) also saw *S. cantharus* as the greatest contributor to dissimilarity (0.23037, p = 0.003). Other contributors included *M. brachydactyla* (0.1468, *p* = 0.013), *Pagarus spp.(0.1016*, *p* = 0.001), *S. canicula* (0.0415, *p* = 0.005), *L. bergylta* (0.0327, *p* = 0.026), and *T. luscus* (0.0289, *p* = 0.033). Additional species accounted for part of the dissimilarity, although these differences were not significant between sites (see Table S10b).

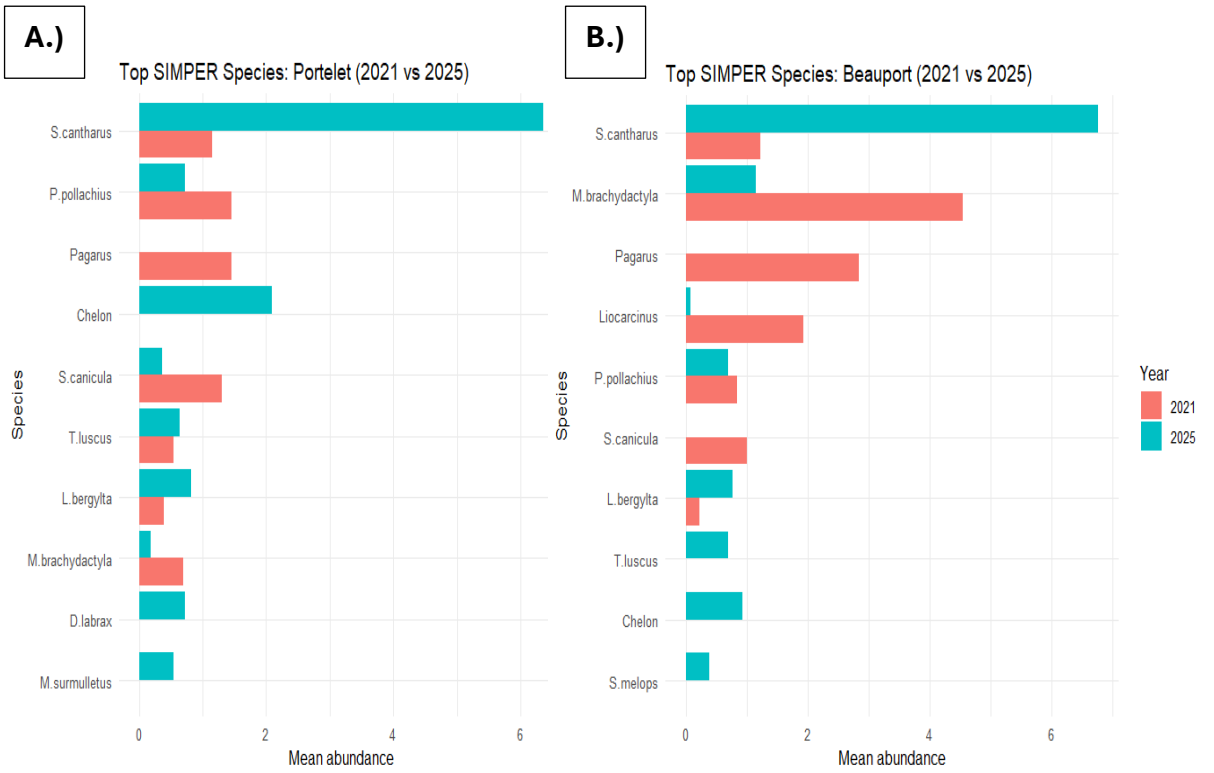


Figure 16, Mean abundances of species contributing most to community dissimilarity (SIMPER analysis) in A.) Portelet and B.) Beauport in 2021 and 2025. Bars showing mean abundance of each species, with years distinguished by colour.

Portelet vs Beauport

PERMANOVA analyses comparing community composition between PTLNTZ and Beauport showed a marginal significant effect in 2021 ($R^2 = 0.073$, $p = 0.067$) and a non-significant effect in 2025 ($R^2 = 0.03$, $p = 0.729$) (see Table S11a and S12a). SIMPER analysis of 2021 (Figure 17a) saw *M. brachydactyla* account for most dissimilarity (0.1842, p = 0.002). The only other species contributing > 2% that was also significantly different between sites was *Liocarcinus* spp. (0.0775, $p = 0.035$), although several other species contributed > 2% to dissimilarity, but were not statistically significant. SIMPER Analysis of 2025 (Figure 17b) found no species significantly different between PTLNTZ and Beauport (see Table S11b and S12b).

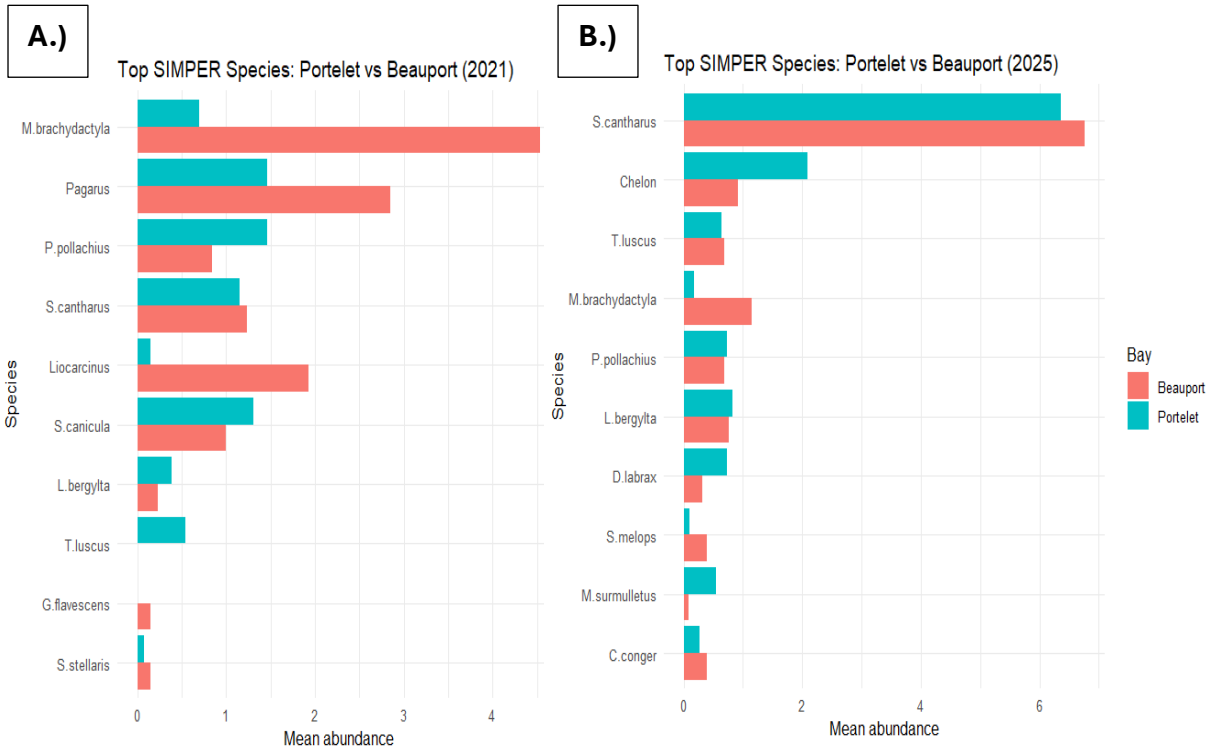


Figure 17, Mean abundances of species contributing most to community dissimilarity (SIMPER analysis) between PTLNTZ and Beauport in A.) 2021 and B.) 2025. Bars showing mean abundance of each species, with years distinguished by colour.

3.3 Nocturnal BRUVS

During 2025, 18 successful Nocturnal BRUVs were deployed, capturing a total of 36 hours of usable footage. Throughout this time, 30 species were recorded, of which 19 were *Actinopterygii*, 2 were *Elasmobranchii*, 4 were *Malacostraca* and 3 were *Cephalopoda* (Table 4).

Table 6, Total counts of each species recorded across the 2025 Nocturnal BRUV surveys in brackets, grouped by class.

Actinopterygii	***Elasmobranchii***
T. luscus (98)	*R. microocellata (1)*
T.tracharus (32)	*S. canicula (1)*
A. presbyter (25)	
P. pollachius (15)	***Malacostraca***
C. conger (8)	
S. cantharus (8)	*M. brachydactyla (14)*
Chelon spp. (5)	*H. gammarus (5)*
S. solea (5)	*Liocarcinus spp. (5)*
A. tobianus (4)	*Pagarus spp. (4)*
L. bergylta (4)	*C. pagarus (2)*
M. merlanguis (4)	
M. surmulletus (2)	***Cephalopoda***
D. labrax (1)	
L. mixtus (1)	*L. vulgaris (18)*
E. vipera (1)	*S. officianis (6)*
G. flavescens (1)	*S. atlantica (5)*
S. Melops (1)	
T. bulbalis (1)	
Z. punctatus (1)	

3.3.1 Nocturnal vs Diurnal

Visual inspection of species composition indicates a clear difference in species assemblage between day and night recordings (Figure 18). Seven species were consistently only observed during the day BRUV recording. Ten species were only recorded during night BRUV recording.

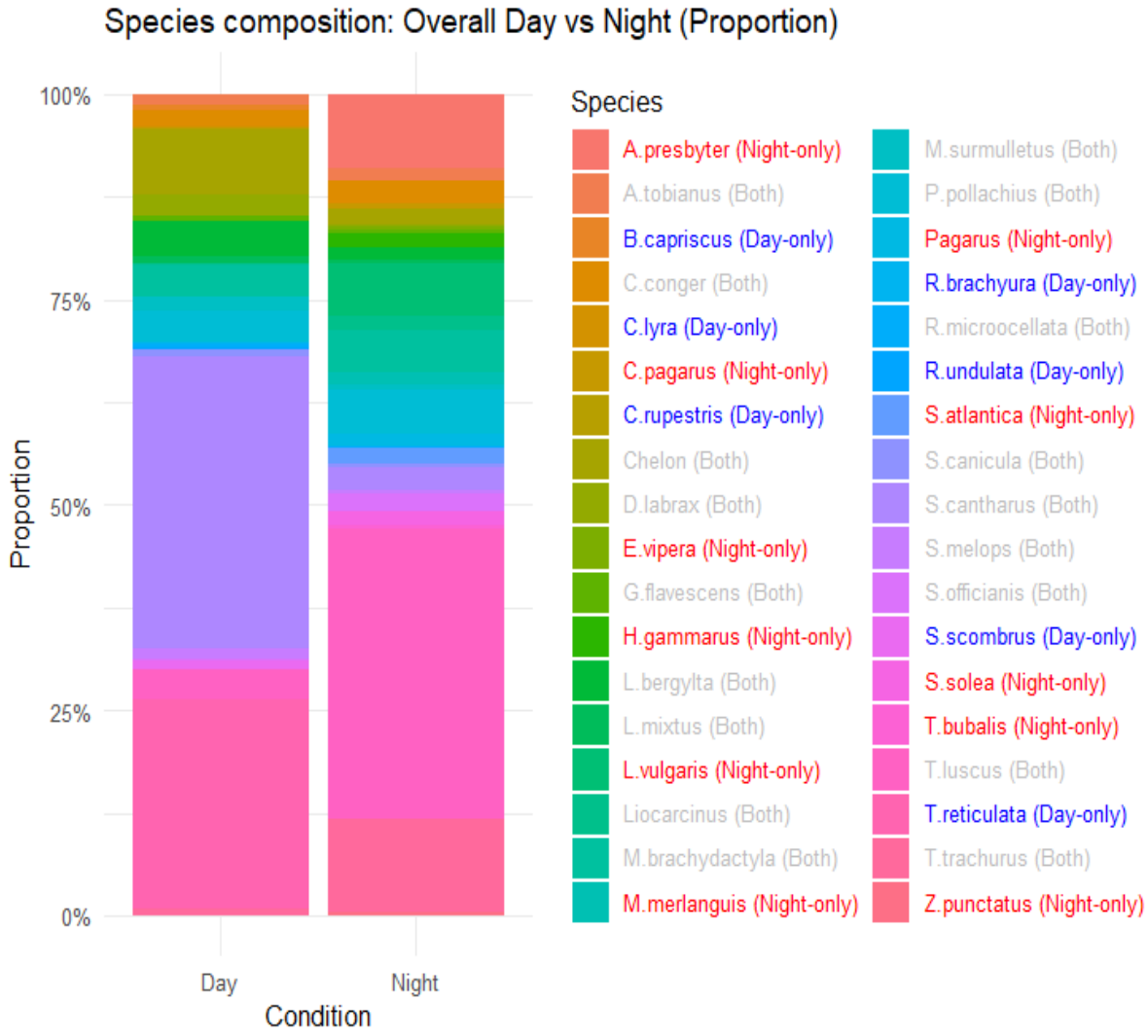


Figure 18, Proportional composition of species observed during diurnal and nocturnal conditions. Bars show relative contribution of each species to total abundance for each condition, with species color-coded and labelled according to their occurrence category: Day-only (blue), Night-only (red), or Both (grey).

3.3.2 Nocturnal Overall Abundance

For overall abundance, the GLM found Habitat had a significant effect, with higher counts in sand habitats over kelp ($p < 0.001$), but bay showed no significant effect ($p = 0.790$) (Figure 19, see Table S13a).

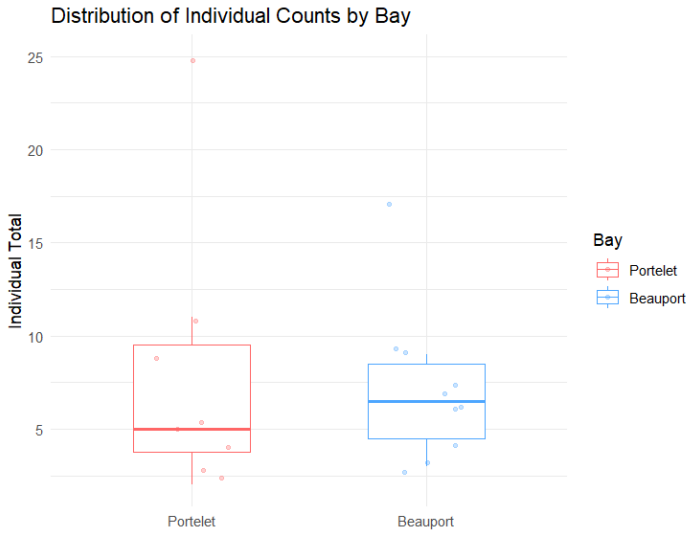


Figure 19, Raw overall abundance counts at Beauport and PTLNTZ in 2025. Boxplots showing median and interquartile ranges, with jittered points representing individual BRUV deployments.

3.3.3 Nocturnal Species Richness

For species richness, the GLM found neither Bay ($p = 0.906$) neither Habitat ($p = 0.442$) had a significant effect (Figure 20, see Table S13b).

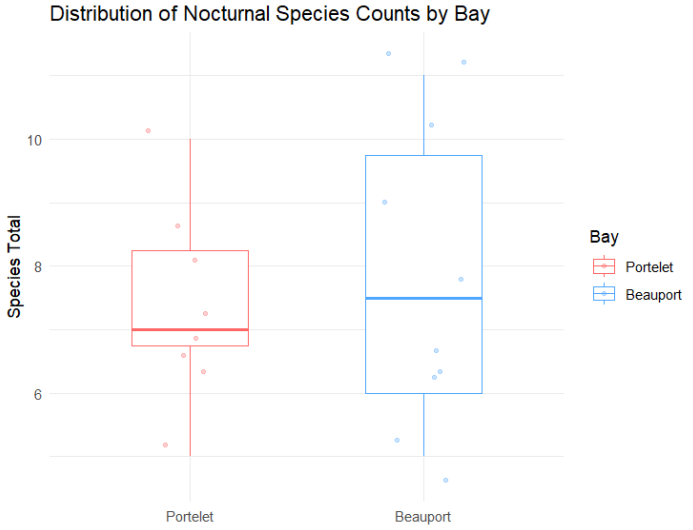


Figure 20, Raw overall species counts at Beauport and PTLNTZ in 2025. Boxplots showing median and interquartile ranges, with jittered points representing individual BRUV deployments.

3.3.4 Species Assemblage

Portelet vs Beauport

A PERMANOVA comparing fish community composition between Portelet and Beauport in 2025 found no significant difference (R^2 = 0.079, p = 0.21) (see Table S14a). Subsequent SIMPER analysis indicated that *T. luscus* contributed the most to dissimilarity between bays but was not significant (0.1948, p = 0.195). Overall, no species were significant but both *S. cantharus* (0.0268, p = 0.056) and *C. conger* (0.0260, p = 0.062) saw marginal significance between bays (Figure 21, see Table S14b).

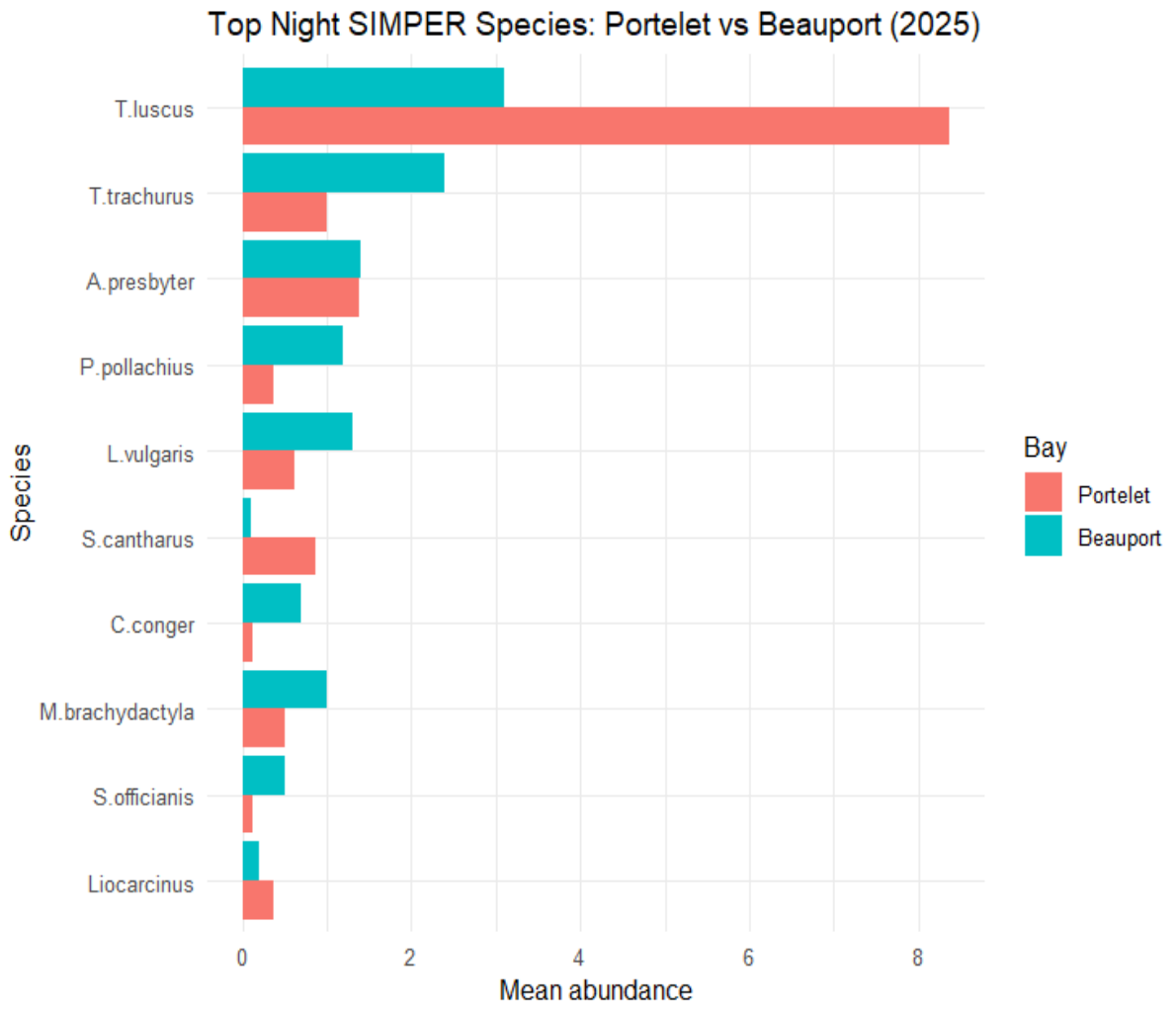


Figure 21, Mean abundances of species recorded during nocturnal deployments contributing most to community dissimilarity (SIMPER analysis).

4. Discussion

In 2022, the Portelet NTZ (PTLNTZ) was designated in Jersey to assess its benefit for both biodiversity and local fisheries via monitoring for an initial five-year period. This report marks the 5th year of data recording within PTLNTZ and provides an opportunity to assess 5 years of data. Results from four years of lobster pot trials suggest that commercially important *H. gammarus* and *M. brachydactyla* within PTLNTZ are increasing in size and their abundance is significantly changing. Findings from five years of BRUV deployments indicate overall abundance of individuals found within the NTZ is increasing and species richness is remaining stable. Species assemblage at PTLNTZ saw a significant shift over the 5 years with certain species becoming more abundant in 2025. Species assemblage between PTLNTZ and Beauport became more comparable across the study period. Results from 1 year of nocturnal BRUV surveys recorded ten species not observed during the five years of diurnal monitoring. Whilst differences emerged between day and night assemblages, nocturnal communities did not differ from those at the control site, and neither species richness nor overall abundance varied significantly between PTLNTZ and the control site.

4.1 Lobster pot Trials

Size and Abundance of H. Gammarus

Since monitoring began in 2022, *H. gammarus* individuals found within PTLNTZ have consistently been larger than those found in OPTLNTZ and Ouaisné, with 2025 for the first time having significantly larger specimens within the PTLNTZ. The effectiveness of NTZs in increasing *H. gammarus* mean size is likely linked to their limited home ranges (Smith et al., 2001), meaning that small reserves such as PTLNTZ can provide effective protection.

Larger H. Gammarus specimen size within PTLNTZ follows similar patterns observed in Kåvra (Bergström et al., 2022), Flamville (Amelot et al., 2024) and Lundy NTZ's (Hoskin et al., 2011), whereby cessation of fishing activity has allowed a reduction in harvest selection and a reversal in size structure truncation (Fernández-Chacón, et al., 2020). The increase in size structure within PTLNTZ will have positive ecological effects. Firstly, the reproductive output from this *H. Gammarus* population will increase as larger female individuals contribute

disproportionately to reproduction, with egg number and quality greater (Moland et al., 2010; Bergström et al., 2022). Crucially, larval export beyond PTLNTZ will provide a significantly greater contribution to the recovery of surrounding exploited populations in comparison to adult spillover (Diaz et al., 2011). Secondly, a return to an unexploited population size structure could, in turn, improve the ability of the *H. gammarus* population to withstand predicted increases in environmental variability, derived from increasing anthropogenic pressures (Planque et al., 2010), ensuring that even under sub-optimum conditions, this site can still provide a strong larval supply.

Regarding benefits outside the protected area, OPTLNTZ did not follow the upward trend in carapace size, but instead followed the slight downward trend also recorded at Ouaisné. This suggests adult spillover was low, a trend mirrored in Kåvra NTZ in Sweden, whereby it was noted that larger individuals displayed greater residency (Bergstrom et al., 2022). Further supporting this, a large, berried v-notched female was recaptured within PTLNTZ during the 2025 potting trials, one year after initial marking, indicating strong residency (Pers obs., 2025). The decrease in *H. gammarus* size in OPTLNTZ may be explained by a greater capacity for spillover of sublegal *H. gammarus*, as observed in the Lundy NTZ (Hoskin et al., 2011), or by intense fishing-the-line at OPTLNTZ (Pers Obs., 2025), which could be masking spillover effects. Without records of catches from fishers at this site, it is difficult to determine the extent of spillover of larger individuals from PTLNTZ into adjacent fished areas.

H. gammarus CPUE, meanwhile, initially underwent a large and significant increase from OPTLNTZ and Ouaisné in 2023, but against expectations decreased both in 2024 and again in 2025. CPUE, although still greater within PTLNTZ, was no longer significantly different from the other sites.

The initial spike of CPUE in 2023, one year on from NTZ designation, follows a similar trend of that in Lundy's NTZ (Hoskin et al., 2011). However, subsequent declines in *H. gammarus* abundance contrast with findings from larger reserves such as Kåvra NTZ (2.4 km^2) and Lundy's NTZ (4 km^2). A plausible explanation is that, owing to the considerably smaller size of PTLNTZ (0.26 km^2), carrying capacity may have been reached more rapidly. Intra-specific competition may have intensified earlier than other NTZs, as larger *H. gammarus* individuals, requiring more territory, often outcompete smaller individuals (Goñi et al. 2014), ultimately resulting in a self-thinning of population size. This theory is complemented by the consistent increases in mean *H. gammarus* carapace size found inside PTLNTZ. OPTLNTZ, on the other

hand, saw small decreases in CPUE from 2022 to 2025 with no evidence of *H. Gammarus* abundance spillover, although, like the decreases in size high high-intensity fishing-the-line could be an explanation behind the lack of evidence for spillover.

Size and Abundance of M. Brachydactyla

The observed patterns in carapace length suggest that *M. brachydactlya* populations are not responding uniformly across sites or years. Whilst PLTNTZ showed a clear trend of increasing mean size over time, both OPLTNTZ and Ouaisné exhibited comparatively stable or even slightly declining mean carapace length.

The increase in *M. brachydactyla* size within PTLNTZ contrasts with findings from both Flamville (Amelot et al., 2024) and Lundy NTZs (Hoskin et al., 2011), where mean carapace size remained stable or declined slightly. In 2022, *H. gammarus* within PTLNTZ were significantly smaller than those outside the reserve. One explanation could be that PTLNTZ's shallow, sheltered environment plays a role, as juveniles preferentially settle on sandy substrates in low-exposure habitats (Corgos et al., 2011). Following protection, a higher proportion of adults likely remained within PTLNTZ and boosted average size. The 2022-2023 difference was likely furthered due to the timing of the 2022 survey. *M. brachydactlya* undertake seasonal migrations into shallow waters each spring (Corgos et al., 2006; Bodin et al., 2007). In 2022, the PTLNTZ closure only began on May 2nd, by which point a substantial portion of the migratory *M. brachdactlya* population had likely already been harvested. Furthermore, only 3 weeks later, in June, potting trials commenced, unlikely to show any benefits of NTZ protection. By contrast, in 2023, the NTZ was fully enforced throughout the migration and breeding season, preventing further harvesting. As a result, larger individuals will have persisted within the population, explaining the marked increase in recorded mean size.

In 2022, mean carapace length at Ouaisné and OPTLNTZ did not differ; however, by 2025, these sites had diverged significantly, with both OPTLNTZ and PTLNTZ exhibiting greater mean carapace lengths than Ouaisné. This may reflect adult spillover into adjacent areas, therefore indicating that the absence of fishing pressure within the NTZ has allowed OPTLNTZ to retain larger individuals, a trend not observed at the fully fished control site, Ouaisné.

The creation of the PTLNTZ has coincided with the steady increase in *M. brachydactyla* CPUE inside the NTZ, a trend that was not matched by OPTLNTZ, where CPUE remained stable, with a slight decreasing trend. This, therefore, implies a lack of population benefits extending beyond the NTZ, a trend expected, as despite seasonal migrations to deeper waters, during their residence in shallow water systems, they, like *H. gammarus*, also display high residency (Bernandez et al., 2003). Increases in CPUE were also recorded at the control site, Ouaisné, reflecting broader increases in *M. brachydactyla* populations in Jersey's waters (Marine Resources Annual Report, 2024). This reduces the apparent signal of NTZ benefits and better aligns with observations from other NTZs, where no clear changes in abundance were detected. In contrast, OPTLNTZ did not follow this upwards trend, maybe attributed to intense fishing pressure at this site (Pers obs., 2025), with high removal rates of *M. brachydactlya* potentially masking the regional trend.

4.2 Diurnal BRUVs

Overall Abundance

Baseline abundance was higher at Beauport compared with PTLNTZ, yet the findings suggest that this initial difference between sites has diminished. Overall abundance at Portelet showed significant departures from its baseline in 2022, 2024, and 2025, whereas Beauport abundances remained consistent across the study period. This could indicate that PTLNTZ species populations are gradually catching up to the higher levels found within Beauport following protection, a trend expected due to community compensatory dynamics (Gonzalez and Loreau, 2009). Such increasing trends have also been observed in NTZs (Williamson et al., 2004; Sköld et al., 2022). The increase in abundance at Portelet also extended to the commercial species filter, with 2024 and 2025 showing a departure from the baseline (2021). This similarity is primarily due to the more abundant species recorded, which are also present within the commercially filtered dataset (e.g., *S. cantharus and Chelon spp.).*

S. cantharus was largely responsible for the increased abundance within PTLNTZ by 2025, with its mean abundance more than quadrupling from its 2021 levels. *S. cantharus* also increased markedly in Beauport, inferring that the NTZ had little impact on its populations. This increase was therefore likely recording the projected rises expected in the stock size of *S.*

cantharus in English waters (Pinnegar et al., 2023). This increase did not contribute to a significant rise in Beauport overall abundance over time due to the simultaneous decreases in Malacostraca spp. abundance. Species exhibiting greater mean abundance gains at PTLNTZ relative to Beauport included *Chelon spp., D. labrax, M. surmulletus and B. capriscus.* Interestingly, all apart from *B. capriscus* are primary targets of inshore gillnetting activity (Plaster et al., 2023), so greater increases in abundance within PTLNTZ could be a direct result of inshore netting cessation. However, this is difficult to confirm, as data on inshore netting effort is unavailable; such activity occurs on small vessels under 12 m, which, prior to NTZ implementation, were not required to carry Vessel Monitoring Systems (Marine Resources Annual Report, 2024).

Some species saw declines at both Beauport and Portelet. *Pagarus spp.* was expected due to earlier data collectors citing most *Malacostraca spp* as *Pagarus*. Only *P. pollachius* showed a larger decline at Portelet compared to Beauport, which, rather than being negatively impacted by the NTZ, is more likely reflecting the broader, ongoing declines in pollack populations (ICES, 2025). Both sites exhibited similar declines in *S. canicula*, suggesting a general decline in this species that the NTZ does not appear to mitigate. Interestingly, *M. brachydactyla* declined more at Beauport than at Portelet, despite potting trials at the control site showing increases in CPUE in a similar area. This discrepancy could be due to differences in the timing of BRUV deployments, which were conducted in late May in 2021 but only in mid-July in 2025, potentially capturing different stages of *M. brachdactyla* migration. Many of the non-commercial species, such as *Labridae spp*. saw no change from 2021 to 2025, but these are not commercially targeted species, so change should not be expected (Claudet et al., 2010).

Species Richness

Although overall abundance increased relative to Beauport, species richness showed only a minor increase and remained comparable to Beauport throughout. Only in 2024 did Portelet show substantially higher species richness; however, this year had fewer replicates, which may have contributed to Portelet's significant difference. This trend was replicated in the commercial species filtered dataset. These trends suggest that increases in overall abundance did not translate into an increase in species, but instead a rise in the numbers of already present species, leading to the higher abundance levels without an increase in species richness (Roswell et al., 2021).

Global analyses of NTZs have shown overall biomass as a strong responder to protection, whereas species richness has shown smaller and less distinct increases (Lester et al., 2009). The absence of significant changes in species diversity between PTLNTZ and Beauport may be explained by the intermediate disturbance hypothesis (Connell, 1978). Increased fishing effort increases disturbance and creates more opportunity for settlement of novel species; however, before designation, PTLNTZ experienced relatively low fishing pressure, limited mainly to lobster potting (Pers obs., 2025), which targets only *Malacostraca* spp. Consequently, the lack of fishing pressure on *Actinopterygii* spp. may have contributed to the observed similarities following NTZ implementation.

This interpretation is supported by studies of fish communities along gradients of human disturbance, confirming that as disturbance decreases, biomass rises steadily. Species richness, on the other hand, increases only until top predators reach a threshold, after which prey populations may decline, causing a plateau (Sandin et al., 2008). Within PTLNTZ, predatory species may already be at this threshold before NTZ designation, explaining why species richness has remained stable and comparable to Beauport despite relative increases in abundance. This suggests that overall abundance would serve as a more reliable indicator of recovery within PTLNTZ.

Species Assemblage

Species assemblage changed over time at both PTLNTZ and Beauport, but differences in species assemblage between Portelet and Beauport were not observed in the baseline year (2021) and remained similar by 2025. This suggests that overall assemblage shifts within PTLNTZ were driven by regional population changes as opposed to site-specific.

This trend matches some NTZs (Varnes and Olsen, 2023) but also contrasts with other findings observing significant shifts in species assemblage from control sites (Barrett et al., 2007; Guidetti et al., 2014). A key distinction is that sites exhibiting change were historically subjected to intense destructive fishing, which will have substantially altered community structure prior to protection. In contrast, Beauport and PTLNTZ experience relatively low levels of such activity, as their complex seabed limits the use of destructive mobile fishing gear. This, as with trends observed in species richness, may explain why assemblage composition at PTLNTZ has remained comparable to Beauport, given that the site was already in a somewhat

healthy state before NTZ implementation. Additionally, the small size of PTLNTZ means that species with a greater home range venture beyond the protected area and are therefore exposed to similar conditions within the control site (Sköld et al., 2022), reducing the effect the NTZ implementation has on these species.

In 2021, there was a marginal difference between sites, whereas in 2025, there was no difference. This suggests there is some evidence that the PTLNTZ species assemblage indeed changed, with mean abundances of species noticeably closer to Beauport in 2025 (Figure 17a and b). Potentially, the change in species assemblage is occurring, but the amount of time required is currently beyond the scope of this study (5 years), with community-level shifts requiring more time before becoming evident (Babcock et al., 2010). Consequently, these results highlight the importance of continued monitoring beyond the initial five-year period to capture longer-term ecological responses to NTZ protection.

4.3 Nocturnal BRUVs

Nocturnal vs Diurnal

Nocturnal abundance levels at both Portelet and Beauport were lower than diurnal data, despite the increased recording period for nocturnal data, this follows a similar trend as nocturnal and diurnal BRUVs deployed by Harvey et al., (2012). This pattern may be driven by heightened predation risk at night (Rickel and Genin, 2005) and a behavioural shift from foraging to sheltering as nocturnal conditions begin (Hobson, 1972). Nocturnal species richness on the other hand saw higher levels in comparison to diurnal surveys, likely driven by the increased time of nocturnal recording.

Consistent with prior research, our nocturnal BRUVs recorded species assemblages that differed from those observed in diurnal BRUVs (Harvey et al., 2012). Differences may reflect ecological patterns such as diel shifts in foraging activity (Helfman, 1986; Dos Santos Silva Amaral et al., 2022) and habitat use (Fergueson et al., 2013). This pattern was most evident in the increased occurrence of *T. luscus*, a species known to forage primarily at night (Fowler et al., 1999), as well as the presence of several predominantly nocturnal feeders; *S. solea (*Lagardere et al., 1987*)*, *L. vulgaris* (Cabanellas-Reboredo et al., 2012), *H. Gammarus (*Smith

et al., 1998), and *S. atlantica* (Jardas et al., 2004), recorded only during nocturnal BRUV deployments.

Despite this, some species known to increase foraging activity during darkness did not show the expected rise in presence. For example, *C. conger* and *D. labrax*, which predominantly forage at night (Xavier et al., 2010; Rodriguez-Garcia et al., 2024), exhibited lower relative abundance during nocturnal recordings. One possible explanation is that the artificial lighting used on the rigs may affect certain nocturnal species (Gordon et al., 2002; Marchesan et al., 2005), discounting more light-averse species and ultimately masking true nocturnal assemblages.

Nocturnal Overall Abundance

In 2025, no significant difference in overall abundance between PTLNTZ and Beauport was reported. Based on this information alone, it could be concluded that the nocturnal abundance has not changed in PTLNTZ relative to Beauport 5 years on, yet the difference between the two sites is comparable to the diurnal 2025 data. This could indicate that nocturnal assemblages are following a similar recovery trajectory to the diurnal community at PTLNTTZ, although due to a lack of comparable nocturnal data, this interpretation remains speculative.

Nocturnal Species Richness

2025 saw no significant difference in species richness between sites. In the absence of prior nocturnal recordings, this data can only serve as a baseline and does not allow us to determine whether protection at PTLNTZ has had a measurable impact on nocturnal species richness. Nevertheless, like the patterns observed in overall abundance, the 2025 diurnal and nocturnal surveys produced a comparable difference between sites. This suggests that, despite capturing different species assemblages, diurnal surveys, despite only capturing a subset of the assemblage within PTLNTZ, may provide a reasonably reliable indication of overall community response to protection.

Nocturnal Species Assemblage

Given that protection at Portelet was only established in 2022 and 2025 represents the first year of nocturnal surveys, it is therefore not possible to conclude the effects PTLNTZ designation has had on fish community composition. What can be stated is that, in 2025, the two bays exhibited similar assemblages, but without pre-protection data, it is unknown whether this similarity reflects a natural baseline or an early response to management.

4.4 Future Methodological Considerations

Across the deployment of 48 newly designed BRUVs in 2025, only one bait arm was lost, and one was damaged, with all torches, cameras and rig frames remaining in good condition. Both cases of damage involved metal bait cages becoming caught in rocks. In contrast, the plastic cages never became stuck, although they were more prone to damage from C. conger feeding. Increasing the distance from the seabed would reduce the potential of the bait cage getting stuck, but it would also decrease the visibility of demersal species. Some bait cages lost all bait before the 40 minutes; this only occurred in the metal bait cages with larger holes, therefore, future use of a mesh bag inside would lengthen the period individuals remain attracted to the BRUV (Whitmarsh et al., 2017). Variation in the flexibility of the carbon fibre bait arms made it hard at first to correctly angle the camera pre-deployment, with some failed BRUV deployments linked to camera positioning. Future modifications to the BRUVs should include a standardised bait arm diameter and camera positioning to reduce this error.

It was suspected that the bright white lights on the nocturnal BRUVs contributed to decreased abundance and absence of species of timid nature; furthermore, in some nocturnal deployments, white light attracted high levels of zooplankton, obscuring the view. Red light is therefore a viable option, particularly as it attracts fewer phototactic species (Harvey et al, 2012) and has reduced detectability by coastal fish (Von der Emde et al., 2004). The torches used have a strong red-light setting, making this methodological shift straightforward, although it is important to note that visibility would be reduced due to attenuation of red light underwater (Harvey et al., 2012).

5. Conclusion

Monitoring of PTLNTZ and adjacent unprotected areas has revealed clear ecological responses to protection from fishing pressure. The commercially important crustaceans *H. gammarus* and *M. brachydactyla* both showed increases in body size, consistent with reduced harvest pressure, leading to greater reproductive potential. For *M. brachydactyla*, CPUE rose steadily within PTLNTZ, whilst for *H. gammarus* CPUE initially increased before declining, suggesting PTLNTZ may have reached carrying capacity, with benefits expressed more strongly in size structure than in sustained abundance. However, at the boundary site (OPTLNTZ), *H. gammarus* CPUE decreased, suggesting intensified fishing pressure and potential displacement effects.

Diurnal fish communities recorded revealed increases in overall abundance at PTLNTZ towards the control sites' levels, suggesting a gradual recovering trajectory. Species richness remained stable and species assemblage saw subtle shifts. This indicates NTZ implementation is benefiting a limited number of key species within PTLNTZ and is not currently driving broad-scale shifts in community composition. The inclusion of Nocturnal BRUVs saw a selection of species recorded for the first time in this study, along with markedly different species assemblage recorded, yet despite this difference, the outcomes mirrored 2025 diurnal BRUV results.

Overall, five years of monitoring at PTLNTZ demonstrate that NTZ protection in Jersey can deliver measurable benefits, especially in increased body size and CPUE of crustaceans and gradual recovery of overall abundance. At the same time, community-wide changes remain limited as not enough time has passed within PTLNTZ. The findings of this report underscore the great potential and challenges of NTZs in Jersey’s waters, providing a valuable evidence base for conservation and fisheries management strategy.

6. Recommendations

Given the findings of 5 years of monitoring PTLNTZ, several recommendations emerge for future study of NTZs in Jersey. Ecological change within NTZs often occurs over periods longer than 5 years (Babcock et al., 2010), therefore, non-significant findings within this report should not imply total absence of ecological response. Continued monitoring beyond the 5-year compulsory period is therefore strongly advised.

Future policy should also address the intense potting observed in OPTLNTZ (fishing-the-line), which may have rendered this area more exploited than pre-NTZ designation. This is reflected in the decline of *H. gammarus* CPUE at OPTLNTZ, in contrast to the relatively stable levels recorded at Ouaisné. The suggestion of a buffer zone for potting extending beyond the NTZ to the reefs immediately outside could help mitigate this pressure (Di Lorenzo et al., 2020; Ohayon et al., 2021).

Further research is recommended to understand the implications of protection, particularly the increase in reproductive potential of *H. gammarus*. Quantifying increased larval export and modelling of larval transport will provide evidence to commercial fishers about the hidden benefits of NTZ implementation. Secondly, if Sauvage reef is to become an NTZ, comparative studies between two unique sites would be beneficial in understanding the impact of NTZ designation in two ecologically different systems.

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8. Appendix

This appendix presents supplementary information supporting the main analyses in this report, including raw statistical outputs from GLM, GLMM, PERMANOVA, SIMPER, and Pairwise Comparisons.

Appendix Contents

Lobster Pot Trials

A.)

Coefficients:

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	83.6552	1.7631	47.447	<2e-16	***
String_PositionOPLTNTZ	-3.7266	2.5156	-1.481	0.139	
String_PositionOuaisne	-3.0302	2.9569	-1.025	0.306	
Year2023	0.9213	2.0419	0.451	0.652	
Year2024	-2.4766	2.1722	-1.140	0.255	
Year2025	1.6782	2.3281	0.721	0.471	
String_PositionOPLTNTZ:Year2023	-0.2499	3.1567	-0.079	0.937	
String_PositionOuaisne:Year2023	-1.5463	3.6619	-0.422	0.673	
String_PositionOPLTNTZ:Year2024	-0.1827	3.3772	-0.054	0.957	
String_PositionOuaisne:Year2024	1.8516	4.2267	0.438	0.662	
String_PositionOPLTNTZ:Year2025	-4.7317	3.5208	-1.344	0.180	
String_PositionOuaisne:Year2025	-6.7432	3.8289	-1.761	0.079	.

B.)

Coefficients:

	Estimate	Std. Error	t value	Pr(>|t|)	
(Intercept)	110.059	4.288	25.664	< 2e-16	***
PositionOPLTNTZ	17.078	4.684	3.646	0.000284	***
PositionOuaisne	16.922	4.940	3.426	0.000645	***
YEAR2023	11.389	5.401	2.109	0.035279	*
YEAR2024	16.980	5.515	3.079	0.002150	**
YEAR2025	18.459	4.896	3.770	0.000175	***
PositionOPLTNTZ:YEAR2023	-10.932	5.998	-1.823	0.068755	.
PositionOuaisne:YEAR2023	-11.544	6.211	-1.859	0.063464	.
PositionOPLTNTZ:YEAR2024	-21.149	6.252	-3.383	0.000754	***
PositionOuaisne:YEAR2024	-19.785	6.351	-3.115	0.001904	**
PositionOPLTNTZ:YEAR2025	-15.323	5.620	-2.726	0.006545	**
PositionOuaisne:YEAR2025	-23.755	5.696	-4.170	3.38e-05	***

Table S1, Results of Interactive GLM's examining the effects of sampling position (String_Position), year (Year), and their interaction on carapace size for A.) H. Gammarus and B.) M. brachydactlya. PTLNTZ set as the reference site and 2022 as reference year

A.)

```
Year = 2022:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ     3.727 2.52 388   1.481  0.3010
 PLTNTZ - Ouaisne     3.030 2.96 388   1.025  0.5616
 OPLTNTZ - Ouaisne   -0.696 2.98 388  -0.234  0.9703

Year = 2023:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ     3.976 1.91 388   2.085  0.0942
 PLTNTZ - Ouaisne     4.576 2.16 388   2.119  0.0874
 OPLTNTZ - Ouaisne    0.600 2.49 388   0.241  0.9684

Year = 2024:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ     3.909 2.25 388   1.735  0.1935
 PLTNTZ - Ouaisne     1.179 3.02 388   0.390  0.9195
 OPLTNTZ - Ouaisne   -2.731 3.31 388  -0.824  0.6883

Year = 2025:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ     8.458 2.46 388   3.434  0.0019
 PLTNTZ - Ouaisne     9.773 2.43 388   4.018  0.0002
 OPLTNTZ - Ouaisne    1.315 2.71 388   0.485  0.8786
```

B.)

```
YEAR = 2022:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ   -17.078 4.68 789  -3.646  0.0008
 PLTNTZ - Ouaisne   -16.922 4.94 789  -3.426  0.0019
 OPLTNTZ - Ouaisne    0.156 3.09 789   0.050  0.9986

YEAR = 2023:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ    -6.145 3.75 789  -1.640  0.2294
 PLTNTZ - Ouaisne    -5.378 3.77 789  -1.428  0.3268
 OPLTNTZ - Ouaisne    0.768 2.58 789   0.298  0.9524

YEAR = 2024:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ     4.071 4.14 789   0.983  0.5878
 PLTNTZ - Ouaisne     2.863 3.99 789   0.717  0.7533
 OPLTNTZ - Ouaisne   -1.208 3.01 789  -0.402  0.9149

YEAR = 2025:
 contrast          estimate   SE  df t.ratio p.value
 PLTNTZ - OPLTNTZ    -1.755 3.11 789  -0.565  0.8387
 PLTNTZ - Ouaisne     6.833 2.84 789   2.409  0.0428
 OPLTNTZ - Ouaisne    8.588 2.55 789   3.363  0.0023
```

Table S2, Results of emmeans Pairwise Comparisons of carapace length between years for A.) H. Gammarus and B.) M. brachydactlya

A.)

```
Coefficients:
                                Estimate Std. Error z value Pr(>|z|)
(Intercept)                      2.67415    0.18570  14.401  < 2e-16 ***
String_PositionOPLTNTZ          -0.03509    0.26495  -0.132  0.89463
String_PositionOuaisne          -0.59471    0.31142  -1.910  0.05618 .
Year2023                         0.66989    0.21505   3.115  0.00184 **
Year2024                         0.25259    0.22878   1.104  0.26956
Year2025                        -0.10920    0.24520  -0.445  0.65607
String_PositionOPLTNTZ:Year2023 -0.85221    0.33247  -2.563  0.01037 *
String_PositionOuaisne:Year2023 -0.62907    0.38568  -1.631  0.10287
String_PositionOPLTNTZ:Year2024 -0.73216    0.35569  -2.058  0.03955 *
String_PositionOuaisne:Year2024 -0.94574    0.44517  -2.124  0.03363 *
String_PositionOPLTNTZ:Year2025 -0.45042    0.37082  -1.215  0.22449
String_PositionOuaisne:Year2025  0.15002    0.40327   0.372  0.70988
```

B.)

```
Coefficients:
                           Estimate Std. Error z value Pr(>|z|)
(Intercept)                 2.14007    0.24254   8.824  < 2e-16 ***
PositionOPLTNTZ             1.23866    0.26493   4.675 2.93e-06 ***
PositionOuaisne             1.11803    0.27938   4.002 6.29e-05 ***
YEAR2023                    0.12862    0.30546   0.421  0.67371
YEAR2024                    0.42488    0.31191   1.362  0.17313
YEAR2025                    0.78667    0.27691   2.841  0.00450 **
PositionOPLTNTZ:YEAR2023 -0.04161    0.33924  -0.123  0.90239
PositionOuaisne:YEAR2023  0.03646    0.35129   0.104  0.91733
PositionOPLTNTZ:YEAR2024 -0.79135    0.35361  -2.238  0.02523 *
PositionOuaisne:YEAR2024 -0.39957    0.35919  -1.112  0.26596
PositionOPLTNTZ:YEAR2025 -0.92020    0.31785  -2.895  0.00379 **
PositionOuaisne:YEAR2025 -0.29919    0.32216  -0.929  0.35303
```

Table S3, Results of Interactive GLM's examining the effects of sampling position (String_Position), year (Year), and their interaction on CPUE for A.) H. Gammarus and B.) M. brachydactlya. PTLNTZ set as the reference site and 2022 as reference year.

A.)

```
Year = 2022:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ    0.0351 0.265 Inf   0.132  0.9904
 PLTNTZ - Ouaisne    0.5947 0.311 Inf   1.910  0.1359
 OPLTNTZ - Ouaisne   0.5596 0.313 Inf   1.786  0.1744

Year = 2023:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ    0.8873 0.201 Inf   4.418  <.0001
 PLTNTZ - Ouaisne    1.2238 0.228 Inf   5.379  <.0001
 OPLTNTZ - Ouaisne   0.3365 0.262 Inf   1.285  0.4037

Year = 2024:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ    0.7673 0.237 Inf   3.233  0.0035
 PLTNTZ - Ouaisne    1.5404 0.318 Inf   4.843  <.0001
 OPLTNTZ - Ouaisne   0.7732 0.349 Inf   2.216  0.0686

Year = 2025:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ    0.4855 0.259 Inf   1.871  0.1470
 PLTNTZ - Ouaisne    0.4447 0.256 Inf   1.736  0.1919
 OPLTNTZ - Ouaisne  -0.0408 0.286 Inf  -0.143  0.9888
```

B.)

```
YEAR = 2022:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ   -1.2387 0.265 Inf  -4.675  <.0001
 PLTNTZ - Ouaisne   -1.1180 0.279 Inf  -4.002  0.0002
 OPLTNTZ - Ouaisne   0.1206 0.175 Inf   0.690  0.7695

YEAR = 2023:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ   -1.1971 0.212 Inf  -5.649  <.0001
 PLTNTZ - Ouaisne   -1.1545 0.213 Inf  -5.421  <.0001
 OPLTNTZ - Ouaisne   0.0426 0.146 Inf   0.292  0.9542

YEAR = 2024:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ   -0.4473 0.234 Inf  -1.910  0.1358
 PLTNTZ - Ouaisne   -0.7185 0.226 Inf  -3.183  0.0042
 OPLTNTZ - Ouaisne  -0.2712 0.170 Inf  -1.595  0.2477

YEAR = 2025:
 contrast          estimate    SE  df z.ratio p.value
 PLTNTZ - OPLTNTZ   -0.3185 0.176 Inf  -1.813  0.1652
 PLTNTZ - Ouaisne   -0.8188 0.160 Inf  -5.105  <.0001
 OPLTNTZ - Ouaisne  -0.5004 0.144 Inf  -3.464  0.0015
```

Table S4, Results of emmeans Pairwise Comparisons of CPUE between years for A.) H. Gammarus and B.) M. brachydactlya

Diurnal BRUVs

A.)

Fixed effects:

	Estimate	Std. Error	z value	Pr(>|z|)	
(Intercept)	2.53011	0.09991	25.324	< 2e-16	***
BayPortelet	-0.56201	0.12570	-4.471	7.79e-06	***
Year2022	-0.04936	0.12521	-0.394	0.693408	
Year2023	0.03173	0.12147	0.261	0.793918	
Year2024	0.08090	0.12564	0.644	0.519677	
Year2025	0.05279	0.10809	0.488	0.625298	
BayPortelet:Year2022	0.56517	0.19007	2.974	0.002944	**
BayPortelet:Year2023	0.17512	0.19961	0.877	0.380309	
BayPortelet:Year2024	0.90328	0.18982	4.759	1.95e-06	***
BayPortelet:Year2025	0.60223	0.16733	3.599	0.000319	***

B.)

Fixed effects:

	Estimate	Std. Error	z value	Pr(>|z|)	
(Intercept)	2.07944	0.09806	21.206	< 2e-16	***
Year2022	-0.03175	0.16045	-0.198	0.843145	
Year2023	-0.72490	0.20463	-3.542	0.000396	***
Year2024	0.25131	0.15331	1.639	0.101162	
Year2025	0.31144	0.12906	2.413	0.015821	*
BayPortelet	-0.45474	0.15737	-2.890	0.003858	**
Year2022:BayPortelet	0.19880	0.24845	0.800	0.423618	
Year2023:BayPortelet	0.42195	0.30060	1.404	0.160409	
Year2024:BayPortelet	0.55759	0.23706	2.352	0.018669	*
Year2025:BayPortelet	0.50225	0.19936	2.519	0.011761	*

Table S5, Results of Interactive GLMM's examining the effects of sampling position (Bay), year (Year), and their interaction on Overall Abundance for A.) All individuals and B.) Commercial species filtered individuals. Beauport set as the reference site and 2021 as reference year

A.)

```
Year = 2021:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  0.56201 0.126 Inf   4.471  <.0001

Year = 2022:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.00316 0.142 Inf  -0.022  0.9822

Year = 2023:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  0.38689 0.154 Inf   2.511  0.0121

Year = 2024:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.34128 0.142 Inf  -2.400  0.0164

Year = 2025:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.04023 0.110 Inf  -0.364  0.7155
```

B.)

```
Year = 2021:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.4547 0.157 Inf   2.890  0.0039

Year = 2022:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.2559 0.192 Inf   1.331  0.1831

Year = 2023:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.0328 0.256 Inf   0.128  0.8981

Year = 2024:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  -0.1029 0.177 Inf  -0.580  0.5618

Year = 2025:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  -0.0475 0.122 Inf  -0.388  0.6979
```

Table S6, Results of emmeans Pairwise Comparisons of Overall abundance between years for A.) All individulas and B.) Commercial species filtered individuals

A.)

Fixed effects:

	Estimate	Std. Error	z value	Pr(>|z|)	
(Intercept)	1.460e+00	1.336e-01	10.929	<2e-16	***
Year2022	1.737e-01	2.055e-01	0.845	0.3980	
Year2023	2.668e-01	2.002e-01	1.333	0.1826	
Year2024	1.772e-01	2.136e-01	0.830	0.4068	
Year2025	3.054e-01	1.761e-01	1.734	0.0829	.
BayPortelet	-2.366e-14	1.890e-01	0.000	1.0000	
Year2022:BayPortelet	1.151e-01	2.861e-01	0.402	0.6875	
Year2023:BayPortelet	-2.247e-02	2.840e-01	-0.079	0.9369	
Year2024:BayPortelet	4.418e-01	2.975e-01	1.485	0.1375	
Year2025:BayPortelet	-2.054e-02	2.545e-01	-0.081	0.9356	

B.)

Fixed effects:

	Estimate	Std. Error	z value	Pr(>|z|)	
(Intercept)	1.03945	0.19482	5.335	9.53e-08	***
Year2022	0.25767	0.25822	0.998	0.31834	
Year2023	0.28213	0.25807	1.093	0.27430	
Year2024	0.41104	0.25512	1.611	0.10715	
Year2025	0.61532	0.21335	2.884	0.00393	**
BayPortelet	0.01966	0.24262	0.081	0.93541	
Year2022:BayPortelet	0.14242	0.35716	0.399	0.69007	
Year2023:BayPortelet	0.14672	0.35830	0.410	0.68217	
Year2024:BayPortelet	0.43388	0.35588	1.219	0.22278	
Year2025:BayPortelet	-0.01254	0.30315	-0.041	0.96700	

Table S7, Results of Interactive GLMM's examining the effects of sampling position (Bay), year (Year), and their interaction on Species Richness for A.) All individuals and B.) Commercial species filtered individuals. Beauport set as the reference site and 2021 as reference year

A.)

```
Year = 2021:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.0000 0.189 Inf   0.000  1.0000

Year = 2022:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  -0.1151 0.215 Inf  -0.536  0.5921

Year = 2023:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.0225 0.212 Inf   0.106  0.9156

Year = 2024:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet  -0.4418 0.230 Inf  -1.923  0.0545

Year = 2025:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet   0.0205 0.170 Inf   0.121  0.9040
```

B.)

```
Year = 2021:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.01966 0.243 Inf  -0.081  0.9354

Year = 2022:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.16209 0.261 Inf  -0.621  0.5346

Year = 2023:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.16639 0.261 Inf  -0.637  0.5243

Year = 2024:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.45354 0.260 Inf  -1.744  0.0812

Year = 2025:
 contrast            estimate    SE  df z.ratio p.value
 Beauport - Portelet -0.00712 0.181 Inf  -0.039  0.9687
```

Table S8, Results of emmeans Pairwise Comparisons of Overall species richness between years for A.) All individulas and B.) Commercial species filtered individuals

`

A.)

```
adonis2(formula = species_sub ~ Year, data = df_sub, permutations = 999, method = "bray'
         Df SumOfSqs      R2     F Pr(>F)
Model     1   1.4448 0.19396 5.294  0.001 ***
Residual 22   6.0041 0.80604
Total    23   7.4489 1.00000
```

B.)

Contrast: 2021_2025

	average	sd	ratio	ava	avb	cumsum	p	
S.cantharus	0.24730	0.14742	1.67750	1.15380	6.36400	0.294	0.003	**
P.pollachius	0.08009	0.11569	0.69230	1.46150	0.72700	0.389	0.993	
Pagarus	0.08001	0.08988	0.89020	1.46150	0.00000	0.484	0.200	
Chelon	0.06589	0.12969	0.50800	0.00000	2.09100	0.562	0.003	**
S.canicula	0.06084	0.06857	0.88720	1.30770	0.36400	0.634	0.998	
T.luscus	0.05236	0.09941	0.52670	0.53850	0.63600	0.696	0.809	
L.bergylta	0.04503	0.04476	1.00620	0.38460	0.81800	0.750	0.099	.
M.brachydactyla	0.03864	0.04811	0.80320	0.69230	0.18200	0.796	0.975	
D.labrax	0.03441	0.05279	0.65170	0.00000	0.72700	0.837	0.001	***
M.surmulletus	0.02294	0.03601	0.63710	0.00000	0.54500	0.864	0.005	**
C.conger	0.01312	0.02319	0.56580	0.00000	0.27300	0.879	0.003	**
A.tobianus	0.01243	0.02078	0.59820	0.00000	0.27300	0.894	0.005	**
C.lyra	0.01222	0.02517	0.48530	0.15380	0.09100	0.909	0.983	
B.capriscus	0.01077	0.02535	0.42470	0.00000	0.27300	0.921	0.005	**
Liocarcinus	0.00906	0.02294	0.39490	0.15380	0.00000	0.932	0.997	
L.mixtus	0.00861	0.02158	0.39900	0.07690	0.09100	0.942	0.267	
S.melops	0.00856	0.02828	0.30280	0.00000	0.09100	0.953	0.002	**
S.scombrus	0.00632	0.01436	0.44020	0.00000	0.18200	0.960	0.008	**
G.flavescens	0.00537	0.01737	0.30920	0.00000	0.09100	0.966	0.007	**
R.undulata	0.00503	0.01895	0.26550	0.07690	0.00000	0.972	0.993	
Raja	0.00468	0.01746	0.26810	0.07690	0.00000	0.978	0.995	
S.stellaris	0.00412	0.01514	0.27190	0.07690	0.00000	0.983	0.995	
R.brachyura	0.00377	0.01210	0.31180	0.00000	0.09100	0.987	0.008	**
R.microocellata	0.00377	0.01210	0.31180	0.00000	0.09100	0.992	0.008	**
Asterioda	0.00368	0.01341	0.27460	0.07690	0.00000	0.996	0.996	
T.trachurus	0.00323	0.01032	0.31250	0.00000	0.09100	1.000	0.006	**

Table S9, Diurnal Species assemblage analysis of Portelet 2021 vs 2025 with A.) PERMANOVA result and B.) Subsequent SIMPER table formed.

A.)

```
adonis2(formula = species_sub_beau ~ Year, data = df_sub_beau, permutations = 999, method = "bray"
         Df SumOfSqs     R2      F Pr(>F)
Model     1   2.0065 0.2577 8.3321  0.001 ***
Residual 24   5.7795 0.7423
Total    25   7.7860 1.0000
```

B.)

Contrast: 2021_2025

	average	sd	ratio	ava	avb	cumsum	p	
S.cantharus	0.23037	0.13435	1.71460	1.23100	6.76900	0.270	0.003	**
M.brachydactyla	0.14683	0.12226	1.20100	4.53800	1.15400	0.443	0.012	*
Pagarus	0.10155	0.09769	1.03960	2.84600	0.00000	0.562	0.002	**
Liocarcinus	0.06231	0.11222	0.55520	1.92300	0.07700	0.635	0.157	
P.pollachius	0.05671	0.11276	0.50290	0.84600	0.69200	0.702	0.487	
S.canicula	0.04153	0.05349	0.77640	1.00000	0.00000	0.751	0.003	**
L.bergylta	0.03268	0.03184	1.02670	0.23100	0.76900	0.789	0.024	*
T.luscus	0.02893	0.05920	0.48860	0.00000	0.69200	0.823	0.026	*
Chelon	0.02665	0.06656	0.40050	0.00000	0.92300	0.854	0.322	
S.melops	0.01861	0.03770	0.49360	0.00000	0.38500	0.876	0.064	.
C.conger	0.01725	0.02433	0.70910	0.07700	0.38500	0.896	0.148	
G.flavescens	0.01247	0.02390	0.52170	0.15400	0.15400	0.911	0.541	
D.labrax	0.01116	0.02159	0.51700	0.00000	0.30800	0.924	0.116	
L.mixtus	0.01042	0.01803	0.57760	0.07700	0.23100	0.936	0.377	
A.tobianus	0.00853	0.01616	0.52790	0.00000	0.23100	0.946	0.139	
T.trachurus	0.00830	0.02961	0.28040	0.00000	0.23100	0.956	0.506	
Macropodia	0.00755	0.02818	0.26810	0.15400	0.00000	0.965	0.470	
S.scombrus	0.00669	0.01634	0.40970	0.00000	0.23100	0.973	0.286	
C.rupestris	0.00652	0.01799	0.36230	0.07700	0.07700	0.981	0.611	
S.stellaris	0.00549	0.01973	0.27820	0.15400	0.00000	0.987	0.598	
R.undulata	0.00299	0.01072	0.27930	0.00000	0.07700	0.990	0.523	
S.officianis	0.00299	0.01072	0.27930	0.00000	0.07700	0.994	0.523	
M.surmulletus	0.00277	0.00987	0.28040	0.00000	0.07700	0.997	0.504	
R.microocellata	0.00241	0.00853	0.28200	0.00000	0.07700	1.000	0.537	

Table S10, Diurnal Species assemblage analysis of Beauport 2021 vs 2025 with A.) PERMANOVA result and B.) Subsequent SIMPER table formed.

A.)

```
adonis2(formula = species_data ~ Bay, data = df_2025, permutations = 999, method = "bray")
         Df SumOfSqs      R2      F Pr(>F)
Model     1   0.5552 0.07275 1.8829   0.07 .
Residual 24   7.0773 0.92725
Total    25   7.6325 1.00000
```

B.)

Contrast: Beauport_Portelet

	average	sd	ratio	ava	avb	cumsum	p	
M.brachydactyla	0.18425	0.13288	1.38660	4.53800	0.69230	0.239	0.003	**
Pagarus	0.12395	0.09607	1.29030	2.84600	1.46150	0.400	0.656	
P.pollachius	0.09361	0.16657	0.56200	0.84600	1.46150	0.521	0.980	
S.cantharus	0.09080	0.11775	0.77110	1.23100	1.15380	0.639	0.934	
Liocarcinus	0.07749	0.12678	0.61120	1.92300	0.15380	0.740	0.034	*
S.canicula	0.07314	0.06929	1.05560	1.00000	1.30770	0.834	0.994	
L.bergylta	0.02852	0.04398	0.64860	0.23100	0.38460	0.871	0.917	
T.luscus	0.02798	0.10219	0.27380	0.00000	0.53850	0.908	0.989	
G.flavescens	0.01115	0.02697	0.41330	0.15400	0.00000	0.922	0.007	**
S.stellaris	0.01039	0.02645	0.39270	0.15400	0.07690	0.936	0.918	
Macropodia	0.00968	0.03435	0.28190	0.15400	0.00000	0.948	0.013	*
C.lyra	0.00810	0.02075	0.39040	0.00000	0.15380	0.959	0.990	
L.mixtus	0.00682	0.01738	0.39250	0.07700	0.07690	0.968	0.448	
C.conger	0.00557	0.01988	0.28040	0.07700	0.00000	0.975	0.010	**
C.rupestris	0.00557	0.01988	0.28040	0.07700	0.00000	0.982	0.010	**
R.undulata	0.00521	0.01960	0.26610	0.00000	0.07690	0.989	0.984	
Raja	0.00484	0.01801	0.26860	0.00000	0.07690	0.995	0.986	
Asterioda	0.00378	0.01375	0.27500	0.00000	0.07690	1.000	0.991	

Table S11, Diurnal Species assemblage analysis of Beauport 2021 vs Portelet 2021with A.) PERMANOVA result and B.) Subsequent SIMPER table formed.

A.)

```
adonis2(formula = species_data ~ Bay, data = df_2025, permutations = 999, method = "bray"
         Df SumOfSqs    R2      F. Pr(>F)
Model     1   0.1406 0.029 0.6571   0.71
Residual 22   4.7063 0.971
Total    23   4.8469 1.000
```

B.)

Contrast: Portelet_Beauport

	average	sd	ratio	ava	avb	cumsum	p	
S.cantharus	0.18809	0.14343	1.31140	6.36400	6.76900	0.306	0.837	
Chelon	0.07441	0.11757	0.63290	2.09100	0.92300	0.427	0.363	
T.luscus	0.04293	0.05769	0.74420	0.63600	0.69200	0.497	0.562	
M.brachydactyla	0.04254	0.04771	0.89170	0.18200	1.15400	0.567	0.221	
P.pollachius	0.04198	0.05435	0.77250	0.72700	0.69200	0.635	0.683	
L.bergylta	0.03200	0.03387	0.94500	0.81800	0.76900	0.687	0.325	
D.labrax	0.03064	0.04216	0.72680	0.72700	0.30800	0.737	0.208	
S.melops	0.02059	0.03709	0.55520	0.09100	0.38500	0.770	0.725	
M.surmulletus	0.01997	0.03004	0.66470	0.54500	0.07700	0.803	0.059	.
C.conger	0.01841	0.02376	0.77480	0.27300	0.38500	0.833	0.758	
A.tobianus	0.01435	0.01965	0.73030	0.27300	0.23100	0.856	0.599	
S.canicula	0.01336	0.02385	0.55990	0.36400	0.00000	0.878	0.035	*
L.mixtus	0.01124	0.02007	0.55990	0.09100	0.23100	0.896	0.723	
S.scombrus	0.01060	0.01798	0.58980	0.18200	0.23100	0.913	0.861	
T.trachurus	0.01049	0.02926	0.35840	0.09100	0.23100	0.931	0.774	
G.flavescens	0.00920	0.01887	0.48780	0.09100	0.15400	0.946	0.786	
B.capriscus	0.00898	0.02152	0.41710	0.27300	0.00000	0.960	0.127	
R.microocellata	0.00511	0.01241	0.41160	0.09100	0.07700	0.969	0.438	
C.lyra	0.00473	0.01631	0.29010	0.09100	0.00000	0.976	0.166	
R.brachyura	0.00312	0.01023	0.30530	0.09100	0.00000	0.981	0.229	
Liocarcinus	0.00293	0.01051	0.27880	0.00000	0.07700	0.986	0.769	
R.undulata	0.00293	0.01051	0.27880	0.00000	0.07700	0.991	0.769	
S.officianis	0.00293	0.01051	0.27880	0.00000	0.07700	0.996	0.769	
C.rupestris	0.00271	0.00969	0.27990	0.00000	0.07700	1.000	0.787	

Table S12, Diurnal Species assemblage analysis of Beauport 2025 vs Portelet 2025 with A.) PERMANOVA result and B.) Subsequent SIMPER table formed.

Nocturnal BRUVs

A.)

```
Coefficients:
            Estimate Std. Error z value Pr(>|z|)
(Intercept)  2.38086    0.12050  19.758  < 2e-16 ***
BayPortelet -0.03286    0.12351  -0.266     0.79
HabitatSand  0.55224    0.13884   3.978 6.96e-05 ***
```

B.)

```
Coefficients:
            Estimate Std. Error z value Pr(>|z|)
(Intercept)   2.1205     0.1406  15.082   <2e-16 ***
BayPortelet  -0.0211     0.1786  -0.118    0.906
HabitatSand  -0.1375     0.1790  -0.768    0.442
```

Table S13, Results of GLM examining the effects of sampling position (Bay), Habitat, and their effect on A.) Overall abundance and B.) Species richness. Beauport set as the reference site.

A.)

```
adonis2(formula = species_data2 ~ Bay, data = spec_2025, permutations = 999, method = "bray"
         Df SumOfSqs      R2     F Pr(>F)
Model     1   0.3281 0.07893 1.371  0.203
Residual 16   3.8284 0.92107
Total    17   4.1565 1.00000
```

B.)

Contrast: Portelet_Beauport

	average	sd	ratio	ava	avb	cumsum	p	
T.luscus	0.19476	0.17238	1.12990	8.37500	3.10000	0.286	0.178	
T.trachurus	0.07492	0.05985	1.25180	1.00000	2.40000	0.396	0.488	
A.presbyter	0.05621	0.07183	0.78260	1.37500	1.40000	0.478	0.841	
P.pollachius	0.04911	0.08920	0.55050	0.37500	1.20000	0.550	0.768	
L.vulgaris	0.03219	0.05058	0.63640	0.62500	1.30000	0.597	0.830	
S.cantharus	0.02680	0.03523	0.76050	0.87500	0.10000	0.637	0.061	.
C.conger	0.02596	0.02773	0.93590	0.12500	0.70000	0.675	0.055	.
M.brachydactyla	0.02453	0.02422	1.01270	0.50000	1.00000	0.711	0.198	
S.officianis	0.01909	0.02499	0.76380	0.12500	0.50000	0.739	0.652	
Liocarcinus	0.01682	0.02322	0.72440	0.37500	0.20000	0.763	0.410	
H.gammarus	0.01671	0.02161	0.77330	0.37500	0.20000	0.788	0.177	
A.tobianus	0.01579	0.02069	0.76330	0.00000	0.40000	0.811	0.069	.
S.atlantica	0.01562	0.02028	0.77030	0.25000	0.30000	0.834	0.787	
L.bergylta	0.01362	0.01972	0.69060	0.12500	0.30000	0.854	0.411	
Chelon	0.01344	0.01807	0.74330	0.25000	0.30000	0.874	0.730	
M.merlanguis	0.01313	0.02080	0.63120	0.25000	0.20000	0.893	0.427	
S.solea	0.01268	0.01915	0.66190	0.25000	0.30000	0.912	0.863	
Pagarus	0.01157	0.01645	0.70310	0.25000	0.20000	0.929	0.500	
C.pagarus	0.00750	0.01575	0.47610	0.00000	0.20000	0.940	0.320	
M.surmulletus	0.00687	0.01430	0.48060	0.12500	0.10000	0.950	0.426	
L.mixtus	0.00520	0.01405	0.37000	0.12500	0.00000	0.957	0.227	
S.melops	0.00520	0.01405	0.37000	0.12500	0.00000	0.965	0.227	
T.bubalis	0.00426	0.01343	0.31690	0.00000	0.10000	0.971	0.622	
Z.punctatus	0.00390	0.01224	0.31850	0.00000	0.10000	0.977	0.662	
D.labrax	0.00376	0.01010	0.37240	0.12500	0.00000	0.982	0.274	
E.vipera	0.00376	0.01010	0.37240	0.12500	0.00000	0.988	0.274	
G.flavescens	0.00302	0.00938	0.32240	0.00000	0.10000	0.992	0.752	
R.microocellata	0.00284	0.00880	0.32320	0.00000	0.10000	0.997	0.788	
S.canicula	0.00234	0.00625	0.37420	0.12500	0.00000	1.000	0.343	

Table S14, Nocturnal species assemblage analysis of Beauport 2025 vs Portelet 2025 with A.) PERMANOVA result and B.) Subsequent SIMPER table formed.