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Sonar, so good: Tree-reefs drive net gain in fish size, abundance, and diversity

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ABSTRACT: Global land use changes have greatly reduced the flow of driftwood to the sea. This driftwood is critical habitat for many sessile species that require hard substrates, and provides shelter, spawning, rearing, and foraging areas for mobile species. (Re)placement of large wood as a habitat restoration measure is established in rivers and freshwater lakes but has seldom been done in estuaries and marine environments where sunken driftwood occurred in the past. In the subtidal Dutch Wadden Sea, we placed 32 'tree-reefs' to mimic this historical sunken wood to assess the effects on mobile (i.e. fish and crustaceans) faunal biodiversity over a medium-term timescale of 2.5 yr. Because of the challenging environment, where traditional monitoring methods are limited due to low visibility, strong tidal currents, wave action, and unstable sediments, we modified a Garmin Panoptix Livescope (a fishfinder used by sport fishermen) to monitor pelagic fish populations around the reef sites. These efforts were supplemented by benthic fish traps. Within 2.5 yr of deployment, trap fishing showed that at reef sites, fish were 6 times more abundant, larger in size, and more diverse, while crab foraging activity was 77% lower compared to control sites. Sonar showed 215–359% greater pelagic fish abundance at reef sites compared to control sites across 3 size classes of fish monitored. Results indicate that (1) sport-fishing fishfinder technology is a valuable tool to monitor fish in shallow turbid waters at moderate costs, and (2) tree-reefs can be a cost-effective, scalable, natural, and biodegradable approach to restore marine biodiversity and trophic complexity.

KEY WORDS: Tree-reef · Sonar · Active acoustics · Artificial reef · Wood · Driftwood · Wadden Sea · Marine restoration

1. INTRODUCTION

Reefs are vital to marine ecosystems, providing habitat, food, and nursery grounds for countless species while supporting global fisheries and livelihoods. They form from both geogenic (bedrock, stone) and biogenic (corals, shellfish, worms, driftwood, shells) origins, (Hendy et al. 2014, Tan et al. 2020), offering hard surfaces for colonization and enhancing biodiversity (Nauta et al. 2023). Some, like bivalve reefs,

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also filter pollutants and improve water quality (Newell 2004). Regardless of form and origin, reefs worldwide provide essential ecosystem services whilst underpinning healthy marine systems (Commito et al. 2014, Perkol-Finkel & Sella 2015, Alam et al. 2020, Cobacho et al. 2020, Dickson et al. 2023, Nauta et al. 2023, Hickman et al. 2024).

Despite their importance, human activities such as bottom trawling, dredging (Beck et al. 2011), and global change impacts like heatwaves (Spalding & Brown 2015) and eutrophication (Zu Ermgassen et al. 2020) are causing massive reef declines. Trawling not only removes marine life but also disturbs or removes hard substrates (Buhl-Mortensen & Buhl-Mortensen 2018) needed for reef formation, with 75% of global shelf seas already trawled, often repeatedly, for decades (Amoroso et al. 2018). This benthic disruption hinders new reef establishment and accelerates the loss of existing ones (Watling & Norse 1998, Kaiser et al. 2002, Thrush & Dayton 2002).

Humans have also reduced natural substrate supply, especially large wood historically delivered by rivers to coastal seas (Wohl & Iskin 2021). Once a critical foundation for biogenic reefs, this input has nearly ceased due to deforestation, agriculture, dams, and river/coastal engineering (Wohl & Iskin 2021). Fossil records show wood has supported marine life for over 200 million years, offering hard, colonizable surfaces (Kaim 2011), nutrients, and shelter. While wood reintroduction is common in freshwater restoration, its ecological role and its possibility to contribute to restoration in marine systems remain largely overlooked.

The North Sea and Wadden Sea systems are prime examples of systems where reefs were once extensive but have dramatically declined or disappeared with increasing exploitation. Here, European flat oyster *Ostrea edulis* reefs covered vast areas: 1.75 million ha pre-exploitation (Thurstan et al. 2024). Now, the European flat oyster is functionally extinct due to centuries of exploitation (Thurstan et al. 2024). Within the Wadden Sea, historic biogenic structures such as these flat oyster beds, along with subtidal seagrass meadows and *Sabellaria* reefs are now locally extinct (Riesen & Reise 1982, Giesen et al. 1990, Thurstan et al. 2024), while subtidal shellfish (i.e. Pacific oyster *Magallana gigas* and blue mussel *Mytilus edulis*) beds have dramatically declined (Ricklefs et al. 2022). Indeed, 70.2% of extinctions are caused by habitat loss in the Wadden Sea (Lotze et al. 2005), including those of transitional estuarine habitats such as saltmarshes, coastal oak forests, and natural estuarine deltas. Now, the Wadden Sea has hardened, dyked coastlines, which have eliminated these soft transitional habitats as well as eliminated the possibility of rivers delivering wood and peat chunks to the sea (Franken et al. 2023, Dickson et al. 2025). This lack of large driftwood input likely hampers biogenic reef establishment.

Following developments in recent large wood reintroductions in freshwater systems that have restored and improved local fish populations (e.g. Whiteway et al. 2010, Foote et al. 2020), we constructed and deployed 32 pyramidal tree-reefs, each ~3 × 3 × 3 m in the subtidal Dutch Wadden Sea. See Dickson et al. (2023) for details on site selection and methodology. Initial tree-reef monitoring showed that the response by benthic fish communities was, in ecological terms, immediate. Within 6 mo, 5 times as many fish were found at reef sites versus nearby sandy control sites; the fish were also bigger when contrasted vs. control catches (Dickson et al. 2023). Furthermore, common prawn *Palaemon serratus* were found to be 10 times as numerous on-reef and, like the fish, were also larger (Dickson et al. 2023). In 2023, thousands of fertilized cuttlefish *Sepia officinalis* eggs were found attached to the reefs J. Dickson pers. obs.), while many juvenile fish (rock gunnel *Pholis gunnelus*) were found within the reefs (J. Dickson pers. obs.). The fish and prawns observed are important food sources for mid-level and apex predators such as cod, cuttlefish, seals (de la Vega et al. 2016), and sharks. These readily available food sources for predatory fish communities, and their relatively high abundance at reef sites, should have important implications for top-down control of benthic predators, such as sea stars and crabs, both of whom prey on juvenile shellfish (Byers et al. 2017). Unfortunately, difficult environmental conditions such as strong tidal currents (up to 1.8 m s^{-1}, van der Molen et al. 2022), low to no visibility, and smothering amounts of drifting macroalgae (*Ulva* sp.) make monitoring of pelagic species challenging. Hence, our prior study (Dickson et al. 2023) focused on short-term effects (6 mo) of benthic communities on tree-reefs. Thus, we know of the short-term reef function for benthic communities but lack knowledge on both pelagic communities in general as well as the long-term response of benthic communities. Therefore, the interaction between tree-reefs and pelagic fish communities still required further elucidation.

This paper highlights benthic community development from 6 to 30 mo and offers a first look at pelagic fish community use of tree-reef sites. To achieve this, we used a combination of traditional fishing gear and the novel approach of sonar platforms. Specifically, to study the benthic community, we examined the development of fish and European shore crab *Carcinus maenas* populations from 2022 to 2024 around these tree-reefs and associated control sites using barrel-fyke nets (hereafter 'traps'). Pelagic fish abundance between reef and control sites was studied in 2024 using active acoustics (sonar). To do this, we developed a new approach based on an existing fishfinding sonar used by sport fishermen, the Garmin Panoptix Livescope, and a customized floating platform. We used the combined approaches of both sonar and traps to reveal (1) how benthic communities changed from 6 to 30 mo after tree-reef deployment, and (2) how pelagic fish communities differ between tree-reef blocks and nearby control sites.

The application of sonar was necessary because monitoring using traditional fishing nets, such as traditional fyke or gill nets, around complex structures in highly energetic, low-to-no-visibility environments proved extremely problematic. Furthermore, the traps deployed in the 2022–2024 monitoring largely select for shelter-seeking benthic species, such as *Ciliata mustela* and *Anguilla anguilla*. While these benthic fish are significantly more abundant at reef sites vs. control sites (Dickson et al. 2023, J. Dickson pers. obs.), a knowledge gap remains with respect to pelagic fish species: these pelagic fish dominate Wadden Sea fish biomass (Maathuis et al. 2024). Other traditional methods such as dive surveys and video are not viable due to environmental conditions and zero visibility, while additional extractive monitoring (such as hook-and-line fishing) is not plausible around highly complex structures in areas with high-energy currents. Furthermore, smothering amounts of drift macroalgae (*Ulva* sp.) for the majority of late spring, summer, and early fall prevent the regular use of nets/hook-and-line fishing (J. Dickson pers. obs.). To be able to monitor the pelagic fish species in and around tree-reefs without extractive fishing, we mounted a Garmin Panoptix Livescope fishfinder onto platforms which were anchored at the perimeter of tree-reef blocks and a paired nearby control site. This was done both as a monitoring method and a trial for this technology, which, to our knowledge, has thus far not been used for scientific purposes. We expected that both trap fishing and sonar imagery results would reinforce the 6 mo monitoring results of Dickson et al. (2023), i.e. fish would be more numerous, biodiverse, and larger at tree-reef sites.

2. MATERIALS AND METHODS

2.1. Site setup and location

In April 2022, 32 tree-reef units of approximately 3 m^3 each, made from felled orchard pear trees, were deployed at the ends of tidal gullies in the Eijerlandse Gat (western Dutch Wadden Sea) at approximately 3–5 m depth, depending on tidal range. Each tree-reef was made of 6 felled pear trees at the end of their viable economic lifespan (20–30 yr); each tree trunk was approximately 10–20 cm thick at its base. As many branches and twigs as possible were retained. Four 'reef blocks' were established; each of these reef-blocks had 8 tree-reefs in a roughly square pattern. For further details on site selection, reef-construction, and deployment methodology, please refer to Dickson et al. (2023).

2.2. Fish traps: equipment and setup

The traps were 100 × 25 × 25 cm with a 15 cm net opening; the nets were made from 14 mm mesh. Each net was attached with a 25 cm line to a 20 kg concrete weight; to this, a 6 m rope was affixed to a 35 cm buoy to locate and retrieve the traps. These traps primarily target benthic fish by creating a velocity shelter in tidal currents. These traps then sit on the bottom, parallel to the current, with the opening facing downcurrent. When the tides shift, the traps pivot around the weight with the tides, so that the opening is always downcurrent; fish searching for shelter from the current then swim into the traps (Fig. 1). Crabs also enter the nets by moving along the bottom and into the net opening. Prior to deployment, traps were soaked in freshwater tanks to ensure negative buoyancy.

Due to unpredictable weather conditions and general difficulties with using nets in a highly tidal area, we were only able to fish near the reefs sequentially using traps in autumn 2022 and 2024. We therefore decided to complement the fishing with active sonar technology in 2024 (see Section 2.3). For each fishing event, 3 fish traps were deployed at each reef and paired control sites. At each reef-block, a bottom-sounder was used to locate the tree-reefs; traps were then deployed approximately 5 m away from any tree-reef at the reef block perimeter, maintaining approximately 10–20 m spacing between nets. A similar spacing was employed at control sites. At T24h (24 h after deployment of the nets), termed 'Lift 1', nets were lifted, the catch was measured and recorded to the nearest 0.5 cm, and then released; the traps were

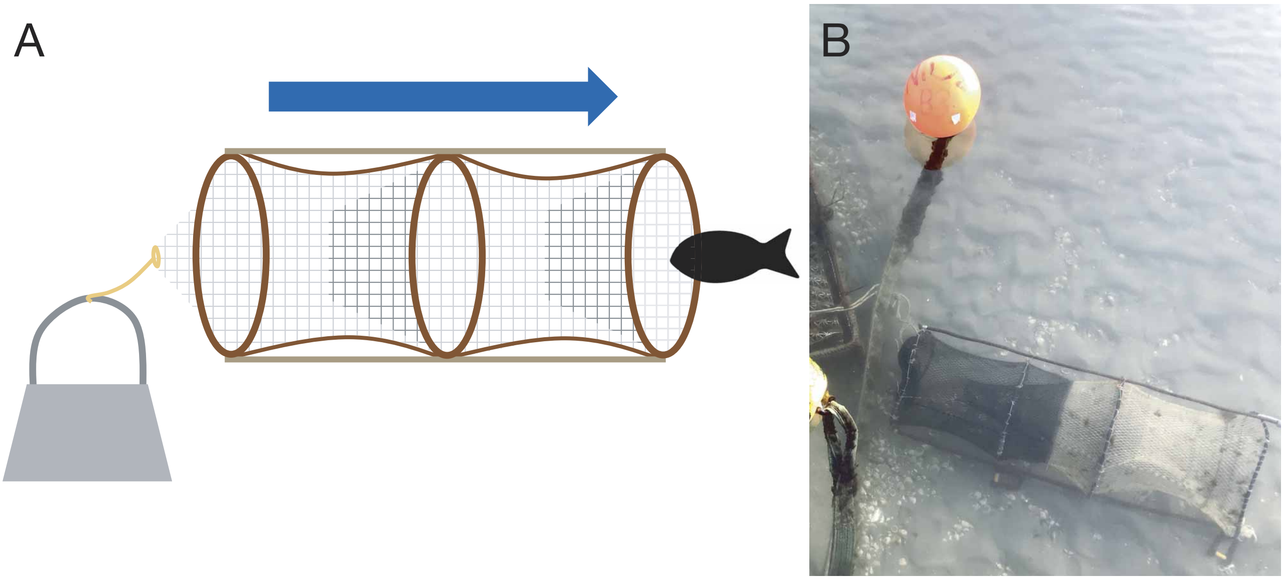

Fig. 1. Fish traps used for catch data. (A) Schematic showing trap opening facing away from current direction (indicated by blue arrow). (B) A trap in shallow water

re-deployed in the same spot. At T48h, this same procedure took place termed 'Lift 2', though the traps were then removed. One net in 2024 that became baited by the presence of a dead fish was removed from the data. All trap fishing was concluded in advance of deploying sonar platforms, so that the sonar platform and larger ship used did not affect fish behaviour and potentially catch data.

2.3. Active acoustics: equipment and setup

The Garmin Panoptix Livescope (hereafter 'fishfinder') is boat-mounted, active acoustic (sonar) equipment that is used by sport fishermen to locate and image fish. In this study, we modified this technology by integrating it onto a buoyant platform for underwater ecological monitoring; to our knowledge, this is the first time that this approach has been used for scientific purposes. The associated transducer, deck unit, and chartplotter (Garmin GPSMAP8410) are normally hardwired to vessels and powered by onboard electricity. Here, we adapted this technology by mounting the electronics inside a Pelican case, while the power source consisted of 4 car batteries in a second Pelican case; these cases were connected with waterproof cables. The fishfinder was set to 'forward' mode, entailing a 20 × 135° cone of sonar 'vision'. Data were recorded using an AverMedia Portable 2, an external card capture unit normally used by gamers to record their games. Video signal was transferred via HDMI cable from the chartplotter to the external card capture where it was written to a microSD card; the card capture unit was powered by a Makita 5.0 aH battery and USB-A adaptor. The fishfinder unit itself used the following settings: 'forward' mode, 'auto' gain, bottom stabilization 'on', and in 'livescope' mode.

The 2 Pelican cases were mounted atop a 1.6 × 1.9 m steel platform; buoyancy was provided by 28 floats of 18 kg buoyancy each. The platform also featured a radar reflector, light, and flag for visibility (Fig. 2). See Fig. S1 in the Supplement (www.int-res.com/articles/suppl/meps15048_supp.pdf) for the engineering diagram. Lastly, 3 anchors (10 kg) were attached to 10 m lines to 3 of the corners of the platform; by deploying all 3 anchors at approximately 120° to each other by use of a rigid inflatable boat, we could 'pin' the platform in place atop/near to reef and control sites, regardless of currents, waves, and wind. Each platform had a lifting sling attached for deployment via ship-mounted crane.

These 2 platforms were deployed at paired reef and control sites simultaneously. Due to risk of platform—anchor entanglement with tree-reefs at the reef blocks, sonar platforms were anchored slightly outside of the reef block by approximately 10—15 m to allow for retrieval of anchors without impacting or snagging the tree-reefs. The platforms were deployed and anchored with sonar running and the card capture unit recording; this allowed 14—24 h, or potentially longer, of sonar imagery to be recorded to a

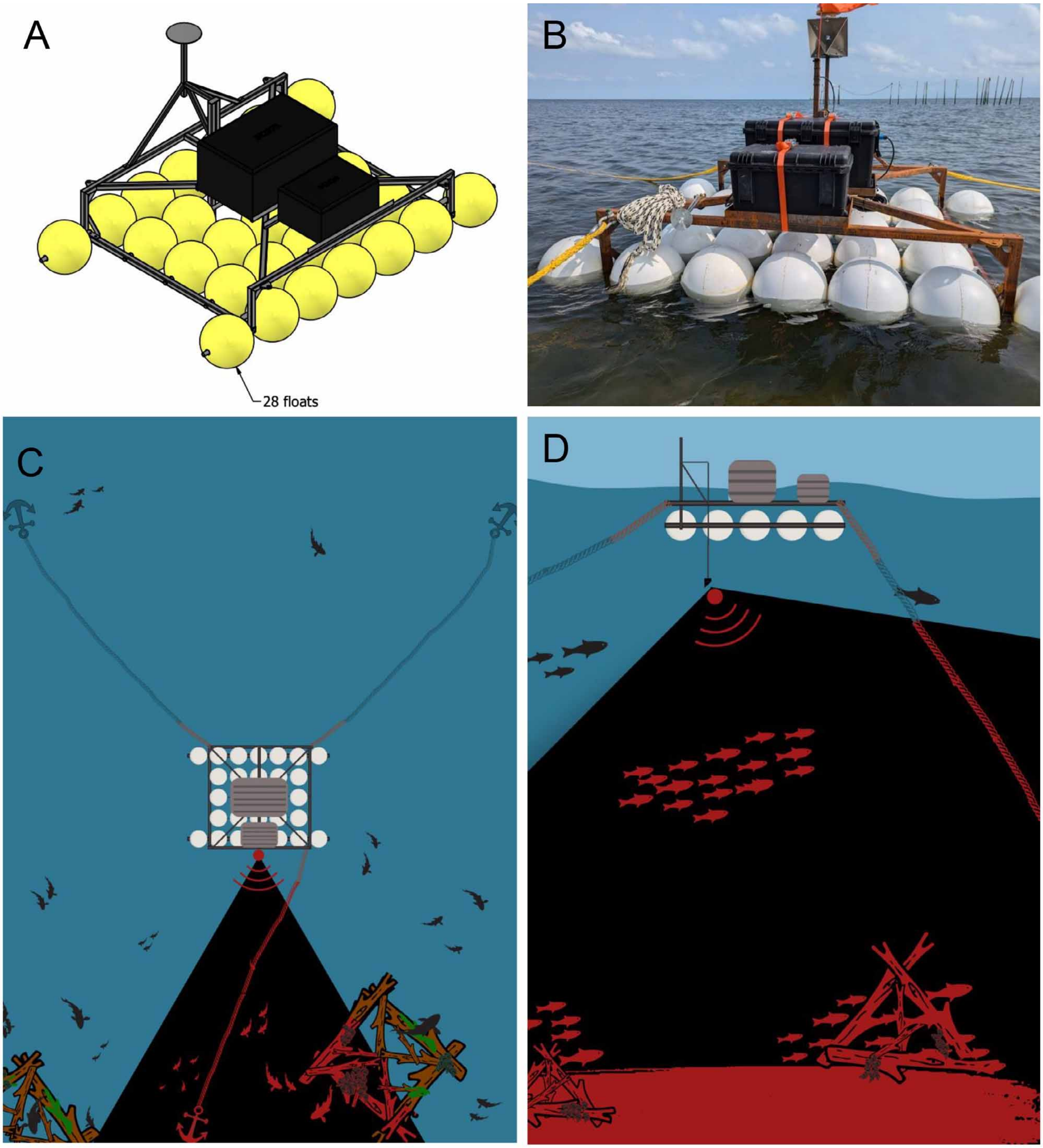

Fig. 2. (A) Sonar platform engineering diagram. (B) Sonar platform deployed for testing. (C) Conceptual diagram of sonar cone, top-down view. (D) Conceptual diagram of sonar cone, side profile view

memory card for later analysis. The size of the memory card (128 GB) supported by the Avermedia Portable 2 is the limiting factor in the length of sonar recordings, rather than the power supply. At 24 h after initial deployment, we returned to the site with a crane-equipped vessel and lifted these platforms on board. On board, we swapped out the batteries and memory cards while sailing to the next site, where the platforms were re-deployed. All 4 reef blocks and paired control sites were surveyed by this method; thus, this study used 14 h of sonar data from each reef-block and paired control.

2.4. Active acoustic analysis

Sonar video imagery was visually analysed in Windows Media Player 2025, Version 11.2501.9.0. The visual output by the fishfinder in 'forward' mode varied depending on water depth; typical output is shown in Fig. 3. In the shallowest depths across the 4 blocks, fish signals are distorted beyond 4 m forward; thus, 4 m forward and maximum range 'back' were the maximum area considered for fish abundance analysis. Targets outside of this range were discarded. The grid marks are at a 1 m scale (Fig. 3); due to the shallow depth of the water, the vertical scale is distorted. The sizes of fish signatures were estimated using these 1 m^2 grids as reference. Ground-truthing of signatures using metal 'dummy' fish was performed in calm water; field monitoring target-returns appeared to underestimate the true size of fish by about 30%. This analysis used 'detected size' without correcting for this approximate 30% underestimation; see Fig. S2 for a visualization of fish targets used in ground truthing compared to sonar returns. Estimates of whether a fish was classified as 'medium' or 'small' were applied conservatively; if a target was of borderline sizing, it was grouped into 'small'.

The ambient signal noise on screen created by wind, current, turbulence, and waves was variable depending on conditions, and had to be accounted for in visual analyses. We created a sonar noise index (SNI) scale from 1 to 10 (Figs S3 & S4, Table S1, Text S1). Fish (target) signatures were indistinguishable from the noise at SNI 9 and SNI 10; sub-samples with this level of noise were removed from the analysis. Because of this, the number of usable sample periods was not consistent across all blocks: reef block 1 had 35 usable sub-sampled windows, block 2 had 52, block 3 had 51, and block 4 had 54. To account for sub-samples deemed too noisy, if a paired reef—control sample had usable reef imagery but unusable control imagery, the paired time windows were both excluded from the analysis. Thus, we used total fish (fish count per size class / # of sub-sample windows / seconds) to produce a 'fish-per-second' result to enable comparison across all reef and control sites.

The sonar video imagery was sub-sampled by analysing 3 min at every factor of 15 min, i.e. 15:00—15:03, 15:15—15:18, 15:30—15:33, 15:45—15:48, 16:00—16:03, and so on. Each of these 3 min windows was split into 5 s sub-windows. Within each 5 s sub-window (n = 36 per 3 min window), fish were categorized by size class (small: 1—30 cm, medium: 31—60 cm, and large: >61cm) and abundance, i.e. numbers of individuals, or numbers within an aggregate or group. Aggregates were defined as fish of similar size and largely exhibiting similar behaviour, but independently of each other (Fig. 3B), whereas groups were defined by fish moving together in close proximity, with schooling behaviour (Fig. 3C). Within these groups, sonar returns blended targets together such that individual counts were not possible. Categories were then converted to a raw total number for analysis. For example, an aggregate of 20 was converted to 20 individual fish for eventual analysis. Conversely, where abundances could not be counted, for example where schools of fish (targets) occluded one another, fish were marked as 'group' (Fig. 3C). Groups were conservatively assumed to have at least 20 fish. Attempts to assign 'number of fish' to a group by watching schools for longer periods than the sub-sampled windows consistently showed at least 20 targets breaking off from the school; it is likely that this semi-arbitrary quantifying of groups as 20 fish is a significant underestimate of the true number of fish within these schools.

Fish target counts per 5 s sub-window were summed over the 36 sub-windows to create 1 overall count per size class per 3 min window. We acknowledge this likely led to some double-counting of individuals as they swam in and out of the sonar beam. For example, if an individual fish swam through the sonar 'cone of vision' and out the other side, only to immediately return, and did this 4 times within 5 s, this would register as 4 individual fish, as we cannot prove that it was the same individual. Conversely, if 4 distinct individuals remain in the sonar cone for the entirety of the 5 s, that also registered as 4 individual fish. This was unavoidable, as sonar cannot identify individuals. However, this possible double-count effect was likely consistent across control and reef sites. Other methods that attempted to mitigate this double-counting by using nMAX per size class per 3 min window led to disproportionate weighting of outliers and were thus unusable.

2.5. Statistics

2.5.1. Fish-trap data analysis

All statistical analyses were conducted in Rstudio (Version 2024.12.0+467), an integrated development environment for R (Version 4.4.2). To test whether fish and crab abundance differed between reef and control sites over time, we fitted separate truncated negative binomial generalized linear mixed-effects models (GLMMs) using the 'glmmTMB' function from the

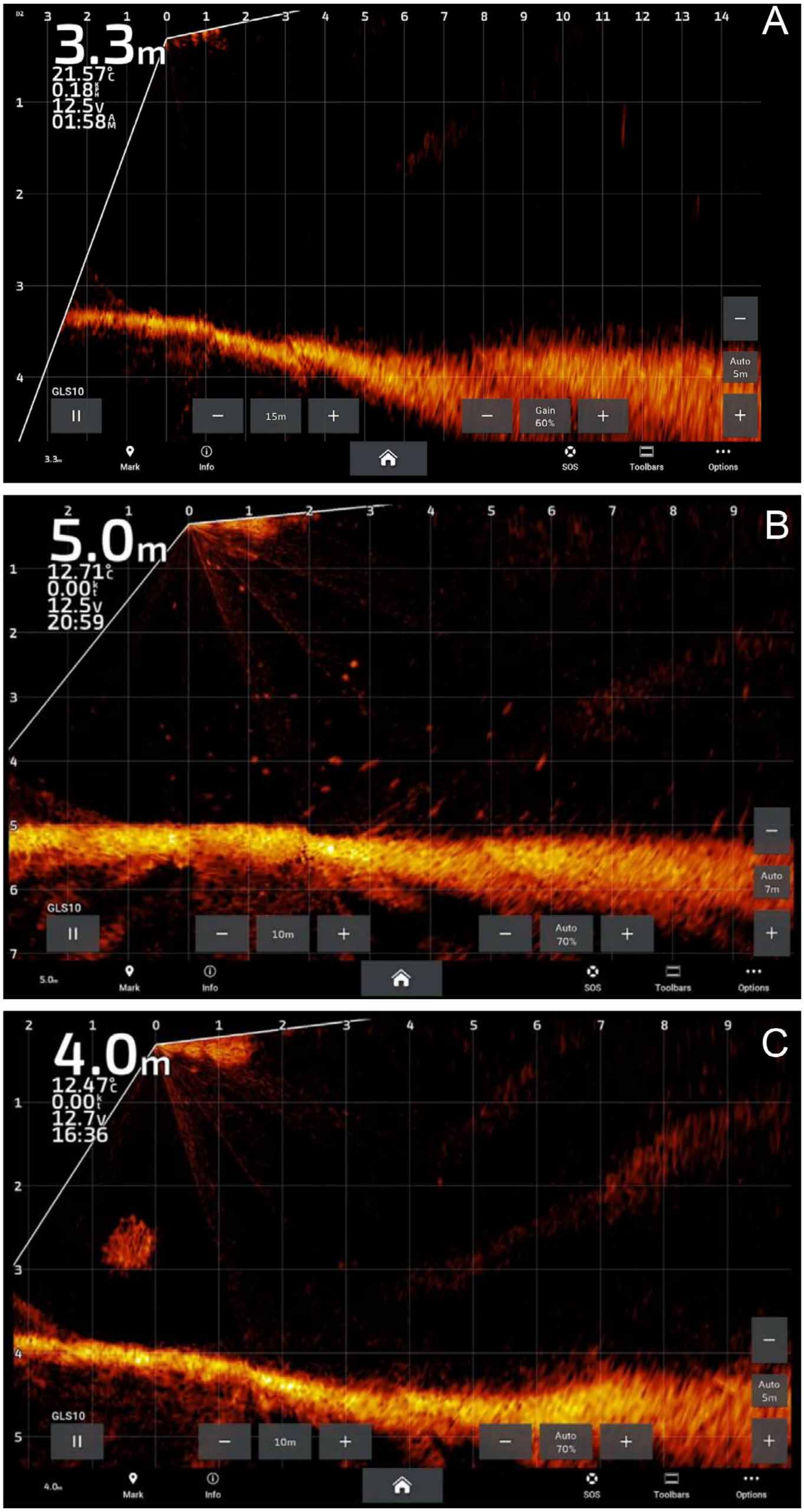

Fig. 3. Chartplotter interface screen captures, showing examples of (A) no fish present, (B) an aggregate of >20 fish, and (C) a group (at image left)

'glmmTMB' package (Brooks et al. 2017). Fixed effects included lift number (Lift 1 and Lift 2), treatment (reef and control), their interaction, and year (2022 and 2024). A nested random effects structure (reef block / lift number) was included to reflect the experimental design. Initial model exploration revealed overdispersion, violating assumptions of a Poisson distribution. To address this, we used a negative binomial distribution, which introduces an additional dispersion parameter to account for increased variability. Additionally, the data set contained no zero counts, a result of the sampling design in which at least 1 individual organism was always observed per lift. Although negative binomial models do not require zeros, standard (non-truncated) count models assume they are possible. Because zeros were structurally absent in our data, we used a zero-truncated negative binomial model to avoid bias in parameter estimates. This choice was supported by improved model fit (Akaike's information criterion, AIC), residual diagnostics, and convergence success, all of which favored the zero-truncated negative binomial model. For fish abundance, the full model including random effects was retained. For crabs, the random effects structure was excluded due to negligible variance estimates and convergence issues. To test for differences in the fish size distributions between treatments, each year was analysed separately using the Mann-Whitney *U*-test (Mann & Whitney 1947), which compares median fish sizes between reef and control treatments. Data were examined throughout using $\alpha = 0.05$, save for the five-bearded rockling *Ciliata mustela* size analysis which used $\alpha = 0.025$ to account for multiple tests with a Bonferroni correction.

2.5.2. Sonar data analysis

A linear mixed-effects model ('lme4' package in R; Bates et al. 2015) was applied using a $\log(x + 1)$ transformation to the fish-per-second counts for each designated size class to reduce skewness in the count data and to accommodate zero values that could otherwise interfere with statistical analyses; size classes were small individuals (1–30 cm), medium individuals (31–60 cm), and groups of small individuals. Counts of large individuals (≥61 cm) were excluded due to low abundance (n = 4). Reef vs. control was used as a fixed effect, while block and time window were included as random effects to account for variability between blocks and within time windows, factors not central to the primary research. The model investigated whether fish abundance differed between reef and control treatments across the specified size classes. Model residuals were assessed using a histogram, Q–Q normal plot, and the Shapiro-Wilk test to evaluate normality. To further analyse fish abundance by size class, an ANOVA was conducted using a Box-Cox transformation to normalize fish-per-second counts across all sizes when comparing reef and control.

3. RESULTS

3.1. Fish traps

3.1.1. Fish

Total fish-trap catches and species richness were consistently higher at reef sites compared to control sites across both years. In 2022, the average catch per unit effort was 3.4 times higher at reef sites (10 ± 1.22 at reef, 2.93 ± 0.43 at control, mean ± SE), and in 2024, reef catch was 5.5 times higher (20.8 ± 3.94 at reef, 3.76 ± 0.40 at control) (Fig. 4A). Reef fish-trap catches also had significantly higher species richness in both years, with an average of 1.39 ± 0.14 species per net at reefs and 1.07 ± 0.07 at control nets in 2022, and 1.4 ± 0.13 at reefs and 1 ± 0 at controls in 2024 (Fig. 4B).

Rockling *Ciliata mustela* comprised the vast majority of the total fish catch in both years (n = 748, 97.4%); the only other fish species found in numbers >1 within individual autumn sampling seasons were European eel *Anguilla anguilla* (2022: n = 1, 2024: n = 2), and pouting *Triscopterus luscus* (2022: n = 8, 2024: n = 2). In 2022, a total of 6 fish species (*C. mustela*, *T. luscus*, *A. anguilla*, shorthorn sculpin *Myoxocephalus scorpius*, rock gunnel *Pholis gunnelus*, and common goby *Pomatoschistus microps*) were caught at reef sites, while 2 species were found in controls (*C. mustela* and *M. scorpius*) (Dickson et al. 2023). In 2024, a total of 7 fish species were found at reef sites (*M. scorpius*, *A. anguilla*, *T. luscus*, European flounder *Platichthys flesus*, greater pipefish *Syngnathus acus*, *P. gunnellus*, and *C. mustela*), with 1 species found at control sites (*C. mustela*).

C. mustela were significantly more abundant at reef sites across both years, (GLMM: $z = 3.30$, $p < 0.001$, 2022: n = 261 at reef, n = 43 at control, and 2024: n = 487 at reef, and n = 79 at control). Neither lift number nor the interaction with treatment or year was significant. Median *C. mustela* size also proved significantly larger at reef sites than controls in both 2022 (Mann-Whitney $U = 3550.5$, $p < 0.05$; 14.5 and 14 cm at reef and control, respectively) and 2024 (Mann-Whitney

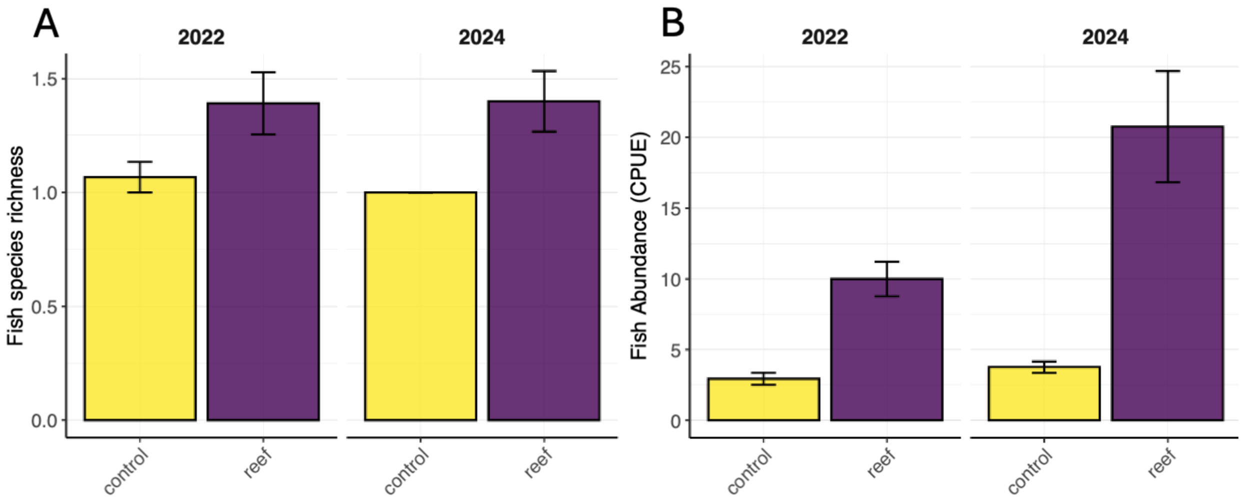

Fig. 4. Fish catches in traps across reef and control sites. (A) Fish species richness per trap and (B) fish per trap catch per unit effort (CPUE). Bars represent the averages and error bars indicate the standard error of the means

U = 11664, $p < 0.005$; 15 and 13.5 cm at reef and control, respectively; Fig. 5). See Tables S2–S5 for statistical outputs. A model including the treatment × year interaction did not improve fit ($\Delta AIC = 2$), and the interaction term was non-significant, indicating that while absolute catches were higher in 2024, the relative effect of reef vs. control did not change significantly between years.

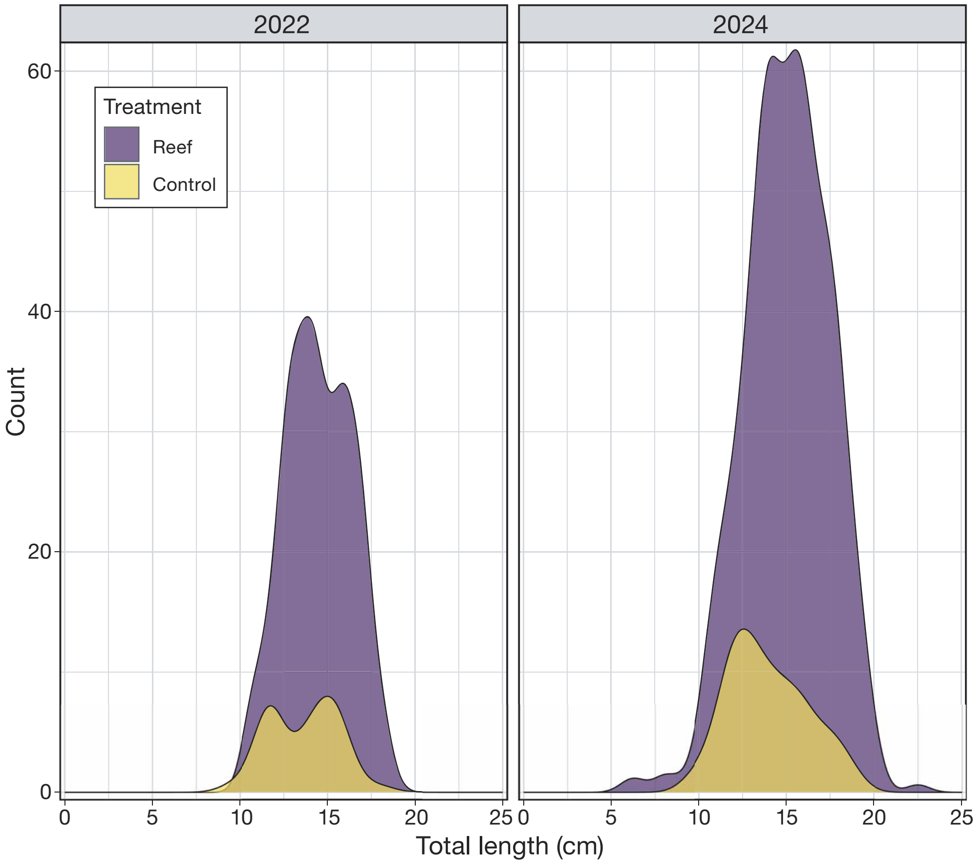

Fig. 5. Size and abundance of individual five-bearded rockling *Ciliata mustela* caught in fish traps in 2022 and 2024 at reef and control sites

3.1.2. Crabs

Carcinus maenas counts increased significantly over time from 2022 to 2024 (GLMM: $z = 6.69$, $p < 0.001$), with an average increase of approximately 88% over this period. Counts were also significantly lower — by about 77% — in the reef treatment compared to the control in 2024 (GLMM: $z = -2.74$, $p = 0.006$). In contrast to fish catch, we found a significant interaction between lift number and treatment (GLMM: $z = 2.02$, $p = 0.043$). This indicates that the effect of lift number varied by treatment, with reef sites showing a more positive lift response than in the control site (Fig. 6). A model including the treatment × year interaction did not improve fit ($\Delta AIC = 3$), and the interaction term was non-significant, indicating that while absolute catches were higher in 2024, the relative effect of reef vs. control did not vary significantly between 2022 and 2024.

3.2. Active acoustics

Detections of fish were significantly greater at reef sites compared to controls across the 3 categories (Fig. 7). Counts of small individual fish at reef sites (mean: 1.9945 s^{-1}) were 300% that of controls (0.6643 s^{-1}; LMER: $F_{1,191} = 284.64$, $p < 0.001$). Counts of medium individuals at reef sites (0.0079 s^{-1}) were 359% that of controls (0.0022 s^{-1}; LMER: $F_{1,191} = 11.81$, $p = 0.001$). Concurrently, counts of groups of small individuals at reef sites (0.0409 s^{-1}) were 215% that of controls (0.0190 s^{-1}, LMER: $F_{1,191} = 27.29$, $p < 0.001$). See Table S5 for statistical outputs.

4. DISCUSSION

In this study, we explored the longer-term (i.e. 3 growing seasons across 2.5 yr) response of benthic and pelagic fish communities to the introduction of tree-reefs constructed from discarded felled fruit trees. Tree-reef blocks show a higher abundance and diversity of fish using both the fish trap and sonar methodologies, while the traps also show lessened *Carcinus maenas* foraging behaviour at reef sites. Our findings further highlight that sonar can be a valuable tool for monitoring pelagic fish abundance and size classes in relatively shallow, turbid waters.

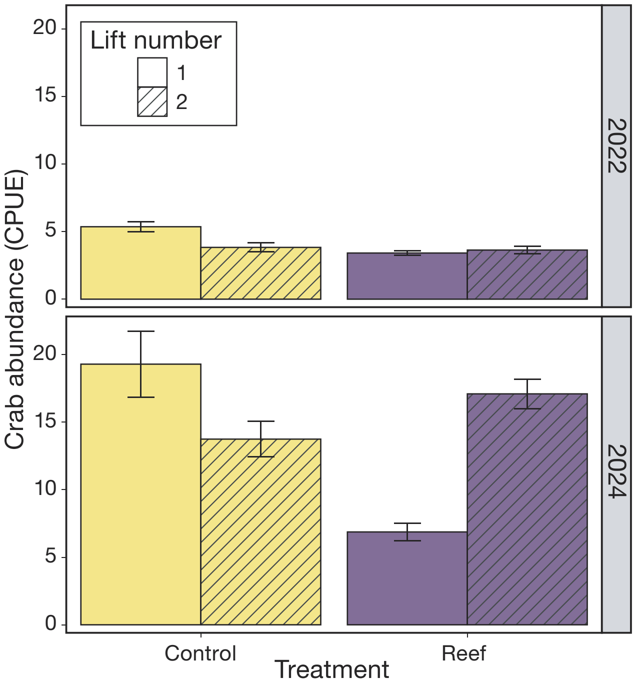

Fig. 6. *Carcinus maenas* catch per unit effort/trap (CPUE) by lift number, control vs. reef, and 2022 vs. 2024. Bars represent the average CPUE, calculated as the total number of crabs divided by the total number of traps sampled. Error bars indicate the standard error of CPUE, derived from the variation in crab counts among individual nets within each lift

4.1. Biological communities

4.1.1. Fish traps: benthic fish

Fish trap results show that rockling *Ciliata mustela* were larger and more numerous at reef sites versus nearby control blocks in both 2022 and 2024. More-

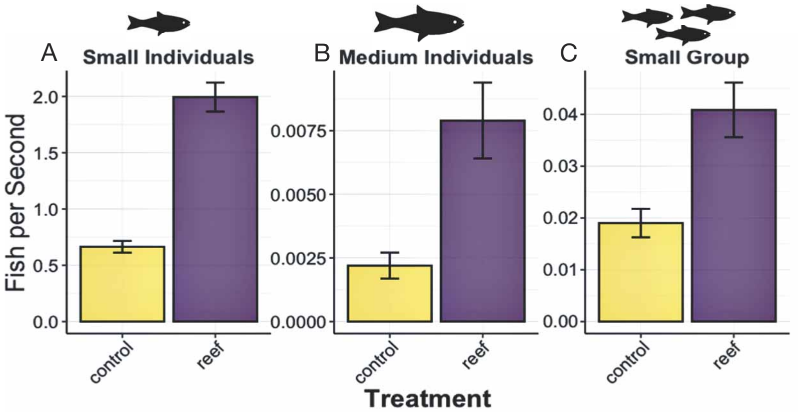

Fig. 7. Sonar results of fish-per-second (non-transformed data) of (A) small individuals (1–30 cm), (B) medium individuals (31–60 cm), and (C) groups of small individuals (≥20 fish per school). Bars indicate the average fish s^{-1} for each size class and error bars indicate the standard error of the means

over, we found that rockling generally increased in number and size from 2022 to 2024. However, as these increases were stronger at tree-reefs than at controls, these results suggest that a reef effect rather than a wider-spread environmental effect may be driving this trend. Indeed, the relatively short distance between control and reef sites (~200 m) lends weight to this hypothesis. It is conceivable that the increase in sessile primary and secondary producers on the tree-reefs (Dickson et al. 2023) became both food and habitat for fish and other consumers. Though we were unable to survey sessile life in 2024, initial measurements in 2022 showed up to 270% coverage of organisms on substrate, with both biodiversity and abundance increasing with height off the sea floor (Dickson et al. 2023). Indeed, anecdotal visual observations on the tree-reefs in 2023 revealed tens of thousands of tunicates (*Styela clava*) that had been 'sliced' open with 'guts' sucked out (J. Dickson pers. obs.). Clearly, something was preying on the sessile communities growing on these reefs, though exactly which species remains unclear. This predation on sessile communities may have led to fish in both greater abundance and larger size (a proxy for health; Ricker 1975), as seen within the rockling catch.

This pattern of larger healthier fish around reef sites was similarly demonstrated within reef-enhanced seagrass beds in the Bahamas, where white grunt *Haemulon plumierii* showed both higher abundance and better health in seagrass meadows proximal to artificial reefs (Layman & Allgeier 2020) when contrasted to non-structure-enhanced seagrass meadows. It is plausible that increased secondary production from sessile benthic organisms on these artificial reefs allows bottom-up effects of nutrient supply to outweigh population-level competition for resources (Yeager et al. 2014, Layman & Allgeier 2020). In other words, we hypothesize that the tree-reefs may in fact be producing fish, as suggested by Layman & Allgeier (2020) from other studies, and not only attracting them from elsewhere. The facts that rockling exhibit site fidelity (Gibson 1967) and were significantly larger and more numerous at reef sites seem to support this hypothesis; anecdotal observations of caught eel at reef sites also align with this pattern. Furthermore, multiple tree-reefs have been observed with thousands of attached cuttlefish *Sepia officinalis* eggs (Fig. 8; J. Dickson pers. obs.), lending further weight to the idea that the tree-reefs may be producing and/or supporting free-swimming aquatic organisms, via structure/substrate attachment points and habitat, and/or increased primary and secondary production. In addition, anecdotal reef-lifting operations in the springs of 2023–2025 showed dozens of <2 cm juvenile *Pholis gunnelus* falling out of the tree-reefs (J. Dickson pers. obs.); these were estimated to be <10 d old, showing that the reefs likely provide both spawning and rearing habitat for at least 2 species of fish.

Fig. 8. Cuttlefish *Sepia officinalis* eggs attached to a tree-reef branch in 2023

4.1.2. Fish traps: crabs

Crab populations can only be accounted for in the fish trap data. In 2022, 6 mo after deployment of tree-reefs, no difference was found in crab catches between reef and control sites. In 2024, however, crab catches were 77% greater at control sites vs. reef sites within the first 24 h set, while there was no difference in the second set. This is curious, as crabs should, in theory, be more numerous at reef sites (Schlotterbeck & Henze 2021), as reefs typically offer greater food availability and habitat for crabs (Young & Elliott 2020). Indeed, it is well documented that crab larvae and juveniles recruit to structurally complex habitats at a higher density than open sandy bottoms (Hedvall et al. 1998, Moksnes 2002, Ellrich et al. 2025). We hypothesize that a seascape and/or soundscape of fear around the tree-reefs could be suppressing crab foraging during the first 24 h set. Other experiments have shown that visual (Byers et al. 2017), acoustic (Hughes et al. 2014), and chemical (Hazlett 2010) signals from predatory fish and sharks suppress crab foraging behaviour, which can have significant positive effects on survival of sessile on-reef organisms (Byers et al. 2017). At the tree-reef sites, catch, sonar, and passive acoustic data showed that smaller generalist fish such as rockling, larger predatory fish such as sea bass and eels, along with seals are active around the tree-reefs in higher abundance than nearby controls (Dickson et al. 2023, Watson et al. 2025, Watson et al. unpubl.; J. Dickson pers. obs.). It is plausible that these multi-trophic populations around tree-reefs are emitting predatory cues that suppress crab foraging due to seascape of fear effects.

We hypothesize that the suppression of crab activity is lost during the second 24 h set as the nets may have become functionally 'baited' by fish scent from the first set, where traps caught many fish, which created higher motivation for crabs to forage more openly during the second set at reef sites. Conversely, at control sites, where there are fewer predatory pelagic animals, crab behaviour is likely not influenced by a seascape of fear nor by scent effects as there were lower fish catches and fewer sonar target returns. We suggest that further research is required to assess this 'seascape of fear' hypothesis, potentially using scent trials in aquaria.

From 2022 to 2024, crab populations increased overall at both reef and control sites. The reason for this increase remains unclear. We hypothesize that this may be due to a Wadden Sea-wide trophic cascade caused by mass settlement of blue mussels *Mytilus edulis*, a preferred food source for shore crabs (Young & Elliott 2020). We witnessed this trophic cascade through 2024 fieldwork: mussel-predating crabs along with sea stars (*Asterias rubens*) showed clearly higher abundances at both reef and control sites than in prior years. Regardless of the underlying reason, crab populations increased overall. Nevertheless, relative crab populations proved less abundant at reef sites versus control sites, in spite of the wide-scale availability of increased crab prey at the reefs (Dickson et al. 2023). Given the abundance of hard substrate provided by the tree-reefs, and high abundance of mussels and thus crab prey, this is quite curious indeed; one might expect crab populations to be higher in a place with more prey and more shelter (Young & Elliott 2020, Schlotterbeck & Henze 2021).

4.1.3. Pelagic fish

The 3 size classes of fish analysed show that small to medium fish and groups of small individuals are much more abundant at tree-reef sites. Although our sonar methodology could not account for species, it provided several advantages over our fish traps and other types of nets. Firstly, sonar can assess fish numbers, presence, rough sizing, and behaviour (i.e. schooling) over time (Warren 2012), whereas nets can only provide an aggregated snapshot over a fixed time period. A further advantage conferred by the use of sonar versus extractive net monitoring is that sonar is significantly less affected by drift algae that tend to clog nets to the point where they become unable to fish, or even retrieve (J. Dickson pers. obs.). Lastly, sonar is non-invasive and able to sample throughout the water column, as opposed to barrel-fyke fish traps, which only select for shelter-seeking benthic species.

However, sonar is generally unable to detect benthic fish because they blend in with sea bottom noise (Francisco & Sundberg 2019). Therefore, we assume that the majority of fish recorded by the sonar platforms are pelagic fish, though we acknowledge that pelagic fish do utilize benthic habitats, and benthic fish, for example rockling, use the water column. In addition, it is plausible that the presence of the sonar platforms themselves, as 'floating shelter', may have influenced fish behaviour by acting as a 'fish attraction device'.

Using the fish-trap methodology, only 3 fish caught in traps over the 2 sampling seasons would count as 'medium individuals' (31–60 cm) or 'large individuals' (>61 cm); these fish, all eels, were caught at reef sites. Conversely, sonar targets of medium fish numbered 349 in the single sampling season, 72% of them detected at reef sites. This, then, suggests that autumnal fish traps — presumably due to their relatively small size — are likely underestimating the overall abundance of larger pelagic and benthic fish, but to what degree is uncertain. Moreover, in order to ensure that our anchors and their lines did not entangle within the tree-reefs, the 'reef' sonar platforms were anchored 10–15 m away from edges of tree-reef units. As larger, often piscivorous, fish tend to prefer to remain closer to reefs (Whitfield et al. 2014, Paxton et al. 2019, 2017), and our usable sonar 'window' measured a maximum of 6 m forward (and back), this may have artificially depressed medium individual observations (Paxton et al. 2019). Finally, the relative proportion of small fish versus medium fish also may be a function of local geospatial ecology, as the Wadden Sea is an important nursery area (Maathuis et al. 2025); quite often, larger, sexually mature fish enter the area after spring from the North Sea (van der Veer et al. 2015), while sampling in this study took place in autumn. We thus suggest that medium and large fish could make a proportionally greater fraction of fish detections if sonar monitoring was conducted in late spring or summer as opposed to autumn.

Despite the promising results of increased fish size and abundance over 2.5 yr at tree-reef sites, the fish community at tree-reef sites has almost certainly not reached a climax community. Most artificial reef studies do not persist in monitoring schemes for multi-decadal time scales, which may be required to determine when an artificial reef has reached climax community and therefore become equivalent to natural reefs. For example, 2 artificial reef installations monitored in New South Wales, Australia, at 2 and 12 yr post deployment showed a more than 2-fold increase in species richness, while ~85% of fish species increased in abundance (Becker et al. 2022). Interestingly, fish abundance generally increased or remained stable at both artificial and nearby natural reef sites over this period in the majority of species monitored (Becker et al. 2022). However, most (68%) artificial reef monitoring programs end within 2 yr or less (Lima et al. 2019), thereby shrouding longer-term ecological community development in mystery. We expect tree-reefs to display similar patterns over multi-decadal timescales as evidenced by the study of Becker et al. (2022) in New South Wales, though only further research and monitoring can reveal this.

4.2. Management implications

4.2.1. Sonar monitoring

Our sonar platforms have proven useful in establishing fish abundance by size class. Moreover, we suggest that with further analysis, these data could be used to assess behaviour as well. As such, this method may be able to elucidate time and tide-specific fish behaviour that nets, eDNA, and dive surveys cannot. The equipment itself is relatively affordable given the generally high costs of marine monitoring; however, data processing and ship time to move the platforms is unfortunately time consuming and potentially expensive. Overall, these Garmin Panoptix Fishfinder units have proven to be highly useful and cost-effective tools for monitoring fish populations in a turbid, high-flow environment with significant 3D structure where we found no other methods to work (i.e. weighted GoPros, mounted long-term cameras, baited cameras, and pelagic fishing gear). Yet, sonar for fish monitoring is not a silver bullet; several potential problems exist. Sonar returns, particularly in shallow water, can be negatively affected by inclement weather which induces 'noise' on the sonar returns (Robinson & McConnell 1983, Zou & Badiey 2018, Francisco & Sundberg 2019), thereby allowing fish to hide within the noise. Furthermore, structures induce turbulence, which also increases noise (Lhermitte & Lemmin 1993, Hou et al. 2013). Lastly, if a fish is hiding within a porous structure such as a tree-reef, it is likely that the target will blend in with the reef structure and thus be undetectable. This study took place in shallow water (max depth 3–5 m); we expect this sonar to be increasingly effective in deeper waters down to 60 m, the maximum forward and down range of the Garmin Panoptix Fishfinder (https://www.garmin.com/en-GB/marine/live-sonar/panoptix/). Overall, however, the fishfinder platforms yielded more data and new insights into fish abundance around the tree-reefs than any other method we have attempted; it is a highly effective monitoring tool in the violent turbid waters of the Dutch Wadden Sea where nets are selective, ineffective, or impossible to deploy/retrieve.

4.2.2. Loss and restoration of historic sunken wood habitat

The flow of large, complex wood from coastlines and rivers to estuaries and the sea has been heavily disrupted by human-induced landscape change and river cleaning (Wohl & Iskin 2021). Indeed, global forest cover has been reduced from 40 to 27% (Wohl & Iskin 2021) and continues to decrease in most countries (Busch & Ferretti-Gallon 2017, Farrokhi et al. 2024). Driftwood historically provided and, where contemporarily present, continues to provide numerous ecosystem benefits (Dickson et al. 2025). Large wood in the marine sphere acts as structure for settlement of shellfish, sponges, bryozoans, seaweeds, and other sessile organisms (Thiel & Gutow 2005, Schwabe et al. 2015) and fish habitat (Forget et al. 2020, Dickson et al. 2023), and facilitates trophic transfer of terrestrial nutrients to marine consumers. Indeed, it has been found that 1 m^3 of softwood will produce 7–25 kg of protein via shipworm (Japan Fisheries Agency 2006), a boring bivalve that actively tunnels into and consumes wood (Turner 1966, Wolff 1979). Boring isopods graze at the surface of the wood (Cragg et al. 2007, Borges 2014), eventually exposing shipworm tunnels and allowing predatory fish and worms to consume shipworms and make use of shipworm tunnels. Empty shipworm tunnels are used by fish to shelter in and lay eggs (Hendy et al. 2013, 2014). Marine fungi and bacteria colonize wood and are consumed as an energy source by a series of marine wood-boring organisms (Kohlmeyer et al. 1995, Distel et al. 2011, Garzoli et al. 2015), along with their host wood. Wood-boring organisms excrete waste and form the genesis of a detrital ecosystem, where detritivores such as sea cucumbers consume the waste of wood-borers and other species (Harmon

2021); these detritivores become prey for predatory fish (Purcell et al. 2016). Throughout the decomposition cycle of marine wood, it continues to provide shelter for fish from predators (Nishimoto et al. 2015), spawning grounds (Hendy et al. 2013), foraging areas (Wohl & Iskin 2021), and energy-saving zones (Hafs et al. 2014) while hosting its own sessile communities (Bersoza Hernández & Angelini 2019). The degradation rate of marine wood depends on numerous factors such as species of wood, ratio of heartwood to sapwood, water temperature and associated marine boring community, and presence of bark (Murphy et al. 2021), amongst many other factors; see Dickson et al. 2025, for an overview of ecological considerations of aquatic and marine wood.

Complex woody 3D structures, for example these tree-reefs, may be a useful tool for coastal managers and restoration practitioners to increase and enhance local biodiversity using natural material that was historically present in large amounts, as many fish tend to prefer complex structures (Bevelhimer 1996, Smokorowski & Pratt 2007, Matis et al. 2018, Dye et al. 2024). Managers may be able to use tree-reefs as a tool to safely secure suitably scaled quantities of sunken wood to the seafloor to kickstart shellfish reef formation, promote heterogeneity of historically present seafloor habitats, and increase fish populations and diversity. Natural wood export from rivers historically occurred at a vast scale (Wohl & Iskin 2021); this sheer amount of driftwood is not realistically possible to recreate. However, by using engineered tree-reefs, managers may be able to partially compensate for this loss of driftwood in a safe and controlled way. This may potentially induce a push to return to more biodiverse food webs by attracting and then producing multiple trophic levels of fish communities as well as hard substrate sessile communities. Tree-reefs are not a one-size-fits-all approach: efficacy and suitability depend on the local environment. However, they provide numerous ecological benefits that concrete/metal/plastic artificial reefs cannot, simply due to the natural material and inherent ecosystem services provided by aquatic large wood.

Artificial reefs are generally made of metal, concrete, and plastic (Lima et al. 2019); wooden artificial reefs are not widespread. To our knowledge, limited scientific studies have only been conducted on wooden artificial reefs in the Netherlands (Dickson et al. 2023) and Japan (Masuda et al. 2010, Alam et al. 2020). This scarcity of research makes contrasting the performance of tree-reefs to concrete or metal artificial reefs difficult. However, it is probable that concrete/metal reefs will continue to provide usable settlement substrate and shelter for the duration of their lifespan, while tree-reefs will be partially consumed by marine wood-borers (Hendy et al. 2013, 2014, Borges et al. 2014), softened by marine microbes (Kohlmeyer et al. 1995), and eventually collapse on their way to complete degradation.

It is possible that shellfish colonization is significant enough upon the wood surface that when the wood fully biodegrades a natural shellfish bed will remain in place. However, due to their medium-term expected lifespan (estimated at 10–40 yr, depending on the local environment), tree-reefs or other wooden artificial reefs are likely better deployed in calmer, deeper environments, though they have proven to withstand significant stress in energetic shallow waters with minimal damage after 2.5 yr at the time of writing. As tree-reefs are built of natural, biodegradable materials, it is expected — and desired — that they eventually break down and decompose. As they degrade via marine-wood borers, wood increases in internal complexity due to the borer tunnels. Wood-based reefs are the only artificial reef material that actively provides not just structure, but also nutrients via decomposing wood, while undergoing changes in complexity and dimension during the natural degradation process. Manufactured substrates such as concrete and plastic cannot provide this lignocellulose-based food, wood-based detrivorous ecosystem, or increasing internal complexity due to lack of woody material.

While we cannot yet compare the ecological performance of tree-reefs vs. traditional artificial reefs (e.g. reef balls) due to lack of study, some clear differences have emerged: tree-reefs are significantly lighter (~500 kg), taller (~1.8×), and wider (~1.5×) than the largest reef ball (Goliath, ~2750 kg) (https://www.reefball.org/technicalspecs.htm). Furthermore, they also allow flow-through and thus settlement of filter feeding organisms on the inside of the reef, something that is not seen with artificial reef-like objects such as scour protection (Hansen et al. 2007) or, presumably, reef balls. Lastly, since the orchard trees used are a waste product that is generally turned into biofuel or burned in their fields, the material is relatively low cost, which may allow tree-reef blocks to be scaled to significant size. All this being said, further research is suggested to directly compare long-term performance of concrete/metal reefs versus tree-reefs.

4.3. Conclusion

After 2.5 yr of monitoring, tree-reefs have fostered a large improvement in fish abundance, size, and biodiversity when contrasted to nearby control areas. Furthermore, the fish recruited, retained, and seemingly produced within novel tree-reef ecosystems appear to be exerting top-down predation pressure on benthic predators such as crabs, which would have important implications for restoration efforts where over-predation of sessile species is problematic. Overall, the cost-effective engineering principles and environmentally sound approach of tree-reefs appears to be a highly useful tool for bolstering fish populations and assisting in managing the biodiversity crisis.

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