DOI: 10.1002/agc.4216

Revised: 24 May 2024

ARTICLE

WILEY

Cobble reef restoration in the Baltic Sea: Implications for life below water

E. Casabona^{1,2} | T. Wilms³ | M. Moltesen¹ | J. L. Bertelsen¹ | B. M. Kruse⁴ H. Flávio⁵ | P. Holloway^{2,6} | J. C. Svendsen¹

¹National Institute of Aquatic Resources, DTU Aqua, Technical University of Denmark, Silkeborg, Denmark

²Department of Geography, University College Cork, Cork, Ireland

³Environment and Sustainability Unit, Vattenfall A/S, Copenhagen, Denmark

⁴HavNatur, Sønderborg, Denmark

⁵Department of Biology & Laurier Institute for Water Science, Wilfrid Laurier University, Waterloo, Ontario, Canada

⁶Environmental Research Institute, University College Cork, Cork, Ireland

Correspondence

J. C. Svendsen, Technical University of Denmark, National Institute of Aquatic Resources, DTU Aqua, Vejlsøvej 39, 8600 Silkeborg, Denmark. Email: jos@aqua.dtu.dk

Funding information

This research was funded by 1) the European Union, 2) the European Maritime and Fisheries Fund (EMFF), 3) the Danish Fisheries Agency, 4) the Velux Foundation, 5) the energy company Vattenfall, and 6) the Danish Rod and Net Fish License Funds. In terms of funding sources 1–3, project numbers are 33113-B-16-059 and 33113-B-19-142. The EMFF funding ensures that Denmark and Europe invest in sustainable fisheries and aquaculture.

Abstract

- Many coastal ecosystems are impacted by human pressures. Rocky reefs are structurally complex habitats that often support elevated fish abundance and marine biodiversity. In the Baltic Sea, rocky reefs have suffered from extraction for decades, leading to a decrease in hard substrata and complex habitat availability.
- 2. This study is the first to restore cobble reefs and examine the biological effects.
- 3. Baited and unbaited underwater video systems (BRUVS and UBRUVS, respectively) were employed across five years to monitor fish communities before reef deployment in 2017 and after reef deployment in 2018 and 2021.
- Using a before-after control-impact (BACI) study design with replicates, relative abundances of Atlantic cod (*Gadus morhua*), herring (*Clupeidae* sp.), goldsinny wrasse (*Ctenolabrus rupestris*), two-spotted goby (*Pomatoschistus flavescens*), shore crab (*Carcinus maenas*), and flatfish (*Pleuronectiformes* spp.) were compared across time and test sites. Comparisons were conducted across 1) restored cobble reefs, 2) natural cobble reefs, and 3) sand-bottom test sites.
- 5. This study found positive reef restoration effects revealed consistently by BRUVS and UBRUVS in three species: Atlantic cod, goldsinny wrasse and two-spotted goby. These findings indicate that A) it is possible to restore cobble reefs and the associated mobile fauna, but also that B) continued marine extraction of cobble degrades complex habitats to the detriment of various marine species.
- 6. To preserve Atlantic cod, and other sensitive species, we emphasize ecosystem restoration and warn against marine cobble reef extraction in vulnerable areas. Restoration of marine habitats may contribute to achieving the UN sustainable development goal covering life below water.

KEYWORDS

BACI design, Baltic Sea, BRUVS, coastal, cobble reef, ecosystem restoration, fish abundance, MaxN, sustainable development goal

This is an open access article under the terms of the Creative Commons Attribution-NonCommercial-NoDerivs License, which permits use and distribution in any medium, provided the original work is properly cited, the use is non-commercial and no modifications or adaptations are made.

1 | INTRODUCTION

Habitat loss is one of the main human impacts on coastal ecosystems (Lotze et al., 2006; Bulleri & Chapman, 2010) and fish populations (Arthington et al., 2016; Yan et al., 2021). Over half of the vulnerable marine habitats are already lost (Duarte et al., 2020), including rocky reefs and seagrass areas (Dafforn et al., 2015; Elliott et al., 2016; Folpp et al., 2020; Hastings et al., 2020). Structurally complex habitats are decreasing across temperate marine environments (Airoldi et al., 2008), causing degradation of nearshore marine systems and the associated biodiversity (Lotze et al., 2006; Singh et al., 2021).

Rocky reefs are one of the marine ecosystems subjected to one of the fastest rates of human cumulative impact, exceeded only by coral reefs, seagrass meadows, and mangroves (Halpern et al., 2019). This habitat degradation is especially significant in coastal areas (Schwartzbach et al., 2020). In the Baltic Sea, one of the most human impacted regions in the world (Halpern et al., 2008), extraction of non-living resources (e.g., cobble) remains a significant anthropogenic stressor to coastal habitats (Dailianis et al., 2018). In Denmark alone, extraction of rocky material eliminated at least 55 km² of exposed rocky reef surface across the period 1900–2000 (Helmig et al., 2020; Kristensen et al., 2017; Støttrup et al., 2014). Significant extraction of rocky material also happened before 1900, but the documentation is limited. As a result, vast marine areas have undergone a depletion in hard substrata and structurally complex habitat availability, turning the seafloor into bare sandy bottom (Airoldi et al., 2008; Støttrup et al., 2014). This habitat loss may result in declining species diversity and overall biomass (Flávio et al., 2023; Parsons et al., 2016; Støttrup et al., 2019; Wilms et al., 2021).

Rocky reefs are structurally complex habitats that provide shelter from predators, spawning and nursery areas, a feeding source for many species and hard substrata for algae to grow (Schwartzbach et al., 2020; Støttrup et al., 2014). By supporting fish foraging, breeding, and growth to maturity (Schwartzbach et al., 2020; Støttrup et al., 2019), rocky reefs are considered vital to support local fish stocks (Flávio et al., 2023; Katara et al., 2021). Rocky reefs in temperate waters may support high biomass and biodiversity (Støttrup et al., 2017). In temperate reefs, macroalgae are the main enhancing habitat complexity (Trebilco foundation species, et al., 2015). Macroalgae host significant numbers of invertebrates constituting food resources for fish species (Christie et al., 2009). Importantly, rocky reefs are currently under protection by the Natura 2000 network, which is a network of natural protected areas in the European Union and constitutes one of the largest protected areas in the world (European Commission, 2013; Kristensen et al., 2017).

Because of the significant declines in marine abundance and biodiversity, as well as the U.N. Sustainable Development Goal 14, calling for a conservation of our oceans, different restoration initiatives have emerged (Duarte et al., 2020; Lotze et al., 2006; Støttrup et al., 2017; Wilms et al., 2021). Ecological restoration is the process of intervening in the recovery of an ecosystem that has been previously degraded, damaged, or destroyed by human activities resulting in improved ecosystem services and biodiversity (Elliott et al., 2016; Seaman, 2007; Silliman et al., 2024). Restoring reefs could increase the availability of structurally complex habitats in areas where complex habitat is limited, potentially increasing the carrying capacity of the ecosystem (Bohnsack, 1989; Folpp et al., 2020). Restored reefs may increase fish production (Schwartzbach et al., 2020; Wilhelmsson et al., 1998), resulting in higher fish density and biomass (Cresson et al., 2019; Folpp et al., 2020; Hunter & Sayer, 2009; Wilms et al., 2021), as well as increased ecosystem productivity (Rouse et al., 2020; Schwartzbach et al., 2020).

Despite rocky reef protection under Natura 2000, reef restoration effects remain relatively unknown in comparison with other artificial marine infrastructures (Perkol-Finkel et al., 2012), especially in terms of higher trophic levels (Bulleri & Chapman, 2010; Bergström et al., 2013; Holloway & Field, 2020). To date, the consensus is that an increase in fish abundance and fish diversity may be expected after reef deployment (Folpp et al., 2011; Folpp et al., 2020). Conservation evidence of rocky reef restoration is not yet summarized as a practical management intervention, and previous studies differ in terms of their sampling methods (invasive vs passive; Hunter & Sayer, 2009; Kristensen et al., 2017; Cresson et al., 2019, Rhodes et al., 2020), target species (Folpp et al., 2020), duration of sampling events (Folpp et al., 2020; Lowry et al., 2014), use of pre-deployment assessments (Lowry et al., 2014), and statistical method (Holloway & Field, 2020). Therefore, questions remain concerning the impact of reef restoration on fish abundances over time, especially in temperate latitudes. Scientific monitoring using before-after control-impact (BACI) study designs is key to accurately assess the restoration success, by investigating both reef deployment sites and suitable control sites (Wilms et al., 2021).

To the best of our knowledge, the present study is the first to explore biological effects of cobble reef restoration in the Baltic Sea. This study investigated cobble reef restoration effects on the abundance of mobile fauna, using underwater cameras deployed on the seabed. The approach included both baited and unbaited cameras, similar to previous studies (Rhodes et al., 2020). Focal species included: Atlantic cod (Gadus morhua), herring (Clupeidae sp.), goldsinnv wrasse (Ctenolabrus rupestris), two-spotted goby (Pomatoschistus flavescens), and shore crab (Carcinus maenas) as reef associated species, and flatfish (Pleuronectiformes spp.) as sand bottom-associated species. Increasing the availability of rocky reef habitat in our study area provided hard substrata for the attachment of macroalgae and other organisms. It was hypothesized that Atlantic cod, goldsinny wrasse and two-spotted goby would respond positively to the reef restoration. Specifically for herring, it was hypothesized that the restored reefs would be used for spawning and therefore increase herring abundance during the spawning season (spring). It was further hypothesized that the abundance of sand-bottom associated species may decrease locally after reef restoration.

2 | MATERIALS AND METHODS

2.1 | Study site

The study area is located in Sønderborg Bay in Flensborg Fjord (Figure 1), a Natura 2000 protected area located in the Western Baltic Sea. Flensborg Fjord forms part of the border between Germany and Denmark. Extraction of rocky material is well documented across the Sønderborg Bay (Stenberg & Kristensen, 2015), reducing the rocky reef covered seabed areas.

This study restored cobble reefs experimentally in January 2018 and examined the effects in terms of the abundance and diversity of marine organisms. To this end, six study sites were identified (Figure 1): two control sites with a seabed consisting mainly of sand (Viemose and Kegnæs Ende); two sites with a seabed covered by natural cobble reefs (Spar Es and Vesterhage), and two sites with a seabed consisting mainly of sand covered by cobble after reef deployment (Hvide Mur and Stenholt). All the sites were at water depths between 6 to 7 m.

The cobble reefs were built using small rocks with a diameter ranging between approximately 6 and 30 cm. The applied range deviates slightly from the traditional cobble definition (Udden-Wentworth scale; 6.4-25.6 cm). Cobbles were obtained from a guarry in Lyngdal, Norway. In January 2018, cobble reefs were deployed at two dedicated sites (Hvide Mur and Stenholt). At each site, reef outlines consisted of two rows of 11 individual mounds. Each mound is a flat-topped pyramid with an overall bottom area of 11x11 m and a top area of 5.5x5.5 m. In every row, the height of the mounds alternated between 1.3 m and 0.6 m using 87 and 40 m³ of cobble, respectively, per mound (Annex 1).

2.2 | Study species

Fish are often considered reef restoration success indicators at higher trophic levels (Kristensen et al., 2017; Lewis et al., 2021). Several focal species of this study rely on rocky reefs at different stages of their ontogeny. The use of reefs for shelter from predators that enhances juvenile survival and growth to maturity has been reported in 1) juvenile cod (Gotceitas et al., 1995; Schwartzbach et al., 2020; Støttrup et al., 2017; Støttrup et al., 2019), 2) goldsinny wrasse (Støttrup et al., 2014), 3) two spotted goby that use rock crevices to nest (Utne-Palm et al., 2015), and 4) shore crab (Moksnes, 2002). Reefs can also be used as a spawning and nursery area by Atlantic herring. Despite being a pelagic species, herring is the only cluepoid to lay demersal eggs (Maravelias et al., 2000). Spring spawning occurs inshore (Frost & Diele, 2022), and eggs are usually attached to benthic substrates including gravel, rocky areas or macroalgae e.g., Fucus vesiculosus (De Groot, 1980; Von Nordheim et al., 2018; Flávio et al., 2023). The present study deployed cobble reefs in areas with known historical records of herring spawning and sampled the area during the herring spawning season (spring). We expected herring to utilize the cobble reefs and the associated vegetation for spawning and subsequently as a nursery area (Flávio et al., 2023; ICES, 2023b).

2.3 | Data collection and experimental design

2.3.1 | Sampling campaign

Sampling was carried out daily from April to May in 2017, 2018, and 2021. Any seasonality-related variability in species abundance and diversity was minimized by sampling during the same season (spring) each year (Folpp et al., 2020; Wilms et al., 2021). Our



FIGURE 1 Study area with sites of different seabed-treatments: Sand-bottom (red circles), natural cobble reef (light blue triangles), and restored cobble reef (dark blue squares). Each seabed-treatment included two different sites.

2.3.2 | Camera setups

Recordings were obtained using Baited Underwater Video Systems (BRUVSs) and Unbaited Underwater Video Systems (UBRUVSs). Go Pro cameras (HERO 3, 3+, 4, and, 7) at 720p resolution and 30 frames per second were used. Remote Underwater Video System (RUVS) setup consisted of a concrete base $(45 \times 45 \times 5 \text{ cm})$; length \times width \times height) and a metal pole (3 cm diameter; 100 cm high). Cameras and underwater cases were attached to the vertical pole 25 cm from the seabed and filmed horizontally. BRUVS included a bait, which consisted of 500 g chopped herring (Clupea harengus), and was attached to a horizontal bait arm. 80 cm from the camera. The bait arm in BRUVS had markings every 10 cm for visibility estimates. UBRUVS setup was similar to BRUVS, but instead of a bait arm, a rope was attached to the concrete base and laid flat on the seabed in front of the camera's Field of View (FOV). The rope had meter markers for visibility estimates and was secured with an anchor at the end. Both BRUVS and UBRUVS included a rope connecting the anchor to a floating buoy with a flag for identification.

2.3.3 | Deployment and recording characteristics

This study used a side scanner (Lowrance Elite-7 Ti; Lowrance Electronics; https://www.lowrance.com/) to scan the seabed for the desired substrate (e.g., cobble reef or sand) and subsequently deploy the cameras. Deployment procedure and camera placement was similar to previous studies (Langlois et al., 2018; Rhodes et al., 2020). BRUVS recordings started immediately after deployment and ran for 60-120 minutes, subjected to later video analysis. Deployment of BRUVS was undertaken between 10:00 and 18:00. UBRUVS were intervalometer (BlinkX equipped with an or Time Lapse Intervalometer https://cam-do.com/). Intervalometers were programmed to turn on the UBRUVS every hour and record for 2 min and turn off. Each 2 min recording provided a video sequence, subjected to later video analysis. This setting ran through the entire diel cycle, recording for >24 h, however, only daytime footage was included in the analysis. At each deployment, temperature measurements were conducted.

2.4 | Video analysis

This study analysed the recordings using the VLC Media Player[®] software. For each camera deployment (BRUVS) and 2 min sequence

(UBRUVS), data on species present, macroalgae coverage, functional visibility and Field of View (FOV) were estimated. Macroalgae coverage (% seabed) was visually quantified directly from the footage (Rhodes et al., 2020). Macroalgae species identification or community analysis was not within the scope of this study. Functional visibility (m) was defined as the greatest distance at which fish were identifiable. Functional visibility was estimated with the bait arm (BRUVS) or rope markers (UBRUVS). FOV was expressed as a percentage of the total available view, and was calculated by subtracting the percentage of visually estimated seabed structures that obstructed the view from 100%. Lastly, recording duration or soaktime was annotated in each BRUVS recording.

Fish species abundances were assessed using the MaxN metric, which is the greatest number of certain fish species (or group of species) in a single frame (Cappo et al., 2004; Smith et al., 2012; Wilms et al., 2021). Flatfish (*Pleuronectiformes* spp.) were treated as a functional group at the order level, containing MaxN from right and left-eyed species including the following species: *Pleuronectiformes* sp., *Platichthys flesus*, *Pleuronectes platessa*, *Pleuronectidae* sp., *Limanda limanda*, *Scophthalmidae* sp., *Scophthalmus maximus*, *Scophthalmus rhombus*. Herring included counts (MaxN) of individuals labelled as *Clupeidae* sp. covering *C. harengus* and *Sprattus sprattus* because, it was difficult to differentiate the two species morphologically using the underwater footage.

2.5 | Data analysis

Generalized linear mixed models (GLMMs) were used to analyse the abundance (MaxN) of our six focal species groups (i.e., Atlantic cod, herring, two-spotted goby, shore crab, goldsinny wrasse, and flatfish).

Model explanatory variables included the factors "Year" (three levels: 2017 (before), 2018 (four months after) and 2021 (three years after)), "Treatment" (three levels: Sand, Natural reef and Restored reef) and their interaction. This study included the additional covariates "Temperature", "Macroalgae coverage", "Visibility", "(FOV)" for all models to account for variation. BRUVS data models included "Soaktime" as fixed effects to account for the duration of the recording. BRUVS, and UBRUVS models were analysed separately and included "Site" (e.g. "Hvide Mur") as random effects (Smith, 2002). To allow the models to account for potential correlations between sampling units (2-minute sequences) from the same deployment or site, UBRUVS models incorporated random effects for "Deployment number" nested within "Site" (Wilms et al., 2021). Continuous recording-related variables such as: "Visibility", "FOV", "Soaktime" (latter only in BRUVS) were standardized by adding the natural logarithm to account for differences in exposure (Zuur & leno, 2016).

We parameterized GLMMs for 2017, 2018, and 2021 that had uneven sample sizes across the period (Annex 2). To avoid excluding and/or removing data unnecessarily, we retained all data in the models, with the assumption that the smaller sample sizes had data Missing at Random (Kain et al., 2015). The incorporation of random **FIGURE 2** Diagram showing the four different before-after control-impact (BACI) scenarios where positive effects would be detected. In our case, the impact is cobble reef restoration.



effects in our GLMM models was considered a reliable approach to accommodate variation between groups caused by the unbalanced sample sizes (Kain et al., 2015; Magnusson et al., 2017).

Our response data exhibited various distributions. For BRUVS models, we fitted a negative binomial distribution for Atlantic cod, goldsinny wrasse, and two-spotted goby and a quasi-Poisson distribution for herring, shore crab, and flatfish. For UBRUVS models, we fitted a negative binomial distribution for Atlantic cod, herring, and two-spotted goby, and a Poisson distribution for goldsinny wrasse, shore crab, and flatfish. The goldsinny wrasse BRUVS model had a higher outlier frequency than expected (p < 0.0001), which was thought to be due to the large volume of data, potentially inducing minimal deviation to become significant. Residuals were plotted against predictors, and no pattern was observed (results shown in Annex 3), and the model was therefore accepted following previous studies (Hartig, 2022).

Data analysis and modelling were carried out with R version 4.2.2 through RStudio. Model selection and backward elimination of the covariates was undertaken based on the Akaike Information Criterion (AIC) values (Burnham & Anderson, 2002). We opted for backward elimination against other methods to avoid the removal of structural variables inherent to our study design such as "Year" or "Treatment" and to consider variations caused by exposure terms such as "Soaktime" at all times. We only included one interaction (Year x Treatment) because it forms the basis of the BACI comparison and to ensure we did not include interactions that may not be ecologically valid, spurious and nonlinear (Duncan & Kefford, 2021). Models were fitted through a log-link function with the "glmmTMB" (Magnusson et al., 2017). Model validation was carried out with 10,000 simulations using the function "simulateResiduals" from the DHARMa package. Residual assumptions were tested with the "DHARMa" package (Hartig, 2022). Interaction analysis of the models' outputs was conducted with two post-hoc methods: pairwise comparisons (estimated marginal means comparisons) and interaction contrasts (slope comparison) of the previous ones, both using the two controls

(natural reef and sand bottom sites). Both were carried out with the "emmeans" package (Lenth, 2023). Figures were produced using the "ggplot2" package (Wickham, 2016).

Results were interpreted using the Before After Control Impact (BACI) Framework (Figure 2). Four different scenarios can account for a positive effect of restoration (or another impact). The different scenarios include A) an increase of the response variable in restored (impact) sites (Figure 2A), B) a decrease in the response variable at the control site but remaining constant at a restored (impact) site (Figure 2B), C) an overall increase in the response variable with a steeper trend in restored (impact) sites (Figure 2C), or D) a general decrease in the response variable with less sharp slopes in restored (impact) sites (Figure 2D).

3 | RESULTS

3.1 | Model outputs

"Temperature" as an environmental explanatory variable exhibited different significant correlations across the species groups (Figure 3). BRUVS and UBRUVS models coincided in reporting significant correlations of temperature on all focal species except cod. Goldsinny wrasse, two-spotted goby, shore crab and flatfish abundances correlated positively with temperature while there was a negative, significant correlation in herring. Thus, wrasse, two-spotted goby, shore crab, and flatfish were more abundant in warmer water, whereas herring were more abundant in colder water. Cod abundance was the only species where the correlations with temperature reported by BRUVS and UBRUVS significantly differed, with significant negative and positive correlations reported, respectively (Figure 3).

The effect of "Macroalgae" was also consistent between BRUVS and UBRUVS, showing a significant positive correlation in two-spotted goby and a significant negative correlation in shore



FIGURE 3 Explanatory variables significance in the GLMMs. *Colour ramp uses green or red colours depending on the value of the estimate. If estimate>0, the colour ramp is green, showing a positive correlation with the response variable. If estimate<0, the colour ramp is red, representing negative correlation with the response variable. For example, in the intersection between goldsinny wrasse and temperature, dark green colour represents a significant positive correlation between the response variable (wrasse abundance) and the explanatory variable temperature. Blank squares represent covariates eliminated throughout the backwards elimination model selection process.

crab and flatfish. Thus, two-spotted goby abundance increased with more macroalgae, whereas shore crab and flatfish decreased with more macroalgae. In several UBRUVS models, "Macroalgae" was removed during the model selection process. In UBRUVS models, macroalgae had significant positive effects in two-spotted goby (Figure 3). Models revealed strong statistical evidence of the positive relationship between macroalgae and the abundances of cod, goldsinny wrasse, and two-spotted goby. This is in agreement with macroalgae being a key contributor to habitat structural complexity and abundances of reef associated species as well as biodiversity indicators.

To interpret the potential correlations of reef restoration and focal species abundance, the interactions between "Restored reef" and the years posterior to the restoration event, that is 2018 (four months after) and 2021 (three years after) were inspected.

In both BRUVS and UBRUVS models, abundances of Atlantic cod and goldsinny wrasse were positively correlated with the two restoration interactions (2018 and 2021) (Figure 3) (Annexes 4 and 5). The effect of the described interactions in the remaining species models was not as consistent across methods. In herring models there was only one significant negative correlation in the UBRUVS model with the interaction of restored reefs and 2021 (Figure 3). The twospotted goby models only revealed positive correlation with restoration interactions both four months and three years post restoration in UBRUVS (Figure 3). Only BRUVS shore crab models showed statistically significant positive correlations with the interaction of restored sites in 2018 and 2021. Flatfish models showed by both BRUVS and UBRUVS a significant negative correlation with restored reefs four months after restoration. After three years, only BRUVS reported negative correlation with the restoration interaction.

3.2 | Focal species abundance response

Significant absolute positive abundance changes in restored sites from 2017 to 2021 were only evident in goldsinny wrasse, according to both BRUVS and UBRUVS (Figures 4C and 5C). The goldsinny wrasse increase in restored sites highlights a correlation between goldsinny wrasse abundance and topographically complex habitats such as cobble reefs. In two-spotted goby, absolute significant positive changes from 2017 to 2018 were reported by both RUVS (Figures 4D and 5D). The rest of the species abundances in restored reef sites either underwent significant negative changes (Figures 4A, 4F and 5B) or they did not undergo any significant changes (Figures 4B, 4E and 5E). Cod abundance underwent a severe depletion at all sites (Figures 4A and 5A). These findings suggest an overall decrease in Atlantic cod abundance, caused by larger scale factors affecting the whole marine area rather than the reef deployment itself. Cobble reef restoration might have countered some of this detrimental development. Restored sites abundance comparisons in 2021 across sand and natural sites, revealed no significant differences with natural reef sites in all species (and sampling methods) (Figures 4 and 5) except in the UBRUVS cod model (Figure 5A). All coefficient tables are included in the supplementary material: Annexes 8 and 9.

3.3 | Restoration effects from interaction contrasts

The BACI framework (Figure 2) contains several scenarios that can be considered as positive restoration effects. Interaction contrasts (slope comparisons) were conducted using both control types, A) natural cobble reef sites (positive control) and restored sites, and B) sand (negative control) and restored sites in 2017 and 2021. Results of



Treatment + Sand + Natural_Reef + Restored_Reef

FIGURE 4 Estimated marginal means of the BRUVS species groups abundances (MaxN) with a confidence interval level of 0.95. Lines with circles (red) represent sand bottom sites (control), triangles (light blue) represent natural reef sites, and squares (dark blue) restored reef sites. Note that the Y-axis range was adjusted to the data for each panel. Estimated marginal means and confidence intervals are available in Annex 6. Dashed lines indicate the cobble reef restoration event

both BRUVS and UBRUVS slope comparisons showed significant positive effects of reef restoration in three out of the six focal species, when compared with natural reef sites, the species being: cod (BRUVS: p < 0.001, UBRUVS: p < 0.001), goldsinny wrasse (p < 0.001, p < 0.001), and two spotted goby (p = 0.0011, p = 0.0035) (Figure 6A and B). Positive restoration effects were also reported in the same species across BRUVS and UBRUVS models when compared with sand bottom sites: cod (p = 0.0016, p < 0.001), goldsinny wrasse (p = 0.0014, p < 0.001). Lastly, in two spotted goby, UBRUVS reported significant positive restoration effects (p = 0.0019) (Figure 6D), while BRUVS showed positive non-significant restoration effects (p = 0.3394) (Figure 6C).

Shore crab abundances revealed changes in abundance in response to restoration compared with natural reef sites (Figure 6A and B), although these results were not statistically significant (Annexes 10 and 11). When compared with sand bottom sites, only BRUVS data revealed significant positive restoration effects for shore crab (p < 0.001) (Figure 6C).

Results revealed negative restoration effects for two out of the six focal species in only one of each sampling method (i.e. BRUVS and UBRUVS). The response of flatfish to the restoration was inconsistent across sampling methods. BRUVS reported significant negative restoration effects both compared with natural reef sites (p < 0.001) and sand sites (p < 0.001) (Figure 6A and C). UBRUVS reported effects that were not statistically significant compared with natural reef sites (p = 0.6158) and sand sites (p = 0.2857) (Figure 6B and D). Herring responses to the restoration were also inconsistent among sampling methods. BRUVS did not report any significant response (p = 0.4094, p = 0.8469) (Figure 6A and C) while UBRUVS revealed a significant negative restoration response (Figure 6B and D).

These findings suggest that when examining the results through the BACI lens, statistical evidence of positive restoration effects was shown by both sampling methods in three out of the six focal species. Results showed two negative restoration effects in herring and flatfish. However, these effects were not consistent across sampling methods.

4 | DISCUSSION

To the best of our knowledge, this is the first study examining biological effects of cobble reef restoration. Across five years, this

WILEY 7 of 13



Treatment 🔶 Control 📥 Natural_Reef 🖶 Restored_Reef

FIGURE 5 Estimated marginal means of the UBRUVS species groups abundances (MaxN) with a confidence interval level of 0.95. Lines with circles (red) represent sand bottom sites (control), triangles (light blue) represent natural reef sites and squares (dark blue) restored reef sites. Note that the Y-axis range was adjusted to the data for each panel. Estimated marginal means and confidence intervals are available in Annex 7. Dashed lines indicate the cobble reef restoration event

study found that three out of six focal species were positively affected by the reef restoration. No species revealed negative restoration effects consistent across BRUVS and UBRUVS sampling methods. Our study shows that cobble reef restoration is feasible, and that mobile fauna is responding quickly to such restoration. Cobble reef restoration is therefore a viable tool for coastal restoration of complex habitats and the associated mobile species. This is useful information for ongoing and future nature protection and restoration programs such as the Natura 2000 program within the European Union and the sustainable development goals of the UN. Importantly, our study indicates that cobble extraction likely eliminates complex benthic habitats, with negative consequences for threatened species like the Atlantic cod.

Goldsinny wrasse and two-spotted goby showed the most evident positive effects of our cobble reef restoration. Goldsinny wrasse is often associated with rocky reefs (Almada et al., 1999; Støttrup et al., 2014) and shows high site fidelity (Cresci et al., 2022; Sayer et al., 1993) Thus, with increased cobble reef availability, our study showed clear positive abundance responses. Two-spotted goby, one of the most abundant demersal species of the Northeast Atlantic Ocean (Utne-Palm et al., 2015) relies on structurally complex habitats (Wilhelmsson et al., 2006) and also responded positively to the restoration.

According to trend comparison within the BACI framework (Figure 2D), Atlantic cod was also positively affected by the cobble reef restoration. The use of rocky reef by juvenile cod (Støttrup et al., 2017; Støttrup et al., 2019) as shelter and nursery grounds (Beisiegel et al., 2019) and an increased cod abundance after rocky reef deployment have previously been reported (Kristensen et al., 2017; Støttrup et al., 2014). In our study, cod abundance decreased from 2018 to 2021, across all treatments, showing that the decrease was unrelated to the restoration. Atlantic cod stocks are severely depleted (ICES, 2023a) partly due to eutrophication caused by nutrients from land (especially from the farming industry) leading to large demersal areas depleted of oxygen in the Western Baltic Sea (ICES, 2023b; Nikulina & Dullo, 2009; Svendsen et al., 2022). Other factors include past overfishing, unsustainable management of fisheries, and bottom trawling (Receveur et al., 2022) causing physical damage to the seabed. These factors have played important roles for the collapsing fisheries for Atlantic cod in the western Baltic Sea.



FIGURE 6 Interaction contrast ratios of natural reef sites and restored cobble reef sites in the years 2017 and 2021 of focal species group abundances from BRUVS (A) and UBRUVS (B). Interaction contrasts ratios of sand sites and restored cobble reef sites in the years 2017 and 2021 of focal species abundances from BRUVS (C) and UBRUVS (D). *Colour ramp being green or red depends on the ratio value. If ratio>1 colour ramp is green showing positive restoration effects, if ratio<1 colour ramp is red representing negative restoration effects. The higher the significance of the p-values the darker the colour in the scale. *p*-Values can be found in Annexes 10 and 11.

Based on our quantitative data covering several years, our analyses suggest that the deployed cobble reefs are supporting local Atlantic cod abundances through the provision of habitat. Specifically, our BACI analyses revealed that Atlantic cod responded positively to the reef restoration, which partly countered the negative stock developments across the years. Therefore, cobble reefs represent a vital habitat for Atlantic cod. Importantly, cobble is still extracted in many marine areas together with other sediment types. Our research shows that ongoing cobble reef extractions could exacerbate Atlantic cod declines in the local areas.

We expected herring to use the restored cobble reefs for spawning, and therefore, we hypothesized an increased herring abundance in the restored cobble reefs sites. Sampling was consequently conducted during the spring (April–May) across the years. Contrary to our hypothesis, herring abundance was not positively affected by reef restoration, and neither herring spawning activity nor eggs were identified in the recordings. BRUVS reported no significant restoration effects while UBRUVS reported significant negative restoration effects. The factors causing this apparent discrepancy between sampling methods might be related to either potential bait effect or the increased exposure (soaktime) in BRUVS recordings.

Several factors may explain the absent effect of reef restoration for herring spawning including 1) depleted herring abundance in the area (ICES, 2023b), 2) herring preference for specific vegetation (e.g., macroalgae species) for spawning and egg deposition, 3) herring preference for suitable spawning substrate at a water depth different from 6–7 m where the cobble reefs were deployed, 4) strong site fidelity of herring spawning across generations, and 5) nocturnal spawning activities undetected by our diurnal underwater footage. The vegetation growing on the cobble reefs was not identified, making it impossible to scrutinize what vegetation herring possibly prefer or avoid. Further studies are needed to reveal how herring spawning habitats are restored.

Shore crab response to reef restoration compared with sand sites was positive, according only to BRUVS. However, trend comparison with natural reef sites as a control site showed unclear effects. These findings might indicate a potential BRUV attraction effect or suggest that shore crab might be an early colonizer of restored reefs that later undergo density-driven intraspecific spatial competition (Moksnes, 2004).

Flatfish response to our cobble reef restoration, compared with both sand and natural reef sites, was negative according to the BRUVS data (Figure 6A and C). Flatfish more often associate with sand and gravel bottom substrates over rocky substrates (Florin et al., 2009). Cobble reef restoration caused a decline in soft bottom availability, negatively affecting flatfish (Wilms et al., 2021) although this was only indicated by our BRUVS data. Interestingly, UBRUVS data showed no negative restoration effects (Figure 6B and D). Locally, flatfish may partially benefit from reef restoration because of reef spillover effect along reef edges, providing additional foraging resources (Posey & Ambrose, 1994; Wilms et al., 2021). Further studies are needed to better understand reef spillover effects for various species. In temperate reefs, macroalgae are often the main habitat engineer species, increasing habitat complexity and ecosystem functioning (Geist & Hawkins, 2016). Reef fish species often depend on the structural complexity provided by canopy forming macroalgae, which provides refuge, especially for juvenile fishes (Eggertsen et al., 2017; Fulton et al., 2020), including Atlantic cod (Lorentsen et al., 2010). Macroalgae require hard substrata to attach to (Blanfuné et al., 2019), and according to Geist & Hawkins (2016), suitable areas for active restoration in coastal systems include ecosystems, which have lost their biogenic structure. In our models, macroalgae was a significant positive predictor in cod, two-spotted goby, and goldsinny wrasse (demersal species showing the strongest positive restoration responses) (Figure 3; Figure 6). Thus, restoring macroalgae forests will likely support and increase the abundances of these species.

Temperature was overall a significant positive predictor of fish abundances (Figure 3), however, herring abundance correlated negatively with temperature, revealed by both BRUVS and UBRUVS data. In the Western Baltic Sea, stocks of cod and herring are below the safe biological limits, caused mainly by overfishing, changes in salinity and declining oxygen concentration in the water (ICES, 2023a). Water temperature will likely vary with climate change (ICES, 2023a). Herring migration and reproduction patterns are shaped by water temperature. Increasing temperatures advance the arrival of Western Baltic Spring Spawning herring stock to the Kiel Fjord spawning grounds by up to 3.5 months (Ory et al., 2022). Warmer waters generally have also been linked with decreased herring reproductive success (Polte et al., 2021). The absent response of herring to cobble reef restoration might have been partially related to increasing water temperature. Data collection started in early April during all the sampling years (2017, 2018 and 2021). The sampling time was scheduled to detect Atlantic herring spring spawning, however, it remains possible that the fish were spawning earlier, perhaps in March, when no RUVS were in the water.

The interpretation of the data from this study might have been limited by a series of factors. Adding measures of oxygen concentration in water as an explanatory variable in the models might have been useful to account for the effects of external environmental factors that could have affected abundance shifts in the study area. Year to year comparisons in temperature records would have been a reasonable way to further describe the role of temperature for the changes in abundance of Atlantic cod and Atlantic herring in the study area. To better identify Atlantic herring spawning in future studies further sampling techniques should be considered, including sampling earlier in the spring, nocturnal sampling, e.g. visual census or searches for herring eggs (Polte et al., 2021).

Both BRUVS and UBRUVS successfully identified three out of six focal species that were positively affected by reef restoration as well as detecting the same effects of temperature (with the exception of Atlantic cod) and macroalgae. However, no negative abundance effects were consistently detected via both sampling methods, indicating limited, but uncertain, negative effects of the cobble reef restoration. BRUVS might be the right technique to reduce sampling effort without obscuring correlations of fish abundance and reef restoration. Alternatively, UBRUVS might be the better choice for detecting restoration effects as well as species natural behaviour.

4.1 | Implications for conservation

Findings of this study provide new evidence of the effects of habitat restoration in temperate latitudes on fish abundances. By providing additional availability of structurally complex habitat (i.e., reef area), three species, Atlantic cod, goldsinny wrasse and two-spotted goby, were positively affected by the restoration.

Our findings underline the significant contribution of canopy forming macroalgae to habitat structural complexity and the positive effects on the fish community. The species that benefited the most from the restoration are the ones that positively correlated with the explanatory variable "macroalgae" in the models. We advise that future reef restoration projects enhance the development of the canopy forming macroalgae community, but also conduct continued monitoring across multiple years, because macroalgae development requires time.

Rocky reef restoration effects might be altered by larger scale environmental factors such as temperature or oxygen concentration and/or human pressures (e.g.,fishing). Therefore, by adopting a holistic, ecosystem-based approach for the management of coastal areas, fish populations might benefit additionally from reef restoration.

Reef restoration contributes to the implementation of the UN Sustainable Development Goal 14, by restoring habitats for various fish species. The results of this study underline the ecological importance of cobble reefs and their potential to become useful conservation tools in areas that have been subjected to cobble extraction. However, in many dedicated areas, ongoing extraction of cobble, and other seabed substrates, is still impacting coastal ecosystems and the associated marine life (e.g., fish abundance). We warn against this anthropogenic impact in vulnerable areas and emphasize the importance of ecosystem restoration efforts.

ACKNOWLEDGEMENTS

We thank all the fieldwork participants including Carl Baden, Charlotte Bourdon, Florence Ghestem, Johan Mølgård, Gesine Ramm, Nicole Rhodes and Océane Pirou who contributed to the data collection. We would also like to thank chief journal editor Heidi Burdett for support and the reviewers, including Pete Nelson, for their constructive inputs that improved this paper. This research was funded by the European Union, the European Maritime and Fisheries Fund, and the Danish Fisheries Agency. Project numbers are 33113-B-16-059 and 33113-B-19-142. The funding supports sustainable fishing practices and coastal communities in terms of diversifying economies, and improving quality of life along European coasts. The project was also funded by the Danish Rod and Net Fish License Funds, the Velux Foundation, and the energy company Vattenfall. All funding sources are greatly appreciated.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in Harvard Dataverse at https://doi.org/10.7910/DVN/L3WZE9.

ETHICS AND PERMIT APPROVAL STATEMENT

No ethical approvals or permits were required for this study.

ORCID

E. Casabona D https://orcid.org/0009-0001-6252-1974

REFERENCES

- Airoldi, L., Balata, D. & Beck, M.W. (2008). The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366(1–2), 8–15. https://doi.org/10.1016/j.jembe.2008.07.034
- Almada, V.C., Henriques, M.I.G.U.E.L. & Gonçalves, E.J. (1999). Ecology and behaviour of reef fishes in the temperate North-Eastern Atlantic and adjacent waters. Behaviour and conservation of littoral fishes. Lisboa: ISPA, pp. 33–69.
- Arthington, A.H., Dulvy, N.K., Gladstone, W. & Winfield, I.J. (2016). Fish conservation in freshwater and marine realms: status, threats and management. Aquatic Conservation: Marine and Freshwater Ecosystems, 26(5), 838–857. https://doi.org/10.1002/aqc.2712
- Beisiegel, K., Tauber, F., Gogina, M., Zettler, M.L. & Darr, A. (2019). The potential exceptional role of a small Baltic boulder reef as a solitary habitat in a sea of mud. Aquatic Conservation: Marine and Freshwater Ecosystems, 29(2), 321–328. https://doi.org/10.1002/aqc.2994
- Bergström, L., Sundqvist, F. & Bergström, U. (2013). Effects of an offshore wind farm on temporal and spatial patterns in the demersal fish community. *Marine Ecology Progress Series*, 485, 199–210.https://doi. org/10.3354/meps10344
- Blanfuné, A., Boudouresque, C.F., Verlaque, M. & Thibaut, T. (2019). The ups and downs of a canopy-forming seaweed over a span of more than one century. *Scientific Reports*, 9(1), 5250. https://doi.org/10. 1038/s41598-019-41676-2
- Bohnsack, J.A. (1989). Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? *Bulletin of Marine Science*, 44(2), 631–645.
- Bulleri, F. & Chapman, M.G. (2010). The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology*, 47(1), 26–35. https://doi.org/10.1111/j.1365-2664. 2009.01751.x
- Burnham, K.P. & Anderson, D.R. (2002). Model selection and inference: a practical information-theoretic approach. 2nd edition: Springer-Verlag.
- Cappo, M., Speare, P. & De'ath, G. (2004). Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the great barrier reef Marine Park. *Journal of Experimental Marine Biology and Ecology*, 302(2), 123–152. https://doi.org/10.1016/j.jembe.2003. 10.006
- Christie, H., Norderhaug, K.M. & Fredriksen, S. (2009). Macrophytes as habitat for fauna. *Marine Ecology Progress Series*, 396, 221–233. https://doi.org/10.3354/meps08351
- Cresci, A., Larsen, T., Halvorsen, K.T., Durif, C.M., Bjelland, R., Browman, H.I. et al. (2022). Goldsinny wrasse (Ctenolabrus rupestris) have a sex-dependent magnetic compass for maintaining site fidelity. *Fisheries Oceanography*, 31(2), 164–171. https://doi.org/10.1111/fog. 12569

- Cresson, P., Le Direach, L., Rouanet, E., Goberville, E., Astruch, P., Ourgaud, M. et al. (2019). Functional traits unravel temporal changes in fish biomass production on artificial reefs. *Marine Environmental Research*, 145, 137–146. https://doi.org/10.1016/j.marenvres.2019. 02.018
- Dafforn, K.A., Glasby, T.M., Airoldi, L., Rivero, N.K., Mayer-Pinto, M. & Johnston, E.L. (2015). Marine urbanization: an ecological framework for designing multifunctional artificial structures. *Frontiers in Ecology* and the Environment, 13(2), 82–90. https://doi.org/10.1890/140050
- Dailianis, T., Smith, C.J., Papadopoulou, N., Gerovasileiou, V., Sevastou, K., Bekkby, T. et al. (2018). Human activities and resultant pressures on key European marine habitats: an analysis of mapped resources. *Marine Policy*, 98, 1–10. https://doi.org/10.1016/j.marpol.2018. 08.038
- De Groot, S.J. (1980). The consequences of marine gravel extraction on the spawning of herring, Clupea harengus Linné. *Journal of Fish Biology*, 16(6), 605–611.
- Duarte, C.M., Agusti, S., Barbier, E., Britten, G.L., Castilla, J.C., Gattuso, J.P. et al. (2020). Rebuilding marine life. *Nature*, 580(7801), 39–51. https://doi.org/10.1038/s41586-020-2146-7
- Duncan, R.P. & Kefford, B.J. (2021). Interactions in statistical models: three things to know. Methods in Ecology and Evolution, 12(12), 2287– 2297. https://doi.org/10.1111/2041-210X.13714
- Eggertsen, L., Ferreira, C.E., Fontoura, L., Kautsky, N., Gullström, M. & Berkström, C. (2017). Seaweed beds support more juvenile reef fish than seagrass beds in a South-Western Atlantic tropical seascape. *Estuarine, Coastal and Shelf Science*, 196, 97–108. https://doi.org/10. 1016/j.ecss.2017.06.041
- Elliott, M., Mander, L., Mazik, K., Simenstad, C., Valesini, F., Whitfield, A. et al. (2016). Ecoengineering with ecohydrology: successes and failures in estuarine restoration. *Estuarine, Coastal and Shelf Science*, 176, 12–35. https://doi.org/10.1016/j.ecss.2016.04.003
- European Commission. (2013). Interpretation manual of European Union habitats. EUR 28. Appendix 1- marine habitat types definitions. Brussels, BE: European Commission.
- Flávio, H., Seitz, R., Eggleston, D., Svendsen, J.C. & Støttrup, J. (2023). Hard-bottom habitats support commercially important fish species: a systematic review for the North Atlantic Ocean and Baltic Sea. *PeerJ*, 11, e14681. https://doi.org/10.7717/peerj.14681
- Florin, A.B., Sundblad, G. & Bergström, U. (2009). Characterisation of juvenile flatfish habitats in the Baltic Sea. *Estuarine, Coastal and Shelf Science*, 82(2), 294–300. https://doi.org/10.1016/j.ecss.2009. 01.012
- Folpp, H., Lowry, M., Gregson, M. & Suthers, I.M. (2011). Colonization and community development of fish assemblages associated with estuarine artificial reefs. *Brazilian Journal of Oceanography*, 59(spe1), 55–67. https://doi.org/10.1590/s1679-87592011000500008
- Folpp, H.R., Schilling, H.T., Clark, G.F., Lowry, M.B., Maslen, B., Gregson, M. et al. (2020). Artificial reefs increase fish abundance in habitat-limited estuaries. *Journal of Applied Ecology*, 57(9), 1752–1761. https://doi.org/10.1111/1365-2664.13666
- Frost, M. & Diele, K. (2022). Essential spawning grounds of Scottish herring: current knowledge and future challenges. *Reviews in Fish Biology and Fisheries*, 32(3), 721–744. https://doi.org/10.1007/ s11160-022-09703-0
- Fulton, C.J., Berkström, C., Wilson, S.K., Abesamis, R.A., Bradley, M., Åkerlund, C. et al. (2020). Macroalgal meadow habitats support fish and fisheries in diverse tropical seascapes. *Fish and Fisheries*, 21(4), 700–717. https://doi.org/10.1111/faf.12455
- Geist, J. & Hawkins, S.J. (2016). Habitat recovery and restoration in aquatic ecosystems: current progress and future challenges. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26(5), 942–962. https://doi.org/10.1002/aqc.2702
- Gotceitas, V., Fraser, S. & Brown, J.A. (1995). Habitat use by juvenile Atlantic cod (Gadus morhua) in the presence of an actively foraging

CASABONA ET AL.

0990755, 2024.8, Downloaded from https://onlinelibrary.wiley.com/doi/10.1002/aqc.4216 by Dalhousie University Dalhousie, Wiley Online Library on [23/12/2024]. See the Terms

and Conditions

(http:

elibrary. wiley

-and

on Wiley Online Library for rules

of use; OA

are gov

emed by the applicable Creative Commons

12 of 13 WILEY-

and non-foraging predator. *Marine Biology*, 123(3), 421-430. https://doi.org/10.1007/BF00349220

- Halpern, B.S., Frazier, M., Afflerbach, J., Lowndes, J.S., Micheli, F., O'Hara, C. et al. (2019). Recent pace of change in human impact on the world's ocean. *Scientific Reports*, 9(1), 11609. https://doi.org/10. 1038/s41598-019-47201-9
- Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., d'Agrosa, C. et al. (2008). A global map of human impact on marine ecosystems. *Science*, 319(5865), 948–952. https://doi.org/10.1126/ science.1149345
- Hartig, F. (2022). DHARMa: Residual Diagnostics for Hierarchical (Multi-Level/Mixed) Regression Models. R Package Version 0.4.6
- Hastings, R., Cummins, V. & Holloway, P. (2020). Assessing the impact of physical and anthropogenic environmental factors in determining the habitat suitability of seagrass ecosystems. *Sustainability*, 12(20), 8302. https://doi.org/10.3390/su12208302
- Helmig, S.A., Nielsen, M.M. & Petersen, J.K. (2020). Andre presfaktorer end næringsstoffer og klimaforandringer – vurdering af omfanget af stenfiskeri i kyst-nære marine områder DTU Aqua-rapport nr. 360-2020.: Institut for Akvatiske Ressourcer, Danmarks Tekniske Universitet. pp. 24.
- Holloway, P. & Field, R. (2020). Can rock-rubble groynes support similar intertidal ecological communities to natural rocky shores? *Land*, 9(5), 131. https://doi.org/10.3390/land9050131
- Hunter, W.R. & Sayer, M.D.J. (2009). The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science*, 66(4), 691–698. https:// doi.org/10.1093/icesjms/fsp058
- ICES. (2023a). Cod (Gadus morhua) in subdivisions 22-24, western Baltic stock (western Baltic Sea): ICES Advice: Recurrent Advice. Report. https://doi.org/10.17895/ices.advice.21820494.v1
- ICES. (2023b). Stock Annex: Herring (Clupea harengus) in subdivisions 20– 24, spring spawners (Skagerrak, Kattegat, and western Baltic). Browse ICES Content by Type. Report. https://doi.org/10.17895/ices.pub. 23898678. v1
- Kain, M.P., Bolker, B.M. & McCoy, M.W. (2015). A practical guide and power analysis for GLMMs: detecting among treatment variation in random effects. *PeerJ*, 3, e1226. https://doi.org/10.7717/peerj.1226
- Katara, I., Peden, W.J., Bannister, H., Ribeiro, J., Fronkova, L., Scougal, C. et al. (2021). Conservation hotspots for fish habitats: a case study from English and Welsh waters. *Regional Studies in Marine Science*, 44, 101745. https://doi.org/10.1016/j.rsma.2021.101745
- Kristensen, L.D., Støttrup, J.G., Svendsen, J.C., Stenberg, C., Højbjerg Hansen, O.K. & Grønkjær, P. (2017). Behavioural changes of Atlantic cod (Gadus morhua) after marine boulder reef restoration: implications for coastal habitat management and natura 2000 areas. *Fisheries Management and Ecology*, 24(5), 353–360. https://doi.org/10.1111/ fme.12235
- Langlois, T., Williams, J., Monk, J., Bouchet, P., Currey, L., Goetze, J., Harasti D, Huveneers C, lerodiaconou D, Malcolm H & Whitmore, S. (2018). Marine sampling field manual for benthic stereo BRUVS (baited remote underwater videos). In: Field Manuals for Marine Sampling to Monitor Australian Waters.
- Lenth R (2023). _emmeans: Estimated Marginal Means, aka Least-Squares Means_. R Package Version 1.8.4–1. Available from: <<u>https://CRAN.</u> R-project.org/package=emmeans> https://doi.org/10.1080/ 00031305.1980.10483031
- Lewis, D.M., Durham, K.E., Walters, L.J. & Cook, G.S. (2021). A resident fish guild as a higher trophic level indicator of oyster reef restoration success. *Sustainability*, 13(23), 13004. https://doi.org/10.3390/ su132313004
- Lorentsen, S.H., Sjøtun, K. & Grémillet, D. (2010). Multi-trophic consequences of kelp harvest. *Biological Conservation*, 143(9), 2054– 2062. https://doi.org/10.1016/j.biocon.2010.05.013

- Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C. et al. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science*, 312(5781), 1806–1809. https:// doi.org/10.1126/science.1128035
- Lowry, M.B., Glasby, T.M., Boys, C.A., Folpp, H., Suthers, I. & Gregson, M. (2014). Response of fish communities to the deployment of estuarine artificial reefs for fisheries enhancement. *Fisheries Management and Ecology*, 21(1), 42–56. https://doi.org/10.1111/fme.12048
- Magnusson, A., Skaug, H., Nielsen, A., Berg, C., Kristensen, K., Maechler, M., van Bentham K, Bolker B, Brooks M, & Brooks, M. M. (2017). Package 'glmmtmb'. R Package Version 0.2. 0, 25.
- Maravelias, C.D., Reid, D.G. & Swartzman, G. (2000). Seabed substrate, water depth and zooplankton as determinants of the prespawning spatial aggregation of North Atlantic herring. *Marine Ecology Progress Series*, 195, 249–259. https://doi.org/10.3354/meps195249
- Moksnes, P.O. (2002). The relative importance of habitat-specific settlement, predation and juvenile dispersal for distribution and abundance of young juvenile shore crabs Carcinus maenas L. *Journal of Experimental Marine Biology and Ecology*, 271(1), 41–73. https://doi. org/10.1016/S0022-0981(02)00041-2
- Moksnes, P.O. (2004). Interference competition for space in nursery habitats: density-dependent effects on growth and dispersal in juvenile shore crabs Carcinus maenas. *Marine Ecology Progress Series*, 281, 181–191. https://doi.org/10.3354/meps281181
- Nikulina, A. & Dullo, W.C. (2009). Eutrophication and heavy metal pollution in the Flensburg Fjord: a reassessment after 30 years. *Marine Pollution Bulletin*, 58(6), 905–915. https://doi.org/10.1016/j. marpolbul.2009.01.017
- Ory, N. C., Gröger, J. P., Lehmann, A., Mittermayer, F., & Clemmesen, C. (2022). Early arrival of spring-Spawner Atlantic herring at their spawning ground in the Kiel Fjord, Western Baltic, Relates to Increasing Sea Water Temperature. Western Baltic, Relates to Increasing Sea Water Temperature. https://doi.org/10.2139/ssrn. 4164412
- Parsons, D.F., Suthers, I.M., Cruz, D.O. & Smith, J.A. (2016). Effects of habitat on fish abundance and species composition on temperate rocky reefs. *Marine Ecology Progress Series*, 561, 155–171. https://doi. org/10.3354/meps11927
- Perkol-Finkel, S., Ferrario, F., Nicotera, V. & Airoldi, L. (2012). Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology*, 49(6), 1457–1466. https://doi.org/10.1111/j.1365-2664. 2012.02204.x
- Polte, P., Gröhsler, T., Kotterba, P., Von Nordheim, L., Moll, D., Santos, J. et al. (2021). Reduced reproductive success of western Baltic herring (Clupea harengus) as a response to warming winters. *Frontiers in Marine Science*, 8, 10. https://doi.org/10.3389/fmars.2021.589242
- Posey, M.H. & Ambrose, W.G. (1994). Effects of proximity to an offshore hard-bottom reef on infaunal abundances. *Marine Biology*, 118, 745– 753. https://doi.org/10.1007/BF00347524
- Receveur, A., Bleil, M., Funk, S., Stötera, S., Gräwe, U., Naumann, M. et al. (2022). Western Baltic cod in distress: decline in energy reserves since 1977. ICES Journal of Marine Science, 79(4), 1187–1201. https://doi. org/10.1093/icesjms/fsac042
- Rhodes, N., Wilms, T., Baktoft, H., Ramm, G., Bertelsen, J.L., Flávio, H. et al. (2020). Comparing methodologies in marine habitat monitoring research: an assessment of species-habitat relationships as revealed by baited and unbaited remote underwater video systems. *Journal of Experimental Marine Biology and Ecology*, 526, 151315. https://doi.org/ 10.1016/j.jembe.2020.151315
- Rouse, S., Porter, J.S. & Wilding, T.A. (2020). Artificial reef design affects benthic secondary productivity and provision of functional habitat. *Ecology and Evolution*, 10(4), 2122–2130. https://doi.org/10.1002/ ece3.6047

- Sayer, M.D.J., Gibson, R.N. & Atkinson, R.J.A. (1993). Distribution and density of populations of goldsinny wrasse (Ctenolabrus rupestris) on the west coast of Scotland. *Journal of Fish Biology*, 43(sA), 157–167. https://doi.org/10.1111/j.1095-8649.1993.tb01185.x
- Schwartzbach, A., Behrens, J.W. & Svendsen, J.C. (2020). Atlantic cod Gadus morhua save energy on stone reefs: implications for the attraction versus production debate in relation to reefs. *Marine Ecology Progress Series*, 635, 81–87. https://doi.org/10.3354/ meps13192
- Seaman, W. (2007). Artificial habitats and the restoration of degraded marine ecosystems and fisheries. In: Biodiversity in enclosed seas and artificial marine habitats: proceedings of the 39th European marine biology symposium, held in Genoa, Italy, 21-24 July 2004: Springer Netherlands, pp. 143-155. https://doi.org/10.1007/s10750-006-0457-9
- Silliman, B.R., Hensel, M.J., Gibert, J.P., Daleo, P., Smith, C.S., Wieczynski, D.J. et al. (2024). Harnessing ecological theory to enhance ecosystem restoration. *Current Biology*, 34(9), R418–R434. https://doi. org/10.1016/j.cub.2024.03.043
- Singh, V., Shukla, S. & Singh, A. (2021). The principal factors responsible for biodiversity loss. Open Journal of Plant Science, 6(1), 011–014. https://doi.org/10.17352/ojps.000026
- Smith, E.P. (2002). BACI design. Encyclopedia of Environmetrics, 1, 141–148. https://doi.org/10.1002/9780470057339.vab001.pub2
- Smith, T.M., Jenkins, G.P. & Hutchinson, N. (2012). Seagrass edge effects on fish assemblages in deep and shallow habitats. *Estuarine, Coastal and Shelf Science*, 115, 291–299. https://doi.org/10.1016/j.ecss.2012.09.013
- Stenberg, C. & Kristensen, L.D. (2015). Stenrev som gyde-og opvækstområde for Fisk (Revfisk). DTU Aqua.
- Støttrup, J.G., Dahl, K., Niemann, S., Stenberg, C., Reker, J., Stamphøj, E.M. et al. (2017). Restoration of a boulder reef in temperate waters: strategy, methodology and lessons learnt. *Ecological Engineering*, 102, 468–482. https://doi.org/10.1016/j.ecoleng.2017.02.058
- Støttrup, J.G., Kokkalis, A., Brown, E. J., Vastenhoud, B., Ferreira, S., Olsen, J., & Dinesen, G. E. (2019). Essential fish habitats for commercially important marine species in the inner Danish waters.
- Støttrup, J.G., Stenberg, C., Dahl, K., Kristensen, L.D. & Richardson, K. (2014). Restoration of a temperate reef: effects on the fish community. *Open Journal of Ecology*, 4(16), 1045. https://doi.org/10. 4236/oje.2014.416086
- Svendsen, J. C., Kruse, B. M., Wilms, T., Dahl, K., Buur, H., Andersen, O. G., Bertelsen JL & Kindt-Larsen, L. (2022). The importance of reef habitats for fish, harbor porpoise and fisheries management.
- Trebilco, R., Dulvy, N.K., Stewart, H. & Salomon, A.K. (2015). The role of habitat complexity in shaping the size structure of a temperate reef

fish community. Marine Ecology Progress Series, 532, 197–211. https://doi.org/10.3354/meps11330

- Utne-Palm, A.C., Eduard, K., Jensen, K.H., Mayer, I. & Jakobsen, P.J. (2015). Size dependent male reproductive tactic in the two-spotted goby (Gobiusculus flavescens). *PLoS ONE*, 10(12), e0143487. https:// doi.org/10.1371/journal.pone.0143487
- Von Nordheim, L., Kotterba, P., Moll, D. & Polte, P. (2018). Impact of spawning substrate complexity on egg survival of Atlantic herring (Clupea harengus, L.) in the Baltic Sea. *Estuaries and Coasts*, 41(2), 549–559. https://doi.org/10.1007/s12237-017-0283-5
- Wickham, H. (2016). ggplot2: elegant graphics for data analysis. New York: Springer-Verlag.
- Wilhelmsson, D., Öhman, M.C., Ståhl, H. & Shlesinger, Y. (1998). Artificial reefs and dive tourism in Eilat, Israel. Ambio, 27(8), 764–766.
- Wilhelmsson, D., Yahya, S.A. & Öhman, M.C. (2006). Effects of high-relief structures on cold temperate fish assemblages: a field experiment. *Marine Biology Research*, 2(2), 136–147. https://doi.org/10.1080/ 17451000600684359
- Wilms, T.J., Norðfoss, P.H., Baktoft, H., Støttrup, J.G., Kruse, B.M. & Svendsen, J.C. (2021). Restoring marine ecosystems: spatial reef configuration triggers taxon-specific responses among early colonizers. *Journal of Applied Ecology*, 58(12), 2936–2950. https://doi. org/10.1111/1365-2664.14014
- Yan, H.F., Kyne, P.M., Jabado, R.W., Leeney, R.H., Davidson, L.N., Derrick, D.H. et al. (2021). Overfishing and habitat loss drive range contraction of iconic marine fishes to near extinction. Science. *Advances*, 7(7), eabb6026. https://doi.org/10.1126/sciadv.abb6026
- Zuur, A.F. & Ieno, E.N. (2016). A protocol for conducting and presenting results of regression-type analyses. *Methods in Ecology and Evolution*, 7(6), 636–645. https://doi.org/10.1111/2041-210X.12577

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Casabona, E., Wilms, T., Moltesen, M., Bertelsen, J.L., Kruse, B.M., Flávio, H. et al. (2024). Cobble reef restoration in the Baltic Sea: Implications for life below water. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 34(8), 34(8), e4216. <u>https://doi.org/10.1002/aqc.4216</u>