SEAFLOOR HETEROGENEITY: ARTIFICIAL STRUCTURES AND MARINE ECOSYSTEM DYNAMICS

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and Victoria Louise Georgia Todd

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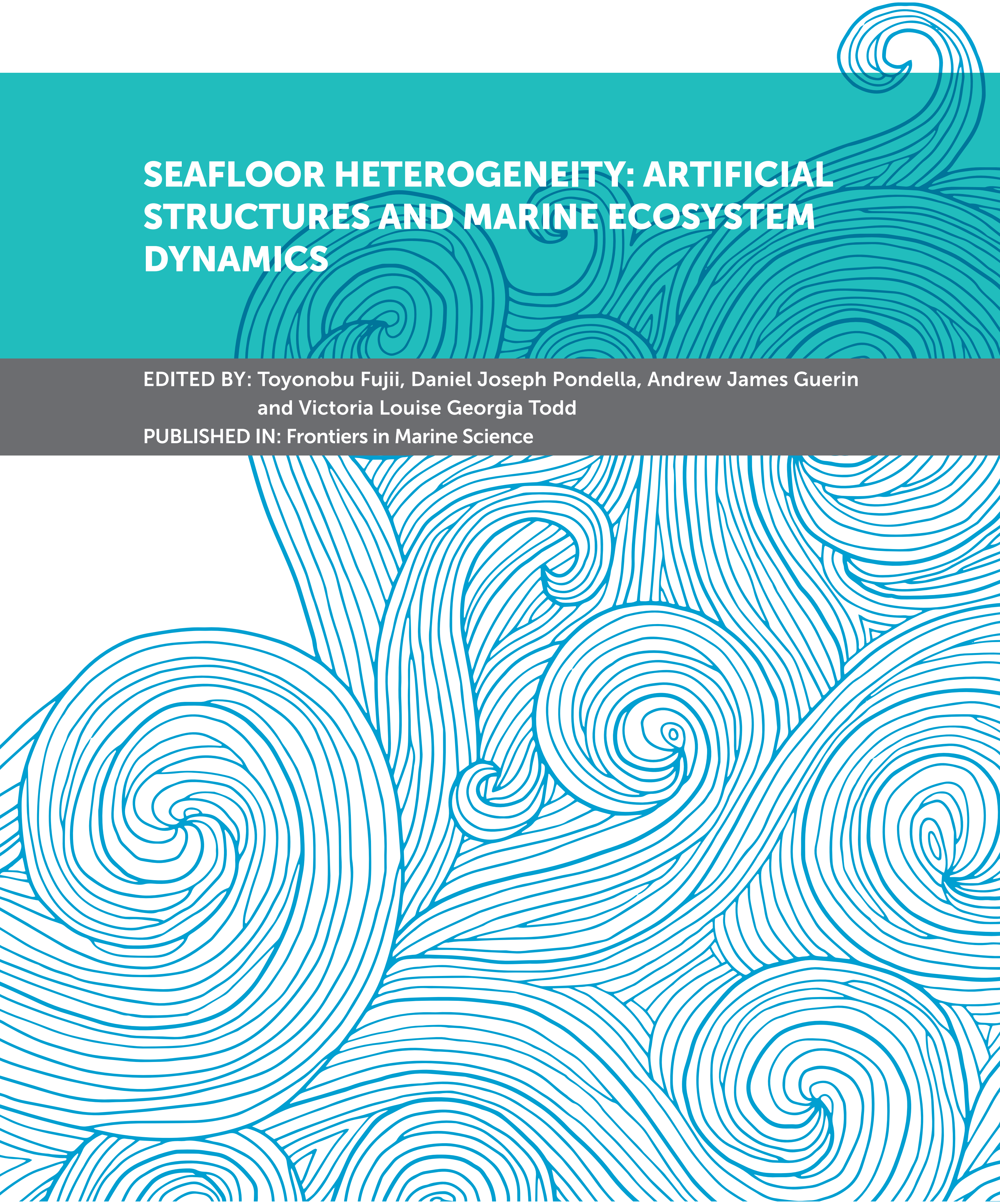

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SEAFLOOR HETEROGENEITY: ARTIFICIAL STRUCTURES AND MARINE ECOSYSTEM DYNAMICS

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Editorial: Seafloor Heterogeneity: Artificial Structures and Marine Ecosystem Dynamics

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Editorial on the Research Topic

Seafloor Heterogeneity: Artificial Structures and Marine Ecosystem Dynamics

Some of the most productives and biodiverse communities occur on "reefs" (Birkeland, 2015). Many species benefit from physical presence of habitat-forming reefs which provide complex three-dimensional hard substrates and a greater number of ecological niches (Loke et al., 2015). Although reefs are often exemplified by "corals," they also include other seafloor features such as biogenic substrates, natural bedrock, and man-made sub-sea structures (Steimle and Zetlin, 2000). Installation of sub-sea infrastructure is often considered to have negative impacts on surrounding marine ecosystems (Halpern et al., 2008; Benn et al., 2010; Bullieri and Chapman, 2010), although some studies show that such structures can also have beneficial effects by acting as "artificial reefs" (Gass and Roberts, 2006; Claisse et al., 2014).

Marine ecosystems are changing at alarming rates as a result of increasing anthropogenic influences (Halpern et al., 2008; McCauley et al., 2015; Duarte et al., 2020), and artificial structures are becoming ubiquitous. The sphere of influence, and effects of these artificial habitats on marine ecosystem dynamics, are poorly understood. This Research Topic assembles 11 articles investigating relationships between marine ecosystem dynamics and various types of anthropogenic structures globally. Here we present an overview of these contributions and highlight emerging views and future directions in this field.

ARTIFICIAL REEFS FOR ECOLOGICAL ENHANCEMENT

Callaway studied fauna inhabiting interstitial spaces within artificial reef units, built from rock or bivalve shell material, on an intertidal sand flat in Swansea Bay, UK. The reef units hosted greater biodiversity than nearby sediment, and the volume of interstitial space influenced species richness and community composition. Lohrer et al. manipulated artificial patches across experimental sites by inserting pinnid bivalve mimics into the seabed to observe the response of post-settlement stage snapper in Mahurangi Harbor, New Zealand. They showed that access to seafloor features and abundance of zooplankton are of primary importance to the snappers.

COASTAL FEATURES RELATING TO AQUACULTURE OPERATIONS AND RECREATIONAL ACTIVITIES

Fujii et al. investigated dynamics of benthic macrofaunal communities in relation to changes in environmental factors, including the re-building of coastal aquaculture facilities in Onagawa Bay, Japan, after the 2011 Great East Japan Earthquake and tsunami. Coastal aquaculture facilities influenced the occurrence of benthic macrofaunal communities and facilitated recovery of seafloor biota at ecosystem scales. Macolino et al. investigated effects of boat moorings on sediment infauna in Sydney Harbor, Australia, and whether current impact assessment methodologies have sufficient sensitivity to detect such effects. Fine-scale effects of boating infrastructure were detected when considering distance to moorings; however, comparisons at large scales failed to detect ecological change, underlining the importance of sampling at multiple scales during impact assessments.

EMERGING TECHNIQUES FOR MARINE HABITAT ASSESSMENT AND ECOSYSTEM RESTORATION

Zellmer et al. used stacked-species distribution models (s-SDMs) to identify optimal regions for restoration throughout the Southern California Bight, US. Using 21 ecologically important taxa and overlaying the s-SDMs with geospatial layers, they identified optimal areas for restoration. They also found that many man-made reefs in the area were placed in non-optimal locales, highlighting the need to evaluate locations for future restoration activities. van Elden et al. reviewed the role that decommissioned oil and gas platforms play in their environment. Traditionally these structures are removed once decommissioned, but often this is no longer technically or economically feasible. These offshore installations are not directly comparable in many instances to natural ecosystems and, as such, they propose evaluating them as "novel ecosystems" facilitating appropriate assessment and decision-making processes.

ARCHIVAL UNDERWATER IMAGERY FOR USE IN THE ASSESSMENT OF OFFSHORE OIL AND GAS INFRASTRUCTURE

Thomson et al. used ROV-inspection footage to characterize the sessile invertebrates and fishes associated with an oil platform on the North West Shelf of Australia. Depth was a major driver of invertebrate assemblages; the highest densities of commercial species occurred around intermediate depths where small baitfish were abundant, suggesting that mid-depth platform sections had high habitat value. Rouse et al. examined northern North Sea inspection footage to assess abundance of organisms on and around offshore pipelines. They observed almost 60 taxa, including 12 that represented "features of conservation importance," demonstrating that even pipelines can have reef effects. Gates et al. reported results from visual inspection and physical sampling of an offshore structure decommissioned from an oil field in the North East Atlantic. They showed that structures enhanced the biomass of epifauna which, in turn, supported diverse associated macrofauna, providing a food source for motile invertebrates and fishes in an area where background hard substratum had been lost through the impacts of drilling. Todd et al. examined commercial ROV/diver imagery from global industrial partners and YouTube, and identified 17 species of marine megafauna, most of which displayed foraging and/or interaction with structures. They also reported the first confirmed visual sighting of a seal following a pipeline, and the deepest confirmed record of a sleeper shark, demonstrating the utility of online data sources to quantitatively elucidate relationships between offshore infrastructure and marine species. McLean et al. argue that, through cost-effective enhancements of ROV equipment and survey operations, offshore industry has the potential to contribute to our understanding of the impacts of artificial structures on the marine environment, and to collect invaluable data to support scientific investigation of changing marine ecosystems.

EMERGING OPINIONS AND FUTURE DIRECTIONS

This Research Topic provided a great opportunity to discuss our current understanding of: (1) the diversity and dynamics of human uses of the marine environment; (2) the responses of marine species, populations, communities and ecosystems to sub-sea artificial structures; and (3) the nature of their collective impacts on wider ecological processes. While Lohrer et al. did not identify the specific reason why early life-stage snappers had a positive affinity for structures, other studies reported that species used artificial structures as shelter or feeding ground (Callaway; Gates et al.; Thomson et al.; Todd et al.). Many authors also emphasized that the ecological significance of artificial structures must be considered within the context of the total footprint and the extent to which they contribute to biological connectivity and ecological processes operating at larger scales (Fujii et al.; Macolino et al.; Rouse et al.; Zellmer et al.). Furthermore, the authors who worked on offshore infrastructure were unanimous that access to global industry datasets can drive a better understanding of the changing ocean in areas impacted by anthropogenic activity. This, however, requires establishment of industry-academia partnerships to facilitate understanding of needs, priorities, and limitations of all parties (McLean et al.). With regard to issues surrounding decommissioning, van Elden et al. proposed a "novel ecosystem" approach to facilitate decision-making. Overall, this Research Topic provided a renewed insight into how distributional responses of different marine species are related to seafloor heterogeneity and shed new light on the broader role of marine artificial habitats.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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ORIGINAL RESEARCH
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Interstitial Space and Trapped Sediment Drive Benthic Communities in Artificial Shell and Rock Reefs

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Enhancing habitat complexity and thereby biodiversity is a main motivation for the creation of artificial reefs in the marine and coastal environment. Uncertainty remains, however, regarding which types of reef best deliver this aim, and how material properties impact faunal communities. The objective of this study was to assess the macrobenthic infauna in standardized reef-units made from different types of shell and rock and to quantify factors explaining community properties. 70 × 75 × 25 cm reef-units were made from cockle, mussel and oyster shells and rocks. Replicate units were placed on an intertidal sand flat of Swansea Bay (Wales, UK). After 5 months the benthic fauna was washed out of the reef-units and identified to species level. The volume of reef material, interstitial space and trapped sediment in each unit was quantified. A total of 45 invertebrate species were recorded in artificial reef-units compared with 12 species in the reef-free surrounding sands; 37 species were exclusively found in reefs. There was no significant difference between the infauna communities in different reef types in terms of univariate or multivariate diversity descriptors, but multivariate dispersion was lower among rock than shell-reef replicates. Distance-based linear models (DistLM) showed that the volume of interstitial space per reef-unit was the factor best explaining community structure, followed by properties of the trapped sediment. Species richness was significantly correlated with the volume of interstitial space and trapped sediment. Species seemed to use the reef-units fleetingly as shelter during low water, more permanently for protection, or as hunting ground for prey. The study demonstrated that artificial reef-units made of loose shell material and rocks can significantly enhance infauna diversity in sandy coastal environments. The identity of the material seems less relevant as long as it maximizes interstitial space and allows trapping of sediment. This provides practitioners with a degree of creative freedom when designing artificial reefs with the aim to enhance infauna diversity.

Keywords: benthos, biodiversity, biogenic reef, cockle, infauna, mussel, oyster

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INTRODUCTION

Artificial reefs are man-made discrete areas of firm material arising from the surrounding seafloor. OSPAR defines them as "a submerged structure deliberately placed on the seafloor to mimic some functions of a natural reef such as protection, regeneration, concentration and/or enhancing population of living marine resources" (OSPAR CCOMMISSION., 1999). The motivation for designing and constructing reefs include fisheries protection and production, habitat protection and restoration, research and recreation (OSPAR COMMISSION, 2009). In Europe over 200 artificial reefs have been deployed over the past 40 years (Fabi et al., 2011). Projects range from small nature conservation initiatives to major programs developing regional fisheries or protecting large areas from trawling (Jensen, 2002).

Artificial reefs are either designed to enhance habitat complexity and provide space for a generally more diverse epifauna, or they are built to attract specific species (Baine, 2001). They may protect or enhance fish stocks, restore marine habitats or create sites for recreational diving and fishing (Jensen, 2002). Most artificial reefs are inadvertent reefs; they are coastal and marine infrastructure such as coastal defense measures, moorings, breakwaters or increasingly foundations of off-shore wind turbines (Langhamer, 2012; Lawless and Seitz, 2014). Off-shore wind turbines and other marine renewable energy devices are generally constructed on soft bottom substratum, and the submerged parts of the infrastructure, turbine foundation as well as scour protection can potentially turn exposed, biodiversity-poor areas into species rich ecosystems (Inger et al., 2009; Broadhurst and Orme, 2014).

One function of artificial reefs is to provide attachment surfaces for sessile species, and they may offer shelter from wave and current exposure, or from predators. The magnitude and nature of the impact of artificial reefs is influenced by their location, how well they are connected with other reefs, their height, size and complexity and the surrounding habitat (Lenihan, 1999; Grabowski, 2004; Grabowski et al., 2008). Structural complexity of a reef, besides other factors such as reef diversity, is linked with its impact on the abundance of fish and invertebrates as well as species diversity (Hunter and Sayer, 2009). Further, the reef community is determined by the presence of fouling species, in particular the amount of meroplankton in surrounding waters and the bioengineering potential of the fouling species (Ambrose and Swarbrick, 1989; Moffitt et al., 1989; Einbeinder et al., 2006). They can increase or decrease benthic invertebrates or have little effect (Langois et al., 2006). An assessment of European reefs indicated that only 50% of case studies met their objectives while the remainders had little or no effect (Baine, 2001).

Artificial reefs can be created from a plethora of different materials, such as concrete modules, natural rocks and boulders, tires, pulverized fuel ash (PFA), PVC, wood, trees, rope, netting or stabilized quarry dust slurry (Baine, 2001; Jensen, 2002; Fabi et al., 2011; La Peyre et al., 2014). Gravel and boulders may be used to protect offshore energy infrastructure from scour (Langhamer, 2012). OSPAR guidelines advocate the use of inert materials (non-polluting through leaching, physical or chemical weathering and/or biological activity), and they should not be made of wastes (Baine, 2001).

In shallow coastal areas bivalve shells from mussels or oysters are increasingly used as reef material (Fabi et al., 2011). Bivalve reefs such as mussel or oyster beds diversify the habitat, particularly in sedimentary sub- and intertidal areas, and they thereby engineer the resident benthic communities (Jones et al., 1994). Such features are recognized globally as providing ecosystem services disproportionate to their size (Lundquist et al., 2017). Artificial shell reefs mimic the structural element of natural biogenic reefs; they lack functional services such as water filtration. Oyster shells are ubiquitously used to enhance natural oyster populations for harvesting, the stabilization of intertidal sandflats or wave attenuation and coastal defense (Levine et.al., 2017). Bivalve reefs are part of an entire portfolio of coastal eco-engineering concepts for flood defenses (van Loon-Steensma et al., 2014). These are particularly applied at locations that have sufficient space between urbanized areas and the coastline to accommodate the creation of ecosystems, such as shellfish reefs, tidal marshes or mangroves, which have the natural capacity to reduce storm waves and storm surges and can keep up with sea-level rise by natural accretion of mineral and biogenic sediments (Temmerman et al., 2013). Like other bio-engineering species, bivalve reefs form complex structures that provide spaces for feeding and nesting, and they are a refuge from predation and environmental exposure (Bartol et al., 1999; Bartholomew et al., 2000; Gutiérrez et al., 2003; Coen et al., 2007).

Studies on the impact of artificial reefs focus on epifauna and fish communities and infauna is often ignored (Fabi et al., 2011). Many designed reef structures as well as moorings or breakwaters may indeed not provide habitat suitable for small invertebrates which require narrow internal spaces. However, it is possible to consider artificial reef designs that offer an open structural matrix with interstitial space to be exploited by invertebrates. The impact of different reef material on infauna can be difficult to compare due to the variable structures of the reefs they create and challenges for standardized sampling (Baine, 2001).

In this study a field experiment was designed with standardized reef-units made from oyster, mussel and cockle shells and from limestone rocks. The basic design was an adaptation of a commercial ecological engineering product called rock-roll, which is primarily used for river bank stabilization (SALIX, 2018). Artificial shell-reefs resemble natural biogenic reefs such as mussel and oyster beds, and they have potentially similar functions for associated fauna (Largaespada et al., 2012). It was hypothesized that bivalve shells may have a greater benefit for biodiversity than aggregations of limestone rocks. Further, using bivalve shells for artificial reef structures would provide shellfish processors and coastal managers with an opportunity to return shell by-products from the fishing industry to their natural environment.

The objectives of this study were to:

1) Determine the difference between artificial reef-units and sandy intertidal surroundings in terms of macrobenthic species richness and abundance.
2) Compare diversity properties and community structure of the associated benthic fauna among oyster, cockle and mussel shells and limestone rocks.
3) Quantify the degree to which habitat properties of reef materials (trapped sediment, interstitial space) explain variation in the associated benthic community.

MATERIALS AND METHODS

Effects of shell and stone-filled artificial reef-units on benthic fauna were tested in intertidal sandflat areas of Swansea Bay, Wales, UK. Swansea Bay is a shallow embayment on the northern coastline of the Bristol Channel exposed to severe hydrodynamic forces due to strong winds and tides (Callaway, 2016). Reef-units were constructed from polypropylene mesh-tubes with a mesh size of 2.5 × 2.5 cm; the material is generally used in landscape engineering. Mesh-bags were filled with four types of material: Shells of the bivalves *Cerastoderma edule* (cockles), *Mytilus edulis* (mussels), *Ostrea edulis* (oysters) and limestone rock of about 4 cm diameter (**Figure 1**). Cockle and mussel shells were supplied by a local shellfish processor who stores the shells as a waste product, and the oyster shells were collected from Swansea Bay. All shells were dried for 2 months on land to sterilize them from attached fauna, and they were then filled into the prepared mesh bags to form reef-units. Individual reef-units were 70 cm long and had a diameter of 25 cm, therefore covering an area of 1,750 cm^2. Rock-filled units were created and supplied by the landscape engineering company SALIX (UK).

Field Experiment

In June 2014 the reef-units were placed on an intertidal sandflat in Swansea Bay. In the field three units filled with the same material were lined up side by side and stitched together to cover an area of about 75 × 70cm^2. Such a triple-unit would constitute one reef. The reefs were arranged in blocks of four: one of each reef type (oyster, cockle and mussel shells and stones) was placed at a distance of about 20 m to each other so that there was no interaction between them. Altogether six replicate block sets, each composed of the four reef types, were established (**Figure 1**). Sets were located about 50 m apart along the lower intertidal area of the sandflat. No other natural or artificial reef was in close proximity to avoid interaction with the experimental units. Reef-units were fixed in the sand with specialist pegs to avoid their movement or dislocation. However, in the proceeding months two replicates with cockle shells and one unit filled with stones were lost, most likely washed away by currents.

The experimental reefs were left in the field for 5 months. In November 2014 the reef units were collected for laboratory processing. Additionally, sediment core samples were taken from the intertidal sandflat as reference samples. Altogether six core samples were taken in the vicinity of each of the six replicate sets of reef-units; samples covered 300 cm^2, 10 cm deep.

Laboratory Processing

One unit of each reef replicate covering a 25 × 70 cm surface area (1,750 cm^2) was processed for benthos and sediments analysis; of the three units stitched together the sampled unit was always from the outside and never the middle unit. In the laboratory, each mesh-bag was submerged in a seawater filled container and rinsed thoroughly to wash out all sediment and benthic invertebrates. The sediment was then left to settle in the container for approximately 10 min, and subsequently the sample was treated similarly to benthos grab samples. The water was decanted carefully to avoid re-suspension of sediment and to lose as little very fine sediment as possible. It was poured through a 1 mm sieve to retain any suspended benthic fauna. The volume of the sediment was then determined, and a sediment sample (200 g) was taken for grain size analysis. A sample for grain size analysis was also removed from the reference sediment core samples. The remaining sediment of each sample was washed through a 1 mm sieve to retain macroinfauna; in November few juveniles were present in the benthos and most macrofauna had reached a size >1 mm. The sieve residue was fixed in 4% buffered formalin and subsequently preserved in 70% ethanol for benthic infauna analysis. Each rinsed, sediment-free artificial reef-unit mesh bag was then opened and emptied into a container and the volume of the shells or stones was determined. In order to quantify the volume of interstitial space (the space between the reef-unit material), the container was then filled with water until it just covered the shells or stones. The water was drained into a separate container and the total volume of the water was measured, representing the volume of interstitial space.

Shells and stones were checked for attached epibenthos. The infauna was separated from the preserved sieve residue and identified to species level. Sediment samples were air dried and passed through a series of sieves from 2 mm to 63 μm according to the Wentworth-Udden classification scale to determine particle-size.

Data Analysis

Differences between abiotic properties of reef units and reference sediment core samples as well as differences between reef materials were tested with ANOVA, followed by Tukey's pairwise comparison (volume of material per reef unit, trapped sediment, interstitial space). Differences in the structure of the benthic community in the different reef-units and the reference samples were explored by multi-dimensional-scaling (MDS) (PRIMER v6 software) based on a resemblance matrix of Bray-Curtis similarities. To down-weight abundant species and increase the importance of less-abundant ones the data was square-root transformed. Statistical differences in community structure between the different reef types and between blocks (**Figure 1**) were tested with PERMANOVA. In order to test for variation within and between reef types, homogeneity of multivariate dispersion was tested with PERMDISP (PRIMER v6 software). The degree to which different environmental factors explained the variation in the benthic community of reef-units was explored by constructing Distance-based linear models (DistLM in PERMANOVA). These were based on Bray-Curtis similarities of square-root transformed abundance data. R^2 was calculated for each explanatory factor, and the best overall model was selected according to the Akaike Information Criterion (AIC). Results of the DistLM were visualized by distance-based redundancy analysis (dbRDA). A vector overlay was added to the ordination diagram of the dbRDA, with one vector for each predictor variable. Univariate diversity indices were calculated for each sample (species richness S, abundance N, evenness J, Shannon diversity H'), and their link with environmental factors was also tested by DistLM. Differences between diversity indices were tested with *t*-test (reef units vs reference core samples) and ANOVA (differences between reef materials), except for abundances where data were not normally distributed. Differences between abundances were therefore tested with the equivalent non-parametric tests (Mann-Whitney U test and Kruskal-Wallis). The sediment parameters mean grain size (x),

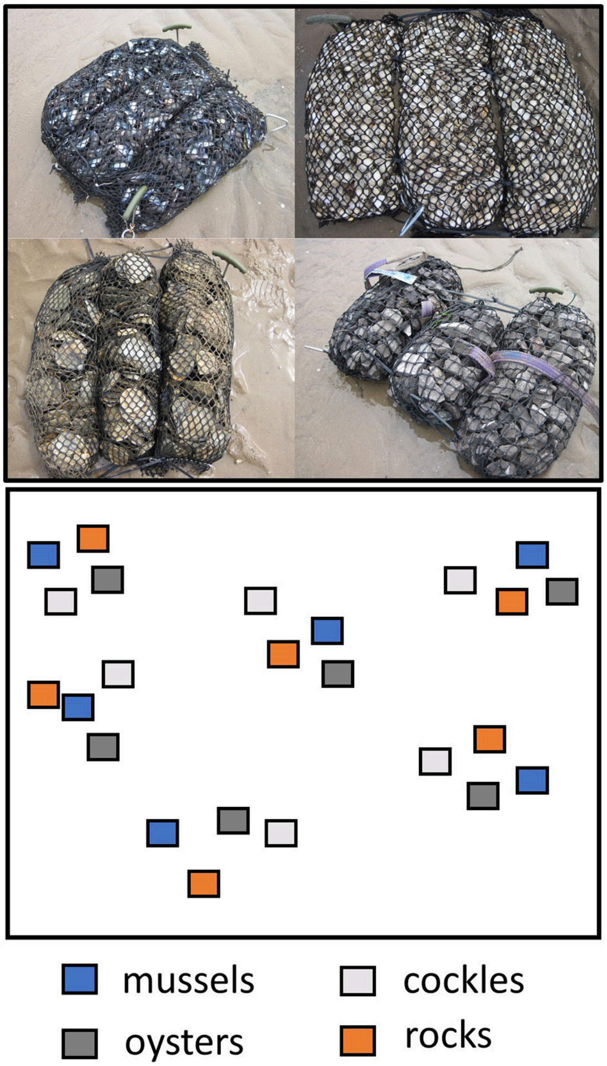

FIGURE 1 | Schematic design of the field experiment with different types of reef-units arranged in blocks, and images of the experimental reefs made of oyster, cockle and mussel shells and limestone rocks.

sorting (σ), skewness (Sk), and kurtosis (K) were calculated with GRADISTAT (Blott and Pye, 2001).

RESULTS

Differences Between Artificial Reef-Units and Reference Sediment Core Samples

Reef units and reference sediment core samples contained similar amounts of sediment; reef units trapped 3.6 ± 1.9 l of sediment ($n = 21$) and core samples contained 3.5 ± 0.5 l of sediment ($n = 6$) (ANOVA between all reef materials and sediment core samples $p = 0.62$).

A total of 45 invertebrate species were recorded in artificial reef-units compared with 12 species in the sediment core samples (**Figure 2**, **Supplementary Table 1**); 37 species were exclusively found in reefs, 4 exclusively in reference sediment cores. Of the 45 species in reef units, 32 were mobile, 5 sessile depending on hard substratum and 8 sessile depending on sediment. In sediment core samples 9 of the 12 species were mobile and 3 sessile. Most of the invertebrates were polychaetes (20 in reefs, 6 in sediment cores), followed by crustaceans (13 in reefs, 3 in sediment cores) and bivalves (8 in reefs, 2 in sediment cores).

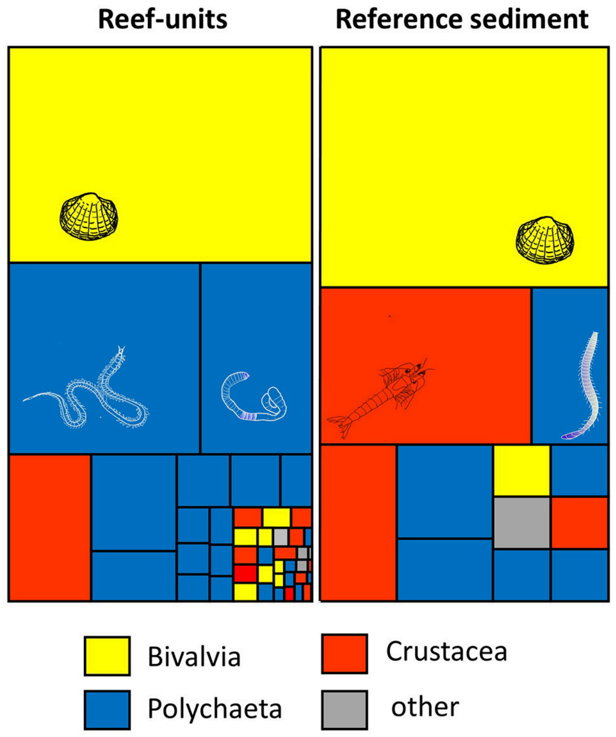

FIGURE 2 | Tree-map illustrating the number of invertebrate species in reef-units and reef-free sediment core samples. The size of tiles reflects the relative abundance of individual species (mean m^{-2}), the colour the taxonomic group; full species list in S1.

Average numbers of species were significantly higher in reef-units (8.1 ± 3.2, mean ± sd, $n = 21$) compared with sediment cores (4.3 ± 1.5, mean ± sd, $n = 6$) (t-test, $p < 0.01$). However, this comparison ought to be viewed with caution: despite the similarity in the amount of sediment in reef-units and reference sediment core samples it needs to be kept in mind that the area covered by reef-units was larger (1,750 cm^2) than that of the reference samples (300 cm^2). Numbers of individuals per unit area were not statistically significantly different: 317 ± 253 m^{-2} in reef-units ($n = 21$) compared with 294 ± 159 m^{-2} ($n = 6$) in sediment core samples (Mann-Whitney test $p = 0.88$).

The most common species in sediment core samples as well as reef units were juvenile <5 mm *Cerastoderma edule* (sediment cores 128 ± 85 m^2, reefs 123 ± 133 m^2). In the experimental reef units, the scavenging polychaete *Phyllodoce maculata* was the second most common species (67 ± 77 m^2), but it was absent from sediment cores. The amphipods *Melita palmata*, *Echinogammarus stoerensis*, and *Gammarus zaddachi* were exclusively recorded in reef units, while the mysid *Praunus inermis* and the sediment dwelling amphipod *Urothoe brevicornis* were predominantly found in the reef-free sediment cores. *Nephtys* spp. were common in reference and reef samples (sediment cores 50 ± 51 m^2, reefs 13 ± 11 m^2; ns difference Mann Whitney test $p = 0.21$).

Differences Among Reef Materials

Due to the manufacturing process rock filled reef-units contained significantly more material than the shell reefs (ANOVA $p < 0.001$, **Figure 3**); there was no significant difference between the volume of material in shell filled reef units (Tukey's multiple comparison test). Despite almost twice as much material in rock-units than in shell-reefs there was no significant difference between the amount of sediment trapped by the four different types of reefs (ANOVA, $p = 0.56$).

The amount of interstitial space was significantly larger in rock-filled units than in cockle shell reefs (ANOVA, $p = 0.013$), but there was no difference between other materials (**Figure 3**). However, the ratio "interstitial space/volume of material" was significantly smaller for rocks than for shell-reefs (rocks 0.40 ± 0.06, cockles 0.57 ± 0.09, mussels 0.68 ± 0.08, oysters 0.68 ± 0.11), meaning that bivalve shells provided significantly more interstitial space per unit volume than rocks (ANOVA $p < 0.0001$; Tukey's test <0.05 between rocks and all shell types).

Species richness ranged from 6.5 ± 3.0 in cockle reefs to 10.2 ± 2.0 in rock reef units and was not statistically significantly different (ANOVA $p = 0.35$). Similarly, abundances per reef unit were not significantly different between the four materials (Kruskal-Wallis $p = 0.09$).

The infauna community composition between the four different reef materials was not significantly different (PERMANOVA between benthic communities in oyster, cockle and mussel shells and rock reef, Bray-Curtis similarities, $\sqrt{}$-transformation of abundances, $p = 0.22$); there was also no significant block effect ($p = 0.1$). The MDS ordination (**Figure 4**) visualizes that the communities in the different reef materials did not form discrete clusters, and the same species colonized all reef materials (**Figure 5**). However, PERMDISP analysis showed significant differences in homogeneity of multivariate dispersion among the four reef types, which are visualized in the MDS ordination (F 23.91, $p = 0.0004$). Dispersion was lowest among rock-reef replicates and largest among mussel and cockle shell-reefs (average deviation from centroid, mussels 49.3, cockles 46.7, oysters 35.0, rocks 21.9). Pairwise comparisons indicated significant differences ($p < 0.05$) between rock-reefs and the three types of shell-reef.

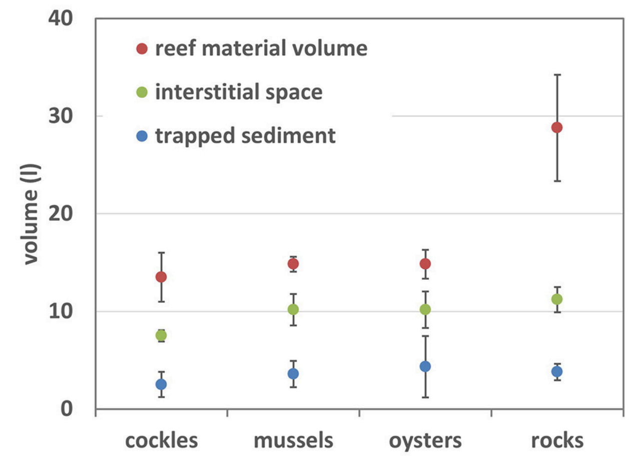

FIGURE 3 | Abiotic properties of experimental reef-units: volume of shell and rock material within each sample unit, interstitial space within the units and amount of trapped sediment.

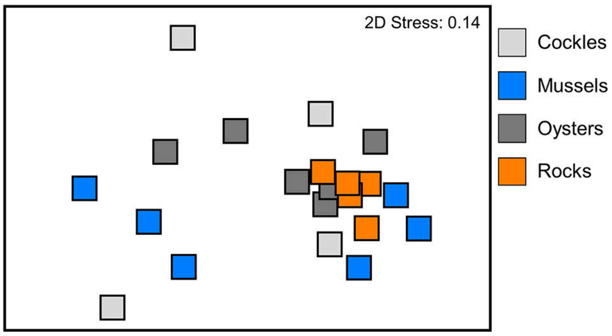

FIGURE 4 | Non-metric multi-dimensional scaling (MDS) of benthic communities within experimental reef units consisting of cockle, mussel and oyster shells as well as rocks. Ordination based on Bray Curtis similarities of square-root transformed abundances of macrobenthic species per reef unit.

Abiotic Factors Driving Artificial Reef Communities

Distance-based linear models (DistLM) were constructed to quantify the degree to which different abiotic factors explained the variation in the benthic communities of reef-units. The explanatory factors entered into the model were (1) volume of interstitial space, (2) volume of reef material (shells or rocks), (3) volume of trapped sediment, (4) mean grain size of sediment, (5) sorting of sediment, (6) sorting skewness of sediment, (7) sorting kurtosis.

The volume of interstitial space provided by a reef unit explained more of the variation in the benthic community than the other factors (**Table 1**). The two factors "interstitial space" and "skewness of sediment sorting" had the lowest AIC, therefore providing the overall best model solution (**Figure 6**). Since *C. edule* happened to be the most common species in the reef units but is generally an unlikely reef dweller, DistLMs were additionally constructed which excluded cockles. The purpose was to establish if the presence of *C. edule* had a profound impact on the model. Again, interstitial space explained more of the variation in the benthic community (cockles excluded) than other factors; interstitial space was the only factor individually explaining significant amounts of variation in the data ($R^2 = 0.10$, $p = 0.041$). The lowest AIC was provided by interstitial space and sediment sorting. All factors combined explained 47% of the variation.

Species richness was significantly correlated with the volume of interstitial space as well as the volume of sediment trapped in the reef units (**Figure 7**). Shannon diversity H' was also significantly correlated with volume of trapped sediment

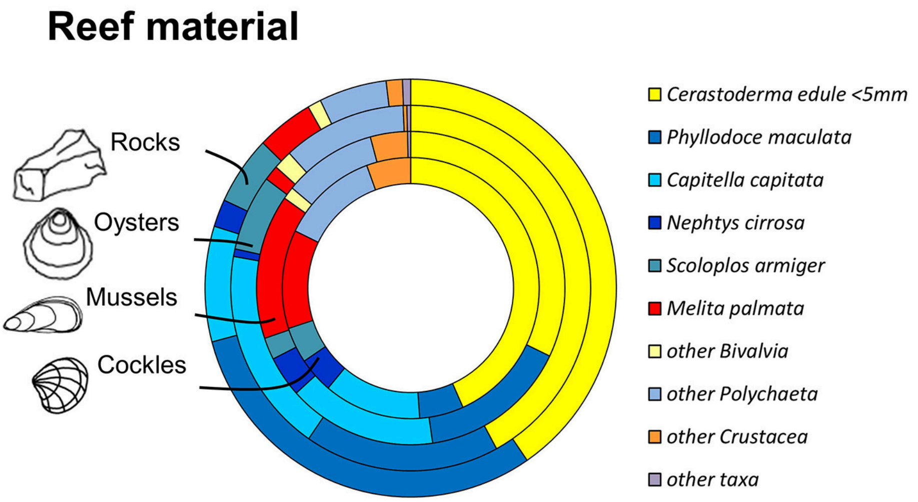

FIGURE 5 | Most abundant species and taxonomic groups in experimental reef-units. From outside to inside: rock-reefs ($n = 5$), oysters ($n = 6$), mussels ($n = 6$), cockles ($n = 4$). Abundances for materials based on mean densities per reef unit.

TABLE 1 | Distance based linear model (DistLM) outputs for relationship between reef properties and benthic infauna in experimental reef units.

	All benthos			**Benthos excluding *C. edule***		
Reef properties (predictor variables)	**Pseudo-F**	***p***	**Prop. Var.**	**Pseudo-F**	***p***	**Prop. Var.**
1. sediment volume	1.29	0.23	7.0	1.57	0.12	8.5
2. reef material volume	1.64	0.12	8.8	1.56	0.12	8.4
3. interstitial space	1.78	0.09	9.5	1.97	0.04	10.4
4. mean grainsize	1.05	0.36	5.8	1.48	0.14	8.0
5. sediment sorting	1.39	0.20	7.6	1.77	0.07	9.4
6. skewness of sediment sorting	1.26	0.26	6.9	1.89	0.29	6.5
7. kurtoises of sediment sorting	1.19	0.31	6.6	1.25	0.27	6.8
	AIC	**R^2**	**Model**	**AIC**	**R^2**	**Model**
Overall Best Solution	146.39	0.19	**3., 6**	146.17	0.21	**3., 5**
All factors	149.23	0.44	1.–7	148.43	0.47	1.–7

($r = 0.54$, $R^2 = 0.29$, $p = 0.015$, $n = 20$), but there were no other significant relationships between univariate diversity indices and abiotic factors.

DISCUSSION

This field experiment demonstrated that artificial reefs made of loose shell material and rocks had a significant impact on the diversity of benthic infauna species in a sandy intertidal environment: the reefs promoted species richness. Generally, complex habitats created by bivalve reefs have the potential to support a suite of species not found in nearby habitats (Coen and Luckenbach, 2000; Tolley and Volety, 2005; Scyphers et al., 2011; Brown et al., 2014). Particularly mobile invertebrates appear to benefit from small-scale habitat complexity of artificial material while sessile species seem to prefer smooth surfaces (Lavender et al., 2017). Whether or not shells or other reef material impact fauna appears to depend on the local assemblage and species-specific responses to the hard substratum (Langhamer, 2012); artificial oyster shell reefs for restoration projects, for example, had different impacts on faunal communities, which appears to be location specific (Schulte et al., 2009; La Peyre et al., 2014; George et al., 2015). Shells do not necessarily enrich the benthic community: in a Brazilian estuary the presence or absence of shells in sediments had no measurable effect on the community structure (Sandrini-Neto and da Cunha Lana, 2014). A review of European artificial reefs indicated a strong interaction between local environmental conditions, the local ecology and the specific reef designs, creating unique systems that may not be reproducible elsewhere (Baine, 2001). Therefore, caution needs to be applied when extrapolating results of this Swansea Bay (Wales, UK) experiment to other locations. Given that the infauna species were typical for intertidal sandflats in North-Western Europe (Hayward and Ryland, 2017), it seems

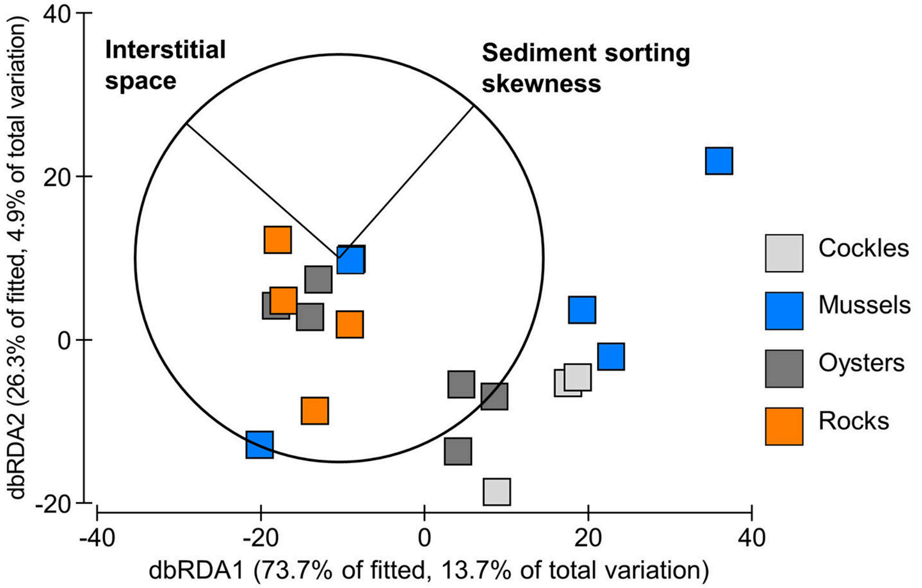

FIGURE 6 | dbRDA ordination of the fitted DistLM model for benthic fauna in experimental reef units filled with shell and rock material (based on Bray Curtis similarities after square-root transformed abundances). Reef characteristics best explaining the variation in the community data are superimposed.

plausible to presume that results would be similar for intertidal sandflats of this geographical region, and broader principles such as the importance of interstitial space may have wider relevance.

Interstitial Space

In this study biodiversity increased with the amount of interstitial space, independent from the reef material, indicating that the space created drove colonization. Differences in the amount of interstitial space were also found for fresh and dredged oyster shells, where material providing more interstitial space was preferentially chosen by crustaceans in mesocosm experiments (Levine et al., 2017). Due to the different ecologies of the infauna species in this field experiment they will have used the space in species-specific ways. Highly mobile crustaceans such as *Nototropis swammerdamei* or *Idotea pelagica* are likely to have used the space as a temporary shelter before migrating back into the water column at high tide (Callaway, 2006). The majority of species were mobile polychaetes that may have used the reefs as refuge from predators like larger crabs, or for protection from challenging environmental conditions. Juvenile cockles *C. edule* were probably passively transported into the reef-units together with mobile sands. Some may have been trapped in the interstitial space, although cockles, particularly juvenile ones, are capable of active movement and migration (Armonies, 1992). It is possible that some pockets of the reef-units offered suitable conditions for the juvenile cockles to survive, meaning enough water circulation to provide oxygen and food.

Some macrobenthic predators may have been attracted to the reefs as hunting ground. The predatory, scavenging *Phyllodoce maculata* was found in significantly larger number in the reef units compared with the surrounding sandflat. The polychaete is highly mobile and likely to have actively explored the interstitial space of the reef-units for meiofauna, juvenile macrofauna or dead organisms. The species appears to be attracted to biogenic reefs and has also been found in significantly higher abundances among *Lanice conchilega* aggregations (Callaway, 2006).

It ought to be kept in mind that the positive correlation between interstitial space and species richness cannot simply be extrapolated; if space between material becomes too large it may no longer fit the needs of the invertebrates. Further research into the interaction of benthic invertebrates with gaps between reef material and how the space is utilized could clarify the positive relationship between interstitial space and biodiversity. It was suggested that traits such as interstitial space of habitats created from various materials should be added to the list of issues considered when natural communities are to be restored in oyster reefs and other environments (Levine et al., 2017).

Reef Material

The study showed that different materials can have a similar positive impact on benthic infauna. This was also found in studies of "accidental reefs": waste material such as wood, metal and tires, which was dumped in an estuary, was colonized at similar rates compared with sandstone reefs (Chapman and Clynick, 2006).

The three bivalve shell types resembled each other in terms of the interstitial space they provided, although oyster shells are considerable bigger than cockle and mussel shells. The surface texture of the shells differs in that oysters are roughly layered, mussels are relatively smooth with only fine groves, and *C. edule* is strongly ribbed; due to the absorbent properties of cockle shells they are used for removing pollutants from wastewater (Kazemi et al., 2016). Still, the differences in texture and size of shells had no significant effect on the infauna they hosted.

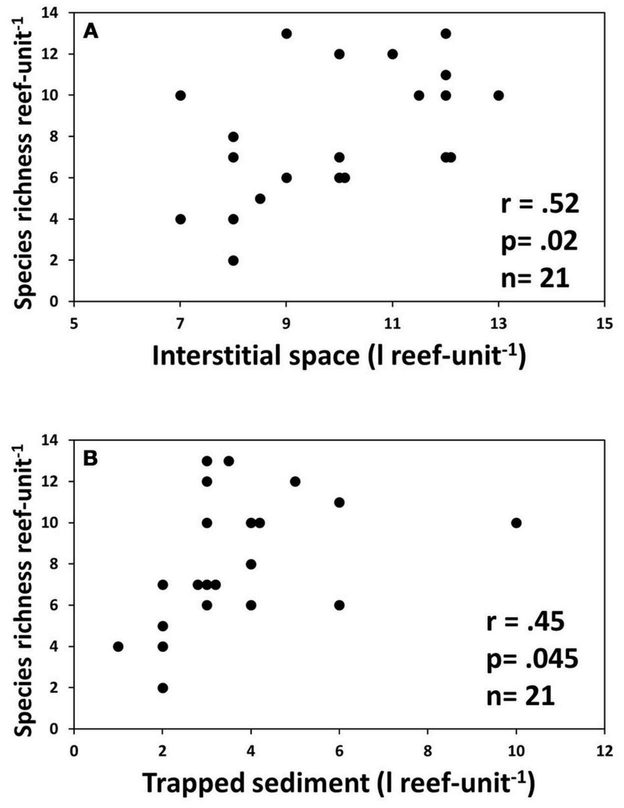

FIGURE 7 | Correlation between abiotic reef properties and species richness. **(A)** Volume of interstitial space within each reef unit; **(B)** Amount of sediment trapped by reef units.

Bivalve shell-reefs showed no greater benefit for biodiversity than limestone rock aggregations. There were, however, significant differences in the multivariate dispersion among the reef types. Rock-reefs showed the lowest dispersion, i.e., the benthic communities in rock-reef replicates were more similar than those of shell-reefs, with the communities in mussel shells-reefs replicates being most different from each other. Homogeneity of multivariate dispersion is an indicator of beta diversity (Anderson et al., 2006) and the results suggest that shell reefs potentially create higher beta diversity. High dispersion levels are, however, also interpreted to reflect stress levels in marine communities (Warwick and Clarke, 1993), and the result may suggest that the shell reefs suffered greater disturbance while rock-reefs offered more constancy. Generally, this result must be viewed with caution since within-group sample size was less than $n = 10$, which limits the power of the multivariate test.

Further, it needs to be considered that the rock-reefs contained significantly more material than the shell-reefs and had a smaller ratio "interstitial space/volume of material," or in other words, due to the larger size of rocks compared with bivalve shells, rock-reefs created less interstitial space per unit volume than shells. In efforts to create habitat for benthic macroinvertebrates, rock-reefs would need to be about double the size of bivalve reefs to produce similar volumes of interstitial space.

Impact of Artificial Shell Reefs Compared With Live Biogenic Reefs

Reefs of living bivalves have different properties than clusters of dead shells. Bivalves filter quantities of water and thereby move particles, excrete feces and pseudofeces, which agglomerate sediments, enriching the organic content and affect nutrient flux (Largaespada et al., 2012). Predators of the bivalves such as whelks or starfish would unlikely be attracted to empty shell reefs (Gosnell et al., 2012). Their live bodies have a different shape and volume compared with empty shells, and therefore the space they create is different. However, while both physical and biological properties of mussels mediate the positive effect of mussels on their recruits (positive intraspecific feedback), physical properties alone can explain the interspecific positive engineering effect of mussels on diversity of associated species (Largaespada et al., 2012).

The results of this study therefore also allow, to a limited degree, a comparison between mussel and oyster banks in terms of their respective impact on biodiversity. Comparisons of different natural bivalve reefs and their associated fauna can be challenging as it may not be possible to standardize sampling methods for different types of reef, or because mussels and oysters occur interlaced (Drent and Dekker, 2013). However, evidence from the Dutch Wadden Sea suggests that oyster and mussel reefs provide habitat for similar macrobenthic communities, except for the barnacle *Elminius modestus*, which was found predominantly with oysters (Drent and Dekker, 2013). This field experiment would support the result from the Wadden Sea: barnacles were found on some of the rocks and oyster shells, and only few individuals on mussel shells.

Epibenthic Species and Birds

The barnacle *Elminius modestus* was the only recorded epibenthic species and therefore epibenthic colonization played a small role in this study. Generally, epibenthic species use bivalve reefs as attachment surface and may also indirectly benefit from shell-reefs. For example, the sea cucumber *Apostichopus japonicus* was found to grow faster on artificial oyster shell-reefs because of increased food supply by diatoms colonizing the shells (Zhang et al., 2014). Whelks were shown to be positively affected by the presence of dense mussel beds that affected their feeding, growth and interaction with other species (Gosnell et al., 2012). Biogenic shell banks provide feeding habitat for coastal birds, but mussel and oyster beds support different species; reefs of the Pacific oyster *Crassostrea gigas* are used by curlews and oyster catchers, while herring gulls prefer mussel beds (Markert et al., 2013); copious footprints of gulls around the reef-units in this experiment indicated that birds were attracted to the features, but no direct observations of the use were made.

Production vs. Aggregation

There is an ongoing discussion if artificial reefs produce new biomass, or if they act as a focal point for existing organisms which aggregate on or in the newly formed reef (Cresson et al., 2014). This is considered to depend on the species present and their limitation by food, refuge, territory, and/or behavioral requirements. Colonisation of new artificial reefs generally follows two main processes: migration of post-larval individuals and settling of larvae. The recruitment success depends on the presence of larvae and mobile invertebrates in the water column, and the nature of the infauna communities near the reef.

The fauna in reef-units of this experiment comprised of species also found in surrounding sand flats, but an additional 37 species were exclusively found in the reefs (of a total of 45 species). Given the relatively short duration of the experiment (5 months), it is likely that they migrated into the reef units as opposed to being new biomass, and therefore the reefs would have re-distributed existing biomass. It seems, however, plausible that over time the reefs could be attractive nurseries for benthic fauna and enhance biomass production; there is evidence that mussel and oyster banks are attractive habitats for juvenile fish and crustaceans (Kochmann et al., 2008; Seitz et al., 2014). The crab *Hemigrapsus sanguineus*, for example, uses mussel beds only during early life stages to be protected from intra and interspecific predation (Pezy and Dauvin, 2014). Artificial reef structures could have a similar nursery function.

CONCLUSION

Material accretions with an open matrix that provide interstitial space and trap sediment have the potential to enhance benthic biodiversity. Like other coastal reefs the main function seems to be protection from severe environmental conditions and predation. However, the field experiment suggested that artificial reefs may also be attractive hunting grounds for polychaete predators, and it would be useful to better understand this function of reefs. Given the likely short-term use of reef structures by some species, it is possible that they interact with surface layers more extensively than with deeper reef areas. Further studies need to clarify this and could have implications for the surface design of coastal infrastructure.

This study indicated that it is possible to create man-made reef units which provide biodiversity enhancing services. Different types of material seem to be suitable for loose-material reefs. Bivalve shells provided suitable structures, but just piles of small rocks were equally effective. It is therefore possible that other materials could achieve a similar effect, such as gastropod shells, different types of natural stone or slate, or recycled material such crushed concrete. Considerations such as the suitability of these materials in a specific coastal environment need to be evaluated, but in terms of infauna-enhancing measure there seems to be creative flexibility.

AUTHOR CONTRIBUTIONS

RC is responsible for the creation of the project, the experimental design, conducting the experiment, data processing and analysis and the interpretation of the results. She is the sole writer of the paper.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00288/full#supplementary-material

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Benthic Structure and Pelagic Food Sources Determine Post-settlement Snapper (*Chrysophrys auratus*) Abundance

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Nursery habitats provide increased survival and growth and are a crucial early life-stage component for many fish and invertebrate populations. The biogenic structures that provide this nursery function, however, are increasingly degraded. Therefore, any effort to conserve, restore or replace habitat with artificial structure should be guided by an understanding of the value provided by that nursery habitat. Here, we experimentally manipulated structure across a number of sites by inserting pinnind bivalve mimics into the seabed and deploying video cameras to observe the response of post-settlement stage snapper, *Chrysophrys auratus* (Forster in Bloch and Schneider 1801). We also collected a range of environmental variables across these sites to determine the relative importance to snapper of benthic vs. pelagic productivity. While the abundance of snapper was low, our results demonstrated a strong association to structure relative to control plots. The environmental variable with the highest correlation to snapper abundance was the abundance of zooplankton eaten by snapper. This result was well supported by the dominance of zooplankton over small benthic invertebrates in snapper gut contents, and the weak influence of benthic infauna in our regression models. These regressions also demonstrated that when combined with zooplankton abundance, turbidity had a negative relationship to snapper abundance. This highlights the importance of relatively clear water in estuaries, which allows post-settlement snapper to more efficiently consume the zooplankton that are present in the water column. The third component that post-settlement snapper require is of course the presence of benthic structure. While benthic habitat structure was the strongest factor affecting juvenile snapper abundance, we did not find any correlations to suggest that this importance was related to energetic sheltering and access to locations with high food flux.

Keywords: ***Pagrus auratus*, juvenile fish nursery, habitat structure, environmental drivers, hydrodynamic variables, zooplankton, benthic infauna, diet**

INTRODUCTION

Nursery habitats are a critical requirement for a number of fish species with separate juvenile and adult life stages (Beck et al., 2001; Heck et al., 2003; Dahlgren et al., 2006). Protection from predation and food availability are the most likely explanations for this juvenile habitat association (Heck et al., 2003). Given the obligate nature of the nursery habitat association and the often degraded state of the benthic habitats that provide it (e.g., Orth et al., 2006), conserving, restoring, or providing artificial habitat structure to ensure that juvenile life stage requirements are satisfied is an important consideration for the maintenance of adult fish populations and the services that they provide (Townsend et al., 2014). Understanding the environmental factors that drive the abundance of nursery habitat occupying juvenile fish, while not the entire picture (Beck et al., 2001), is an important consideration for habitat and fishery management.

Snapper, *Chrysophrys auratus*, (=*Pagrus auratus*) (Perciformes – Sparidae), are a recreationally and commercially important fish species that are abundant in the coastal waters of northern New Zealand (Parsons et al., 2014). Post-settlement stage snapper [<60 mm Fork Length (FL)] occupy shallow estuarine locations for a few months after settling over summer and/or autumn before dispersing to a range of habitats/locations. High abundances of post-settlement stage snapper are usually associated with habitat structure, whereas immediately adjacent bare sediment sites are usually unoccupied (Parsons et al., 2013, 2016). Because interactions between post-settlement snapper and predators have rarely been observed, an alternative explanation to describe their affinity to structure has been developed. This theory relates to energetic sheltering benefits that structure could provide at sites which have high water velocity, but also a high associated flux of the pelagic zooplankton food that post-settlement snapper prefer (Parsons et al., 2015, 2018). In more turbid estuaries this theory may not apply, as visual feeding is more difficult, and benthic prey (which don't have a flux) dominate post-settlement snapper diet (Lowe et al., 2015).

In the present study we conducted a controlled structure manipulation experiment as a means of understanding the response of post-settlement snapper to structure across a number of sites. We also collected a range of environmental and habitat variables from each of these sites, so we could understand how snapper abundance and response to structure was influenced by these variables. Of primary interest was the relative importance of benthic vs. pelagic variables, as the estuary we conducted this experiment within was relatively turbid (Lowe et al., 2015), suggesting that either benthic or pelagic productivity could be important.

MATERIALS AND METHODS

Study Site and Overall Sampling Approach

Sampling was conducted within Mahurangi Harbor, northeastern New Zealand (**Figure 1**). Mahurangi Harbor is a drowned river valley with an area of c. 24.5 km^2 (Feeney and Challis, 1984). While Mahurangi Harbor is relatively turbid and potentially a lower value habitat for juvenile snapper compared to some other estuaries within northeastern New Zealand (Lowe et al., 2015), it still hosts a large population of juvenile snapper (Morrison and Carbines, 2006) and offers one of the largest estuarine habitat areas to juvenile snapper within the Hauraki Gulf, the location of New Zealand's largest snapper population (Parsons et al., 2014).

Different components of sampling were conducted over an almost 3 year period (March 2015 to December 2017), with most sampling events taking place during March, the time period when the post-settlement juvenile snapper occupy sheltered inshore estuaries and harbors such at the Mahurangi (Parsons et al., 2014). Our approach centered around the quantification of post-settlement snapper abundance at five sites, all 5–8 m water depth,within Mahurangi Harbor (**Figure 1**: Site A: −36.4848°S, 174.7165°E, Site B: −36.4910°S, 174.7174°E; Site C: −36.4942°S, 174.7346°E, Site D: −36.5044°S, 174.7229°E; Site E: −36.51282°S, 174.7286°E) and relating this to environmental variables at those same sites. We implemented a structure manipulation experiment at these sites to aid in the quantification of post-settlement snapper abundance and to determine the importance of structure to post-settlement snapper. Animal ethics approval was not required for this study as per the local legislation. All sampling was conducted during daytime hours.

Structure Manipulation Experiment and Post-settlement Snapper Abundance

To assess the effect of structure on post-settlement snapper abundance we created patches of high structure availability at five sites within Mahurangi Harbor (**Figure 1**). The structure element we used to create these patches was intended to mimic the valves of the pinnid bivalve *Atrina zelandica*, which protrude up to 25 cm above the sediment surface, can form extensive biogenic reefs (Cummings et al., 1998), and have been found associated with high abundances of post-settlement snapper at other locations (Parsons et al., 2016). *Atrina* mimics consisted of triangular segments cut from 100 mm diameter polyvinyl chloride (PVC) pipe with approximate dimensions of 40 cm high by 12 cm wide. In December 2015 (the time of peak snapper spawning, Parsons et al., 2014) divers created patches of *Atrina* mimics at each site by marking out a 2 × 2 m area with a quadrat and then haphazardly pushing 25 *Atrina* mimics into the sediment within the quadrat so that c. 10–15 cm of each mimic protruded above the sediment. Divers then measured 15 m from that patch before repeating this process so that $n = 3$ *Atrina* mimic patches were created over a 30 m section of seabed. Following identical spacing, $n = 3$ bare sediment control plots were identified over the next 30 m of seabed, but were unmarked. Previous investigations have demonstrated that post-settlement snapper are resident to patches separated by c. 10 m (Parsons et al., 2013), so 15 m spacing was adequate for us to treat each patch within a site as independent.

The patches of *Atrina* mimics were left in place until March 2016, the time of year when post-settlement snapper abundance should be at its peak (Parsons et al., 2014). At this time we quantified the fish communities associated with *Atrina* mimics (and adjacent control plots) by simultaneously deploying six video cameras at each site. We used GoPro Hero 3 or Hero

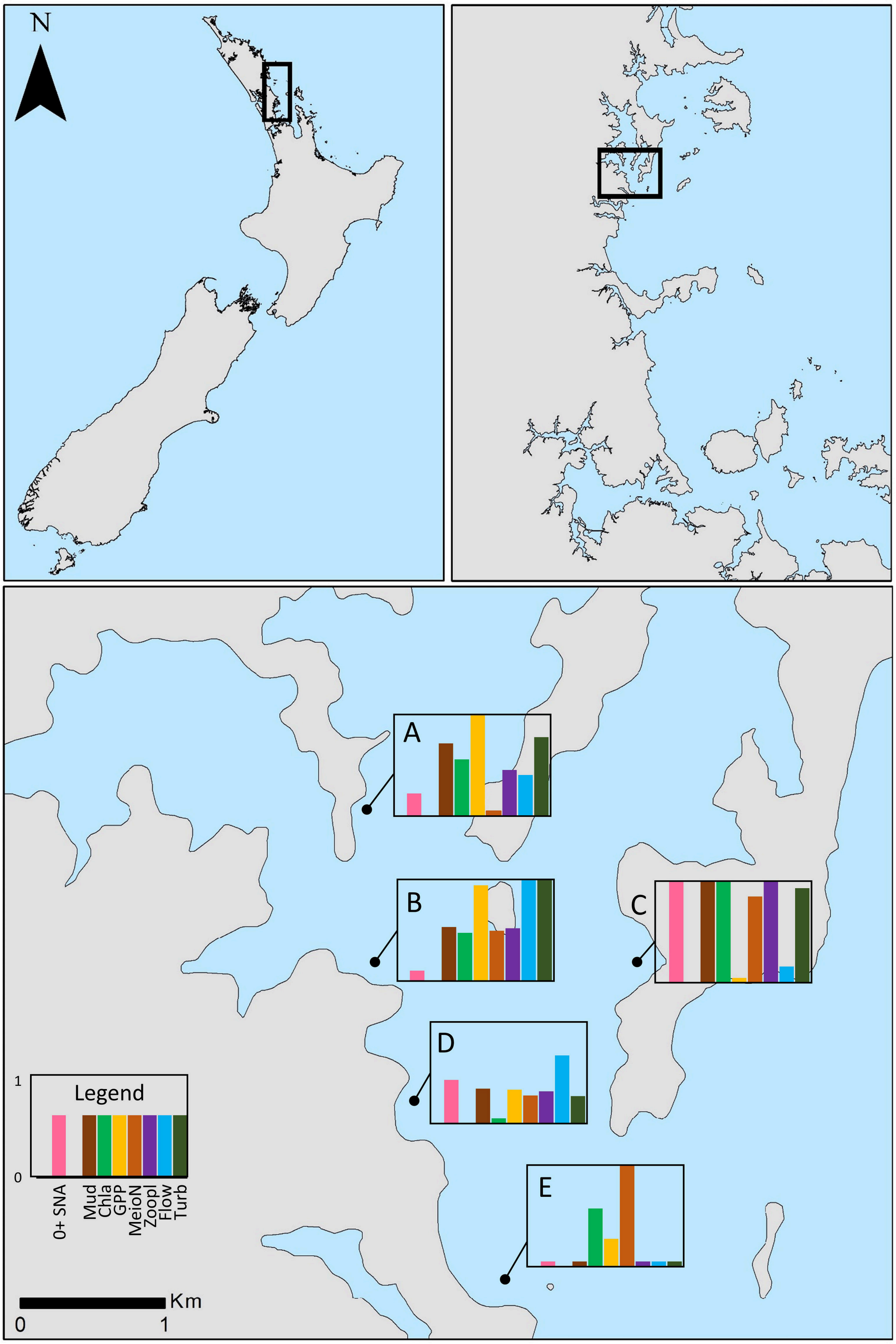

FIGURE 1 | New Zealand (upper left) and the region north of Auckland (upper right) where the study was conducted. Bottom panel shows the positions of the five study sites in Mahurangi Harbor. Water depth at high tide was 5–8 m at all sites; only a small portion of the main harbor channel near the mouth is >10 m water depth. Bar graphs depict the relative importance of habitat structure to post-settlement snapper (pink bars = abundance of snapper amongst structure minus abundance in control plots, SNA diff) and the relative magnitudes of selected environmental variables (see legend) across sites. All variables were standardized to run from 0 to 1 for ease of comparison. Mud, sediment mud content (%); Chl*a*, sediment chlorophyll *a* content (μg g^{-1}); GPP, gross primary production (μmol O$_2$ m^{-2} h^{-1}); MeioN, abundance of meiofauna per core (ostracods, nematodes, and benthic copepods); Zoopl, abundance of zooplankton that dominate post-settlement snapper diet found in bottom water; Flow, average current velocity (m s^{-1}); Turb, light attenuation coefficient, K_d, a proxy for turbidity.

4 cameras mounted on 15 cm high steel stands, which divers deployed facing each *Atrina* mimic patch or bare sediment control plot. Cameras were positioned 1 m from the edge of *Atrina* patches or control plots (the diver would measure 1 m from the camera to indicate where the control plot edge was for the video observer). Divers also took sediment cores ($n = 3$; see more detailed description below) immediately adjacent to *Atrina* patches and control plots to quantify sediment grain size, organic matter content, pigment content and invertebrate community characteristics. Cameras were left to run for a minimum of 60 min before being retrieved via a surface float.

Video Analysis

The first aspect of video analysis was to determine if water visibility was adequate to enumerate fish abundance. For all video replicates water visibility was $\geq$1 m, enabling a clear view of fish over the *Atrina* mimics or control plots. Next, post-settlement snapper abundance was assessed using a subsampling procedure similar to that of Schobernd et al. (2014). Here, 10 random times were selected within the hour of video footage obtained from each camera deployment (excluding the first 10 min to remove the potential of diver disturbance). At each of these times, 2 min of video footage was observed (ensuring that the random times selected were $\geq$2 min apart), counting the maximum number of post-settlement snapper observed at one time. The average of these counts (Avg SNA) was then used to represent snapper abundance for each video replicate. Only snapper that were around the nearest edge of the *Atrina* mimic patch or closer to the camera were counted. For bare sediment sites, a similar depth of field was estimated from footage of the diver making measurements while the camera was set in place. In terms of distinguishing between post-settlement and older aged snapper, it is important to note that this survey was conducted during March. Therefore, post-settlement snapper should be c. 6 cm or c. 100% smaller in length than the next oldest age class (1 + snapper) (Francis, 1994), providing adequate resolution to determine which snapper belonged to the post-settlement and $\geq$1+ age classes.

ENVIRONMENTAL VARIABLES

Benthic Productivity

Incubation chambers were deployed to the seabed at the five sites and sampled by SCUBA divers on two consecutive days during March 2015. A 20 m seabed transect was established at each site, and seven pairs of light and dark chambers were deployed at random positions along each transects length. The chambers (8 cm internal diameter polycarbonate plastic domes with sampling ports) were used to enclose small patches of unmanipulated soft-sediment habitat (0.016 m^2) together with 0.85 L of overlying water. The enclosed chamber waters were sampled initially (one 60 ml water sample collected per chamber at $T = 0$) and again 2–4 h later (i.e., a final sampling at $T =$ end). Dissolved oxygen concentration was measured using a handheld oxygen probe (YSI ProODO) and samples were filtered through a glass fiber filter (Whatman GF/C, pore size 1.2 μm) for later analysis of dissolved inorganic nutrients using standard methods for seawater on an Astoria-Pacific 300 series segmented flow auto-analyser with detection limits of 1 mg m^{-3} for N and P. Changes in dissolved oxygen during the incubation were used to assess rates of microphytobenthic productivity, with the light chambers providing information on net photosynthetic oxygen production in the presence of sunlight, and dark chambers providing information on total community oxygen utilization. Fluxes of dissolved inorganic nutrients (nitrate-plus-nitrite nitrogen, ammonium nitrogen, and dissolved reactive phosphorus) were calculated to assess the influence of photosynthetic uptake on rates of nutrient exchange from sediment to water column. Fluxes of dissolved oxygen and inorganic nutrients were calculated as $(C_{\mathrm{T=end}}\text{-}C_{\mathrm{T=0}}) \times V/(A \times T)$, where C was the solute concentration at the start or end of the incubation, V was the volume of water enclosed in the chamber (constant), A was the area of sediment enclosed (constant), and T was the elapsed time between initial and final water samplings.

Benthic Infauna and Sediment Characteristics

In March of 2015 and 2016, benthic soft-sediment habitat characteristics were assessed by SCUBA divers at the five study sites. On the same days as the chamber incubations (March 2015, described above), 20 small cores of sediment (3 cm internal diameter) were collected along a 20 m seabed transect at each site. Sediment particle size distribution and organic matter content in the top 2 cm of sediment was assessed from five replicate sediment cores site^{-1} time^{-1}; sediment pigment concentrations in the top 2 cm of sediment (chlorophyll *a* and pheophytin) were also assessed from five replicate cores site^{-1} time^{-1}; identities and abundances of sediment invertebrates were quantified from five replicate samples site^{-1} time^{-1}, with each sediment invertebrate sample comprised of two amalgamated 3 cm internal diameter × 5 cm deep cores.

Sediment grain size, organic matter content and pigment samples were kept frozen and in the dark until analysis and processed according to standard protocols (Gatehouse, 1971; Mook and Hoskin, 1982; Sartory, 1982; Lohrer et al., 2012). Briefly, sediment organic matter content was determined from the change in sediment dry weight (%) after combustion in a muffle furnace at 400°C for 6 h. Sediment chlorophyll *a* content (Chl*a*) was determined spectrophotometrically after extracting pigments from sediments by boiling in ethanol and using an acidification step (addition of 1 drop of 1 M HCl to sample cuvette) to separate degradation products (e.g., phaeophytin) from Chl*a*. Sediment particle size distribution (% dry weight of different grain size categories) was determined after wet sieving samples across 2,000, 1,000, 500, 250, and 63 μm mesh screens, and by pipetting to differentiate silt (3.9–63 μm) and clay (<3.9 μm). We operationally define particles <63 μm (i.e., silt + clay) as "mud".

The sediment invertebrate samples were sieved in the field across a 125 μm mesh screen and preserved in 70% isopropyl alcohol until processing. At the laboratory, a red stain (Rose Bengal) was added to the isopropanol solution to facilitate sorting, and all invertebrates per sample (including nematodes, ostracods, and copepods) were identified to the lowest practicable taxonomic level possible under a dissecting microscope (Leica M80, 60× main magnification with 10× magnification binocular eye pieces) and given the condition of the items in the samples. During March 2016, sediment grain size, organic matter content, pigment content and invertebrate community characteristics were again evaluated using identical methodologies, except that 3 replicate samples were collected immediately adjacent to *Atrina* mimic patches and 3 samples were collected from nearby control plots at each site (see above).

Zooplankton

Zooplankton were sampled to describe the potential diet items of post-settlement snapper that may occur within the water column. Sampling was undertaken during the daytime on five different days, at the five sites between December 2015 and March 2016 (this encompasses the seasonal period when post-settlement snapper are present within estuarine waters such as Mahurangi Harbor). Water samples were collected with a plankton pump that consisted of a petrol powered impellor pump connected to a 6.5 cm diameter hose that was long enough to reach the seabed. The hose was lowered to the seabed and held in place with an attached metal pole. A 50 cm long metal arm protruded below the end of the hose. This ensured that the water sample was always taken at a consistent elevation above the seabed. We chose an elevation of 50 cm as we deemed this would allow us to obtain near seabed water samples, which is where post-settlement snapper feeding is likely to occur. The pump was then run until a 1200 l water sample had passed through a 250 μm sieve (measured by repetitively filling a 20 l bucket). Samples were washed off the sieve and fixed in a 10% formalin solution, and then drained and preserved in 70% ethanol within a week. All invertebrates were then sorted, counted, and identified down to the lowest practical taxonomic level possible under a dissecting microscope and given the condition of the items in the samples.

Water Flow and Light Attenuation

Water current velocities were simultaneously measured at the five sites (**Figure 1**) during December 2017 using five Nortek 2 MHz Aquadopp's [Acoustic Doppler Current Profilers (ADCPs)]. ADCPs were deployed on the sea bed facing the water surface in a bottom mounted upward-looking configuration. ADCP acoustic transducers were nominally positioned between 0.2 and 0.35 m above the sea bed. ADCP data were collected in bursts with an interval between bursts of 10 min. Water flow at each of the sites was measured over the same period, which encompassed approximately two complete tidal cycles. Summary statistics of the burst-averaged horizontal current speed in the bin closest to the sea bed were used in subsequent analysis.

Along with each current meter, an Odyssey PAR (photosynthetically active radiation) sensor was deployed to the seabed at each site to gather information on the light available to microphytobenthic primary producers. One PAR logger was also deployed in air to capture data on incident sea surface sunlight radiation. Based on water depth and differences between incident and seabed PAR levels, light extinction coefficients, K_d, were calculated for each site. K_d is a proxy for water column turbidity, as light penetration diminishes quickly with depth in turbid water relative to clear water.

Post-settlement Snapper Diet

In addition to the environmental variables described above we wanted to describe post-settlement snapper diet, which may guide us in identifying particular benthic infauna or zooplankton taxa that may be important variables. Over several days during March 2015 and March 2016 we deployed opera house fish traps (Morrison and Carbines, 2006, length: 100 cm, width: 60 cm, height: 60 cm; 1 cm mesh) to capture post-settlement snapper to assess their diet. Multiple traps were deployed at each site, with some baited (using chopped up pilchard, *Sardinops neopilchardus*, as bait) and others un-baited. Traps would be deployed for a period of up to several hours before being checked and re-deployed. In 2016 traps were only deployed after video deployments for that site had been completed. All juvenile snapper captured were immediately euthanized before administering an injection of a 10% formalin solution (salt water buffered) into the gut cavity and then immersed into the same solution. Gut contents were later obtained by making a ventral incision, removing and opening the fore and hindgut and washing the sample into a vial using a 70% ethanol mixture. All invertebrates were then sorted, counted and identified down to the lowest practical taxonomic level.

Statistical Analyses

Benthic infauna community data from 2016 were visualized using non-metric multidimensional scaling (nMDS) and the effects of site (five levels, random factor) and treatment (i.e., structure vs. control, two levels, fixed factor) on benthic infauna community data were tested in a two-way non-parametric permutational analysis of variance PERMANOVA, PRIMER 7.0.13. The significance of the site × treatment interaction term was also assessed, and *post-hoc* pairwise comparisons were made when required to address specific hypotheses (e.g., significance of structure vs. control within sites). Identical analyses were also conducted for the meiofauna component (i.e., sediment dwelling ostracods, nematodes, and harpacticoid copepods), and sediment chlorophyll *a* content. Because benthic infauna communities in 2016 did not differ by treatment (i.e., structure vs. control), data from all six cores per site collected in 2016 were combined and compared to the 2015 data in a similar analysis (but with a fixed factor for "year" with two levels i.e., 2015 vs. 2016).

Next we assessed the effect of additional structure on post-settlement snapper. The primary response variable of interest was the abundance of post-settlement snapper (Avg SNA), with three replicate data points per site from *Atrina* mimics (i.e., structure), and also from adjacent control plots. Cameras failed on two occasions (i.e., two lost observations of 30 total), but the data set was otherwise well balanced. The effects of site (5 levels) and treatment (structure vs. control) on post-settlement snapper abundance were considered as random and fixed factors, respectively, in a two-way non-parametric permutational analysis of variance (PERMANOVA, PRIMER 7.0.13). The significance of the site × treatment interaction term was also assessed, and *post-hoc* pairwise comparisons were made when required to address specific hypotheses (e.g., significance of structure vs. control within sites).

The final part of our analysis was to relate post-settlement snapper abundance to all the ancillary environmental data collected. Our approach was to average each variable by site, generating a single independent value for each variable at each of the five sites. For snapper, the response variables used were the total of the Avg SNA observations from all replicates within a site regardless of treatment, and the difference in Avg SNA between a site's *Atrina* mimic and control replicates. These new variables were termed "total SNA" and "SNA diff," respectively, with the latter indicating the relative importance of structure to post-settlement snapper at each of the sites. We related the patterns in these response variables to each site's combination of environmental variables using draftsman's plots and the RELATE and BEST procedures in PRIMER 7.0.13. Finally, regression models with a maximum of two independent explanatory variables were used to find the combination of environmental variables that best predicted the response of post-settlement snapper to structure (based on Akaike's Information Criterion, *p*-values, and adjusted r^2 values). Models with more than two independent variables generally suffered from multi-collinearity (variance inflation) and overfitting. All explanatory variables were standardized to run between 0 and 1 prior to inclusion in the models, so that the relative strengths of the contributing variables could be evaluated by comparing the magnitudes of the coefficients.

RESULTS

General Description of Environmental Variables Across Sites

The five sites sampled in the harbor differed significantly in their environmental characteristics. The inner estuary sites A, B, and C had muddy sediments, high sediment organic matter content, and high water column turbidity, relative to sites D and E, which were located closer to the mouth (**Figure 1**). Site B, at the confluence of Pukapuka Inlet and the main harbor channel, had the fastest average tidal current speed, with nearest neighboring sites D and A next fastest. All sites were net autotrophic during the period assessed (i.e., oxygen fluxes in light chambers were positive, with photosynthetic oxygen production exceeding total consumption). Sites A and B had the highest gross oxygen production (1,600 and 1,570 µmol m^{-2} h^{-1}, respectively, relative to 1,110–1,260 µmol m^{-2} h^{-1} at the other sites).

The benthic invertebrate communities investigated in 2016, six months after experimentally adding *Atrina* mimics to the seabed, did not differ between structured and control plots (Site $P_{\text{PERM}} = 0.001$, Treatment $P_{\text{PERM}} = 0.207$, Site × Treatment interaction $P_{\text{PERM}} = 0.775$; **Figure 2B**). Similar results were also obtained for analyses conducted on meiofauna abundance ($P_{\text{PERM}} = 0.349$) and sediment chlorophyll *a* content (P_{PERM} 0.949). Analysis of 2015 and 2016 benthic infauna data demonstrated differences by site and year, with communities at all sites significantly different from one another in both 2015 and 2016 ($P_{\text{PERM}} < 0.05$ for all *post-hoc* pairwise tests, based on Bray-Curtis similarities generated from 4th root transformed community data). The differences between years at a given site were generally greater than the differences among sites within a year (**Figure 2A**).

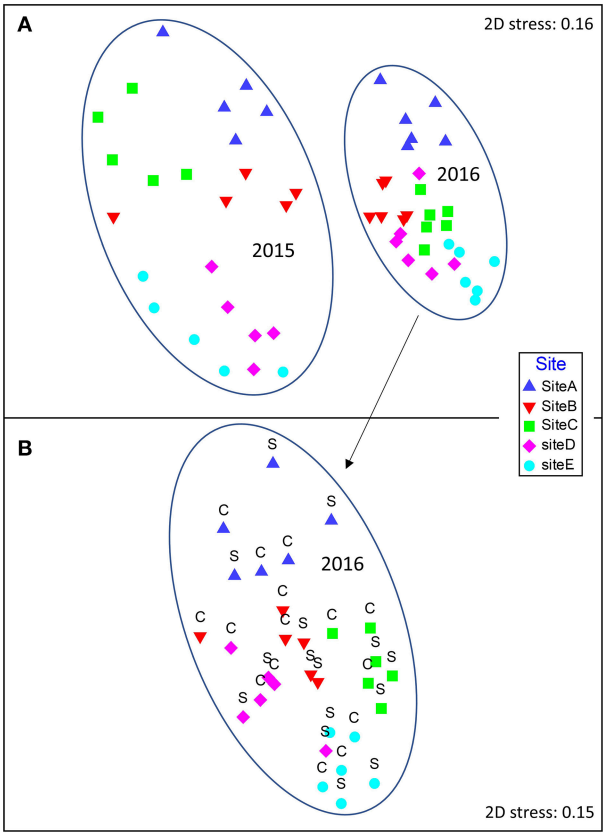

FIGURE 2 | Non-metric multidimensional scaling (nMDS) plot of Bray-Curtis similarities based on log(x+1) transformed sediment invertebrate community sample data (organisms ≥ 0.125 mm). Points closer to one another in ordination space are more similar than those apart. The five sites in Mahurangi Harbor are indicated with different symbol colors and shapes (see legend on figure). **(A)** Data from the five sites in both years, **(B)** data from the five sites in 2016 only, with structured and control plot samples indicated by S and C, respectively. Stress value is an indication of how well the multivariate data are represented in two dimensions.

Nematodes were the most common sediment dwelling invertebrate at all sites in 2016, and other meiofaunal groupings such as ostracods and copepods were among the top five most abundant organisms at all sites. Site F had the highest total abundance of nematodes, total meiofauna, and polychaetes of the genus *Pseudopolydora*. Site A had the lowest abundance of *Pseudopolydora* sp., but had relatively high numbers of bivalves *Arthritica bifurca* and Phoxocephalidae amphipods. *Theora lubrica*, a mud-tolerant bivalve, was relatively common at sites A, B, and C (sites with relatively high sediment mud content). Oligochaetes and small syllid and paraonid polychaetes were other taxa that ranked in the top 10 in abundance at all five sites.

The abundance of zooplankton taxa that dominate snapper diet (here calanoid and cyclopid copepod, and unidentified crustaceans) in bottom water was greatest at the inner estuary sites (A, B, and C), relative to the sites closer to the mouth (D and E). Average sediment mud content and organic matter content were strongly correlated with zooplankton abundance (Pearson's R = 0.87–0.95; **Table 1**). Correlations of all three variables with turbidity were high (Pearson's R = 0.80–0.84; **Table 1**).

Effect of Additional Structure on Post-settlement Snapper

The abundance of post-settlement snapper in Mahurangi Harbor was generally low, with a cumulative count of just 77 post-settlement snapper from 280 two-min video segments analyzed across all sites and treatments (using the maximum number of post-settlement snapper observed during each two-min video segment). The number of post-settlement snapper caught in opera house traps in 2015 and 2016 was also low ($n = 34$ total snapper collected).

The experimental addition of structure (i.e., *Atrina* mimics) was a significant factor explaining the abundance of post-settlement snapper at some sites, with more snapper present next to structure (compared to control patches) on average. Specifically, snapper abundance varied by site ($P_{\mathrm{PERM}} = 0.013$), and there was a significant site × treatment interaction ($P_{\mathrm{PERM}} = 0.0496$). The significant interaction term was the result of greater post-settlement snapper abundance near structure, relative to controls, at sites C, D, and A (**Table 2**). Although there were also more post-settlement snapper next to structure than at controls at Sites B and F, the effect of structure was not significant at these two sites (note there was only one fish recorded at Site E) (**Table 2**). As mentioned above, the addition of structure prior to the 2016 sampling did not have a significant effect at any site on invertebrate communities, meiofauna or sediment chlorophyll *a* content; even at site C, where the response of post-settlement snapper to the added artificial structure was the greatest.

Relationship Between Post-settlement Snapper Abundance and Environmental Variables

Although the resemblance matrix for total SNA present at each site was correlated with the resemblance matrix of environmental variables [Spearman rank correlation method, rho = 0.794, significance level 4.1% (i.e., $\alpha = 0.05$)], the resemblance matrix for SNA diff (i.e., indicative of variation in the relative importance of structure to snapper across sites) was not significantly correlated with the resemblance matrix of environmental variables [Spearman rank correlation method, rho = 0.212, significance level 30.18% (i.e., $\alpha > 0.05$)].

The best single environmental predictor of the response of post-settlement snapper to structure (SNA diff) across sites was the bottom water abundance of zooplankton items found to dominate post-settlement snapper diet (here calanoid and cyclopid copepod, and unidentified crustaceans) ($p = 0.08$, $r^2_{\mathrm{adj}} = 0.59$; **Figures 3**, **4**). Snapper diet was pooled across all sites and both the 2015 and 2016 sampling events due to low sample size.

Overall, a two parameter model with zooplankton abundance and water column turbidity (K_d) had greatest explanatory power ($p = 0.01$, $r^2_{\mathrm{adj}} = 0.98$), with zooplankton abundance (parameter estimate 1.76, $p = 0.0057$) and turbidity (parameter estimate −0.84, $p = 0.0163$) showing positive and negative associations with the response of post-settlement snapper to structure (SNA diff), respectively. A model with organic matter content and GPP also performed well (parameter estimates 0.84 [$p = 0.0036$] and −0.71 [$p = 0.0037$], respectively; model $p = 0.0044$, $r^2_{\mathrm{adj}} = 0.9913$).

DISCUSSION

Understanding the environmental and habitat drivers of abundance of a nursery stage juvenile fish has potential benefits for managers considering the best approaches to sustain fish populations (Adams et al., 2006; Nagelkerken, 2009). In the present study, the overall number of post-settlement snapper observed was low. For example, the average number of post-settlement snapper within 2 min video segments was <1, whereas a similar study using artificial seagrass units in a nearby clear water harbor observed c. 6 snapper per 2 min video segment (Parsons et al., 2018). Despite this low abundance, the influence of the structure that we added to the seabed was strong (65 post-settlement snapper associated with structure vs. 12 from bare controls). This indicates the critical importance of structure as a driver of post-settlement snapper abundance, even within a somewhat degraded estuary (Lowe et al., 2015). The strong affinity to structure is also consistent with previous studies on fishes generally. For example, habitat structural complexity is an important determinant of fish assemblages on Hawaiian coral reefs and artificial reefs in the British Virgin Islands, and the abundance of young of the year rock sole (*Lepidopsetta polyxystr*a) in Alaska (Friedlander and Parrish, 1998; Gratwicke and Speight, 2005; Stoner et al., 2007). Furthermore, multiple other studies conducted on post-settlement snapper have found similar results where higher abundances of snapper are associated with biogenic structures such as sponges or topographic complexity (Thrush et al., 2002; Compton et al., 2012; Lowe, 2013; Parsons et al., 2016), or artificial structures such as seagrass or bivalve mimics (Usmar, 2009; Parsons et al., 2013).

While the response to structure was undoubtedly the strongest driver of post-settlement snapper abundance, other environmental or habitat variables also had some influence. This was indicated by between site variation in the total abundance of post-settlement snapper (Total SNA) and the relative affinity of post-settlement snapper for structure between sites (SNA diff). Some previous studies conducted on snapper in New Zealand and Australia and the similar *Pagrus major* from Japan, have suggested that such variation in juvenile abundance is largely a response to benthic food availability (Tanaka, 1985; Francis, 1995, 1997; Sudo and Azeta, 2001; Hamer and Jenkins, 2004; Saunders et al., 2012). This is unlikely to be true for the

TABLE 1 | Pearson product-moment correlation coefficients for environmental variables, based on averages calculated at each of the five study sites.

	Mud	Org	Chl*a*	S	N	GPP	GPP/Chl*a*	Chl*a*′	S′	N′	MeioN′	Zoopl	Avg flow	Max flow	Food flux	K_d	Tot SNA	SNA diff
Mud		0.94	−0.72	0.16	−0.67	0.04	0.48	0.61	0.03	−0.67	−0.21	0.95	0.01	−0.17	0.48	0.84	0.77	0.74
Org	0.94		−0.90	0.46	−0.49	0.19	0.71	0.31	−0.28	−0.88	−0.47	0.87	0.28	0.15	0.66	0.83	0.70	0.64
Chl*a*	−0.72	−0.90		−0.74	0.09	−0.20	−0.79	0.11	0.54	0.96	0.64	−0.66	−0.53	−0.55	−0.76	−0.65	−0.62	−0.53
S	0.16	0.46	−0.74		0.29	0.45	0.78	−0.57	−0.80	−0.81	−0.64	0.17	0.95	0.91	0.88	0.41	0.13	−0.05
N	−0.67	−0.49	0.09	0.29		−0.36	−0.25	−0.79	−0.19	0.20	0.02	−0.54	0.21	0.65	−0.12	−0.71	−0.16	−0.17
GPP	0.04	0.19	−0.20	0.45	−0.36		0.75	−0.26	−0.75	−0.47	−0.70	−0.18	0.61	0.27	0.46	0.42	−0.54	−0.63
GPP/Chl*a*	0.48	0.71	−0.79	0.78	−0.25	0.75		−0.27	−0.85	−0.93	−0.88	0.31	0.73	0.55	0.77	0.65	0.06	−0.05
Chl*a*′	0.61	0.31	0.11	−0.57	−0.79	−0.26	−0.27		0.72	0.15	0.53	0.64	−0.56	−0.84	−0.13	0.48	0.47	0.51
S′	0.03	−0.28	0.54	−0.80	−0.19	−0.75	−0.85	0.72		0.69	0.93	0.20	−0.79	−0.78	−0.56	−0.19	0.32	0.41
N′	−0.67	−0.88	0.96	−0.81	0.20	−0.47	−0.93	0.15	0.69		0.75	−0.57	−0.67	−0.57	−0.84	−0.73	−0.42	−0.30
MeioN′	−0.21	−0.47	0.64	−0.64	0.02	−0.70	−0.88	0.53	0.93	0.75		0.03	−0.56	−0.59	−0.44	−0.26	0.17	0.19
Zoopl	0.95	0.87	−0.66	0.17	−0.54	−0.18	0.31	0.64	0.20	−0.57	0.03		0.01	−0.15	0.51	0.80	0.89	0.83
AvgFlow	0.01	0.28	−0.53	0.95	0.21	0.61	0.73	−0.56	−0.79	−0.67	−0.56	0.01		0.83	0.87	0.41	−0.11	−0.31
MaxFlow	−0.17	0.15	−0.55	0.91	0.65	0.27	0.55	−0.84	−0.78	−0.57	−0.59	−0.15	0.83		0.60	−0.01	−0.03	−0.16
FoodFlux	0.48	0.66	−0.76	0.88	−0.12	0.46	0.77	−0.13	−0.56	−0.84	−0.44	0.51	0.87	0.60		0.77	0.33	0.13
K_d	0.84	0.83	−0.65	0.41	−0.71	0.42	0.65	0.48	−0.19	−0.73	−0.26	0.80	0.41	−0.01	0.77		0.48	0.35
Tot SNA	0.77	0.70	−0.62	0.13	−0.16	−0.54	0.06	0.47	0.32	−0.42	0.17	0.89	−0.11	−0.03	0.33	0.48		0.98
SNA Diff	0.74	0.64	−0.53	−0.05	−0.17	−0.63	−0.05	0.51	0.41	−0.30	0.19	0.83	−0.31	−0.16	0.13	0.35	0.98	

All data collected from this study can be found in the ***Supplementary material****. Mud, sediment mud content (%); Org, sediment organic matter content (%); Chla, sediment chlorophyll a content (μg g^{-1}); S, sediment invertebrate species richness in 2015; N, sediment invertebrate abundance per core in 2015; GPP, gross primary production (μmol O_2 m^{-2} h^{-1}); GPP/Chla, GPP standardized per unit of sediment Chla; S′, sediment invertebrate species richness in 2016; N′, sediment invertebrate abundance per core in 2016; MeioN, abundance of meiofauna per core in 2016 (ostracods, nematodes, and benthic copepods); Zoopl, abundance of zooplankton items that dominate post-settlement snapper diet (here calanoid and cyclopoid copepods and unidentified crustaceans); Avg flow, average current velocity m s^{-1}; Max flow, maximum current velocity m s^{-1}; Food flux, Zoopl × Avg flow; K_d, light attenuation coefficient (proxy for turbidity); Tot SNA, total abundance of post-settlement snapper at a site; SNA diff, abundance of post-settlement snapper amongst structure minus abundance in control plots (proxy for relative affinity for structure at a site).*

TABLE 2 | Effects of experimentally added structure on the abundance of post-settlement snapper (Avg SNA), assessed using non-parametric permutational multivariate analysis of variance (PERMANOVA, Primer 7).

Source	df	SS	MS	Pseudo-F	P(perm)	Unique permutations
(A) Significance of main effects and interaction term						
Site	4	7.64	1.91	5.78	0.0130**	9967
Structure	1	5.43	5.43	3.95	0.1334	5083
Site × structure	4	5.51	1.38	4.16	0.0496**	9964
Res	20	6.62	0.33			
Total	29	25.20				

Within level	P(MC)
(B) Significance of structure vs. control within "Site" (i.e., pairwise *post-hoc* tests)	
Site A	0.028**
Site B	ns
Site C	0.030**
Site D	0.084*
Site E	0.377

*Site (5 levels, random), Treatment (structured vs. unstructured controls, fixed), and the interaction between Site and Treatment. **P < 0.05, *P < 0.10.*

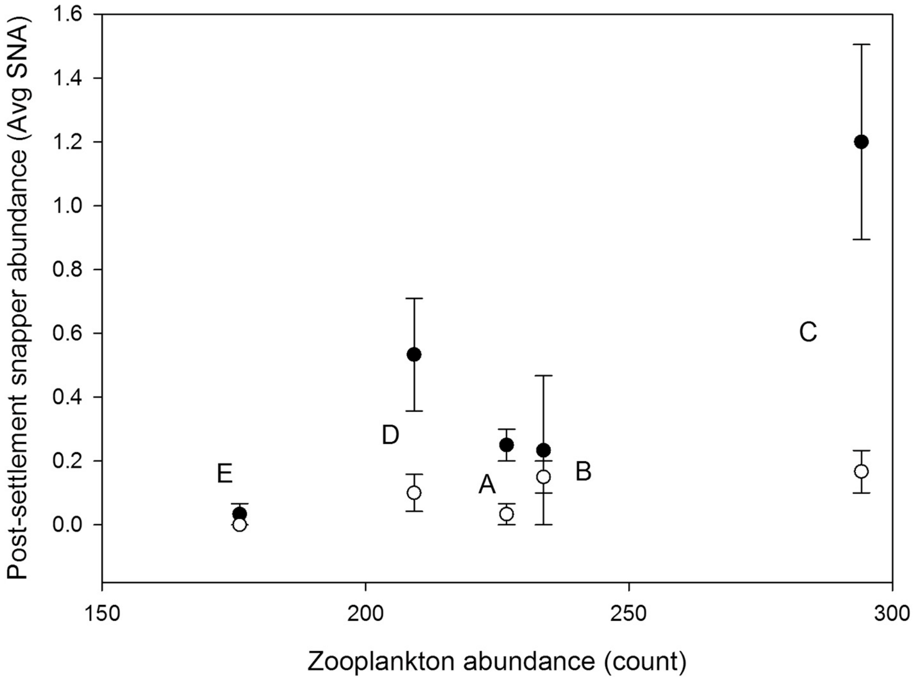

FIGURE 3 | Average abundance of post-settlement snapper (Avg SNA) in structured (filled circle) and control (i.e., unstructured, open circle) habitat patches at each of the five study sites (E, A, D, B, C; labeled next to relevant data points). Error bars are ±1 standard error. The post-settlement snapper data are plotted vs. the average abundance of zooplankton items which dominate post-settlement snapper diet (calanoid and cyclopid copepods, and unidentified crustaceans) collected from near-bed bottom water at each site (see section Materials and Methods and **Figure 4**).

present study, as post-settlement snapper diet (pooled across all sites) was dominated by pelagic zooplankton (not benthic infauna), the abundance of which was the most influential single explanatory variable of snapper abundance across sites. Alternatively, benthic invertebrates were observed to be poor predictors of snapper abundance. Therefore, it appears that the lack of affinity for structure and the preference for benthic food sources in these previous studies may be explained by a focus on larger juvenile snapper, which are known to prefer benthic food sources (Usmar, 2012), or entirely different life history patterns existing for juvenile snapper in different geographic regions.

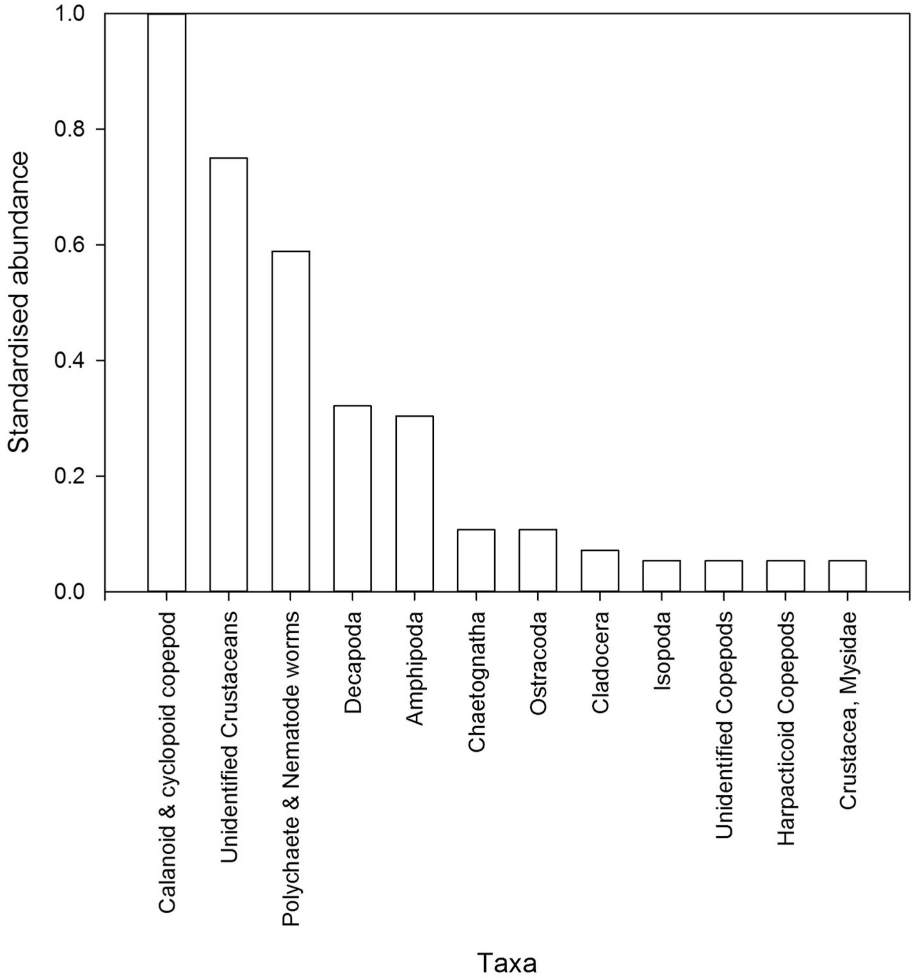

FIGURE 4 | Abundance of invertebrates from post-settlement snapper gut contents ($n = 35$ individual fish) collected from all sites in 2015 and 2016. Abundances are standardized by the most abundant taxa and ordered from most to least abundant. Taxonomic nomenclature presented is not always at the same level, and in some cases has been pooled to likely snapper diet groups.

Previous research has suggested that locations with high water flow may offer post-settlement snapper an increased flux of the pelagic zooplankton they prefer to eat, while benthic structure may help to reduce energy expenditure by providing shelter from fast water flow at these sites (Parsons et al., 2015, 2018). Other research has suggested that sediment induced turbidity is highly influential on post-settlement snapper health and abundance, and can cause a switch from a predominantly pelagic to predominantly benthic diet (Lowe et al., 2015). As such, as part of the present study we sought to quantify environmental characteristics, including benthic and water column food availability, turbidity and current speeds that potentially explain variation in the response of post-settlement snapper to structure across sites.

Our results demonstrated that the abundance of zooplankton (the post-settlement snapper diet component) was highest in muddy, organically enriched, inner estuary sites. While there is no immediate explanation for this relationship, post-settlement snapper abundance was positively related to zooplankton abundance (as described above). Alternatively, variables such as average current speed, maximum current speed, and food flux (zooplankton abundance multiplied by average current speed) were uncorrelated to post-settlement snapper abundance and structure use, and were among the weakest explanatory variables in our dataset. The lack of relationship between post-settlement snapper and food flux could be due to a number of explanations, including the potential for vertical migrations of zooplankton to disconnect water flow and accessible food abundance (Kimmerer et al., 2014). Beyond this, when turbidity was considered together with zooplankton abundance, it had a negative relationship with snapper abundance. Together these results confirm that post-settlement snapper respond to structure and pelagic food abundance (Parsons et al., 2015, 2016), however, at least in Mahurangi Harbor, this relationship isn't mediated by energetic gains related to food flux or hydrodynamic shelter provided by structure as it is for other fish species (Fausch and White, 1981). The negative response to turbidity further emphasizes the importance of pelagic over benthic food sources for the post-settlement stage, and is well aligned with the findings of Lowe et al. (2015), who also found a predominance of pelagic prey in the Mahurangi Harbor snapper analyzed in that study.

It is important to note the limitations of the dataset used in the present study. We collected environmental data between March 2015 and March 2017, which we compared with post-settlement snapper abundance and structure use from March 2016. Moreover, having to condense environmental variables down to simple site averages resulted in a loss of information and weakened our inferential power. A greater number of sites would have been beneficial, as this would have allowed us to include more than two variables within regression models, but was unachievable due to the labor intensive nature of conducting a controlled structure manipulation experiment by SCUBA.

Overall our results provide strong evidence that access to benthic habitat structure and an abundance of zooplankton prey, such as calanoid and cyclopoid copepods, are of primary importance to post-settlement stage snapper. Both the presence of biogenic structure and the ability to visually feed on zooplankton prey require relatively clear water with low sediment induced turbidity (Ellis et al., 2002; Thrush et al., 2004; Waycott et al., 2009), which was also identified as negatively related to post-settlement snapper abundance in the present study. Therefore, the identification and management of the most productive snapper nursery habitats (Parsons et al., 2014) should center on the three variables identified here, namely, turbidity, habitat structure and zooplankton abundance. Because turbidity, and to some extent the presence of habitat structure, can be estimated remotely, these variables could serve as starting points that are later field verified and supplemented with zooplankton abundance data. While the relationship between post-settlement snapper, turbidity and zooplankton are well described, we are now potentially less certain as to why post-settlement have such a high affinity for structure. The positive effect that structure may provide with regard to predation avoidance and food provision are obvious candidates (Beck et al., 2001), but it is the specific mechanisms (Nagelkerken, 2009) which remain elusive and yet may provide important context on the path to more effective nursery habitat conservation or restoration.

AUTHOR CONTRIBUTIONS

AL and DP conceived the study and designed the experiment. AL, DP, LM, DB, and IM collected data and processed samples. AL and DP analyzed the data and wrote the manuscript, LM, DB, and IM wrote sections of the manuscript and provided revisions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00427/full#supplementary-material

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Some Old Movies Become Classics – A Case Study Determining the Scientific Value of ROV Inspection Footage on a Platform on Australia's North West Shelf

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The global oil and gas industry holds a vast archive of Remotely Operated Vehicle (ROV) inspection footage potentially containing useful long-term data on marine biological communities. With the upcoming era of decommissioning of oil and gas structures, it is timely to assess the usefulness of this footage for researching these communities. We used ROV inspection footage to characterize the sessile invertebrates and fishes associated with the Goodwyn Alpha Production Platform (GWA) on the North West Shelf of Australia between depths of 10 and 125 m during 2006 and 2008. Depth was a major driver of invertebrate assemblages, most likely due to specific requirements such as light, and differences between years were most likely from the physical detachment of species by cyclones and internal waves. Phototrophic species were mostly limited to the upper 50 m of the platform, including the hard coral *Pocillopora* sp. and the soft corals *Nephthea* sp. and *Scleronephthya* sp. In contrast, heterotrophic species including sponges, anemones, bryozoans, hydroids, bivalves such as *Lopha folium* and the hard coral *Tubastrea* spp., were distributed across all depths. We observed 1791 fish from at least 10 families and 19 species, including commercial species such as crimson seaperch (*Lutjanus erythropterus*), red emperor (*L. sebae*), saddle-tailed seaperch (*L. malabaricus*), mangrove jack (*L. argentimaculatus*) and trevally (*Caranx* spp.). Fish density increased significantly with depth during 2008, from a mean of 23 fish/50 m^2 between 10 and 25 m to 3373 fish/50 m^2 at 125 m, where small unidentified baitfish were abundant. The highest densities of commercial species occurred between 25 and 75 m depth, suggesting that mid-depth platform sections had high habitat value, a consideration when selecting decommissioning options. The greatest difficulties using the video were the poor lighting and resolution that inhibited our ability to identify sessile species with high taxonomic precision. However, the footage was useful for evaluating high-level biodiversity of the platform, understanding how fish and invertebrate communities changed with depth and comprehending the dynamic nature of the invertebrate community over time. Understanding the habitat value of structures will be necessary for making environmentally sound decommissioning decisions in the future.

Keywords: ROV video, sessile invertebrates, commercial fish species, fish, oil platform, rigs to reef, decommissioning

INTRODUCTION

Subsea infrastructure of the oil and gas industry (hereafter 'oil structures') has proliferated in continental boundaries worldwide, with reportedly more than 7,500 structures around the world (Parente et al., 2006). Over the next two decades, most oil structures will reach end of life and require decommissioning. Decommissioning has historically involved complete removal, due to the assumption that removal represents best environmental practice. However, evidence from some oil and gas-producing nations indicates structures can act as artificial reefs, providing habitat for endangered cold-water corals (Bell and Smith, 1999; Gass and Roberts, 2006) and by developing dense encrusting invertebrate communities (Forteath et al., 1982; Friedlander et al., 2014; Kolian et al., 2017) that provide habitat for other marine organisms, including commercially valuable fish species (Rooker et al., 1997; Stanley and Wilson, 1997; Love et al., 2000). These discoveries suggest that under at least some circumstances, removal of oil and gas infrastructure will result in the loss of valuable habitat for ecologically and commercially important species. In Australia, and most parts of the world, these offshore platforms are not fished and are surrounded by a 500 m exclusion zone making them quasi-marine protected areas or no-take zones. Given some of these platforms have been in place for decades, they have the potential to play a significant role in the population dynamics of key species at a local or even regional scale.

The potential for oil structures to provide important habitat has led to the development of 'rigs-to-reefs' (RTR) programs in some regions. RTR programs operate on a 'win–win' premise, whereby obsolete structures are recycled as artificial reefs, while also providing substantial cost savings to industry (Macreadie et al., 2011). RTR programs have become increasingly popular since their inception in Louisiana in 1986, and have spread to numerous other US states, as well as Southeast Asian nations (Kaiser, 2006). The phrase 'RTR' is now frequently used to describe any decommissioning practice that does not involve complete removal of oil structures, including leaving part of a structure *in situ*.

Despite growing popularity, RTR programs remain controversial. They have historically been viewed as an opportunity for oil companies to avoid their removal obligations, particularly by environmental NGOs and the general public (Jørgensen, 2012). This has resulted in strong opposition to RTR conversions in particular regions (e.g., the North Sea), and heated public debate concerning the instigation of any new RTR program. Before a new program can be established, considerable scientific evidence is required to determine whether oil structures will provide a net benefit to the marine environment (Schroeder and Love, 2004). For example, over a decade of research was required to demonstrate that oil platforms in Southern California provide important habitat for overfished rockfish populations before legislation allowing RTR conversions could be passed in that region (Assembly Bill 2503, 2010).

In Australia, the North West Shelf (NWS) Venture is supported by numerous platforms, well heads, production areas and support facilities connected by hundreds of kilometers of pipelines. Due to a dearth of research, very little is known about the environmental benefits of Australia's offshore oil and gas infrastructure and there is little information to inform policy development or guide upcoming decommissioning decisions. Recent studies suggest that infrastructure on the NWS is well colonized, providing habitat for sessile marine invertebrates and fishes (McLean et al., 2017, 2018b). However, understanding the habitat value of these structures with respect to decommissioning decisions will require an understanding of the variability and transiency of these communities, both with depth and through time. With particular reference to platforms, understanding the distribution of the sessile invertebrate community over depth will be important when considering decommissioning options (Claisse et al., 2015). For example, one decommissioning option for platforms is 'topping,' that is, following the removal offsite of the above-water structure (or topsides), a section of the underwater supporting frame (known as the jacket) is removed down to a safe navigable depth (˜30 m) and placed on the seabed alongside the structure (Macreadie et al., 2011). Would this negatively impact on the habitat value of the structure? Has the most productive and sunlit habitat been lost to greater depths? How will this affect associated fish communities and interactions with surrounding ecosystems? Answers to these questions are critical for determining the habitat value of oil structures and the relative environmental benefit of alternative decommissioning options (Fowler et al., 2014).

An inexpensive approach to assessing the habitat value of oil and gas infrastructure is to use industry inspection footage. Oil structures are regularly inspected for operational and engineering purposes using underwater video cameras mounted on Remotely Operated Vehicles (ROVs). The oil and gas industry worldwide holds a vast archive of inspection footage dating back decades (Macreadie et al., 2018). Although collected in an imperfect scientific manner, this archive contains footage of marine fauna associating with oil and gas structures and surrounding habitats, making it a potential source of long-term data for biological observation (Macreadie et al., 2018; McLean et al., 2018a). The footage also covers a depth range rarely achieved by independent scientific investigations, spanning from the surface to beyond 1,000 m depth. Such data sources are rare in our oceans and particularly valuable, given the impacts of climate change on biological communities and the limited information available on deep sea ecosystems.

To our knowledge, we present here the first study of the biological communities associated with an offshore oil and gas platform in Australia and contribute to the few other studies carried out on NWS oil and gas infrastructure (Fowler and Booth, 2012; Pradella et al., 2014; McLean et al., 2017, 2018b). The aims of this paper are to determine: (1) the composition of the invertebrate and fish communities associated with the platform; (2) how the fish community changes with depth; and (3) how the invertebrate community changes with depth and time. While addressing these aims, we provide a preliminary evaluation of the usefulness of industry ROV footage for researching platform communities. Understanding how these communities change over depth and time will be necessary for assessing the habitat value of this and other platforms, which will be essential for making environmentally sound decommissioning decisions in the near future.

MATERIALS AND METHODS

The Goodwyn Alpha production platform (GWA), operated by Woodside Energy Ltd., is located 135 km north-west of Karratha, WA, Australia, on the NWS in approximately 130 m of water. The platform is located in a subtropical environment and in the path of the southward flowing Holloway Current that transports warm, low salinity water southwards into the Leeuwin Current. Seawater temperature in the upper 100 m ranges between 22 and 30°C, peaking in April each year (Bahmanpour et al., 2016). GWA was commissioned in 1995 and has a complex subsurface structure consisting of horizontal, vertical and diagonal beams or members making up the steel frame or jacket. The jacket is periodically inspected by engineers using ROV footage for structural integrity and the extent of attached invertebrates which can result in increased hydrodynamic load on the structure.

A total pool of 8.5 h of digital footage of the north face or Row G of the GWA jacket was available from 2006 and 2008 (**Figure 1**). This face is on the leeward side of the platform and protected from the prevailing winds and swells from the south west. The footage was collected by ROV during standard industry inspections of the infrastructure. During inspections, the ROV was positioned so that the online inspection engineer had a clear view of the member being inspected, and could determine its condition. This also provided a useful view for surveying both fishes and attached sessile invertebrates. As the quality of the footage was variable and not taken for scientific purposes, the total footage pool was pre-screened to identify footage that adhered to the following criteria: (1) between 3 and 5 m from structure; (2) water visibility ≥ 5 m and; (3) ROV moving at a slow, constant speed. We estimate that 24% of the video pre-screened was unsuitable for our purposes.

Sessile Invertebrate Surveys

Sessile invertebrate communities were surveyed from multiple, replicate quadrats taken from survey footage of the horizontal members of the jacket (**Figure 1**). In 2006, surveys were carried out along horizontal members at depths of 15 m, 50 m, 90 m, and 125 m. Footage was taken between midnight and 03:00 h (Western Standard Time, WST) on August 18th, 2006. In 2008, footage was taken between 15:00 h and 18:00 h on the 20th December, with the exception of footage from 90 m depth, which was taken on the 12th December at 10:00 h. There was no suitable video footage available from the horizontal member at 125 m in 2008. The horizontal members were last cleaned in December 1998, therefore the ‘age’ of the attached community was approximately 10 years by December 2008, notwithstanding changes in composition through natural disturbances (e.g., the passage of cyclones). There was no footage available from Row G between the years we surveyed.

Survey or transect lengths varied according to the length of horizontal members which increased with depth (**Supplementary Table S1a**). In 2006, transects ranged between 30 m and 65 m. In 2008, while there was no suitable video footage available from the horizontal member at 125 m in 2008, at least 70% of the remaining members were surveyed for sessile invertebrates, resulting in survey lengths of between 30 m and 50 m on the upper three members.

Still frames from the transects of the horizontal members were used to set quadrats within which the sessile invertebrates were identified and their coverage quantified. To optimize the number of quadrats from the footage, frames were not random but captured sequentially every 15–30 s as the previous frame of the component was cleared from the field of view. Frames were captured using VLC media player (v. 2.0.5) and then imported into Coral Point Count with Excel Extensions V4.1 (CPCe) (Kohler and Gill, 2006). Within CPCe, a square quadrat border was set for each sample to the approximate diameter of horizontal member taken from engineering diagrams of the platform. This yielded quadrats of approximately 1.25 m^2 at 15 m and 50 m depth, 1.5 m^2 at 90 m and 1.75 m^2 at 125 m. At least 10 replicate quadrats were examined at each depth in each year, with the exception of 125 m depth (**Figure 1** and **Supplementary Table S1a**). Prior to the video analysis, we determined the optimal number of random points per quadrat necessary to most accurately determine sessile invertebrate cover while reducing counting effort. To do so, we chose nine quadrats from GWA with low, moderate and high cover of the easily identified coral species *Tubastrea* sp. We then scored the presence of *Tubastrea* sp. in quadrats within CPCe under an increasing number of random points that increased in 5 point steps to 50, and constructed a performance curve of score standard deviation versus random point number (Brown et al., 2004). We found that from 25 random points per quadrat, increased counting effort did not yield better estimations of *Tubastrea* coverage, thus chose this number as optimal for describing sessile invertebrate cover. To aid identification, the corresponding video was played

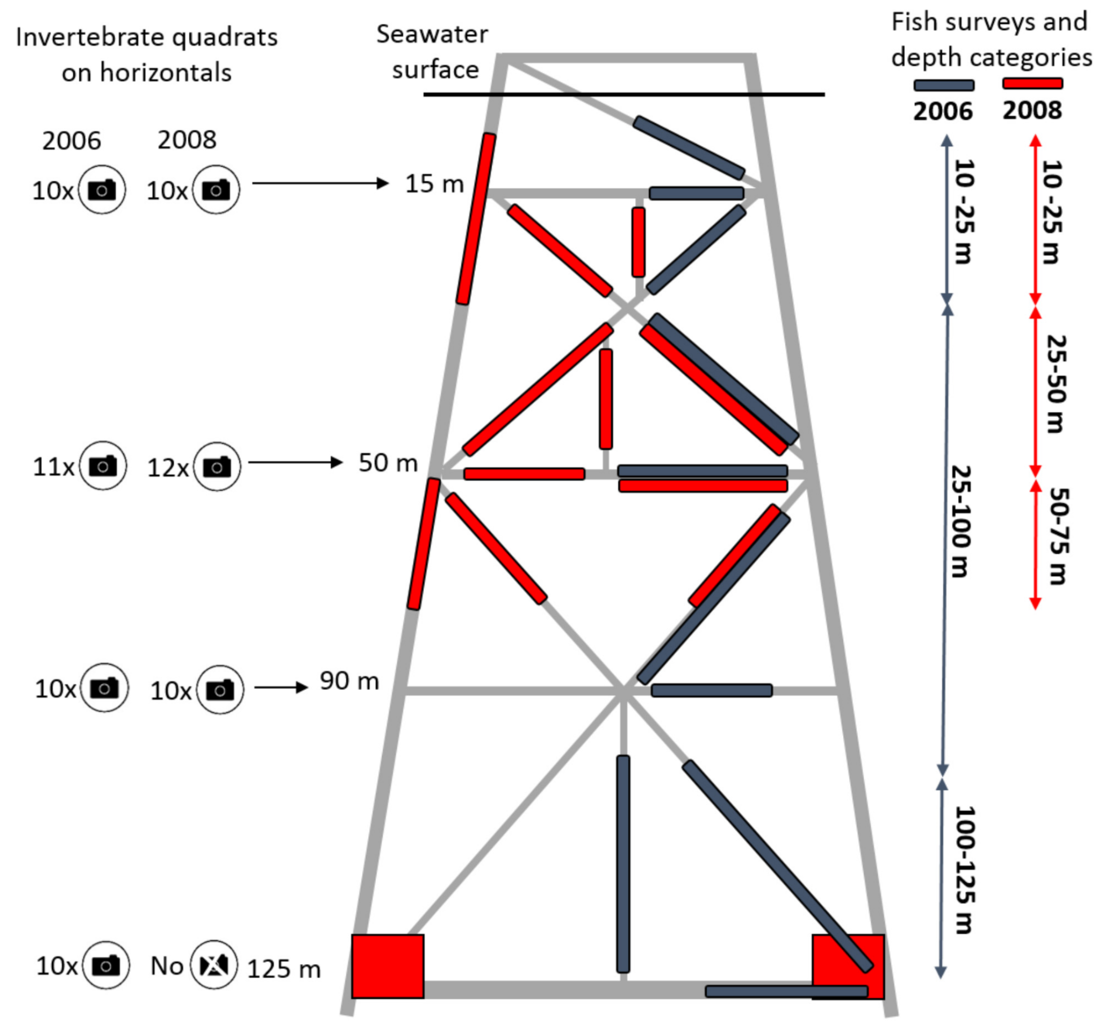

FIGURE 1 | Schematic diagram of Goodwyn Alpha (GWA) platform showing its structure including the horizontal, vertical, and diagonal cross beams surveyed for marine life. Horizontal beams or members at 15, 50, 90, and 125 m were surveyed for attached sessile invertebrates in 2006 and 2008, with the exception of the 125 m depth in 2008 when footage was not available. Fish transects are shown as blue (2006) and red bars (2008). Red squares indicate the location of fish counts from stationary images conducted at the base of the structure during 2008. Vertical arrows indicate the depth zones used for analysis in each year. The number associated with the camera icon 📷 indicates the number of quadrats per horizontal member used in the invertebrate surveys in each year. Drawing not to scale.

concurrently when growth on the still frames was indistinct under the points. Where possible, we used still photographs of members taken during the survey and photographs of samples returned to the surface from related engineering reports from GWA in 2008 to confirm video identifications (Woodside, 2008; Surespek Iss PTY Ltd, 2009). For classifying and identifying the invertebrates, we used the Collaborative and Automated Tools for Analysis of Marine Imagery (CATAMI) system, which introduces an Australian-wide acknowledged, standardized terminology for annotating benthic substrates and biota in marine imagery (Althaus et al., 2015).

Fish Surveys

Video used in assessing fish density included that of horizontal, vertical, and diagonal jacket members over a greater range of depths than used for the invertebrates to increase encounter rates and to account for movement (**Figure 1**). Fishes were surveyed from footage taken on the August 18th, 2006 and December 20th, 2008. Footage from 2006 was taken at night (midnight – 05:30 h, WST) while footage from 2008 was taken during daylight hours (15:30–18:00 h). Transect lengths varied according to member length and the available useable footage, but at least three transects were examined within each depth category (**Supplementary Table S1b**). In 2006, we surveyed approximately 146 m^2 of platform, with transect lengths ranging between 20 and 40 m. In 2008, suitable footage was lacking from depths > 90 m. Therefore we used frames from stationary footage taken at the base of the legs at 125 m on the northeast and northwest corners of Row G (**Figure 1**). This footage was taken 3 months earlier on September 12th, 2008. Consequently, in 2008, we surveyed approximately 366 m^2 of platform with transects of between 19 and 30 m length (**Supplementary Table S1b**).

The depth categories differed between years due to the lack of suitable replicates required for statistical analyses in the 50–75 m category in 2006, requiring the combination of transects between the 25 and 100 m. Some footage along the horizontals used for the invertebrate surveys were reused for surveying fish where suitable.

For each transect, fish counts began from a freeze frame generally at one end of the member. We then advanced the footage frame by frame and counted additional fish as they entered the view in front of, or to the side of the member (Caselle et al., 2002). Individuals were identified to the lowest possible taxonomic level. In 2008, fish were counted from freeze frames of the seafloor infrastructure as this footage was largely stationary. To standardize the reporting of fish densities, surface areas of the members were calculated from known dimensions of the infrastructure (GWA specifications, Woodside Energy Ltd.) and densities are reported as abundance per 50 m^2.

Statistical Analysis

Percentage cover of invertebrate species was calculated for each quadrat within CPCe and each quadrat was used as an individual sample. Invertebrate communities were compared between depths and years using the statistical packages PRIMER (v6.1.14) and PERMANOVA (v1.0.40) (Clarke, 1993; Clarke and Gorley, 2006; Anderson, 2017). Percent coverage was square-root transformed to ensure that the most abundant species did not unduly influence comparisons and a similarity matrix was constructed using the Bray–Curtis similarity coefficient. Permutational multivariate analysis of variance (PERMANOVA) was used to test the significance of differences (alpha: 0.05, 9999 permutations) between community composition at each depth within and between years. Pairwise comparisons between depths within and between years were performed using pairwise PERMANOVA tests which employ a pseudo-*t* statistic. A Multi-Dimensional Scaling (MDS) plot was used to visualize differences in the marine growth among depths and between years. Similarity Percentage analysis (SIMPER) within PRIMER was used to identify which species/groups contributed most to community differences between depths and years. A cut-off of 80% species contribution was used in SIMPER analyses.

As our footage in alternate years was either diurnal or nocturnal, and as time of day is known to affect fish behavior, fish communities were compared among depths within each year separately. Multivariate analyses were similar to those used for the sessile invertebrates. Fish density was $\log(x + 1)$ transformed and a Bray–Curtis similarity matrix was constructed. Community composition was compared among depths using the PERMANOVA routine and SIMPER analysis was used to identify species contributing up to 80% community differences between depths. Total fish density was compared among depths in 2008 using a one-way ANOVA, following inspection of the data for normality and homogeneity of variances. Tukey's *post hoc* tests were used to identify differences between specific depth categories.

RESULTS

Sessile Invertebrates

Community Composition

We identified the invertebrate communities into broad functional and taxonomic categories, genera and species where possible (**Table 1** and **Figure 2**). The broadest functional categories included two underlying encrusting or enveloping invertebrate and crustose algal layers. These layers were overlaid with hard and soft corals, anemones, bivalves and bryozoans and/or hydroids. Few taxa could be identified to higher taxonomic categories due to low camera resolution and lighting; however, we have included the CATAMI codes for each taxa where possible (**Table 1**).

The encrusting layer on the upper 50 m of the structure consisted primarily of a calcite layer of intact bivalves and bleached shells (**Figure 2A**). While the bivalves were not always clearly distinguishable in the footage, physical samples from industry reports in 2008–2009 supported these observations and confirmed that above approximately 60 m depth, the encrusting layer consisted primarily of living and dead bivalves, including species such as *Pinctada* sp. and *Lopha folium* of up to 190 mm diameter (Surespek Iss PTY Ltd., 2009). On the lower structure, from depths of >50 m, the enveloping layer was a colorless mat with filamentous appearance but with few other distinguishing features. Calcareous coralline algae and sponges were also distinguishable but could not be identified further.

Scleractinian stony or hard corals and octocorals (soft corals) were identified in both years (**Table 1**). Hard corals included two species of *Tubastrea* (**Figure 2A**) and one of *Pocillopora* sp. and photographs from industry samples confirm these observations (Surespek Iss PTY Ltd, 2009). Soft corals included two species of *Scleronephthya* and one of *Nephthea* (**Figure 2B**). The remaining cnidaria were the solitary true anemones, consisting of a fleshy stalk and a ring of tentacles surrounding a central mouth (**Figure 2C**). As we could not distinguish between bryozoans and hydroids with any confidence due to the poor resolution and lighting in the footage, we combined them into one category.

Community Changes With Depth and Time

Overall, we found a progressive change in the sessile invertebrate community with depth in both years, with the greatest changes evident between depths less than and greater than 50 m (**Table 1** and **Figures 3A,B**). In 2006, coverage at 15 m depth was dominated by the calcite encrusting layer (42.0%), along with 37.7% of hard coral (mostly 24.1% of *Tubastrea* sp. 2 and 12.7% of *Pocillopora* sp.) and 15.8% soft corals (mostly 12.5% of *Scleronephthya* sp. 1). Coverage at 50 m was dominated by soft corals, including *Scleronephthya* sp. 1 and *Nephthea* sp. (51.4% and 7.8%, respectively), the calcite layer (22.5%) and the hard coral *Tubastrea* sp. 2 (10.5%). At 90 and 125 m, filamentous mat coverage was 51.8% and 65.4%, respectively, hard corals were absent and soft coral cover was 42.0% at 90 m and 28.8% at 125 m. Groups with minor coverage included unidentified hard and soft corals (0.5% and $\leq$8.5%, respectively), sponges at 90 m depth (1.2%) and the bryozoans/hydroids

TABLE 1 | Invertebrate taxa and their mean percent cover on the horizontal members of Goodwyn Alpha platform in 2006 and 2008 (standard deviation shown in brackets).

Invertebrates	**Year 2006 and depth (m)**				**Year 2008 and depth (m)**			**CAAB#**
Encrusting/enveloping	**15**	**50**	**90**	**125**	**15**	**50**	**90**	
Calcite encrusting layer	42.0 (14.9)	22.5 (22.8)			32.5 (15.4)	2.8 (5.0)		
Coralline algae					12.6 (10.7)	0.4 (1.3)		80 300934
Sponges			1.2 (2.7)			5.2 (9.1)	50.0 (17.0)	10 000921
Filamentous mat			51.8 (14.0)	65.4 (27.9)			14.7 (21.9)	
Hard corals								
Tubastrea sp. 1	0.8 (1.7)				6.5 (10.0)			11 290901
Tubastrea sp. 2	24.1 (15.3)	10.5 (9.5)			22.2 (15.3)	45.6 (14.3)		11 290901
Pocillopora sp.	12.7 (13.5)				19.7 (13.9)	2.4 (7.6)		11 290912
Unidentified		0.5 (1.6)			6.5 (10.9)	6.4 (14.0)	3.7 (5.6)	
Soft corals								
Scleronephthya sp. 1	12.5 (14.8)	51.4 (14.3)						11 168911
Scleronephthya sp. 2			39.3 (14.0)	19.9 (30.8)		0.4 (1.3)	0.3 (1.2)	11 168911
Nephthea sp.	3.3 (10.5)	7.8 (14.1)	0.8 (1.7)	0.4 (1.3)		19.2 (18.8)		11 168911
Unidentified			1.9 (2.7)	8.5 (10.1)				11 168909
Others								
True anemones						17.6 (20.3)	30.2 (19.1)	11 229903
Total bryozoan/hydroids	4.2 (10.2)	7.3 (15.3)	5.1 (9.6)	5.6 (7.4)				20 000905/ 11 001000

Collaborative and Automated Tools for Analysis of Marine Imagery (CATAMI) CAAB codes are provided where possible.

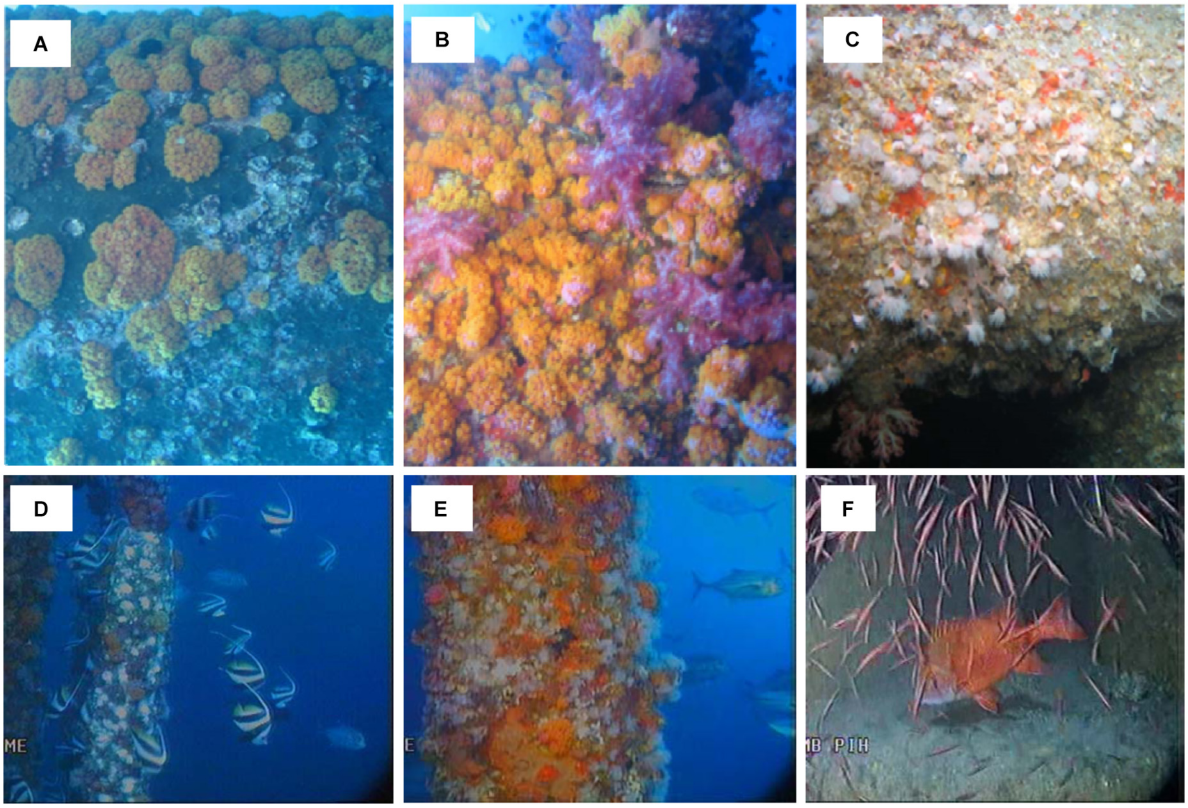

FIGURE 2 | Attached invertebrates on Goodwyn Alpha platform and associated fishes, including **(A)** the white calcite encrusting layer with bleached bivalves and the hard coral *Tubastrea* sp. 2, **(B)** the soft coral *Nephthea* spp. with the orange hard coral *Tubastrea* sp. 2, **(C)** true anemones, sponges and the filamentous layer, **(D)** bannerfish (*Heniochus* sp.), **(E)** trevally (*Caranx* spp.) and **(F)** a school of unidentified juvenile fish and a red emperor (*Lutjanus sebae*).

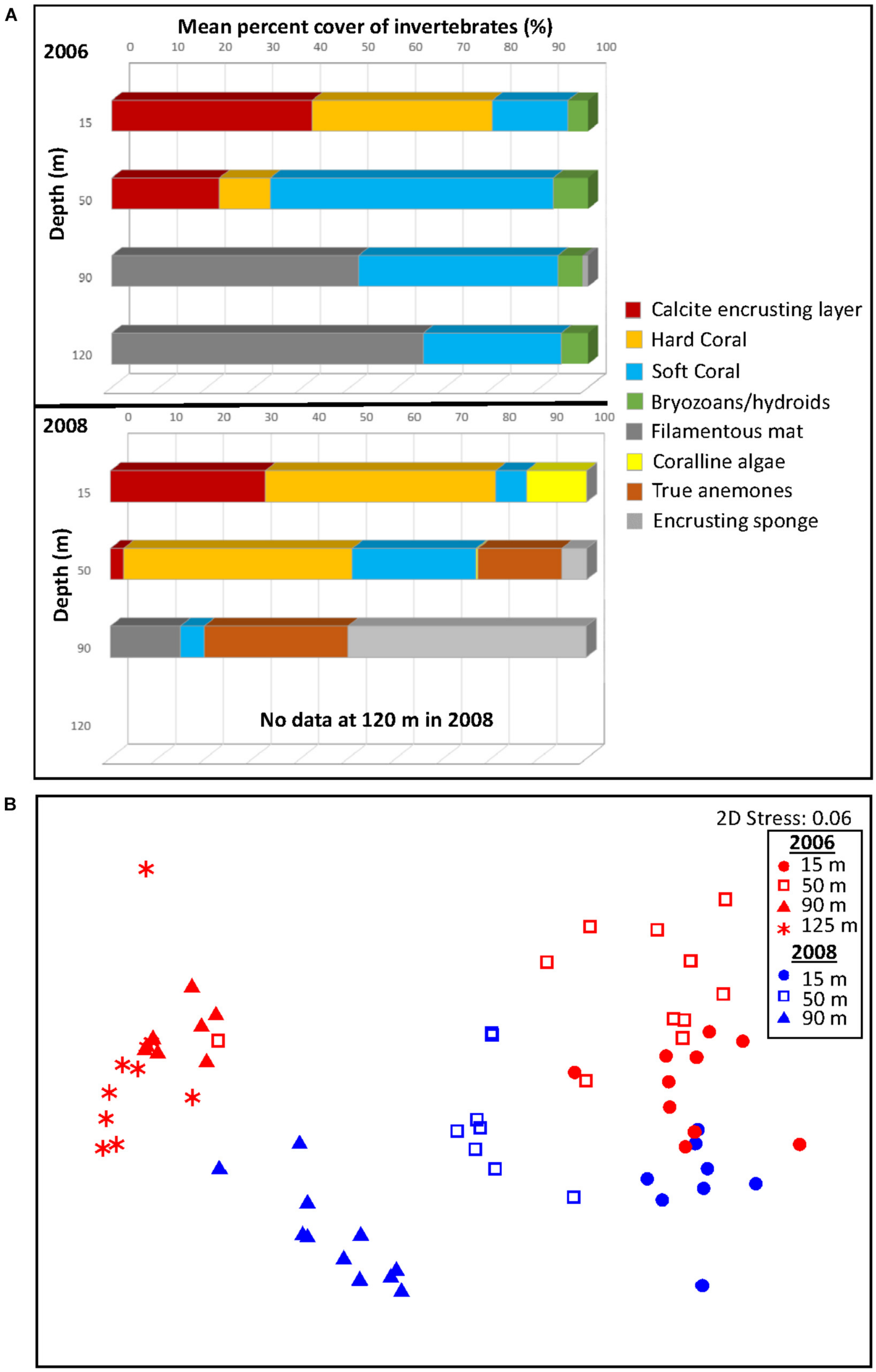

FIGURE 3 | Sessile attached invertebrate coverage and changes in coverage with depth and between years illustrated by **(A)** mean percent cover by the major invertebrate taxa/groups and **(B)** multidimensional scaling plot (MDS) of the sessile invertebrate species on the horizontal members of Goodwyn Alpha platform in 2006 and 2008. In the MDS, each data point represents a replicate quadrat. The MDS is constructed from Bray–Curtis similarities based on the percent coverage of different groups/species. There was no footage available for analysis of the sessile invertebrates from 125 m in 2008.

group which were present over all depths at ≤7.3% coverage (**Table 1**).

The percent coverage of the sessile invertebrate community differed significantly among depths in 2006 (PERMANOVA: pseudo-*F* = 31.461, *P* = 0.0001). Pairwise comparisons between progressive depths indicated that the community differed between depths of 15 m, 50 m, and 90 m. The community coverage between 15 and 50 m was significantly different (pairwise PERMANOVA, pseudo-*t* = 2.6, *P* = 0.0007), primarily due to an increase in coverage of *Scleronephthya* sp. 1 and a decrease in the coverage of the calcite layer and the hard corals *Tubastrea* sp. 2 and *Pocillopora* sp. which together contributed 79% of the dissimilarity between depths (SIMPER, **Table 2**). Community coverage did not differ between 90 and 125 m (pseudo-*t* = 1.2905, *P* = 0.2345). However, the community at 90 and 125 m differed considerably from that at 50 m (PERMANOVA: 50 vs. 90 m, pseudo-*t* = 6.2183, *P* = 0.0001; 50 vs. 125 m, pseudo-*t* = 5.2418, *P* = 0.0001). The difference in communities between 50 and 90 m was primarily due to the increase in coverage of the filamentous mat and the decrease in coverage of the soft coral *Scleronephthya* sp. 1 at 90 m, together contributing 47% of the dissimilarity between depths (**Table 2**). An increase in the cover of *Scleronephthya* sp. 2, a decrease in the cover of the calcite layer and a decrease in the hard coral *Tubastrea* sp. 2 at 90 m contributed to the remaining dissimilarity of 88%.

A similar pattern of community change across depth was observed in 2008 (**Figures 3A,B** and **Table 1**). Overall, at 15 m, coverage was dominated by hard corals at 54.9% (mostly *Tubastrea* sp. 2 and *Pocillopora* sp. at 22.2% and 19.7%, respectively) and the calcite layer (32.5%). At 50 m, the hard corals (mostly *Tubastrea* sp. 2, 45.6%), the soft coral *Nephthea* sp. (19.2%), and the anemones (17.6%) predominated. At 90 m, coverage was dominated by sponges (50.0%), anemones (30.2%) and filamentous mat (14.7%). Bryozoans/hydroids were apparently absent at any depth in 2008.

Community coverage differed significantly among depths in 2008 (PERMANOVA: pseudo-*F* = 39.897, *P* = 0.0001) (**Figures 3A,B**). Like 2006, pairwise comparisons confirmed community differences between each adjacent depth zone. The community coverage at 15 m and 50 m was significantly different (PERMANOVA: pseudo-*t* = 4.6912, *P* = 0.0002), due primarily to a decrease in coverage by the calcite layer which contributed most the dissimilarity between depths (18.8%; **Table 2**). Otherwise, a

TABLE 2 | Percent mean cover and Similarity Percentage results (SIMPER) for sessile invertebrates contributing up to 80% of the dissimilarity in coverage between adjacent depths on the Goodwyn Alpha platform in 2006 and 2008.

Year	Depth comp	Group/Taxa	% mean cover at depth		Contribution to dissimilarity %	Cumulative % contribution
			15 m	**50 m**		
2006	15 vs. 50 m	*Scleronephthya* sp. 1 – SC	12.5	51.4	27.01	27.01
		Calcite encrusting layer	42.0	22.5	19.61	46.62
		Tubastrea sp. 2 – HC	24.1	10.5	16.48	63.1
		Pocillopora sp. – HC	12.7	0	16.01	79.12
		Nephthea sp. – SC	3.3	7.8	9.39	88.51
			50 m	**90 m**		
	50 vs. 90 m	Filamentous mat	0	51.8	24.14	24.14
		Scleronephthya sp. 1 – SC	51.4	0	23.1	47.24
		Scleronephthya sp. 2 – SC	0	39.3	19.81	67.06
		Calcite encrusting layer	22.5	0	12.29	79.35
		Tubastrea sp. 2 – HC	10.5	0	8.18	87.52
			90 m	**125 m**		
	90 vs. 125 m	*Scleronephthya* sp. 2 – SC	39.3	19.3	43.57	43.57
		Filamentous mat	51.8	65.4	23.76	67.33
		Unidentified – SC	1.9	8.5	19.13	86.46
			15 m	**50 m**		
2008	15 vs. 50 m	Calcite encrusting layer	32.5	2.8	18.83	18.83
		Pocillopora sp. – HC	19.7	2.4	14.97	33.8
		Nephthea sp. – SC	0	19.2	14.58	48.39
		True anemones	0	17.6	11.93	60.31
		Coralline algae – Enc	12.6	0.4	10.82	71.14
		Tubastrea sp. 2 – HC	22.2	45.6	10.19	81.33
			50 m	**90 m**		
	50 vs. 90 m	*Tubastrea* sp. 2 – HC	45.6	0	27.77	27.77
		Sponge – Enc	5.2	50.0	22.88	50.65
		Nephthea sp. – SC	19.2	0	15.4	66.05
		True anemones	17.6	30.2	13.69	79.74

SC, soft coral; HC, hard coral.

decrease in coverage by *Pocillopora* sp. and coralline algae and an increase in coverage by *Nephthea* sp., anemones and *Tubastrea* sp. 2 at 50 m compared to 15 m contributed between 10 and 15% each of the remaining dissimilarity between the two depths. Percent coverage by groups again changed significantly between the 50 and 90 m depths (PERMANOVA: pseudo-t = 5.8171, P = 0.0001). The decrease in *Tubastrea* sp. 2 and *Nephthea* sp. at 90 m contributed 43.2% of this dissimilarity with a further 36.6% of dissimilarity explained by an increase in cover of sponges and anemones at that depth.

We found a substantial change in the percent coverage of the attached invertebrates between years (PERMANOVA: pseudo-F = 4.883, P = 0.0002) (**Figures 3A,B**). Overall, this change was evident in the decrease in soft corals and bryozoans/hydroids at all depths and a general increase in cover by hard coral, sponges and true anemones in 2008 (**Table 3** and **Figure 3A**). The community at 15 m differed significantly between years (PERMANOVA: pseudo-t = 2.4481, P = 0.0003) and was due to changes in several groups/species contributing similar percentages to the dissimilarity (between 10 and 19%, **Table 3**). Taxa with greater coverage in 2008 at 15 m included coralline algae and the hard corals, *Pocillopora* sp. and *Tubastrea* sp. 1. Those with less coverage in 2008 included *Scleronephthya* sp. 1, *Tubastrea* sp. 2 and the calcite layer. Coverage also differed between years at 50 m depth (PERMANOVA: pseudo-t = 3.9192, P = 0.0001). At this depth, a large decrease in the mean coverage by the soft coral *Scleronephthya* sp. 1 (which was not observed in 2008) contributed 26.7% to the dissimilarly between years, along with a decrease in the calcite layer coverage which accounted for a further 13% of dissimilarity (**Table 3**). Large increases in mean cover of *Tubastrea* sp. 2, *Nephthea* sp. and anemones contributed 18.2%, 14.2%, and 12. 8%, respectively, to the remaining dissimilarity between years at 50 m depth. Finally, the invertebrate coverage also differed between years at 90 m (PERMANOVA: pseudo-t = 9.0934, P = 0.0001). The large increase in coverage of sponges and anemones during 2008 contributed 45.0% of dissimilarity between years, whilst the apparent disappearance of bryozoans/hydroids and the decrease of the filamentous mat contributed 23.3% and 19.2%, respectively.

Fishes

We identified and counted 1791 individual fish from at least 10 families and 19 species (**Table 4** and **Figures 2D–F**). Where fish could not be identified to species, they were grouped into broader categories; 'small cryptic fish' included those fish sheltering in marine growth cover, and 'baitfish' were those found only at the maximum depth and in large schools. Some fish species could not be reliably distinguished from each other and were grouped together, such as the genera of the unidentified surgeonfish and unicornfish and the bluefin/bigeye trevally (*Caranx melampygus* or *C. sexfasciatus*) which at times could not be distinguished from each other due to varying light levels and speed of movement.

In 2006, abundance and diversity of fish were low, with only 119 individuals from at least six families observed (**Table 4**). Mean fish densities ranged between 8 fish/50 m^2 in the upper water column to a maximum of 14 fish/50 m^2 in the 25–100 m depth category, but we found no significant difference in total fish density among depths (ANOVA, $F_{2,7}$ = 0.06769, P = 0.9351). Unidentified surgeonfish (*Acanthurus* spp.) and false bluefin trevally (*Carangoides orthogrammus*) were most abundant between 10 and 25 m, reaching 5.1 and 1.3 fish per

TABLE 3 | Dissimilarity and percent mean cover of attached invertebrates between depths in 2006 and 2008.

Depth (m)	Group/Taxa	% mean cover		Contribution to dissimilarity %	Cumulative % contribution
		2006	**2008**		
15	*Scleronephthya* sp. 1 – SC	12.5	0	18.59	18.59
	Coralline algae – Enc	0	12.6	18.22	36.81
	Pocillopora sp. – HC	12.7	19.7	16.12	52.93
	Tubastrea sp. 2 – HC	24.1	22.1	13.96	66.89
	Tubastrea sp. 1 – HC	0.8	6.5	10.46	77.35
	Calcite encrusting layer	42.0	32.5	10.09	87.44
		2006	**2008**		
50	*Scleronephthya* sp. 1 – SC	51.4	0	26.71	26.71
	Tubastrea sp. 2 – HC	10.5	45.6	18.18	44.89
	Nephthea sp. – SC	7.8	19.2	14.19	59.08
	Calcite encrusting layer	22.5	2.8	13.41	72.49
	Anemone	0	17.6	12.84	85.33
		2006	**2008**		
90	Sponges	1.2	50.0	25.58	25.58
	Bryozoans/hydroids	5.1	0	23.3	48.87
	Anemone	0	30.2	19.31	68.18
	Filamentous mat	51.8	14.7	19.24	87.42

Dissimilarity comparisons were made within depth categories using SIMPER analyses. Only taxa contributed up to 80% of the in coverage shown. SC, soft coral; HC, hard coral.

50 m^2, respectively. Almaco jack (*Seriola rivoliana*) were the most abundant fish in depths >25 m (reaching 9.1 per 50 m^2) and contributed 80% of the total fish observed in that year, although they were not observed in 2008. Other fish present in 2006 but not observed in 2008 were a unicornfish, a tall-fin batfish (*Platax teira*) and a giant moray (*Gymnothorax javanicus*) which was observed sheltering under a jacket anode.

In 2008, we observed 1671 fish from at least 8 families and 16 species (**Table 4**). Total fish density increased significantly with depth in 2008, from a total of 22.6 fish/50 m^2 at 10–25 m to 3373.6 fish/50 m^2 at 125 m (ANOVA, $F_{3,12}$ = 56.529, P < 0.001). Density at 125 m was significantly greater than all other depths (Tukey's tests, all P < 0.001), however, patterns among other depths were less distinct. Total fish density at 10–25 m was significantly lower than that at 25–50 m (100.8 fish/50 m^2, Tukey's test, P = 0.034), but densities at 10–25 m did not differ from 50 to 75 m (87.5 fish/50 m^2, Tukey's test, P = 0.108). There was also no difference in fish density between 25–50 and 50–75 m (Tukey's test, P = 0.881).

There were also significant differences in fish community composition across the depth categories in 2008 (PERMANOVA, pseudo-$F_{3,12}$ = 5.8658, P = 0.001). Unlike fish density, we found there was no significant difference in the community composition of fishes between the 10–25 m and the 25–50 m (PERMANOVA, pseudo-t = 1.2349, P = 0.372), or between the 25–50 m and 50–75 m depth categories (PERMANOVA, pseudo-t = 1.5156, P = 0.156). However, community composition differed significantly between 10–25 m and 50–75 m (PERMANOVA, pseudo-t = 2.1163, P = 0.018). All upper depth categories (≤75 m) also differed significantly in community composition with the 125 m community (PERMANOVA, pseudo-t ≥ 2.3472, all P ≤ 0.028).

TABLE 4 | Fish taxa and mean densities (fish/50 m^2) identified over three to four depth categories between 10 and 125 m on Goodwyn Alpha platform in 2006 and 2008.

			2006			**2008**			
Depth category (m)			**10–25**	**25–100**	**100–125**	**10–25**	**25–50**	**50–75**	**125**
Transect length (m) (all replicates)			**41**	**51**	**54**	**38**	**55**	**54.3**	+
Transect area (m^2) (all replicates)			**118**	**184**	**192**	**176**	**190**	**393**	**14**
Family	**Genus and species**	**Common name**							
Carangidae	*Seriola rivoliana*	Almaco jack		8.9 (10.3)	9.1 (6.2)				
Carangidae	*Carangoides orthogrammus*	False bluefin trevally	1.3 (1.4)		0.5 (0.8)	5.5 (5.6)	2.4 (2.6)		
Carangidae	*Caranx sexfasciatus or Caranx melampygus*	Bluefin/bigeye trevally				2.2 (3.8)	49.7 (77.0)	19.4 (25.9)	
Lutjanidae	*Lutjanus argentimaculatus*	Mangrove jack				0.2 (0.3)		0.1 (0.1)	
Lutjanidae	*Lutjanus erythropterus*	Crimson sea perch	0.5 (0.8)	0.6 (1.3)	0.2 (0.4)		0.5 (1.0)	0.6 (0.7)	
Lutjanidae	*Lutjanus bohar*	Red bass					1.4 (1.5)	0.1 (0.2)	27.3 (28.1)
Lutjanidae	*Lutjanus sebae*	Red emperor						0.4 (0.8)	5.0 (11.2)
Lutjanidae	*Lutjanus malabaricus*	Saddle-tailed sea perch				0.4 (0.6)		0.1 (0.1)	
Serranidae	*Cephalopholis sonnerati*	Tomato cod						0.3 (0.6)	
Serranidae	*Pseudanthias rubrizonatus*	Red-bar anthias							350.0 (651.9)
Chaetodontidae	*Heniochus diphreutes*	Schooling bannerfish		3.1 (3.2)	0.9 (1.0)		31.0 (21.9)	51.0 (26.9)	
Acanthuridae	*Acanthurus* spp.	Unidentified surgeonfish	5.1 (4.5)			3.2 (1.3)	3.9 (1.4)	0.6 (0.1)	
Acanthuridae	*Acanthurus dussumieri*	Ornate surgeonfish		0.3 (0.6)				0.7 (0.1)	
Acanthuridae	*Naso hexacanthus*	Sleek unicornfish					1.3 (2.5)		
Acanthuridae	*Naso* spp.	Unicornfish	0.5 (0.8)						
Apogonidae	*Apogon* sp.	Cardinalfish							436.7 (736.6)
Pomacanthidae	*Pomacanthus imperator*	Emperor angelfish		0.3 (0.6)			0.3 (0.5)		
Monacanthidae	*Unknown*	Filefish unidentified						0.4 (0.8)	
Ephippidae	*Platax teira*	Tall-fin batfish	0.5 (0.8)					0.4 (0.4)	
Muraenidae	*Gymnothorax javanicus*	Giant moray		0.4 (0.9)					
		Shark unidentified				0.4 (0.6)			
		Small cryptic fish	0.4 (0.6)	0.1 (0.3)		10.9 (11.1)	10.3 (11.0)	14.0 (26.7)	
		Baitfish							2554.7 (2164.6)
Total density depth category (50 m^2)			**8.1**	**13.7**	**10.7**	**22.6**	**100.8**	**87.5**	**3373.6**
Standard deviation			**4.34**	**6.81**	**5.94**	**11.27**	**49.53**	**52.71**	**1397.55**

Standard deviation shown in brackets. + *indicates snapshots of infrastructure at 125 m where transect footage was unavailable.*

SIMPER analysis described the species/groups that contributed most to the similarities within depths and significant dissimilarities between depths during 2008 (**Table 5a,b**). The community at 0–25 m was characterized by small cryptic fish, false bluefin trevally and unidentified surgeonfish, with the former two groups having mean densities of 10.9 and 5.5 fish/50m^2, respectively, and accounting for 80.1% of the similarity at this depth (**Table 5a**). The community at 25–50

TABLE 5 | (a) Mean densities of fish taxa that contributed to similarities of each depth category according to SIMPER analysis, and (b) fish taxa that contributed to dissimilarities between depths where significant differences were found.

(a) Species/group	**Mean density (50 m^2)**	**Contribution to similarity %**	**Cumulative contribution %**
10–25 m			
Small cryptic fish	10.9	47.79	47.79
False bluefin trevally	5.5	32.26	80.05
Surgeon (unidentified)	3.2	19.95	100
25–50 m			
Bluefin/bigeye trevally	49.7	36.38	36.38
Schooling bannerfish	31.0	23.78	60.16
Small cryptic fish	10.3	21.44	81.6
50–75 m			
Schooling bannerfish	51.0	82.34	82.34
Bluefin/bigeye trevally	19.4	11.04	93.38
125 m			
Baitfish	2554.7	70.56	70.56
Red bass	27.3	15.64	86.19

(b) Species/group	**Mean density (50 m^2)**		**Contribution to dissimilarity %**	**Cumulative contribution %**
	10–25 m	**50–75 m**		
Schooling bannerfish	0	51.0	31.07	31.07
Small cryptic fish	10.9	10.3	15.35	46.42
Bluefin/bigeye trevally	2.2	19.4	13.32	59.74
False bluefin trevally	5.5	0	11.47	71.21
Surgeon fish (unidentified)	3.2	0.6	6.01	77.22
Crimson seaperch	0	0.6	3.32	80.54
	50–75 m	**125 m**		
Baitfish	0	2554.7	29.34	29.34
Schooling bannerfish	51.0	0	17.04	46.39
Cardinalfish	0	436.7	10.59	56.97
Red bass	0.1	27.3	10.18	67.15
Red-bar anthias	0	350.0	10.04	77.19
Bluefin/bigeye trevally	19.4	0	7.32	84.52
	125 m	**25–50 m**		
Baitfish	2554.7	0	25.29	25.29
Bluefin/bigeye trevally	0	49.7	12.37	37.66
Schooling bannerfish	0	31.0	10.21	47.87
Cardinalfish	436.7	0	9.34	57.21
Red-bar anthias	350.0	0	8.86	66.07
Red bass	27.3	0	8.81	74.88
Small cryptic fish	0	10.3	7.75	82.44
	125 m	**10–25 m**		
Baitfish	2554.7	0	32.59	32.59
Cardinalfish	436.7	0	11.58	44.17
Red bass	27.3	0	11.27	55.44
Red-bar anthias	350.0	0	10.99	66.43
Small cryptic fish	0	10.9	9.38	75.81
False bluefin trevally	0	5.5	7.05	82.86

Only taxa contributing up to at least 80% are shown.

m was dominated by bluefin/bigeye trevally and schooling bannerfish (*Heniochus diphreutes*), with mean densities of 49.7 and 31.0 fish/50 m^2, respectively. Along with the small cryptic fish, these taxa accounted for 81.6% of the similarity at this depth. Between 50 and 75 m, schooling bannerfish reached 51.0 fish/50 m^2 and accounted for 82.3% similarity within the depth category and bluefin/bigeye trevally were again in high density at this depth, reaching 19.4 fish/50 m^2. The significant difference in the fish communities between 10–25 m and 50–75 m (reported above) was due to greater density of the schooling bannerfish and the bluefin/bigeye trevally (31.1% and 13.3% contribution to dissimilarity) (**Table 5b**) and lower density of the small cryptic fishes and false bluefin trevally (15.4% and 11.5%, respectively) at 50–75 m.

At 125 m, the community was typified by baitfish and red bass (*Lutjanus bohar)* which contributed 70.6% and 15.6%, respectively, of the similarity within the depth category (**Table 5a**). Baitfish were found exclusively at this depth and at a high mean density of 2554.7 fish/50 m^2. Red-bar anthias (*Pseudanthias rubrizonatus*) and cardinalfish (*Apogon* sp.) were also found only at this depth and in high densities (350.0 and 436.7 fish/50 m^2, respectively, **Table 5b**). SIMPER analysis indicated that the significant differences in fish composition between 125 m and all other depth categories (reported above) were largely due to these taxa that were specific to this depth (**Table 5b**). Baitfish contributed up to 32.6% dissimilarity while red-bar anthias (*P. rubrizonatus*) and cardinalfish (*Apogon* sp.) contributed up to 22.6% dissimilarity combined. Bluefin/bigeye trevally and schooling bannerfish were also important contributors to the differences between the 125 m category and all others, being present in relatively high densities at shallower depths but not recorded at 125 m.

DISCUSSION

To our knowledge, this is the first published study of attached invertebrate species and fish communities associated with an operational oil and gas platform on Australia's North West Shelf. Furthermore, this study provides an insight into the usefulness of archival industry ROV footage for assessing marine communities associated with oil and gas structures.

Although not collected for scientific purposes, archived ROV inspection footage is a potentially valuable resource for environmental science as the footage incidentally captures flora and fauna associated with oil structures during inspections. The operator's video archive, for example, contains footage of infrastructure since its establishment in 1995, with multiple visits to the same structures over extended time periods. Some footage dates back to the 1970s, potentially providing rare long-term datasets from an offshore environment that is challenging to study independently. Specific structures on the platform are easily identified and can be revisited over many years, allowing for fine scale studies of biological communities at specific points. Furthermore, footage is available from a range of subsea infrastructure including platforms, risers, pipelines and wellheads at a variety of depths over wide geographical ranges under the influence of different oceanographic regimes. Industry-collected footage may therefore provide an important tool for assessing the biodiversity associated with oil structures, how this diversity may change over time, how infrastructure functions as an artificial reef, and how communities on infrastructure interact with surrounding marine ecosystems. Despite some limitations using industry-collected ROV footage (outlined below), we were able to: (1) broadly identify the major elements of the invertebrate community inhabiting the GWA platform and estimate their coverage on horizontal members; (2) identify changes in the attached community with depth and time, and (3) identify differences in fish density and distribution with depth. It is important to note that our data are meant to illustrate the potential usefulness of ROV archival footage, rather than a comprehensive attempt to assess long term temporal trends on and around this platform.

Attached Invertebrate Communities

Sessile invertebrates could only be placed into broad taxonomic categories, mostly due to the poor resolution of the video. Despite this issue, we were able to determine patterns of community composition with depth along the platform and between the years using data obtained from the industry footage.

It appeared that at least some of the factors affecting depth zonation and community composition of sessile communities on natural reefs, including light availability, prey, competition, and habitat characteristics (Karlson and Cornell, 1998), also influenced their distribution on GWA. On the platform, we found autotrophic species mostly limited to 50 m depth, most likely reflecting light availability necessary for photosynthesis. These included the coralline algae, the hard coral *Pocillopora* sp. and the soft corals. The exception was the soft coral *Scleronephthya* sp. 2 which was unusual in its deeper distribution from 50 m depth, indicating this species requires less light and/or was an azooxanthellate species.

In contrast, heterotrophic species were distributed across all depths. The most significant heterotrophs in terms of surface cover in the upper 50 m of the platform were the bivalves within the calcite encrusting layer. Whilst indistinct in the video, this community appeared to be largely comprised of both living and bleached bivalve shells. Bivalves are common on platforms and large numbers have been previously observed on platforms in the North Sea, the Gulf of Mexico, California coastal waters and on wellheads in the North West Shelf of Western Australia (George and Thomas, 1979; Love et al., 1999; van der Stap et al., 2016; McLean et al., 2018b). The occurrence of these filter feeders on members in the upper 50 m of the platform likely mirrors the occurrence of their planktonic prey in the upper sunlit layers of the water column.

Other abundant heterotrophs included the two species of *Tubastrea*. These hard corals lack photosynthetic zooxanthellae yet contributed much of the coral cover in the upper 50 m of the platform in both years. *Tubastrea* is often found attached to oil and gas platforms in the Gulf of Mexico, West Africa and in the South Atlantic (Friedlander et al., 2014; Sammarco et al., 2015; Creed et al., 2017). At depths $\geq$50 m, heterotrophic species such as sponges, bryozoans, hydroids and anemones with reduced light requirements dominated the coverage. Additionally, the filamentous mat was important from 90 to 125 m depth. Without physical samples, we cannot describe this community further. Based on other platform studies, it is possible that this layer consists of encrusting bryozoans (Forteath et al., 1982), although we could not confirm this from the video footage.

We observed a dramatic change in community composition on the GWA platform where soft corals were largely reduced in cover or lost between 2006 and 2008. Several factors may have caused this change and the most obvious to consider first is jacket cleaning. Removal of attached invertebrates on subsea infrastructure is a semi-regular occurrence necessary to reduce the load exerted against the structure by water currents. However, as the last large scale cleaning event was in 1998 (M. Biczo, pers. comm., 26 August 2013), cleaning was not responsible for the changes we observed. We also discount that biofouling measures impacted the soft corals as the cover of invertebrates we observed would indicate that the antifouling, applied prior to the installation of the platform in 1995, would have previously reached the limit of its effectiveness. It is possible, however, that corrosion of the steel legs may have influenced the change in community composition. Soft corals are known to bioaccumulate metals which can affect their growth and reproduction (Hwang et al., 2018), however, leaching of corrosive metals would likely be gradual and unlikely to cause the severe change in communities we observed over a short period. Natural factors such as predation on and/or disease of soft corals by bacteria, fungus and protozoans (Weil et al., 2015) may also have produced the changes we observed. However, we have no data to support either a role of corrosion or natural impacts in the changes we observed.

The most likely factors that resulted in the changing community composition relate to the oceanography of the NWS and the passage of tropical cyclones. Internal tides or waves are an important and energetic component of the NWS where they are generated by large (10 m) barotropic tides in contact with the continental shelf break (Holloway, 2001). Internal waves displace little water at the ocean surface but result in very large internal, vertical displacements of water (up to 50 m). They can generate strong currents that rapidly oscillate in direction throughout the water column at up to 0.5 ms^{-1}, exerting force on and sometimes damaging oil and gas infrastructure (Bole et al., 1994; Fang and Duan, 2014). The force of these currents is sufficient to physically damage and dislodge soft corals, potentially explaining the decrease in soft coral abundance on the GWA platform between 2006 and 2008 surveys.

Tropical cyclones occur regularly on the NWS between November and April and frequently disrupt oil and gas operations at sea and on land. On natural reefs in shallower water, severe tropical cyclones are important in shaping coral reef communities, and can influence coral cover and species diversity, as well as cause widespread coral mortality (Nott and Hayne, 2001; Beeden et al., 2015). Between August 2006 and December 2008, there were five tropical cyclones on the NWS that disrupted offshore industry activities[1]. Two of these cyclones, Tropical Cyclones Kara and George, were classified as category 4 and 5, respectively, by the Australian Bureau of Meteorology Tropical Cyclone Category System, with category 5 being the most severe[2]. Cyclones cause mixing and cooling throughout the water column to 100 m depth and the generation of very strong, oscillating near-inertial internal waves, with currents reaching up to 1.5 ms^{-1} at the bottom, but typically 0.5 ms^{-1} throughout the entire water column for a period of up to 9 days (Rayson et al., 2015). During the study period, the major difference in the attached communities between years was a decrease in soft coral cover across all depths, especially notable in the two species of *Scleronephthya* and the loss of other soft attached species such as the foliaceous bryozoans/hydroids. Changes in the sessile invertebrate communities and their biomass due to environmental forcing has also been observed on platforms in the Gulf of Mexico due to seasonal flooding from coastal bays and the Mississippi River (George and Thomas, 1979). Thus, it appears that the invertebrate communities of platforms are dynamic in nature, shaped over time by environmental influences. This dynamism may have implications for the effectiveness of platforms to act as artificial reefs if susceptible invertebrates like soft corals play a functional role in the biological communities associated with platforms.

Compared to other studies on platforms (George and Thomas, 1979; Forteath et al., 1982; Friedlander et al., 2014), we did not identify a diverse invertebrate community. This is likely due at least in part to our reliance on video footage and the inability to collect physical specimens. The resolution of video footage was generally too poor to provide quality close-up images that would be required for identifications to lower taxonomic levels. The footage available for this study and our survey approach may also have some inherent biases. Spatial orientation of platform members can influence colonization and this seems particularly important to bryozoan and coral recruitment as these animals seemingly prefer vertical or diagonal surfaces (Forteath et al., 1982; Clark and Edwards, 1999; Perkol-Finkel et al., 2006). These groups may therefore have been under sampled in our study due to the focus on horizontal members. Furthermore, we surveyed only the north facing Row G surfaces of the platform, which is in the lee of the platform and the prevailing south westerly wind. It is possible that communities on this more sheltered aspect of the platform differed to those on more exposed faces. In targeted future studies, we recommend surveys are designed to take member and jacket orientation into account.

Fish Assemblages

Knowledge of the fish communities associated with Australia's offshore oil structures is also essential for understanding their habitat value. We observed 1791 fish from at least 19 species and 10 families associating with the GWA platform during the current study. Like other studies in the region on infrastructure (Pradella et al., 2014; McLean et al., 2017, 2018b), we identified commercially important target species in the Pilbara demersal fishery, including crimson seaperch (*Lutjanus erythropterus*),

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[1] http://www.bom.gov.au/cyclone/history/wa/

[2] http://www.bom.gov.au/cyclone/about/intensity.shtml

red emperor (*L. sebae*), saddle-tailed seaperch (*L. malabaricus*) and mangrove jack (*L. argentimaculatus*) (Fletcher and Santoro, 2012). Trevally (*Caranx sexfasciatus and C. melampygus*) were the most abundant of the commercial and recreational species at GWA and were found in the upper 75 m in 2008.

The diversity of fishes at GWA was lower than on other infrastructure within a 60 km radius. Our sampling effort and time of sampling was most similar to Pradella et al. (2014) who used ROV video to count and identify fish at three obsolete well heads at depths of 85–175 m in June and July 2008. They found 31 species from 14 families, including ten commercially important species. Similarly, McLean et al. (2017) used ROV video to study fish on pipelines between August to December over a 3-year period. This study found 37 species from 21 families on a pipeline at approximately 125 m depth and 74 species from 30 families on another pipeline at 60–80 m depth. The cause(s) of differences in fish diversity among these studies and infrastructure types is unknown. It does not appear related to habitat complexity, because the well heads, and to a lesser extent, the platform would have provided the most complex and high relief habitat and the pipelines the least. Sound and vibration from an operating oil and gas platform such as GWA may have some inhibitory effect on some fish species as underwater sound is known to affect their behavior and the abundance of catches (Engås et al., 1996; Normandeau Associates Inc., 2012). An alternative explanation for the relatively low number of taxa recorded in our study is the short duration of the footage examined, conducted over just 2 days in each year which may have underestimated the state of the fish diversity associated with GWA. Examination of footage from other seasons and years would likely result in a broader range of species being detected at the site.

The low fish abundance recorded during 2006 in the current study was most likely due to footage being taken at night. Although ROV lights illuminated the foreground of footage, low light levels in the background likely reduced detection rates of individuals relative to footage taken during the day. Furthermore, the effect of ROV lighting on fish behavior is variable, whilst some fish may be only slightly affected, others may flee the light the approaching ROV (Laidig et al., 2012), so our results may at least partially reflect this effect. Additionally, many fish species are known to exhibit reduced mobility at night, sheltering in crevices, overhangs and within the invertebrate coverage (Nagelkerken et al., 2000). This behavior makes them difficult to detect with remote survey techniques such as video. In contrast, nocturnal feeders increase mobility at night, and may move away from shelter to forage on surrounding substrata during hours of darkness. This may explain the reduced abundance of lutjanid species observed on GWA in 2006 (**Table 4**), many of which are known to feed at night in areas surrounding their daytime resting sites (Hobson, 1965; Colton and Alevizon, 1981). Seasonal fluctuations in abundance in the study region may also have contributed to the low abundances of fish observed in 2006. Footage in 2006 was taken during late winter (August), whereas footage in 2008 was taken during summer (December). Seasonal investigations which incorporate footage taken during both the day and night are required to identify temporal factors of greatest influence on oil structure communities.

Habitat Value of Structures

Determining the habitat value of oil and gas structures, and the relative value of different sections within large structures like platforms, is important for evaluating decommissioning options involving partial removal. In the case of the 'topping' option, where the upper steel jacket is removed from the surface to approximately 30 m depth, some habitat provided by the photosynthetic portion of the attached community will be lost. However, considering the force of waves at the ocean's surface that minimize settlement and growth and the application of anti-biofouling strategies employed in the upper 15 m (Surespek Iss PTY Ltd, 2009), loss of this habitat over 30 m will likely be minimal. Of possibly greater importance to the habitat value of the jacket of GWA is the continual impact of the passage of cyclones and energetic internal waves on the NWS. If these forces are shaping the attached community through the semi-regular removal of weakly attached species, it may be that the community is not reaching its full habitat potential, unlike the platforms in RTR programs located in more quiescent environments where internal wave energy is quickly dissipated in shallow waters (Rayson et al., 2015). A principle difference between platforms on the NWS shelf and those elsewhere is the occurrence of the strong semidiurnal tides. When the cyclone-induced inertial waves coincide with spring tides, attached communities on structures on the NWS may suffer the additive effect of both forces. It may be that platforms on the NWS are of lower habitat value than those elsewhere because of these impacts.

Alternatively, platforms and other oil structures can be situated where biodiversity is limited and far from the nearest natural reefs. On the NWS for example, mid-shelf to deeper water bottoms are characterized by sandy sediments and muds, often with low biodiversity (McLoughlin and Young, 1985; Lyne et al., 2006). As artificial reefs can create an 'ecological halo' effect up to 15 times the size of the actual structure (Reeds et al., 2018), the biodiversity associated with oil structures such as the GWA platform is likely to remain very high compared to the surrounding benthos, a virtual 'biological oasis' independent of changes in community composition over time. Thus, isolated from the nearest reefs and coastal areas, the sessile invertebrate community we observed on GWA most likely reflects a unique assemblage compared to the diversity of the surrounding marine ecosystem. While the pelagic fish we observed may be indicative of those in the wider environment, other fish may have been recruited to the platform as larvae and will not reflect the surrounding communities. Clearly, more research elsewhere on the NWS is required to understand the habitat value of oil structures in this region.

The increase in fish density with depth on GWA in 2008 suggests deeper sections of the platform have greater habitat value than shallower sections. This result is similar to findings on seven platforms located in Southern California (Martin and Lowe, 2010), and a platform in the northern Gulf of Mexico (Rooker et al., 1997). However, patterns of fish abundance with depth have been found to vary greatly among studies and regions, with some studies finding greatest abundance in the shallowest sections of oil structures (Stanley and Wilson, 1997; Dokken et al., 2000). The generality of our finding on GWA is therefore unclear, and further investigation is required to determine whether other platforms on the NWS share similar patterns. We also offer our conclusions concerning depth associations on GWA cautiously, due to the short duration of investigation and the different time-period of observation for the 125 m depth category (3 months earlier than other depth categories). The high densities of baitfish that drove abundance values at 125 m may represent seasonal recruitment and/or transient movement that may not represent abundance values during other periods.

Community differences between depths on GWA suggest that sections of greatest habitat value are likely to vary among taxa. Similar to findings on other platforms (Rooker et al., 1997), pelagic species (e.g., Bluefin trevally, *C. melampygus*) were most abundant at mid-depth sections of the structure. In contrast, depth associations of benthic species were mixed, with some taxa more abundant at mid-depths (e.g., 'small cryptic fishes'), while others were more abundant near the platform base (e.g., red-bar anthias, *Pseudanthias rubrizonatus*). Mixed patterns for benthic taxa are likely the result of depth preferences or tolerances, and suggest that the prediction of depth associations on other platforms will require knowledge of taxon-specific depth distributions. Depth distributions of reef-associated fishes are one of the most poorly understood ecological aspects of fish communities (Brokovich et al., 2008). Given the deep deployment depths of many oil structures, and the relatively uniform habitat they provide, oil structures may assist scientific understanding of patterns of reef fish distribution with depth.

We found high densities of baitfish in the 125 m depth category exceeding 2,500 individuals per 50 m^2 in three out of five surveys near the platform base, and baitfish were not observed in any other depth category. We termed these as 'baitfish' as we could not determine if they were juveniles of a larger species or the adult stages of a small species. However, this result is consistent with findings of high juvenile or sub-adult abundances on platforms in Southern California, the Gulf of Mexico and on pipelines and wellheads in the region of GWA on the NWS, suggesting that the infrastructure offers suitable habitat for some juvenile fishes (Love et al., 2000; Gallaway et al., 2009; Fowler and Booth, 2012; McLean et al., 2017). Such a conclusion would be unsubstantiated here given the short period of the study, the single platform investigated and our inability to even identify the small fish to family level. However, science-directed studies using industry ROV video on other platforms giving rise to larger datasets would provide vital life history data on fish and on the role of platforms as nurseries and in recruitment. Ideally, such studies would be followed up with direct sampling of fish using traps or nets.

Although not observed in their juvenile stage, the presence of site-attached reef taxa indicates a wide range of species recruited to GWA. Given the isolated location of GWA (outer NWS), and the apparent lack of natural reef in the immediate vicinity (100 km radius), site-attached taxa (e.g., 'small cryptic fishes') most likely settled on the platform following a pelagic larval phase. Our observations of adults of such taxa therefore indicate that GWA provided sufficient habitat for developing and sustaining populations following their recruitment (Fowler and Booth, 2012). The ability of oil structures to provide sufficient habitat for sustaining fish populations is a key aspect of their value as habitat.

Limitations of Industry-Collected ROV Footage

Despite the potential benefits, some challenges and limitations arose when using industry-collected footage for scientific investigation. The video archiving system used by the operator was complex and necessitated a close partnership with industry engineers and access to platform structural plans. Pre-screening to identify useful footage from structures that were repeatedly filmed through time was labor-intensive. Repeat inspections were irregular in terms of frequency and seasons. It was also difficult to find matching footage of components between years taken at similar times of the day. While time of day was largely inconsequential to the invertebrate surveys, it was likely an important factor in describing the fish community (as outlined above). Poor camera resolution and lighting often hampered our ability to identify the attached invertebrates. Similar to another study using ROV footage (van der Stap et al., 2016), our observations underestimated the biodiversity of the community on the GWA platform. For example, from the video we could not reliably distinguish bivalves in the calcite encrusting layer and we did not observe any non-attached motile invertebrates such as worms, snails, crustaceans, etc. which were likely present. Therefore, use of industry archival ROV footage will be useful as a "rapid assessment" tool for biodiversity, which could then be complemented with targeted specimen collection to ensure high levels of scientific rigor.

CONCLUSION AND RECOMMENDATIONS

Offshore oil structures have been identified as important habitat for marine organisms in numerous regions of the world. For example, platforms in Southern California are known to support large abundances of overfished rockfish species (Schroeder and Love, 2004; Claisse et al., 2014), and the expansion of oil infrastructure in the northern Gulf of Mexico has been linked to the establishment of a valuable red snapper (*Lutjanus campechanus*) fishery (Gallaway et al., 2009). These discoveries were the result of extensive research programs spanning decades, and have influenced decommissioning policy and practice in both regions. The recent policy decision to allow 'rigs-to-reefs' conversions of obsolete platforms in California (Assembly Bill 2503, 2010) was contingent on their importance to rockfish populations.

Oil structures in Australia may provide similar habitat functions to those in other regions, yet almost nothing is known about the biological communities associated with Australia's offshore infrastructure. We have demonstrated that archival ROV footage can be used to assess attached invertebrate and fish communities on established offshore platforms, albeit with some limitations in species identification and survey design. Our results indicate that archived ROV footage held by the oil and gas industry is a useful tool for understanding the habitat value of oil and gas infrastructure in Australia. Archival footage also provides a cost-effective means for industry to obtain environmental information on their infrastructure relative to field investigations. Information on the habitat value of oil and gas infrastructure will be essential for informing environmentally sound decommissioning policy in Australia.

Some of the limitations of using industry-collected ROV footage for scientific purposes can be overcome in the future. For example, cameras that provide high resolution video (e.g., 4k) and still frames can be easily fitted to ROVs at a small additional expense. The use of 3D video which allows for precise measurements of organisms and other objects in the field of view would also be valuable (Macreadie et al., 2018). Additionally, ROV research could be complemented by other survey techniques such as baited remote video, light traps and eDNA to help determine how these structures are used by fish (McLean et al., 2017). Finally, ROV-collected specimens would aid greatly in the identification of attached species and cryptic fish. Research using archival ROV footage must be expanded to other platforms and other types of structures (e.g., well heads, pipelines) in order to amass a solid base of evidence for informing decommissioning policy. Research across a broad geographical range is also required to assess variability between different locations under differing environmental regimes.

ETHICS STATEMENT

This study was carried out in accordance with the recommendations of “Animal Welfare Act (2002)” and the requirements of the Australian Code of Practice for the Care and Use of Animals for Scientific Purposes (7th Edition 2004) ‘The Code’ and the UWA Animal Ethics Committee. The protocol was approved by the UWA Animal Ethics Committee.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the manuscript.

AUTHOR CONTRIBUTIONS

DB conceived the idea and secured access to the footage. PT, AF, and DB designed the study. PT, AF, AD, and CP completed the analysis. PT compiled the data and wrote the manuscript with contributions from all other authors.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00471/full#supplementary-material

TABLE S1 | Transect lengths and replicate numbers (in parentheses) according to year and depth for surveys of invertebrates and fishes associated with GWA platform. Replicates for invertebrates refer to quadrats examined along the length of a transect, while replicates for fishes refer to the number of transects completed. Frames examined from 125 m during 2008 were obtained from stationary footage obtained 3 months prior to footage for other depths. Note the different depth strata examined for fishes between 2006 and 2008.

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Spatio-Temporal Dynamics of Benthic Macrofaunal Communities in Relation to the Recovery of Coastal Aquaculture Operations Following the 2011 Great East Japan Earthquake and Tsunami

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The 2011 Great East Japan Earthquake and tsunami wiped out over 1200 shellfish and ascidian culture long-lines and ~120 salmon farm cages that comprised the entire aquaculture installations in Onagawa Bay, Japan, and severely altered the associated ecosystem. A year later, we launched a coordinated monitoring program to measure the extent of the damage caused by the disaster and monitor the change in the state of the marine ecosystem. As part of this effort, we conducted multi-seasonal sampling to characterize spatio-temporal variation in benthic macrofaunal community and a range of environmental parameters across Onagawa Bay between March 2012 and January 2018. The 492 total macrofaunal species recorded included Polychaeta (38.8%), Bivalvia (13.2%), Amphipoda (10.8%), Decapoda (9.6%), Gastropoda (9.3%), and Echinodermata (4.3%). At the outermost reference site, macrofaunal abundance, biomass, and species diversity were all consistently high throughout the study period. Inside Onagawa Bay, macrofauna metrics increased steadily from the lowest values at the beginning of the study to the highest over time. During the same period, the spatial extent of aquaculture facilities for long-lines and fish cages recovered steadily to within 60.8% and 74.8% of the original state, respectively. The significant variables identified by multivariate analysis to explain spatio-temporal variability in benthic macrofaunal communities were: (1) proximity to the nearest aquaculture facilities; (2) wind fetch length (exposure); (3) sediment grain size; and (4) the total area of aquaculture facilities. This study suggests that coastal aquaculture operations may strongly influence the occurrence and distribution of benthic macrofaunal communities and thereby influence the recovery of seafloor biota at ecosystem scales following a catastrophic natural disaster.

Keywords: benthic macrofauna, coastal aquaculture, marine ecosystem ecology, environmental disturbance, 2011 Tohoku earthquake and tsunami, anthropogenic influence, spatio-temporal dynamics, Onagawa Bay

INTRODUCTION

Marine ecosystems have played a vital role in supporting human well-being since ancient times by providing a wealth of resources and services (Katsanevakis et al., 2011; Thurber et al., 2014). However, an array of natural and anthropogenic stressors impacts the global ocean (Halpern et al., 2008; McCauley et al., 2015). Natural environmental stressors or disturbances that affect marine ecosystems may include floods, droughts, hurricanes, volcanic eruptions, earthquakes, and tsunamis (e.g., Whanpetch et al., 2010; Lomovasky et al., 2011). Anthropogenic stressors such as fisheries, aquaculture, land reclamation, eutrophication, marine litter, construction of sea defense walls additionally affect marine ecosystems with increasing frequency (e.g., Benn et al., 2010; Foden et al., 2011; Katsanevakis et al., 2011). These natural and anthropogenic stressors interact, complicating assessment of the consequences of any single stressor (Halpern and Fujita, 2013; McCauley et al., 2015; Gunderson et al., 2016; Liess et al., 2016).

On 11 March 2011, the Great East Japan Earthquake and subsequent tsunami devastated extensive areas of the Pacific coast of the Tohoku region in northern Japan. The major disturbances caused by the disaster included the immediate physical force of the tsunami and associated negative impacts on intertidal biota (Urabe et al., 2013), large amounts of terrestrial debris washed out into the ocean, a severe accident at the Fukushima Daiichi nuclear power plant that caused leakage of radioactive contaminants (Sohtome et al., 2014), spills of toxic chemicals (e.g., heavy oil), and associated negative impacts on subtidal benthic fauna (Abe et al., 2015). Coastal regions also suffered from loss of seagrass/seaweed beds with their associated fauna (Takami et al., 2013), abrupt accumulation of tsunami deposits and resuspended mud on the seafloor (Seike et al., 2013; Toyofuku et al., 2014; Kanaya et al., 2015; Seike et al., 2016), and widespread land subsidence. In order to measure the extent of the damage caused by the disaster and to monitor the subsequent change in the state of the marine environment, we launched the "Tohoku Ecosystem-Associated Marine Sciences" (TEAMS) project in March 2012. As part of this study, we initiated a coordinated monitoring program to obtain a holistic view on the spatio-temporal dynamics of the marine ecosystem of Onagawa Bay (**Figure 1**). To this end, we monitored a range of environmental parameters including biogeochemistry, physical measurements, sediment properties, and community dynamics of benthic and pelagic components potentially impacted by the disaster. This study presents the benthic component and associated environmental factors.

Prior to the earthquake and tsunami, fishing and aquaculture were the primary industries for many towns and villages along the Tohoku Pacific coast. For example, Onagawa Bay supported over 1200 long-lines for culturing scallops, oysters, and edible ascidians as well as around 120 fish cages for farming salmonids. However, the 2011 earthquake and tsunami wiped out entire aquaculture installations. Rehabilitation of coastal aquaculture therefore attracted considerable interest from both marine scientists and policy makers. Recent studies demonstrate varying, but often negative, effects of aquaculture on the environment through changes in the physical, chemical, and biological attributes of sediments below and the water column around aquaculture installations (e.g., Tovar et al., 2000; Borja et al., 2009; Forchino et al., 2011; Sarà et al., 2011; Farmaki et al., 2014; Tomassetti et al., 2016). Most studies demonstrate reduced impacts of shellfish aquaculture than finfish aquaculture because shellfish are grown at comparatively lower intensity and require no external feed (Shumway et al., 2003; McKindsey et al., 2006; Weise et al., 2009). However, shellfish farms typically cover a much greater area than finfish farms and may affect the surrounding environment differently (Weise et al., 2009).

Benthic macrofauna have long been used as indicators of environmental health and trends in coastal marine ecosystems (Borja et al., 2009; Forchino et al., 2011; Tomassetti et al., 2016). We therefore investigated spatio-temporal dynamics of benthic macrofaunal community structure in relation to changes in multiple environmental factors including the rehabilitation of coastal aquaculture facilities in Onagawa Bay following the 2011 Tohoku earthquake and tsunami. Our study addresses four lines of investigation:

(1) characterize spatio-temporal dynamics of benthic macrofaunal community structure;
(2) describe spatio-temporal variation in major environmental variables;
(3) map spatio-temporal distribution of coastal aquaculture facilities to estimate changes in the location, number, and total area of aquaculture facilities;
(4) examine the relationship between macrofaunal community structure and the explanatory environmental variables described above.

In addition, we extracted a subset of an external survey data collected near our sampling locations before the 2011 disaster as a baseline for comparison. Our overall aim is to provide insight into the environmental and anthropogenic factors most strongly related to the recovery of seafloor biota and coastal environmental health in this region.

MATERIALS AND METHODS

Study Site

Onagawa Bay has an area of 27.15 km^2, located at the southernmost part of a ria coastline, known locally as the "Sanriku Coast," on the Pacific Ocean of northern Japan (**Figure 1**). Geologically, this stretch of coast was formed by a combination of sea level rise and the submergence of mountains and river valleys that resulted in an extremely irregular and indented coastline. Onagawa Bay is also a natural sheltered deep-water harbor with a mean depth of 33 m, which makes it suitable for coastal aquaculture.

Onagawa Bay Monthly Survey on *RV Suikoh*

Most of our data were collected during the Onagawa Bay Monthly Survey on board *RV Suikoh* between March 2012 and January 2018. A total of 12 sampling stations (i.e., St.1–8, 11, 12, 16,

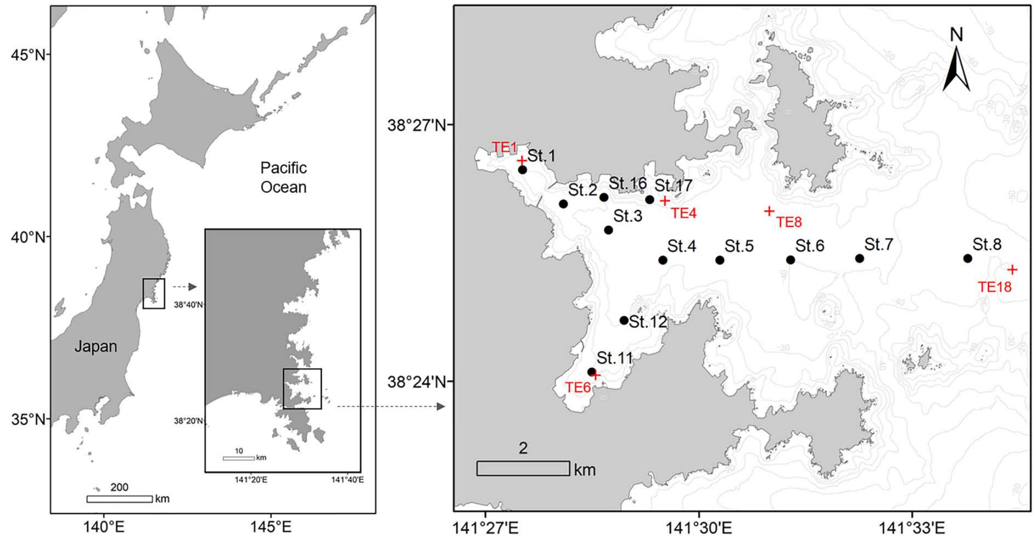

FIGURE 1 | Map of Onagawa Bay, Japan, showing sampling stations for the Onagawa Bay Monthly Survey on board *RV Suikoh* from March 2012 onward (filled black circles: St.1–8, 11, 12, 16, and 17). Red cross symbols (TE1, 4, 6, 8, and 18) indicate locations of five of the sampling stations for the long-term environmental survey program conducted by Miyagi Prefecture and Tohoku Electric Power Company that provided comparative pre-tsunami data.

and 17) were established to monitor multiple environmental parameters (**Figure 1** and **Table 1**). Benthic and physical measurements were made at all 12 stations only in March 2012 (**Supplementary Figure 1**). Thereafter, benthic sampling was conducted mainly at St.1, 3, 4, 6, 8, 11, 12, 16, and 17 (nine stations) on a monthly basis between May 2012 and March 2014, which we reduced to every 2 months from March 2014 onward.

We sampled a total of four sediment samples at each station using a Smith-McIntyre grab (sampling area: 0.05 m^2). For benthic macrofauna, we sieved and merged three of the grab samples on a 1.0 mm mesh with filtered sea water on board. All organisms retained were preserved in 10% buffered formalin for subsequent sorting, species identification, counting, and biomass measurements. Species were identified to the lowest possible taxonomic level using a stereomicroscope and a compound microscope where necessary, expressing abundance and biomass (wet weight) as ind./m^2 and g/m^2, respectively. We then extracted subsamples of the top 5 and 1 cm of the sediment to determine particle size composition and chemical oxygen demand (COD), respectively. Particle size composition was determined using both dry sieving and sedimentation analysis and categorized into three size ranges defined here as "gravel" (GRAV: >2 mm), "sand" (SAND: <2 mm, >75 μm), and "mud" (MUD: <75 μm) (**Table 1**). COD (mg/g) was determined using a standardized solution of potassium permanganate as the oxidant (**Table 1**).

TABLE 1 | Description of environmental variables used in this study.

Category	Variable	Description
Topography	DEPTH	Water depth (m)
	D.LAND	Distance to the nearest land (km)
	FETCH	Wind fetch length as an indication for exposure (km)
Sea water	TEMP	Temperature immediately above the seabed (°C)
	SALI	Salinity immediately above the seabed
	CHLa	Chlorophyll-a immediately above the seabed (μg/l)
	DO	DO immediately above the seabed (mg/l)
	TURB	Turbidity immediately above the seabed (FTU)
Sea bottom	COD	COD of sediment (mg/g)
	GRAV	Gravel (>2 mm) content of sediment (%)
	SAND	Sand (<2 mm, >75 μm) content of sediment (%)
	MUD	Silt and clay (<75 μm) content of sediment (%)
Aquaculture	D.LINE	Distance to the nearest long-line (m)
	D.CAGE	Distance to the nearest fish cage (m)
	A.LINE	Total area occupied by long-lines in the whole bay (km^2)
	A.CAGE	Total area occupied by fish cages in the whole bay (km^2)

TABLE 2 | Description of sampling stations for the Onagawa Bay Monthly Survey (DEPTH, D.LAND, and FETCH as in **Table 1**).

Station	Position		DEPTH	D.LAND	FETCH
	N	E	(m)	(km)	(km)
1	38.4412	141.4586	20.0	0.31	0.8
2	38.4346	141.4682	29.0	0.42	12.5
3	38.4295	141.4788	33.5	0.75	18.6
4	38.4237	141.4915	37.2	1.06	19.1
5	38.4236	141.5049	39.0	0.80	19.4
6	38.4237	141.5216	41.0	1.31	41.7
7	38.4240	141.5378	40.0	1.36	61.6
8	38.4240	141.5632	44.9	2.04	63.8
11	38.4018	141.4749	21.0	0.30	1.3
12	38.4119	141.4824	32.7	0.35	7.2
16	38.4359	141.4777	27.9	0.34	7.4
17	38.4354	141.4884	31.9	0.19	13.2

At each sampling station, we recorded profiles of water temperature (TEMP: °C), salinity (SALI), chlorophyll-a concentration (CHLa: μg/l), dissolved oxygen (DO: mg/l), and turbidity (TURB: FTU) using a CTD RINKO-Profiler (JFE Advantech Co., Ltd., Japan) including measurements immediately above the seabed (**Table 1**). Distance to the nearest land (km) (D.LAND) and wind fetch distance (km) (FETCH) were calculated as indicators of the exposure of coastal locations (**Table 2**). FETCH was determined as an average length (distance to the nearest land) of fetch vectors in 36 equiangular directions with a maximum distance set to 200 km if unobstructed by land, following Burrows et al. (2008) and Seers (2018).

Aquaculture Facilities in Onagawa Bay

Aquaculture operations in Onagawa Bay commonly use two main methods. Hanging culture uses a long-line system to suspend cultured organisms such as oysters, scallops, and ascidians on vertical ropes, typically down to around 15–20 m in depth below surface floats, in contrast to floating fish cages used for farming salmonids. We digitized and mapped these floating aquaculture structures using time-series of high resolution satellite imagery to create distribution maps for the long-lines (polylines) and fish cages (polygons) for the whole of Onagawa Bay from 2011 March to 2017 March (**Table 3**). Each polyline represents a single unit of the long-line system which, in fact, comprises two parallel lines of the same length (typically ∼100 m), approximately 1.5 m apart and connected at regular intervals by floats and/or ropes. We estimated the sea surface area occupied by each long-line by creating a buffer of 0.75 m around each polyline and then computing the area of the buffer zone (polygon).

Data Analysis

We identified structural and spatio-temporal trends in the macrofauna data in Onagawa Bay using multivariate community analysis in PRIMER v6 (Clarke and Warwick, 2001). Following fourth-root transformation of macrofauna counts to down-weight the influence of highly abundant species, we performed cluster analysis (group-average linkage) on a resemblance matrix of the transformed data based on the Bray–Curtis similarity index. A similarity profile (SIMPROF) permutation test on the resemblance matrix identified statistically significant clusters (significance level, $p < 0.01$). SIMPER (similarity percentage) analysis then identified those species most responsible for the similarity within the groups determined by the cluster analysis.

TABLE 3 | A list of satellite data used in this study.

Imagery date	Satellite/source	Resolution (cm)
11 March 2011	WorldView-1	50
6 April 2011	Google Earth, 2018 DigitalGlobe	–
18 July 2011	Google Earth, 2018 DigitalGlobe	–
22 December 2012	Google Earth, 2018 DigitalGlobe	–
28 July 2012	Google Earth, 2018 DigitalGlobe	–
20 October 2012	WorldView-2	50
14 April 2013	GeoEye-1	40
15 April 2014	GeoEye-1	50
19 December 2014	GeoEye-1	40
17 May 2015	WorldView-2	50
19 April 2016	WorldView-3	30
19 March 2017	WorldView-2	50

We assessed general temporal trends by constructing time series for the 16 environmental variables (**Table 1**) and normalized them to generate a resemblance matrix based on Euclidean distance for conducting RELATE routines and BIO-ENV stepwise (BEST) analysis. RELATE routines examined potential differences in macrofaunal community structure between the clusters in relation to each of the 16 environmental variables. The BEST procedure examined rank correlations between the multivariate patterns of macrofauna and environmental factors to identify the subsets of variables that best explained the overall pattern. We then examined the biological metrics of each cluster group (i.e., abundance, biomass, species richness, and Shannon's diversity index H') using box plots. Further, based on the BEST procedures, we assessed those environmental variables that significantly explained the macrofaunal variation using the same approach.

In addition, we extracted a subset of external data collected before the 2011 disaster from annual measurements of warm drainage water from the Onagawa nuclear power plant (Miyagi Prefecture and Tohoku Electric Power Company, 2010, 2011, 2012). This program has monitored a wide range of physical and biological data in Onagawa Bay since 1981. The program conducted benthic surveys twice a year in February and August at 18 sampling stations, of which 5 sites, namely TE1, 4, 6, 8, and 18, were close to sampling stations of St.1, 17, 11, 6, and 8, respectively, in our study (**Figure 1**), reporting the following information: (1) total abundance of macrofauna (ind./m^2); (2) species richness of macrofauna; (3) SAND (%); (4) MUD (%); (5) GRAV (%); and (6) COD (mg/g) (coding of environmental variables as in **Table 1**). The survey reports include details on sampling methods and protocols (Miyagi Prefecture and Tohoku Electric Power Company, 2010, 2011, 2012). We extracted data for August and February in each year from 2008 to 2011 (pre-tsunami: $n = 6$ per variable at TE1, 4, 6, 8, and 18), for comparison of mean values with our corresponding data for July and January from 2015 to 2018 in the present study (post-tsunami: $n = 6$ per variable at St.1, 17, 11, 6, and 8).

We constructed distribution maps for the aquaculture facilities using ESRI ArcGIS 10.5 and calculated values for FETCH using "fetchR" package (Seers, 2018). All the graphics relating to the multivariate analysis were constructed using R version 3.4.1 package (R Development Core Team, 2017).

RESULTS

Benthic Macrofaunal Community Structure

We collected a total of 382 benthos samples and identified 492 species comprising 191 from the class Polychaeta (38.8%), 65 from the class Bivalvia (13.2%), 53 from the order Amphipoda (10.8%), 47 from the order Decapoda (9.6%), 46 from the class Gastropoda (9.3%), 21 from the phylum Echinodermata (4.3%), and 69 others (14.0%). Macrofaunal abundance, species richness, and Shannon index H' were lowest soon after the 2011 disaster, but increased rapidly over subsequent years to peak values toward the end of the study period (**Figures 2A,C,D**). Macrofaunal biomass decreased slightly shortly after sampling began, but then increased toward the end of the study period (**Figure 2B**).

Cluster analysis and a SIMPROF test revealed 11 distinctive macrofaunal groups (Groups A–K) and outliers (OUT) that further subdivided into 40 statistically significant groups (**Figure 3**). While Groups A, D, B, and G comprised only 5 (1.3%), 5 (1.3%), 7 (1.8%), and 8 (2.1%) of the samples, respectively, the majority of samples were categorized into K (100 samples: 26.2%), I (77 samples: 20.2%), H (50 samples: 13.1%), and C (43 samples: 11.3%) groupings (**Figure 3**). Spatio-temporal patterns in benthic groups appeared largely random during the first year (Groups A–H except for Group C) (**Figure 4**). SIMPER indicated lowest average similarities in Groups A, B, and D (32.4, 28.4, and 31.7%, respectively). Relatively few species, e.g., *Paraprionospio coora* (Polychaeta), *Capitella* sp. (Polychaeta), and *Theora fragilis* (Bivalvia), were predominantly responsible for similarity within these groups (**Figure 5**). Groups H–K then began to occur consistently over space and time from the second year onward (**Figure 4**). The highest average percent similarity occurred in Group K (45.3%) and, except for Group C, the number of species responsible for similarities generally increased from Group A toward Group K (**Figure 5**). Group C occurred almost exclusively in the outermost part of Onagawa Bay (St.8) and remained essentially unchanged from the beginning to the end of our study (**Figure 4**). Despite relatively low percent similarity (35.3%), Group C contained high numbers of species,

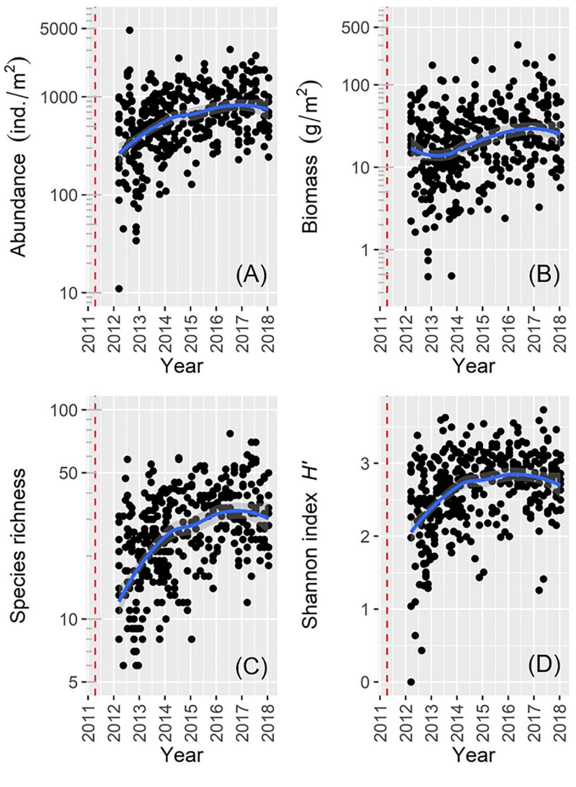

FIGURE 2 | Scatter plots showing general temporal trends of macrofaunal data for: **(A)** abundance; **(B)** biomass; **(C)** species richness; **(D)** Shannon's index H' (the red dotted line denotes the date of the 2011 disaster).

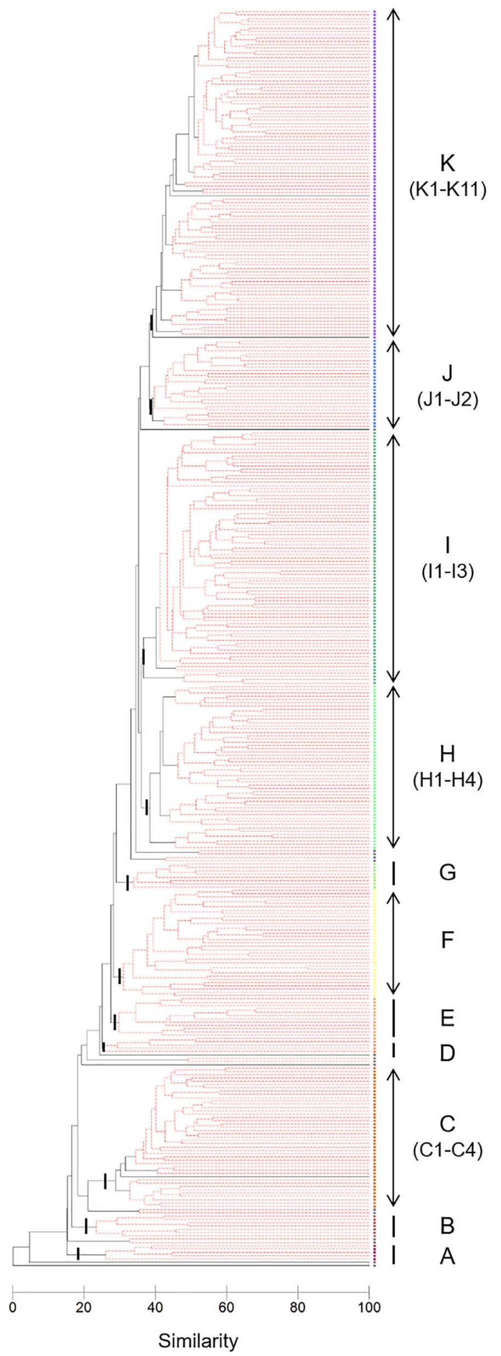

FIGURE 3 | Dendrogram based on fourth-root transformation, Bray–Curtis similarity, and group-average clustering. The SIMPROF test identified 11 main clusters (A–K) with a total of 40 significant sub-divided groups (black branches, $p < 0.01$).

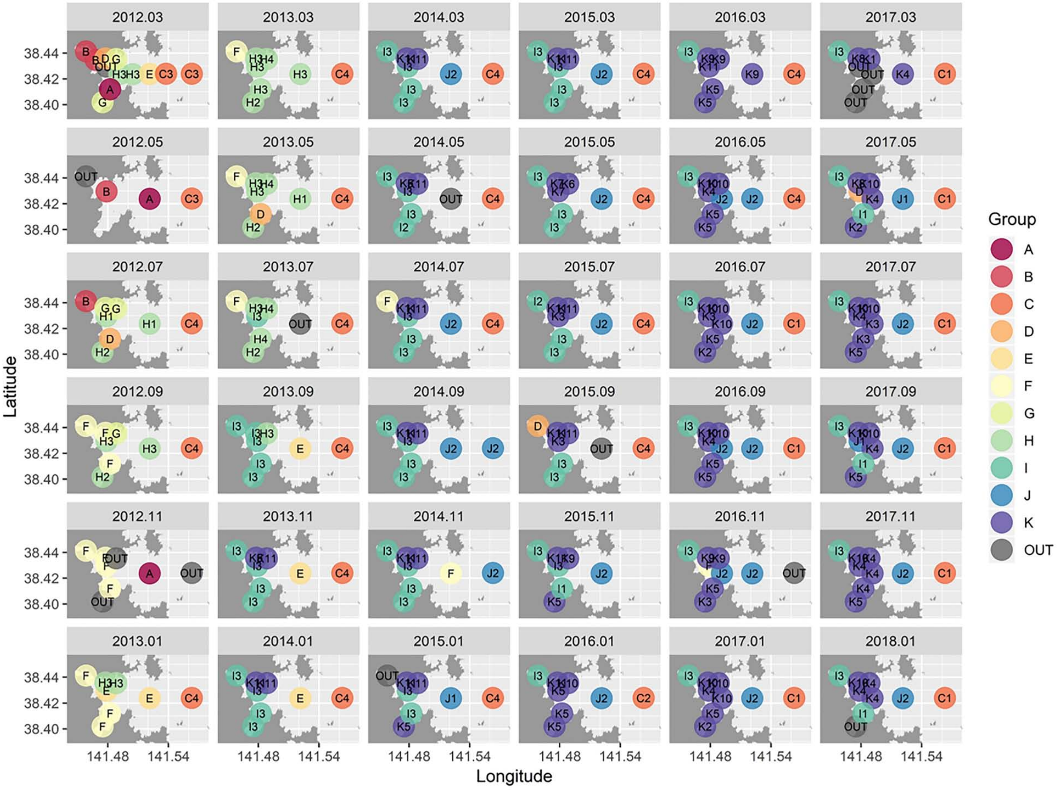

FIGURE 4 | Spatio-temporal distribution maps for benthic macrofaunal community structure in Onagawa Bay from March 2012 (top-left corner) to January 2018 (bottom-right corner). The 11 main clusters (A–K) with a total of 40 significant sub-divided groups (labels in the maps) were identified based on the SIMPROF test ($p < 0.01$).

including several crustaceans, e.g., *Byblis japonicus* (Amphipoda), *Ampelisca naikaiensis* (Amphipoda), and *Iphinoe sagamiensis* (Cumacea), that contributed to average similarity within this group (**Figure 5**).

Environmental Parameters

Temperatures (TEMP), chlorophyll-a concentration (CHLa), and DO clearly demonstrated an annual (seasonal) cycle at all nine stations (**Figures 6A,C,D**). TEMP varied little among stations (**Figure 6A**), whereas CHLa was disproportionately higher at times at several locations (e.g., St.1, 3, 12, and 17) (**Figure 6C**). Further, at St.1 only, summer minimum DO became much lower in recent years (**Figure 6D**). Values for both salinity (SALI) and turbidity (TURB) fluctuated between 33–34 and 0–10, respectively, although these ranges were exceeded in several instances (**Figures 6B,E**). COD and sediment fraction of both sand (SAND) and mud (MUD) varied distinctively among stations, with St.1 and St.8 often at opposite ends of the spectrum (**Figures 6F,H,I**). COD at St.1 was almost always the highest and St.8 the lowest, with some distinct peaks for most stations at the beginning and toward the end of the study period (**Figure 6F**). Sediment grain size at St.8 consistently contained a very high proportion of sand and little mud (**Figures 6H,I**). Nonetheless, a higher fraction of mud and lower fraction of sand characterized all stations at the beginning of our study, but stations gradually became increasingly sandy toward the end (**Figures 6H,I**). Relatively high fractions of gravel recurred at St.17 and, less frequently, at St.1 and 11 (**Figure 6G**).

Aquaculture Facilities in Onagawa Bay

In March 2011 (just before the 2011 disaster), the 1217 long-lines and 119 fish cages occurred mainly along low-energy coastal waters and inlets in Onagawa Bay (**Figure 7A**). After the complete removal of these facilities by the 2011 tsunami, 135 long-lines and 56 fish cages were re-constructed by February 2012 (**Figures 7B,C**), corresponding to 11.1 and 47.1% recovery, respectively (**Figures 8A,B**). By March 2017, the total surface area occupied by the long-lines and fish cages became 0.115 and 0.016 km^2, i.e., 60.8 and 74.8% recovery, respectively (**Figures 7D–H**, **8C,D**).

Macrofaunal Community Structure and Environmental Drivers

The RELATE tests showed that the macrofauna community resemblance matrix correlated most strongly with distance to the nearest long-line culture (D.LINE, $\rho = 0.485$, $p < 0.001$) followed by fetch length (FETCH, $\rho = 0.476$, $p < 0.001$) and distance to the nearest fish cage culture (D.CAGE, $\rho = 0.464$,

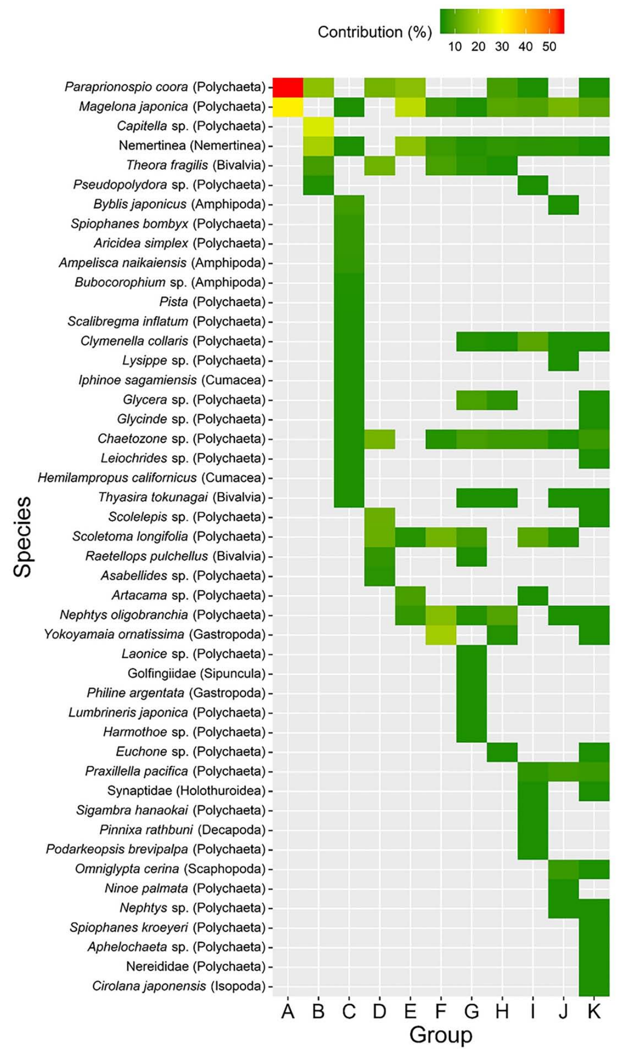

FIGURE 5 | Results of SIMPER analysis showing percentage contributions of macrofaunal species that accounted for most of the similarity (>70%) within each of the 11 cluster groups (A–K).

$p < 0.001$) (**Table 4**). The other aquaculture-related variables, namely A.LINE and A.CAGE, also correlated significantly with macrofauna community structure (**Table 4**). D.LAND, sediment fraction of both sand (SAND) and mud (MUD), and water depth (DEPTH) also correlated significantly with the macrofauna resemblance matrix (**Table 4**). The BEST analysis showed that the combination of FETCH, SAND, and A.LINE produced the highest correlation, followed by FETCH, MUD, and A.LINE (**Table 4**). Although the variables of D.LINE, FETCH, D.CAGE, SAND, MUD, A.LINE, DEPTH, and A.CAGE, appeared repeatedly in the five best subsets selected in the BEST procedure, no other environmental parameters in **Table 4** were selected in a single combination.

We plotted values for macrofaunal abundance, biomass, species richness, and Shannon index H' against cluster groups A–K identified by the SIMPROF test (**Figures 9A–D**). Groups C and G both showed markedly higher values for these biological metrics. Apart from these two groups, all biological metrics generally increased from Group A to Group K which also appeared to correspond to sampling date from the oldest to the newest (**Figures 9A–D**). In terms of proximity to the nearest aquaculture facilities, Group C mostly contained the farthest stations, whereas Group G stations occurred in closer proximity (**Figures 9E,F**). Apart from these groups, values for D.LINE and D.CAGE generally decreased from Group A to Group K with Group K associated with sites closest to the aquaculture facilities (**Figures 9E,F**). Groups C, I, J, and K tended to coincide with sites with increasing or highest cover of aquaculture facilities, whereas Groups A, B, and G characterized low aquaculture sites (**Figures 9G,H**). High fractions of sand characterized Group C and, to a lesser extent, Group G (**Figures 9I,J**). Apart from these two groups, values for SAND generally increased but MUD decreased from Group A to Group K, respectively, with Group K as somewhat intermediate (**Figures 9I,J**). Groups C and J were both significantly associated with exposed and deeper sites, whereas Group I was constrained to more sheltered and shallow locations (**Figures 9K,L**). Both Groups G and K were largely associated with stations with intermediate exposure (**Figures 9K,L**).

Finally, mean values for macrofaunal abundance and species richness between 2015 and 2018 did not differ significantly from before the 2011 disaster (2008–2011) at most of the sampling stations except for St.6/TE8 (**Figures 10A,B**). The mud fraction (MUD) became significantly higher for some stations after the 2011 tsunami (**Figures 10C,D**). The gravel fraction was substantially higher around St.17/TE4 before the 2011 disaster (**Figure 10E**), and values for COD were significantly higher around St.1/TE1 in more recent years compared to pre-2011 values.

DISCUSSION

Our multivariate assessment of benthic macrofaunal community structure following the 2011 Tohoku earthquake and tsunami showed general increases in macrofaunal abundance, biomass, and species diversity during the study period, indicating substantial recovery of seafloor biota and coastal environmental health within 7 years. These community changes correlated with a combination of both natural and anthropogenic factors, adding insights into the mechanisms of community recovery from a catastrophic event.

Of the 11 macrofaunal cluster groups identified, exceptionally high abundances, biomass, and species diversity characterized Groups C and G. Group C occurred almost exclusively at the outermost site of St.8, some distance offshore from the outer boundary of Onagawa Bay. Physical conditions at St.8, particularly sediment grain size and exposure, differed greatly from the sampling stations inside Onagawa Bay. High species diversity and higher proportions of crustaceans (including amphipods and cumaceans), polychaetes, and bivalves characterized Group C community structure; these taxa

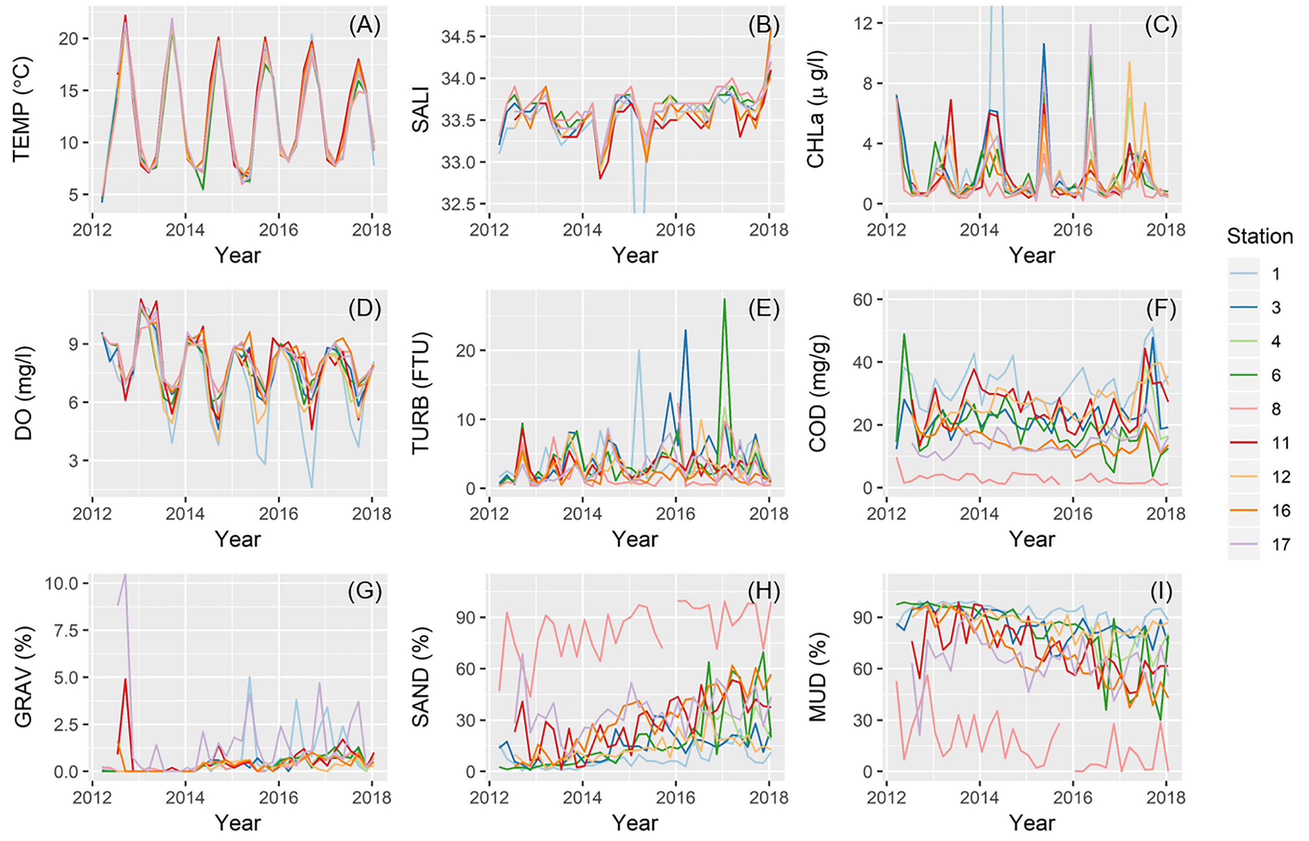

FIGURE 6 | Temporal change in environmental parameters across sampling stations for: **(A)** temperature; **(B)** salinity; **(C)** chlorophyll-a concentration; **(D)** DO; **(E)** turbidity; **(F)** COD; **(G)** gravel content; **(H)** sand content; and **(I)** mud content.

strongly resembled those typical of high-energy sandy bottom environments. Group C persisted at St.8 throughout our study, suggesting that St.8 could act as a reference site with minimal or negligible physical impacts of the 2011 tsunami and the re-establishing process of aquaculture facilities.

In contrast, Group G occurred only in the earliest stage of our study up until September 2012 and only in the presumably low-energy waters at St.11, 16, and 17. Despite their inshore locations, however, sediment granulometry in Group G varied, characterized by high fraction of sand, which was remarkably consistent with Group C at St.8. This resemblance suggests that Group G stations may also be hydrodynamically high-energy sites (Takahashi et al., 2017) that could favor establishment of rich macrofaunal communities. In addition, Group G occurred near sites where aquaculture facilities were first re-installed in Onagawa Bay. Long-lines and fish cages were both re-constructed at and expanded from these locations (i.e., around St.11, 16, and 17). Coastal aquaculture operations can alter pelagic–benthic energy fluxes, enhancing flux of organic matter to the bottom and altering local sediment characteristics and benthic community composition (Crawford et al., 2003). Environmental effects of shellfish and ascidian aquaculture on seafloor environments may differ from those associated with finfish culture in cages (Weise et al., 2009). Cultured shellfish and ascidians feed entirely on naturally occurring phytoplankton in the water column, and captured food and nutrients return to the environment as undigested waste or feces, which falls to the seafloor and may become important food source for benthic deposit feeders (Shumway et al., 2003; McKindsey et al., 2006). Chemicals and excess nutrients from food and feces associated with finfish farming may have similar effects but intense operations can also disturb benthic communities (Borja et al., 2009; Forchino et al., 2011; Tomassetti et al., 2016). For Group G, such organic enrichment generated by both types of aquaculture may have fuelled benthic recovery, particularly if benthic communities were impoverished from the immediate effects of the 2011 disaster via, e.g., vigorous influx of tsunami deposits/resuspended mud (Seike et al., 2013, 2016; Toyofuku et al., 2014; Kanaya et al., 2015) or spills of heavy oil/toxic chemicals (Abe et al., 2015). Group G occurred near the aquaculture facilities (D.LINE and A.LINE), likely explaining the markedly high abundance and biomass in this group.

One key finding of our study was that all the biological metrics for the cluster groups (i.e., abundance, biomass, species richness, and Shannon index *H*′) increased from Group A to Group K (except for Groups C and G). Comparison with the pre-tsunami data (2008–2011) confirmed that the benthic macrofaunal community largely recovered over the

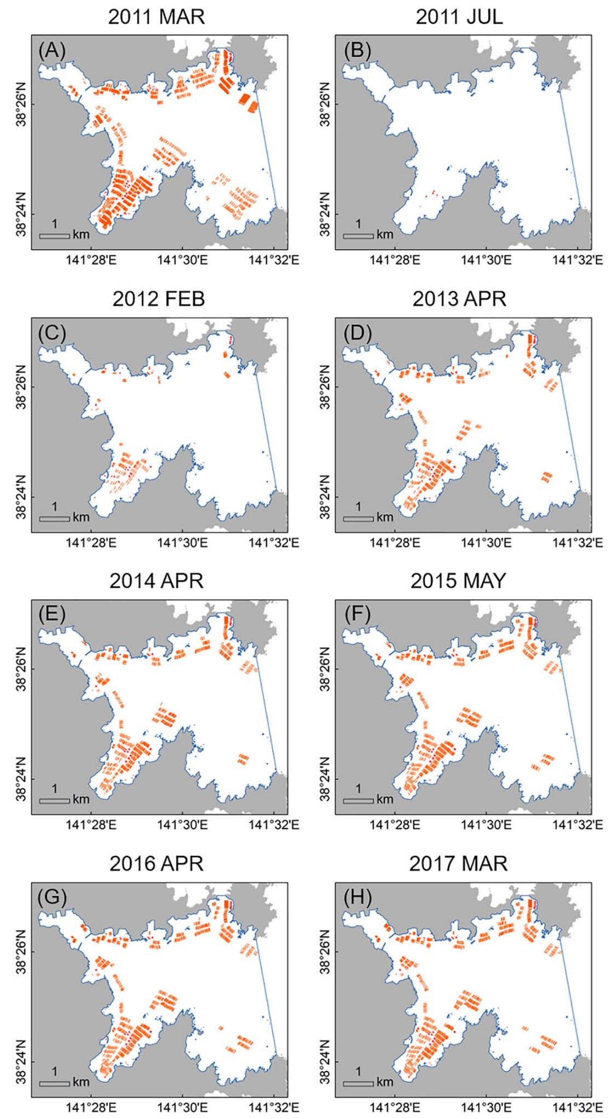

FIGURE 7 | Time series of distribution maps for aquaculture facilities in Onagawa Bay (orange polylines and red polygons represent long-lines and fish cages, respectively, and blue solid line defines the boundary of Onagawa Bay for this study): **(A)** March 2011 (just before the 2011 earthquake and tsunami); **(B)** July 2011; **(C)** February 2012; **(D)** April 2013; **(E)** April 2014; **(F)** May 2015; **(G)** April 2016; and **(H)** March 2017.

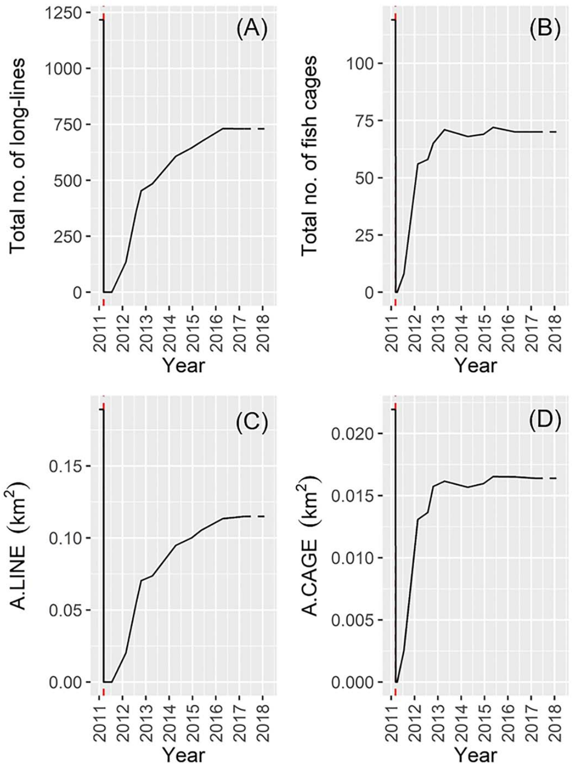

FIGURE 8 | Change in the total number of: **(A)** long-lines; **(B)** fish cages, and changes in the total area of: **(C)** long-lines; **(D)** fish cages (the red dotted line denotes the date of the 2011 disaster). Satellite data for January 2018 were not available and the corresponding values were therefore assumed identical to those estimated in March 2017.

duration of this study. A combination of distance to or total area of the aquaculture facilities, wind/wave fetch length, and sediment granulometry best explained variability in macrofaunal community structure. D.LINE and D.CAGE generally decreased from Group A to Group K suggesting significant, positive effects of the physical presence of aquaculture facilities on benthic macrofaunal populations, acknowledging that aquaculture prior to the disaster may have altered the benthos relative to a pristine condition. A.LINE and A.CAGE generally increased from Group A to Group K suggesting that increasing extent of aquaculture facilities can also positively influence benthic macrofaunal populations. As of March 2017, the total surface area occupied by the long-lines and fish cages were 0.115 and 0.016 km^2, representing only 0.42 and 0.06% of the total area of Onagawa Bay, respectively. However, recent studies suggest that bio-deposits produced from aquaculture operations can disperse and spread through the water column and thereby affect benthic community at a much broader spatial scale than the immediate vicinity of aquaculture facilities (up to several kilometers) (Borja et al., 2009; Weise et al., 2009; Yokoyama, 2010; Sarà et al., 2011). If sphere of influence extends distances of 50, 100, and 150 m from each of the long-lines as suggested, we estimate the potential footprint of aquaculture operations on the seafloor to be 4.70, 6.93, and 8.96 km^2, or 17.3, 25.5, and 33.0% of the total area of Onagawa Bay, respectively. This estimate points to a need to assess accurately the spatial extent of benthic impact resulting from the re-establishment of aquaculture facilities and associated fishery operations in this region.

With respect to changes in macrofauna and sediment grain size following the earthquake, the sand fraction (SAND) and the mud fraction (MUD) clearly increased and decreased, respectively, from Group A to Group K. In estuarine and coastal soft-bottom habitats, sediment granulometry critically influences benthic macrofaunal community (e.g., Constable, 1999; Thrush et al., 2004; Anderson, 2008). The trend of gradually

TABLE 4 | Results of RELATE tests (upper 12 rows) and BIO-ENV stepwise (BEST) analysis (lower 5 rows) showing Spearman's rank correlation coefficients (ρ) between benthic macrofaunal community structure and the environmental variables (coding of environmental variables as in **Table 1**).

Test	Variable(s)	Correlation (ρ)	*p*-value
RELATE	D.LINE	0.485	<0.001
	FETCH	0.476	<0.001
	D.CAGE	0.464	<0.001
	D.LAND	0.434	<0.001
	SAND	0.426	<0.001
	MUD	0.423	<0.001
	A.LINE	0.348	<0.001
	DEPTH	0.347	<0.001
	COD	0.342	<0.001
	A.CAGE	0.266	<0.001
	SALI	0.103	<0.001
	TEMP	0.032	<0.01
BEST	FETCH + SAND + A.LINE	0.660	<0.001
	FETCH + MUD + A.LINE	0.657	<0.001
	D.LINE + FETCH + SAND + A.LINE	0.655	<0.001
	D.CAGE + SAND + A.LINE + DEPTH	0.654	<0.001
	D.LINE + FETCH + SAND + A.LINE + A.CAGE	0.654	<0.001

For the BEST analysis, we present the five best combinations of environmental variables that generated the highest Spearman's rank correlations (ρ).

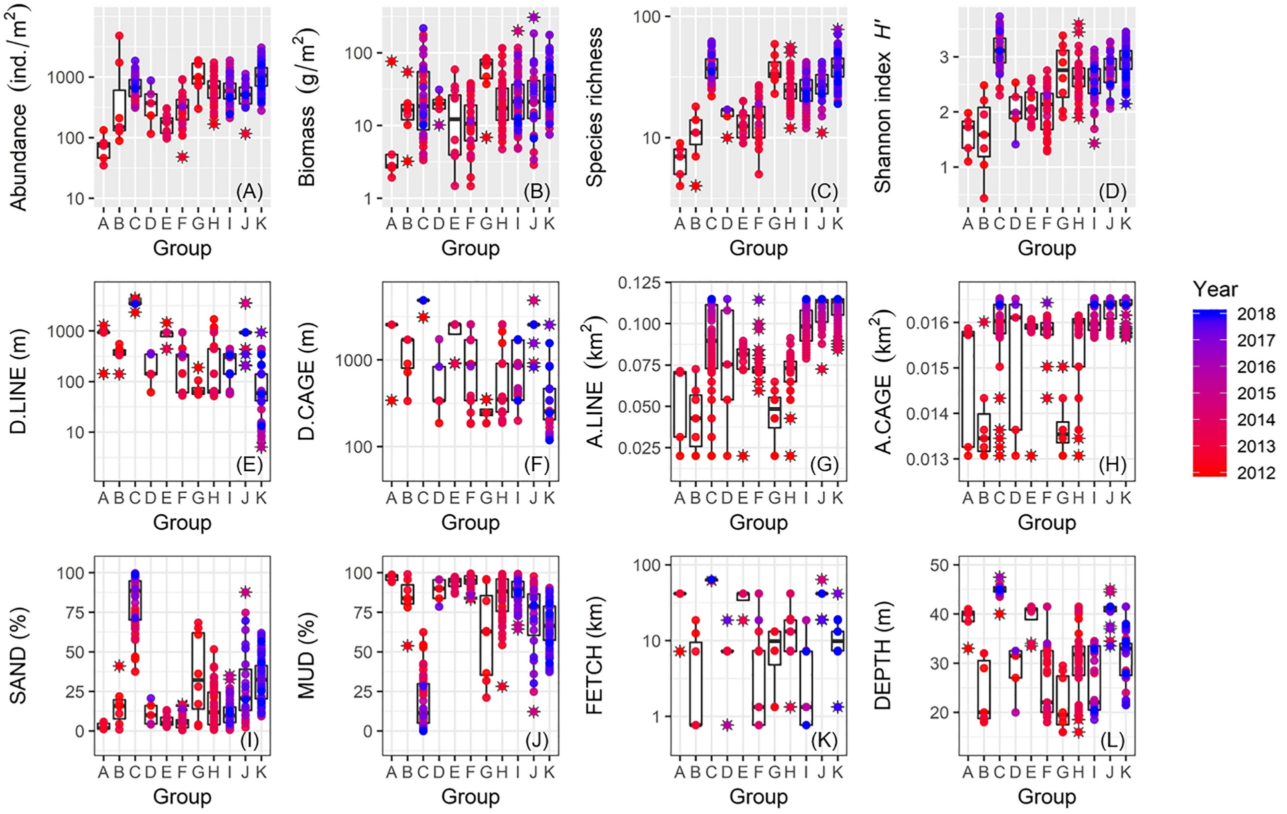

FIGURE 9 | Box plots showing between-group comparisons of macrofauna and environmental data for: **(A)** abundance; **(B)** biomass; **(C)** species richness; **(D)** Shannon's index *H'*; **(E)** D.LINE; **(F)** D.CAGE; **(G)** A.LINE; **(H)** A.CAGE; **(I)** SAND; **(J)** MUD; **(K)** FETCH; and **(L)** DEPTH (coding of environmental variables as in **Table 1**).

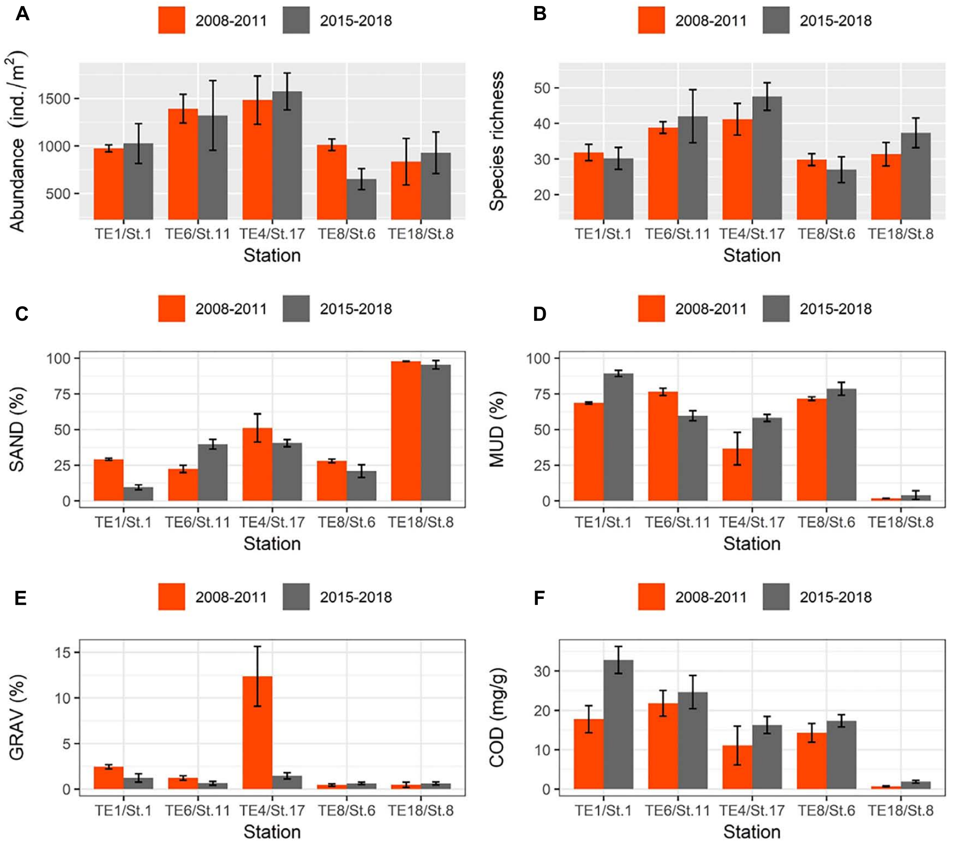

FIGURE 10 | Bar graphs showing changes in mean values (±SE) of selected macrofaunal and environmental data taken "before" (orange bars) and "after" (gray bars) the 2011 disaster: **(A)** abundance; **(B)** species richness; **(C)** SAND; **(D)** MUD; **(E)** GRAV; and **(F)** COD (coding of environmental variables as in **Table 1**).

increasing sand and decreasing mud across Onagawa Bay over time suggests these granulometric changes also affected the dynamics of macrofaunal community structure. Before the 2011 disaster, the sand fractions were significantly higher at some stations prior to anomalous deposition of mud across the whole bay following vigorous erosion and deposition of fine-grained sediments associated with a strong tsunami-induced current (Seike et al., 2016). Abrupt increase in fine-grained sediment could adversely affect benthic community through smothering and clogging feeding structures of suspension-feeders (Thrush et al., 2004), which may have wiped out benthic communities immediately after the 2011 tsunami. Despite gradual decreases in mud fractions, the sand fractions have not yet returned to pre-tsunami levels at some locations. Different rates of removal of fine-grained sediments could reflect different hydrodynamic processes operating at different locations. Our study used wind fetch length (FETCH) as an indicator of exposure and our analysis indicated that Groups I and J were constrained to more sheltered (low-energy) and more exposed (high-energy) locations, respectively, whereas Group K occurred at locations of intermediate exposure (i.e., FETCH: Group J > Group K > Group I). However, the dominance of sand fractions associated with Group K > Group J > Group I suggests that hydrodynamics alone did not control changes in the sediment particle size composition. Links between sediment granulometry and benthic macrofaunal communities have yet to be fully critically evaluated because coastal aquaculture operations can enhance flux of organic matter to bottom sediments which may also alter local sediment characteristics (Crawford et al., 2003). Overall, the relative influence of hydrodynamic processes and the flux of organic matter generated from the aquaculture operations on local variation in sediments is unclear, pointing to a need for future research on how distribution of aquaculture operations relates to sediment characteristics of seafloor habitat and how other environmental parameters such as coastal topography and/or hydrodynamic processes influence benthic–pelagic dependencies across Onagawa Bay.

Our study demonstrated the primary importance of coastal aquaculture operations, along with sediment granulometry and coastal topography (exposure), in determining the occurrence and distribution of benthic macrofauna and thereby influencing recovery in abundance, biomass, and diversity of benthic community at an ecosystem-scale following even a catastrophic natural disaster such as the 2011 Tohoku earthquake and tsunami in Japan. The potential links between the extent of coastal aquaculture operations and spatio-temporal distributional responses of benthic macrofauna as demonstrated in our study illustrate the need to improve understanding of the influence of multiple coastal anthropogenic operations on the structure and dynamics of the marine bentho-pelagic interactions that, in turn, affect the functioning of the whole marine ecosystem. Application of such research to other locations can inform marine policy issues relating to post-tsunami and/or aquaculture operations.

AUTHOR CONTRIBUTIONS

TF led the overall data analysis, writing of the paper, and the statistical analyses. AKi and KK coordinated the project and the survey. KK, TF, MK, YN, AKa, DT, YG, and HA performed the sampling at sea. KK, MK, and YN led the processing and management of the survey data. TF and KK led the analysis of the macrofauna and environmental data. HM, CY, and TF performed the analysis of the satellite imagery.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2018.00535/full#supplementary-material

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Predicting Optimal Sites for Ecosystem Restoration Using Stacked-Species Distribution Modeling

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Habitat restoration is an important tool for managing degraded ecosystems, yet the success of restoration projects depends in part on adequately identifying preferred sites for restoration. Species distribution modeling using a machine learning approach provides novel tools for mapping areas of interest for restoration projects. Here we use stacked-species distribution models (s-SDMs) to identify candidate locations for installment of manmade reefs, a useful management tool for restoring structural habitat complexity and the associated biota in marine ecosystems. We created species distribution models for 21 species of commercial, recreational, ecological, or conservation importance within the Southern California Bight based on observations from long-term reef surveys combined with high resolution (200 m × 200 m) geospatial environmental data layers. We then combined the individual species models to create a stacked-species habitat suitability map, identifying over 800 km^2 of potential area for reef restoration within the Bight. When considering only the 21 focal species, s-SDM scores were positively associated with observed bootstrap species richness not only on natural reefs (linear model: slope = 0.27, 95% CI = 0.17–0.36, $w = 1$), but also this result was supported by two independent test datasets. The predicted richness from this linear model was associated with observed species richness when considering only the focal species on manmade reefs (linear model: slope = 0.52, 95% CI = 0.13–0.92, $w = 1$) and also when considering 204 other non-focal species on both natural and manmade reefs in southern California (slope = 3.65, 95% CI = 2.93–4.37, $w = 1$). Finally, our results demonstrate that the existing manmade reefs included in our study on average are located in regions with habitat suitability that is not only less suitable than natural reefs (t-value = –5.4; $p < 0.05$), but also only slightly significantly better than random ($p < 0.05$), demonstrating a need for more biologically informed placement of manmade reefs. The stacked-species distribution model provides insight for marine restoration projects in southern California specifically, but more generally this method can also be widely applied to other types of habitat restoration including both marine and terrestrial.

Keywords: ecological niche modeling, MaxEnt, rocky reefs, southern California, habitat restoration

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INTRODUCTION

Increasing habitat loss and degradation worldwide threatens many of the world's species (Foley et al., 2005), resulting in population declines (Bender et al., 1998), loss of genetic diversity (Sih et al., 2000), and even species extinctions (Barnosky et al., 2011). Habitat restoration is an important tool for managing degraded ecosystems (Polak and Saltz, 2011), in an effort to restore and prevent species loss (Pavlik, 1996). Yet, habitat restoration initiatives are not always successful (Fischer and Lindenmayer, 2000; Godefroid et al., 2011). One key factor that influences the success of habitat restoration projects is the quality of sites chosen for management (Bottin et al., 2007). For example, manmade habitat structures can fail when placed in areas with non-ideal environmental conditions (e.g., Frissell and Nawa, 1992). To prevent such failures, it is crucial that we develop and test methods for identifying candidate sites for habitat restoration.

Species distribution modeling (a.k.a. ecological or environmental niche modeling) has been proposed as tool for identifying sites for habitat restoration (Pearce and Lindenmayer, 1998) and is increasingly being utilized for this purpose (Rodríguez et al., 2007). Using observation data in conjunction with spatially gridded environmental data, species distribution modeling identifies environmental predictors of species occurrence, creating a model that is then projected across the landscape to identify other areas of suitable habitat (Elith and Leathwick, 2009; Elith et al., 2011). Species Distribution Models (SDMs) have been used to predict optimal sites for restoration for a wide variety of species, including plants (Yang et al., 2013) and animals (Pearce and Lindenmayer, 1998; Wilson et al., 2011). Yet, while many of these efforts focus on single keystone or focal species (e.g., Pearce and Lindenmayer, 1998; Wilson et al., 2011; Yang et al., 2013), for some degraded habitats, restoration is needed for entire communities (Palmer et al., 1997). Stacked-SDMs, where SDMs are first created for individual species and then combined, provide an opportunity to identify suitable habitat across multiple species. Stacked-SDMs have been used for studying spatial patterns of environmental suitability across a range of taxa (Dubuis et al., 2011; Guisan and Rahbek, 2011; Hof et al., 2012; Calabrese et al., 2014; Hof and Svahlin, 2016; da Mata et al., 2017).

Here we implement stacked-SDMs (s-SDMs) to predict optimal locations for the placement of manmade reefs to restore habitat for shallow rocky reef-associated marine fish, invertebrate, and algal communities. To conduct this research, we take advantage of long-term reef survey datasets (Caselle et al., 2015; Pondella et al., 2015a; Zahn et al., 2016) to build SDMs for the entire extent of rocky reefs within the Southern California Bight (SCB). The rocky reefs of the SCB are a habitat of particular interest because this region is on par with some of the most highly productive ecosystems in the world (Hubbs, 1960; Horn and Allen, 1978; Pondella et al., 2005; Horn et al., 2006). Here, cool waters of the California current from the north meet with warm waters from the south to create a set of unique environmental conditions that support a wide variety of marine species (Horn and Allen, 1978; Bograd and Lynn, 2003; Pondella et al., 2005; Horn et al., 2006; Hamilton et al., 2010). Naturally occurring hard substrates make up the base of rocky reef habitats from which wide ranging giant kelp forests (*Macrocystis pyrifera*) grow, providing extensive habitat for many marine fish, invertebrate, and algal species (Graham, 2004; Stephens et al., 2006). Yet at the same time, this productive ecosystem is located next to one of the world's largest megacities, Los Angeles (Nicholls, 1995). As a result, there is intense anthropogenic pressure exerted on this critical ecosystem, including overfishing (Love, 2006; Zellmer et al., 2018), habitat modification due to landslides (Kayen et al., 2002) or development (Ambrose, 1994), and pollution (Schaffner et al., 2015).

Manmade (artificial) reefs have long been used as a successful option for restoration of marine ecosystems (Bohnsack and Sutherland, 1985; Bohnsack et al., 1994). Many hard substrates create manmade reefs, from purposefully designed quarry rock structures to breakwalls, pier pilings, and even sunken shipwrecks (Morris et al., 2018). When standardized, comparisons with natural reefs suggest that manmade reefs can sustain similar levels of species richness and abundance (Carr and Hixon, 1997; Pondella et al., 2002, 2006). Further, some of the best manmade reefs, for example tall quarry rock reefs with high rugosity and steel oil platform structures with extensive spatial coverage, even show evidence of sustaining higher productivity than natural reefs (DeMartini et al., 1994; Johnson et al., 1994; Claisse et al., 2014; Granneman and Steele, 2014; Pondella et al., 2015b). By using environmental data and SDMs to select preferred sites for placement of such manmade reefs, it may be possible to further optimize restoration efforts.

Yet, creating species distribution models for manmade reef restoration poses some unique challenges. Species distribution modeling has been used to study a number of marine ecosystems (Brodie et al., 2018) and for the conservation of marine species (Robinson et al., 2017), including for habitat restoration. For example, Adams et al. (2016) created SDMs for eelgrass restoration. However, rocky reef ecosystems differ from systems like eelgrass communities as they require specific habitat structures that are largely independent of environmental conditions – rocky infrastructure can and is built in many different places (e.g., manmade reefs, breakwalls, jetties; Morris et al., 2018) – and themselves are not constrained by environmental conditions. Further, such projects are time consuming and costly, requiring a significant amount of planning, collaboration, and management. Thus, it is necessary to establish an approach for identifying preferred candidate sites for rocky reef infrastructure by modeling environmental constraints of species found inhabiting these reefs.

To investigate the utility of SDMs for optimizing the placement of manmade reefs, we created individual-SDMs for 21 species and combined them to create stacked-SDMs to identify hotspots for habitat suitability across multiple species. We validate this approach for identifying candidate sites for habitat restoration by assessing whether the s-SDM values for reefs are positively associated with observed richness of the 21 focal species from reef surveys on already established manmade reefs as well as for an independent dataset of non-focal species that includes 204 fish, invertebrate, and algal species on all surveyed reefs. If SDMs provide an accurate tool for identifying candidate sites, then we would expect multi-species habitat suitability from the s-SDM to increase with observed richness of established manmade reefs.

MATERIALS AND METHODS

Species Distribution Modeling

To determine optimal habitat for each of the focal species, we utilized species distribution modeling using a machine learning approach in the program MaxEnt (Phillips et al., 2006a). This approach allows us to develop a model of habitat suitability for each species based on the environment in places where each species has been observed (Elith and Leathwick, 2009). We can then project that model over all other locations to identify additional suitable habitat. To construct SDMs, we first collated observation data for each of the focal species and downloaded and created spatial environmental data layers.

Observation Data

We initially chose 39 fish, invertebrate, and algal species for this analysis that are targeted commercial or recreational species (CDFG, 2001) or of particular ecological concern for the SCB (**Supplementary Table S1**), including representative fish (e.g., Rockfish, *Sebastes* sp.), invertebrate (e.g., Red sea urchin, *Mesocentrotus franciscanus*), and algal species (e.g., Giant kelp, *Macrocystis pyrifera*). Spatial locality information was collected for each of the focal species from long-term monitoring surveys from the Vantuna Research Group (VRG; Pondella et al., 2015a; Zahn et al., 2016), Channel Islands National Park Kelp Forest Monitoring Program (KFM; Kushner et al., 2013), and the Partnership for the Interdisciplinary Studies of Coastal Oceans (PISCO; Hamilton et al., 2010; Caselle et al., 2015; Pondella et al., 2015a). These observations were made from transect surveys on rocky reefs across the entire SCB at 296 sites during 35 years from 1982–2017 (Kushner et al., 2013; Caselle et al., 2015; Pondella et al., 2015a; Zahn et al., 2016). In short, divers conducted subtidal surveys up to 30 m deep with a depth-stratified random sampling design at each site in which randomly located transects were sampled using four methods: (1) fish density and size distribution are recorded along 30 m belt transects on the reef, in the midwater and in the top section of the water column if kelp canopy is present, (2) density of large (>2.5 cm) motile invertebrates and macroalgae recorded along 30 m "Swath" transects, (3) percent cover of sessile invertebrates, turf algae, and habitat characteristics are estimated using uniform point contact along 30 m transects on the reef and (4) size frequency data for commercially and ecologically important invertebrates (Kushner et al., 2013; Caselle et al., 2015; Pondella et al., 2015a; Zahn et al., 2016). We used only presence and absence data from these surveys.

We divided the dataset into localities from *natural* rocky reefs (578 sites) and *manmade* reefs (38 sites). The natural reef data were split into training and test data (described below), whereas the manmade reef data were used only for validating the models. To prevent spatial bias, we used spatial thinning to remove points within 1km of one another. Spatial thinning was completed using the "spThin" R package (Aiello-Lammens et al., 2015). Only species with at least 30 unique observed localities on natural reefs greater than 1km apart were included in subsequent analyses. The SDM method used in this study, MaxEnt (described below), is less sensitive to small sample sizes (10–30) than other distribution modeling methods available, although caution should still be taken in interpreting models with the smallest sample sizes (Wisz et al., 2008). Preliminary analyses of species with fewer than 30 unique observations resulted in SDMs with low support. Of the initial 39 focal species, 21 had at least 30 unique observed locations at least 1 km apart (**Table 1**). These 21 species included 16 fish, six invertebrates, and one algal species.

Environmental Data

Spatially gridded environmental data was collected for the entire SCB. We used six environmental variables at a resolution of 200 m^2, including: aspect, bathymetry, mean annual Chlorophyll-A (ChlA), distance to the 200 m isobath (a proxy for upwelling potential), slope, and mean annual sea surface temperature (SST). For bathymetry, we used a seafloor bathymetry digital elevation model (DEM) which is a product of the California Department of Fish and Wildlife Bathymetry Project. This coastwide 200 by 200 m DEM was clipped to the extent of the Southern California Bight. Seafloor aspect and slope were derived from the bathymetry DEM using the Aspect and Slope tools in ArcMap 10.3. We collated data from MODIS-derived sea surface temperature (SST; degrees Celsius) and Chlorophyll-A (ChlA; $mg \cdot m^{-3}$) from the University of California San Diego, Scripps Institution of Oceanography Photobiology Group[1]. The raw data consists of 15 day averages throughout the California Current Large Marine Ecosystem. We took the mean of each year from 2002–2017 and then took the grand mean of all years. Both SST and ChlA were down-sampled in R using the bilinear method. All data layers were projected to the WGS 1984 UTM Zone 11N coordinate system to limit distortion. We masked each of the environmental layers using the 45 m isobath contour to restrict all analyses to only cells with *average* depths shallower than or equal to 45 m, since all reef survey observation data is limited to this region. We tested for correlations among each of the environmental variables at each of the unique locations from the thinned dataset using Pearson Correlation Coefficient. None of the 15 pairwise comparisons of the six environmental variables were highly correlated at observed focal species localities, with $|r| \leq 0.5$ for all.

Reef presence across the SCB was identified based on a composite of hard-bottom substrate and historical kelp canopy cover (Williams et al., unpublished; Zellmer et al., 2018). We created a second stack of environmental data layers with all variables masked by reef presence. This masked raster stack was used to build individual-SDMs on current established reefs and then the full raster stack was used to project the individual-SDMs across the remaining area in order to identify candidate sites for restoration in the SCB.

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[1] http://spg.ucsd.edu

TABLE 1 | Best fit MaxEnt model selected for each of the 21 focal species.

Species	Common Name	RM	AUC	Mean AUC	Diff. AUC	*w*	*K*	*n*
Sebastes caurinus	Copper Rockfish	3	0.87	0.84	0.04	0.51	15	54
Haliotis corrugata	Pink Abalone	5	0.86	0.80	0.08	0.58	8	63
Paralabrax nebulifer	Barred Sand Bass	4	0.85	0.82	0.07	0.90	11	57
Haliotis fulgens	Green Abalone	4	0.85	0.78	0.10	0.70	9	58
Sebastes serriceps	Treefish	6	0.84	0.81	0.05	0.92	6	69
Sebastes mystinus	Blue Rockfish	3	0.84	0.80	0.05	0.99	13	73
Sebastes serranoides	Olive Rockfish	3	0.82	0.77	0.08	0.91	8	80
Panulirus interruptus	California Spiny Lobster	3	0.82	0.74	0.09	0.46	15	102
Sebastes miniatus	Vermilion Rockfish	5	0.81	0.76	0.05	0.67	8	48
Sebastes carnatus	Gopher Rockfish	5	0.81	0.76	0.05	0.52	8	64
Sebastes atrovirens	Kelp Rockfish	4	0.81	0.77	0.05	0.82	11	111
Sebastes chrysomelas	Black and Yellow Rockfish	7	0.80	0.77	0.05	0.53	2	59
Sebastes auriculatus	Brown Rockfish	5	0.80	0.76	0.07	0.99	8	41
Megathura crenulata	Giant Keyhole Limpet	7	0.80	0.78	0.06	0.34	8	117
Kelletia kelletii	Kellet's Whelk	6	0.80	0.74	0.09	0.20	10	112
Semicossyphus pulcher	California Sheephead	4	0.79	0.76	0.05	0.41	12	128
Macrocystis pyrifera	Giant Kelp	4	0.79	0.76	0.05	0.33	14	123
Scorpaenichthys marmoratus	Cabezon	7	0.78	0.73	0.06	0.40	5	56
Paralabrax clathratus	Kelp Bass	7	0.78	0.76	0.05	0.43	9	126
Ophiodon elongatus	Lingcod	6	0.78	0.69	0.11	0.71	4	42
Mesocentrotus franciscanus	Red Sea Urchin	6	0.78	0.76	0.05	0.30	8	120

All candidate models tested allowed for all feature classes to be used (LQHPT) but varied in the regularization multiplier (RM). AUC is the Area Under the Curve for the full dataset. Mean AUC is averaged across each of the iterations for only the training data. Diff AUC is the mean difference in the AUC values between the training data and the test data. w is the Aikaike weight. K is the number of parameters included in the MaxEnt model. n is the total number of unique observations of each species on natural reef sites prior to splitting into training and test datasets.

Individual-SDMs

Individual SDMs were developed for each species using MaxEnt v. 3.4.1, a presence-only machine learning approach to modeling species distributions (Phillips et al., 2006b) called through the R programming language (R Core Team, 2015). MaxEnt includes two options, feature classes and a regularization multiplier, to customize models and control overparameterization. Feature classes are a transformation of the environmental variables to enable modeling of complex relationships and include linear, product, hinge, threshold, and quadratic (Elith et al., 2010), whereas the regularization multiplier adds a penalty for overparameterization (Elith et al., 2010; Shcheglovitova and Anderson, 2013). By default, MaxEnt allows all feature classes to be selected in training the model and uses a regularization multiplier of one as determined by optimization from empirical studies across a variety of species (Phillips and Dudík, 2008). However, these parameters need to be optimized for each species to prevent overly simplified or overly complex models (Radosavljevic and Anderson, 2014; Morales et al., 2017). Thus, we utilized a model selection approach to compare models based on the corrected Akaike information criterion (AICc) approach for SDMs developed by Warren and Seifert (2011) and implemented in the "ENMeval" R package (Muscarella et al., 2014). For each species, we tested a set of 12 candidate models each with a different regularization multiplier (1–12, increasing by one) and allowed all feature classes in each model. We used the "block" method for model evaluation to account for spatial autocorrelation. This approach divides the data into four spatial blocks. The model is then run four times with three blocks set as training data and one block set as test data for each iteration and evaluation metrics are then summed across the iterations (Muscarella et al., 2014).

Models were evaluated first by comparing the mean Receiver Operating Characteristic Area Under the Curve (AUC) for the training data to the test data. This value measures the true positive rate to the false positive rate at varying thresholds for classifying habitat suitability. AUC values close to one indicate good fit of the model to the data whereas an AUC value of 0.5 indicates the model is no better than random. Comparing the AUC values for the training to test data allows us to validate how well the models fit an independent dataset, thus smaller differences in the training and test data AUCs indicate better transferability. In addition, we calculated AICc scores to compare the 12 candidate models for each species, allowing us to evaluate the fit of each model while accounting for the number of parameters in each model. The model with the lowest AICc score was considered a best model of the candidate models (Burnham and Anderson, 2002) and was used for subsequent analyses. To identify suitable habitat for reef restoration, the best-fit model was then projected across the entire study area using the complementary log-log link (cloglog) function, which is more appropriate for estimating probability of presence than the previous MaxEnt default, a logistic transformation (Phillips et al., 2017).

Stacked-SDMs

To create a model for predicting preferred locations for manmade reefs that optimizes suitability across most of our focal species, we constructed a stacked-species distribution model (s-SDM) by combining each of the individual-SDMs. To do this, we simply added together each of the individual-SDMs as derived by Calabrese et al. (2014). We selected this approach as opposed to combining thresholded binary habitat suitability classification, since combining thresholds has been shown to result in biased s-SDMs (Calabrese et al., 2014).

S-SDM and Species Richness

To assess the quality of the SDMs for identifying high quality habitat for restoration, we evaluated the extent to which the s-SDM is associated with species richness. We calculated observed species richness at each reef site and tested for a positive linear relationship with the s-SDM score for each reef site using linear regression. We performed this analysis first at all of the natural reef sites for the 21 focal species and then used the linear regression to predict species richness at already established manmade reef sites (test dataset 1) and at all reef sites for 204 other non-focal fish, invertebrate, and algal species (test dataset 2).

Species richness was calculated using only the VRG reef survey data (Pondella et al., 2015a; Zahn et al., 2016) to ensure standardized sampling. To quantify species richness, we estimated species richness using the R "vegan" package using the bootstrap estimator, since sites were surveyed an uneven number of times. We calculated bootstrap species richness for all fish and swath (algae and invertebrate) surveys separately then added the estimates together for each site. We used linear regression to statistically evaluate the relationship between observed bootstrap species richness and s-SDM scores. We include two covariates in the model to account for variation in quality of individual reef sites, depth zone and standard deviation (SD) of reef relief. Depth zone describes the different depths at which a reef was surveyed: inner (∼5 m), middle (∼10 m), outer (∼15 m) and deep (∼25 m). Reef relief was measured at 31 points along each survey transect and a higher standard deviation of these relief measurements indicates greater fine scale habitat heterogeneity. The linear regression was evaluated using AICc by comparing to a null model with SD relief and depth zone alone. The linear model was then used to calculate predicted species richness at manmade reef sites. Observed versus predicted bootstrap richness at manmade reef sites was evaluated using linear regression and AICc by comparing to a null model. In addition, we tested whether predicted values from the linear model were correlated with observed bootstrap richness for the 204 other non-focal fish, algae, and invertebrate species surveyed at all reef sites. This additional independent dataset allows us to test not only the validity of the model but also whether the focal species list is sufficient to predict restoration sites for the reef-associated communities or if only applicable to the species included in the model.

Identifying Candidate Restoration Sites

To identify candidate sites for reef restoration, we isolated regions where there is high predicted habitat suitability across multiple species but no existing reefs using the reef data layer. High predicted habitat suitability was defined as the s-SDM score at or above which the linear model predicts species richness as equal to half the number of focal species. We calculated the proportion of cells with an s-SDM score above this threshold for the entire study region as well as for only cells outside of existing reef areas. Cells outside of existing reef areas with greater than the s-SDM threshold are considered candidate regions for installation of manmade reefs, whereas cells within existing reef areas with greater than the s-SDM threshold are considered candidate regions for restoration or rehabilitation of existing reef habitat.

Existing Manmade Reef Quality Assessment

We further assessed the predicted habitat suitability of already established manmade reef sites in the SCB to determine the current quality of restored habitats. We extracted s-SDM scores for all manmade reefs in our study region ($n = 21$) and for all natural reef survey sites ($n = 250$) and calculated the mean for both. We compared mean s-SDM scores for manmade and natural reef sites with a t-test. We then conducted a permutation analysis by randomly sampling sites across the study region ($n = 21$) and calculating the mean value of the s-SDM at those sites iterated 1000 times. We then compared the mean s-SDM value of the manmade sites as well as the mean s-SDM value of the natural sites to the distribution to quantify significance. As habitat restoration may be limited to areas where reefs do not currently exist and therefore random selection of sites may be artificially biased as being more suitable, we recalculated the null distribution from only areas in the SCB where there is no existing reef habitat and reran the analyses.

RESULTS

Individual-SDMs

For each species, we selected the best fit model (ΔAIC = 0) from the 12 candidate models with varying values for the regularization multiplier (**Supplementary Figures S1–S21**). Based on mean AUC values for the test data, all models predicted test observations well (mean AUC range: 0.69–0.84; **Table 1**) and were not overfit (difference between training and test AUC range: 0.04–0.11; **Table 1**). The optimal regularization multiplier selected for each species was higher than the default value in MaxEnt (1) and ranged from 3–7 (**Table 1**). Of the nine environmental predictors, Slope (52%), Distance to 200m (22.6%), and Bathymetry (8.7%) on average contributed the most to each of the individual-SDMs (**Figure 1**). This pattern was consistent among fish and algae species, although for invertebrate species, Distance to 200 m contributed more on average than Slope (**Figure 1**).

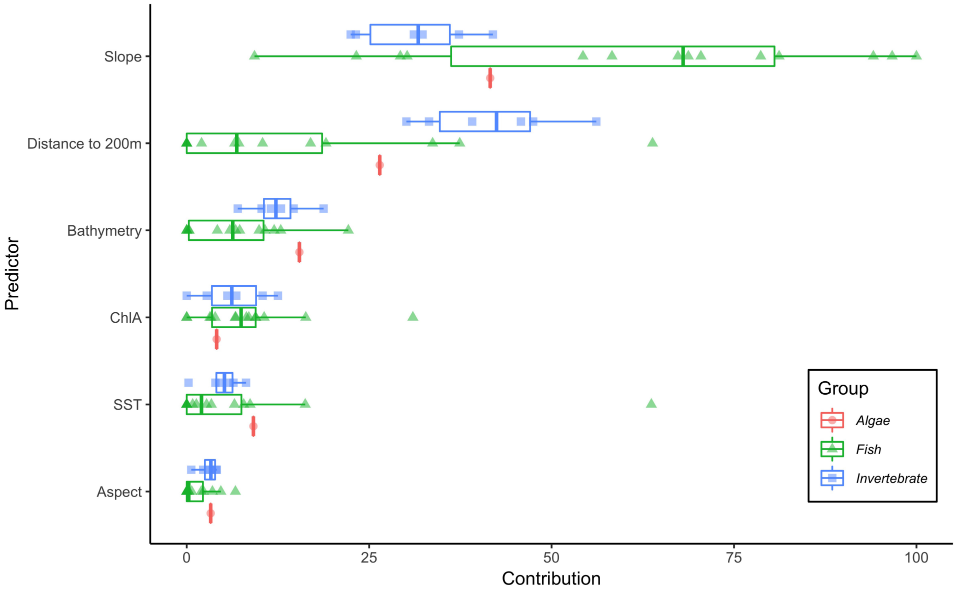

FIGURE 1 | Percent contribution of each of the six predictor variables to the individual-SDMs. Interquartile ranges shown for the three taxonomic groups, algae (red circles), fish (green triangles), and invertebrates (blue squares).

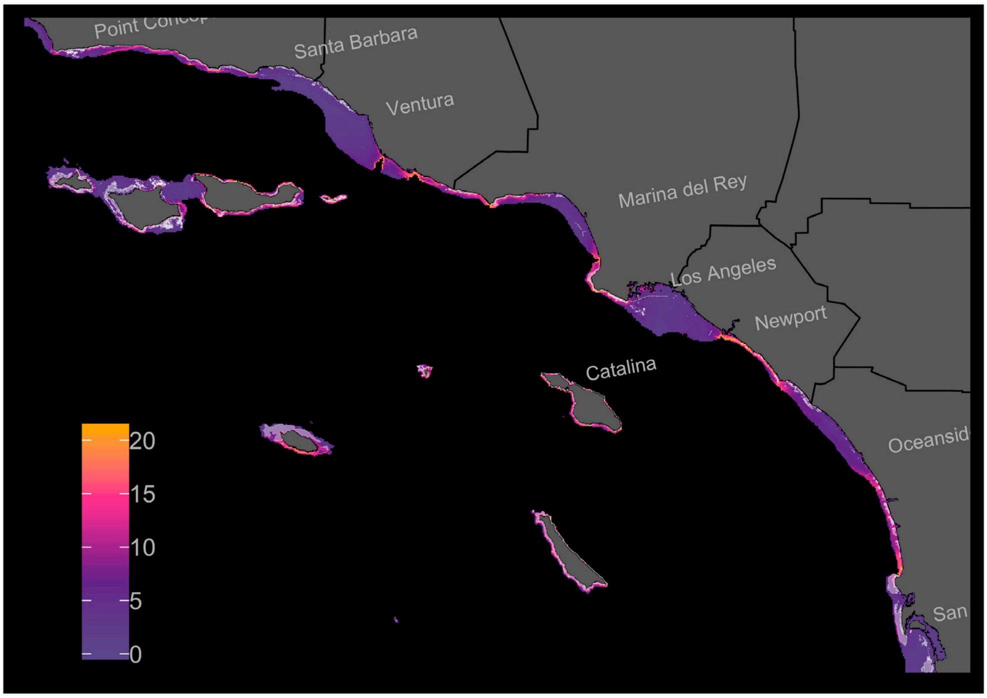

FIGURE 2 | Stacked-Species Distribution Model (s-SDM) showing predicted multi-species habitat suitability at a resolution of 200 × 200 m and within the 45 m isobath in the Southern California Bight. Individual-SDMs were created for 21 focal fish, invertebrate, and algal species across existing reef habitat and then projected across the entire study region. The individual-SDMs were then added together to create an s-SDM. Existing reef habitat shaded with white. Color ramp shows s-SDM scores.

Stacked-SDM

The s-SDM showed high variation in predicted multi-species habitat suitability across the Southern California Bight coastline for the 21 focal species (**Figure 2**). Approximately 38.7% (1132 km^2) of the studied region included habitat that is predicted to be suitable for many of the focal species. In general,

TABLE 2 | Linear regression results comparing s-SDM score with observed bootstrap species richness.

Dataset	Predictor	Estimate	SE	2.5% CI	97.5% CI	AICc	ΔAICc	*w*
Natural Reef Focal	Intercept	7.10	0.79	5.55	8.64	1650.44	0	1
	s-SDM Score	0.27	0.05	0.17	0.36			
	SD Relief	2.39	0.89	0.64	4.14			
	Depth Zone - Inner	–4.09	0.59	–5.25	–2.93			
	Depth Zone - Middle	–3.30	0.60	–4.49	–2.12			
	Depth Zone - Outer	–2.10	0.61	–3.31	–0.90			
Manmade Reef Focal	Intercept	2.71	1.59	–0.83	6.26	49.07	0	1
	Predicted Bootstrap Richness	0.52	0.18	0.13	0.92			
All Reef Non-Focal	Intercept	–3.80	3.28	–10.25	2.64	2569.91	0	1
	Predicted Bootstrap Richness	3.65	0.37	2.93	4.37			

The s-SDM model used was based on 21 focal species. The linear model was calculated first for only the natural reef sites for just the 21 focal species (Natural Reef Focal), and the linear model was then used to calculate predicted species richness values for already established manmade reef sites (Manmade Reef Focal). Predicted and observed bootstrap species richness were compared with linear regression. The association between predicted and observed bootstrap species richness was also evaluated for 204 additional non-focal fish, algae, and invertebrate species (All Reef Non-Focal). ΔAICc is reported relative to the null model tested for each dataset.

average habitat suitability was lower inside bays and higher along points and around islands. The model identified multiple regions with high average predicted habitat suitability that do not already contain reef habitat (**Figure 2**). After removing cells with existing reef habitat, there remained approximately 33% (804 km^2) of the remaining study region that included habitat that is predicted to be suitable for many of the focal species.

For the focal species on natural reefs, the model including s-SDM scores and the two covariates, SD relief and depth zone, better predicted observed bootstrap species richness than the null model with only the two covariates (w = 1; **Table 2** and **Figure 3**). Observed bootstrap species richness increased with increasing s-SDM scores (R^2 = 0.22, slope = 0.27, 95% CI = 0.17–0.36). When this model was used to predict species richness for the established manmade reefs, there was high support for a relationship between observed and predicted species richness (slope = 0.52, 95% CI = 0.13–0.92, w = 1; **Table 2** and **Figure 4A**). Similarly, predicted species richness values from this linear model were positively associated with observed bootstrap species richness when considering 204 other non-focal fish, invertebrate, and algal species that were surveyed on both natural and manmade reefs (slope = 3.65, 95% CI = 2.93–4.37, w = 1; **Table 2** and **Figure 4B**).

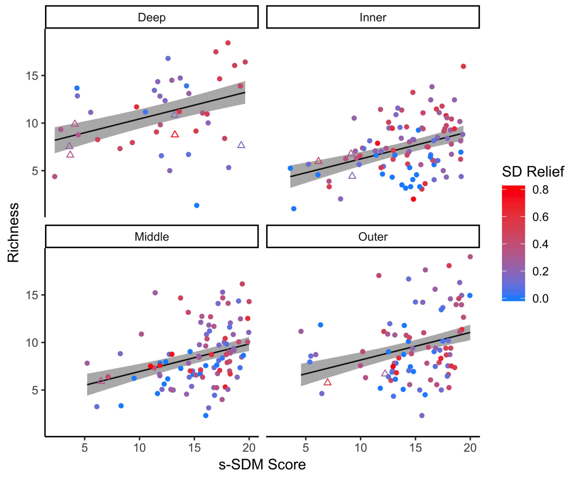

FIGURE 3 | Linear relationship between predicted s-SDM score and observed bootstrap species richness for the 21 focal species. The linear regression was completed for the 21 focal species on all natural reef sites (circles) at four depth zones: inner (∼5 m), middle (∼10 m), outer (∼15 m) and deep (∼25 m). Established manmade reef sites (triangles) for reference. Colors indicate standard deviation (SD) of relief at each reef site, from low (blue) to high (red) fine scale habitat heterogeneity. 95% confidence intervals of the regression lines are shown in gray.

Quality of Existing Manmade Reefs

To assess the habitat suitability of previously established manmade reefs, we extracted the s-SDM score for each survey site and compared s-SDM scores among natural and manmade reefs to one another and relative to a random distribution. Manmade reefs (mean = 8.7) significantly differed on average from natural reefs (mean = 14.2) in s-SDM scores (t-value = –5.4, P-value = 1.8e–05; **Figure 5A**). Manmade reefs had an average s-SDM score that was slightly although significantly greater than randomly selected sites, both when the entire study region was considered (**Figure 5B**) and when only areas with no existing reef was considered (P-value < 0.01) (**Figure 5C**).

DISCUSSION

Improving the success of habitat restoration projects is a necessity as ecosystems worldwide continue to face increasing anthropogenic pressures and habitat loss. This need is especially great in marine ecosystems, due to increasing coastal urbanization (Dafforn et al., 2015; Morris et al., 2018). Species distribution modeling can be used as an important tool in identifying the best places where habitat restoration is likely to be successful by identifying suitable habitat (Pearce and Lindenmayer, 1998). Here we apply this method to the ecosystem level by calculating individual species distribution models for 21 focal species from shallow rocky reefs and stack

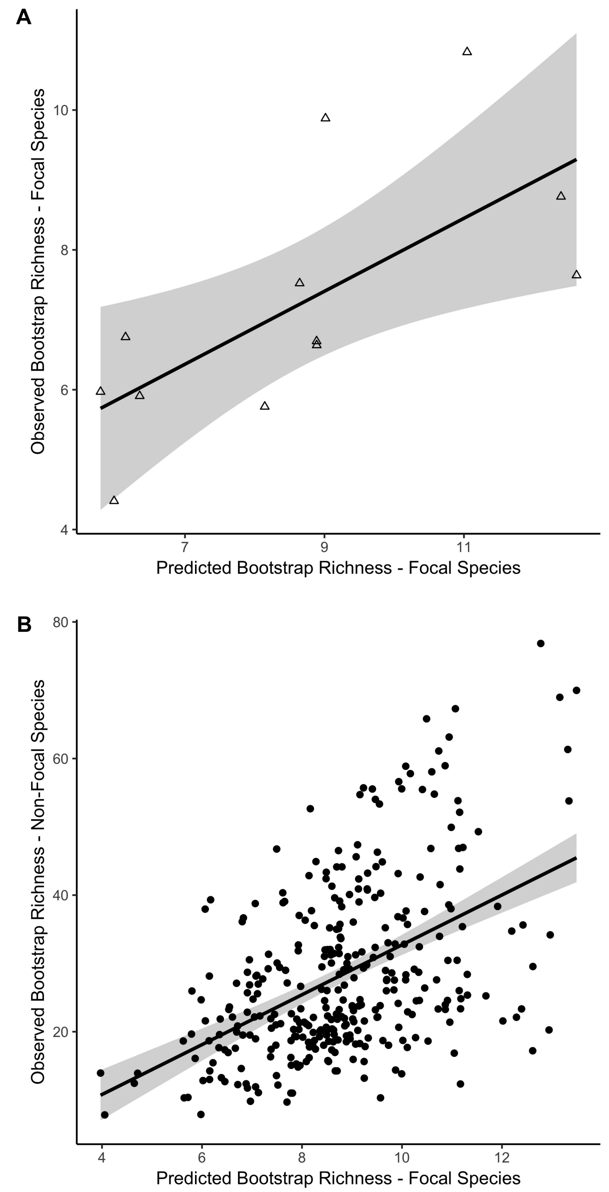

FIGURE 4 | Linear relationship between predicted and observed bootstrap species richness for two independent datasets based on the best fit model for natural reefs. Predicted versus observed species richness for **(A)** the 21 focal species on established manmade reefs and **(B)** the 204 non-focal fish, algae, and invertebrate species on all reef sites, including natural and manmade, at four depth zones. Colors indicate mean standard deviation (SD) of relief at each reef site, from low (blue) to high (red) relief. 95% confidence intervals of the regression lines are shown in gray.

these SDMs to identify areas with suitable habitat for a majority of the species. Our results illustrate a number of potential areas within the Southern California Bight – an area with immense human pressure due to proximity to major metropolitan areas – where habitat is predicted to be suitable for the majority of our focal species, including many areas that do not already contain natural or manmade reefs (**Figure 2** and **Supplementary Figures S1–S21**). This approach allows us to identify sites for habitat restoration using organism-based habitat considerations rather than simply landscape-(or seascape-)based considerations, which is crucial when restoring habitat for multiple species (Miller and Hobbs, 2007).

Moreover, when the individual-SDMs were combined together as the s-SDM, there was a positive linear relationship between s-SDM scores and observed bootstrap species richness on the natural reefs when considering only the 21 focal species (**Table 2** and **Figure 3**), and this relationship was validated by two independent datasets. First, the predicted richness values from this linear model were associated with observed bootstrap species richness at manmade reefs when considering only the 21 focal species (**Table 2** and **Figure 4A**). Second, the predicted richness values from this linear model were also correlated with increases in observed species richness when considering all other 204 fish, invertebrate, and algae species surveyed on Southern California shallow rocky reefs (**Table 2** and **Figure 4B**). Thus, by identifying crucial focal species and combining distribution models for each of these species, it is possible to identify areas that may support greater species richness. For restoration projects in which species diversity or richness is a primary goal (Wortley et al., 2013), this method may provide an opportunity for managers to successfully select more-ideal locations for restoration.

While species richness generally increases with increasing s-SDM scores, there is a high degree of variability, particularly for sites with the highest s-SDM scores. This pattern suggests that additional factors influence suitability of a site beyond just environmental suitability. While our approach identifies environmental suitability and potential locations for habitat restoration, suitability does not guarantee success on its own (Higgs, 1997). Additional factors that need to be considered when selecting sites, include: habitat design (Baine, 2001), species relationships (e.g., Jude and Deboe, 1996), cultural needs (Higgs, 1997), and public participation and socioeconomic factors (Wortley et al., 2013). With Species Distribution Models, species relationships are especially important to consider as SDMs do not inherently account for ecological relationship such as competition and predation (Freeman and Mason, 2015). Thus, some variation could be explained by which species are present at particular reefs.

Additionally, and possibly alternatively, the variation in species richness of high quality sites may instead reflect habitat degradation. For instance, highly suitable sites may be overfished (Zellmer et al., 2018) or exposed to pollution (Schaffner et al., 2015). In fact, some of the natural sites for which there is lower than expected species richness despite high s-SDM scores include some of the more degraded reef sites in our study (high s-SDM score, low richness; **Figures 3**, **4B**). Since many of the best predicted s-SDM scores are on or near existing reefs, our results suggest that there may be immense opportunity for restoring natural reefs as opposed to simply building manmade structures in areas where rocky reefs did not previously exist. In other words, it is important to consider the difference between habitat "restoration" or "rehabilitation" versus habitat "conversion" (Erftemeijer and Lewis, 1999). Restoring previously existing reefs

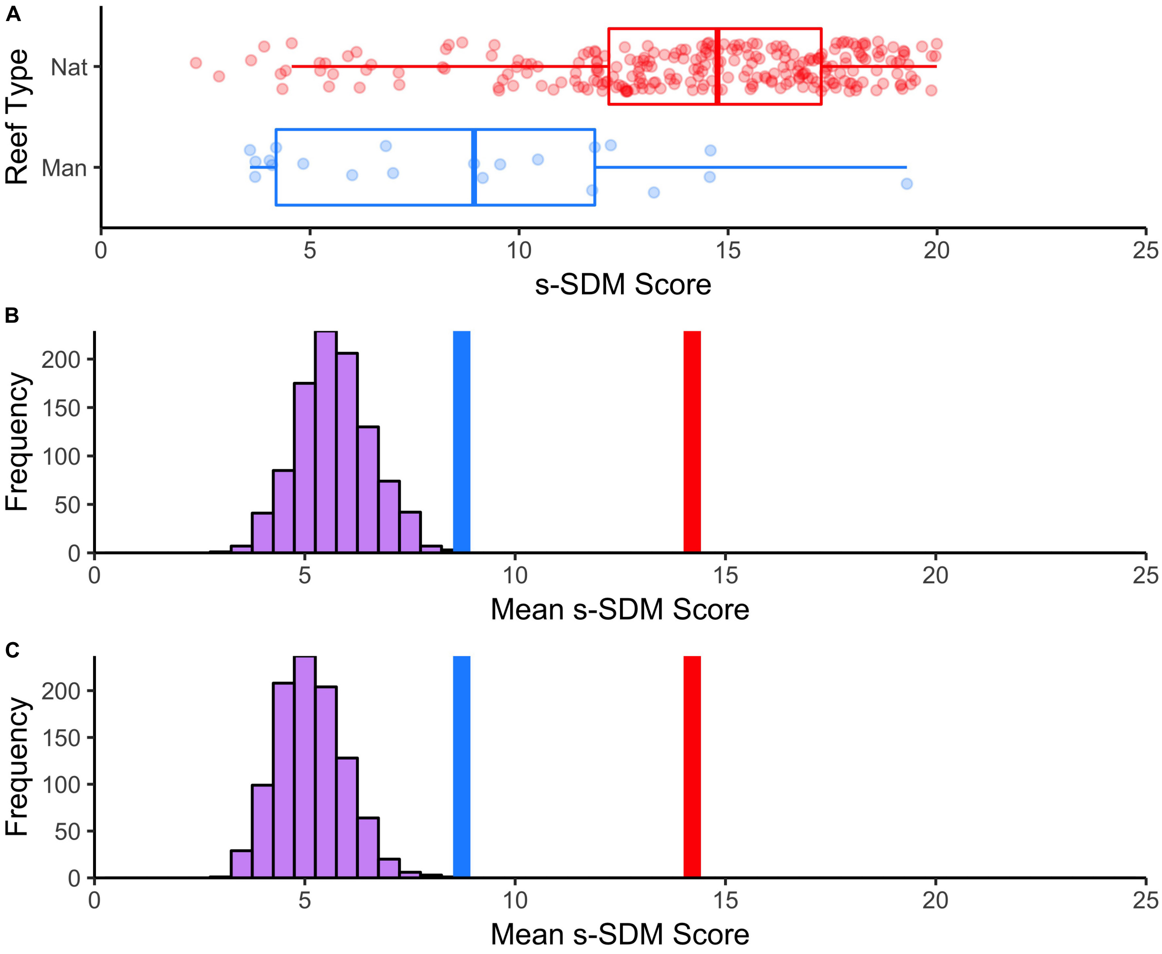

FIGURE 5 | Mean s-SDM scores of existing manmade reefs relative to natural reefs and compared to random. **(A)** Boxplot of differences in s-SDM scores for natural (blue) and manmade (red) reef sites. Mean s-SDM score was calculated for the already existing solitary manmade reefs within the SCB (n = 21, blue line) and for each of the natural reef sites (n = 250, red line). The means were compared to a permuted distribution of 21 randomly selected sites iterated 1000 times. **(B)** Random sites chosen from anywhere in the SCB. **(C)** Random sites chosen from only areas with no existing reef.

may not only be more cost effective, but as our results suggest, may also be more likely to succeed based on habitat suitability.

Further, the variability observed could also be explained by the physical structure and design of reefs. Previous research has shown that reef structure is an important component of restoration success (Baine, 2001; Pondella et al., 2006). Consistent with this previous research, our analyses suggest that even when environmental conditions are suitable, reef structure may influence species presence as predicted species richness was more accurate for purposefully designed manmade reef structures as opposed to unintended manmade reef structures. Thus, once candidate sites are selected based on habitat suitability, restoration should be done in conjunction with expert opinion as to the specific design of manmade reefs.

Regardless of the specific causes of the variability, our model provides an estimate of areas that are predicted to be suitable for multiple species, suggesting that at least some of the focal species could persist in these locations. Conservation managers should consider the locations identified by this model as a set of candidate locations from which they can then select sites after considering these other factors. Thus, this approach adds an additional tool to help managers consider holistic success of habitat restoration. However, while the s-SDM identifies sites where there is high habitat suitability across a majority of the focal species, for some species, such as rare or endangered species (e.g., Abalone, *Haliotis* sp.), more directed conservation measures may be necessary. For such species, the individual-SDMs can be used to help in identifying diverse sites for ecosystem restoration.

Interestingly, the manmade reefs included in this study that are already established in the SCB are in regions that are on average not only less suitable than natural reefs, but are in regions only slightly more suitable than sites selected at random (**Figure 5**). For example, some manmade reefs are placed in gently sloping, sandy-bottom regions. If these reefs had been placed in areas with higher predicted habitat suitability, then it is possible more species could be observed. While we do still observe some species at these locations (Pondella et al., 2015a; Zahn et al., 2016), the lack of habitat suitability suggests that these manmade reefs may be hosting sink populations (van Horne, 1983; Smallwood, 2001). Based on these results, there is strong evidence that habitat restoration may have the most potential when completed at sites with degraded reefs (e.g., inundated by landslides) as opposed to constructing reefs far from existing reef structures. Not only are these more distant sites potentially less suitable, but also have lower connectivity with existing, productive reef habitat (Pondella et al., 2018). With clear predictions for habitat suitability across multiple species, managers can be best prepared to advocate for selection of appropriate sites.

Future Directions

To ensure that habitat restoration is successful in these locations, future studies should focus on continued monitoring and follow up research. While habitat restoration has become an essential tool in conservation biology, long-term assessments of restoration success remain limited (Godefroid et al., 2011; Wortley et al., 2013). Further, future research should consider species specific differences in how they contribute to community biodiversity and success. Finally, species distribution modeling also offers an opportunity to assess how habitat suitability might vary under future global environmental change (Peterson et al., 2002). As global environmental changes continue to occur, it is crucial to consider how those changes influence the goals of habitat restoration (Higgs et al., 2014). Future research should focus on assessing changes in habitat restoration priorities based on potential changes in habitat suitability across multiple species under numerous possible climate scenarios.

AUTHOR CONTRIBUTIONS

AZ envisioned the project, ran the computational analyses, and wrote the manuscript. JC envisioned the project, contributed to field work, data analysis, and writing. CW and SS contributed to the data analysis, field work, and writing. DP envisioned the project, managed and contributed to the field work, and contributed to data analysis and writing.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00003/full#supplementary-material

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Fine-Scale Effects of Boat Moorings on Soft Sediment Communities Masked in Large-Scale Comparisons

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Population growth is driving the demand for recreational marine infrastructure, resulting in extensive coastal habitat modification. Boat moorings, for example, are popular for vessel storage and are known to damage seagrass communities, yet little is known about how they influence unvegetated sediment habitats. Here we investigate the effects of boat moorings on sediment infauna using metrics of community composition, diversity, total abundance and abundances of individual functional groups and dominant taxa. Metrics were compared at fine and larger spatial scales, to investigate how spatial variability affects the ecological assessments in soft-sedimentary environments. Fine-scale models revealed changes in community composition and mollusk abundance with the distance from moorings, while sediment grain size was also an important predictor for composition, bivalve and polychaete abundances, although the direction of effects varied. When the same metrics were compared at larger scales (i.e., boating infrastructure present or lacking) we found that spatial variability among locations was detected, but no effect for moorings. With increased urbanization and industrialization of coastal areas there is a clear need to account for the scale of potential ecological effects in investigations of coastal infrastructure developments.

Keywords: moorings, soft sediment communities, monitoring, spatial variability, marine infrastructure, boating

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INTRODUCTION

Human-induced ecological change has increased over recent decades (Glasson et al., 2013; Lewis and Maslin, 2015; Waters et al., 2016), prompting governments to use legislation to manage user-environment interactions (e.g., Christodoulou and Stamatelatou, 2016; Feng and Liao, 2016; Gunningham and Holley, 2016; Horne et al., 2016; Pettipas et al., 2016; Harvey et al., 2017). In order to conform to these legislative requirements, and to maintain a social license to operate, both industry and government are applying formal impact and monitoring assessments to existing and proposed developments (Glasson et al., 2013). During the past few decades such assessments have become more rigorous, more reliant on formal statistical principles, and are now better able to detect change (Downes et al., 2002; Briggs and Hudson, 2013; Drayson et al., 2017). However, refinement of analytical frameworks must continue and new methodologies and technologies should build on existing techniques to understand ecological changes (Morgan, 2012; Briggs and Hudson, 2013).

Ecological impact assessments (EcIAs) are a component of environmental impact assessments (EIAs) and provide a formal test for ecological change, at a scale relevant to a proposed or existing development (Glasson et al., 2013). EcIAs aim to (1) evaluate whether the activity in question impacts surrounding ecosystems and (2) if present, estimate the magnitude and scale of the disturbance (Downes et al., 2002). All infrastructure is expected to have some effect on natural systems (McKinney, 2002; Benítez-López et al., 2010; Dafforn et al., 2015), so it is often the magnitude and scale of impact that is more pertinent to policy development. The most appropriate EcIA approaches must therefore be designed to test, not only for Impact/No Impact (a simple binary test), but also for the scale (in both space and time) and direction (positive or negative) of such effects. Inappropriately designed assessments may fail to provide policy-makers and decision makers with the most appropriate information they need to manage marine infrastructure development (Legg and Nagy, 2006).

Studies that investigate human-induced ecological change are often geographically restricted, and consist of a few replicate observations taken at relatively large spatial scales, where potentially impacted and reference locations may be 1000s of meters apart (Piersma et al., 2001; Lewis et al., 2002; Dauvin et al., 2006; Wildsmith et al., 2011; Minshall et al., 2014). Such designs may not provide a representative description of the impacts associated with the disturbance in question; particularly the scale of its effects. These approaches also fail to consider important sources of variation (Schmitt and Osenberg, 1996; Downes et al., 2002) especially those occurring at small spatial scales. Many communities are known to exhibit natural small-scale spatial variability (Dexter, 1984; Thrush, 1991; James and Fairweather, 1996; Herman et al., 1999; Fraschetti et al., 2005), but if sampling to infer impact is assessed at larger spatial scales (100s to 1000s of meters), then important information on fine scale species distributions may remain hidden. Similarly, small-scale anthropogenic impacts (restricted developments) on population and/or community level measures will be obscured. EcIAs in these environments may benefit from assessing impact at multiple spatial scales.

Coastal developments that require EcIAs include the local and regional installation of infrastructure such as groins and breakwaters. These are relatively well studied (Connell and Glasby, 1999; Connell, 2001; Martin et al., 2005; Bulleri and Chapman, 2010; Johnston et al., 2011; Rivero et al., 2013; Clark et al., 2015; Sim et al., 2015), but there a paucity of research exists on the ecological and environmental changes associated with estuarine recreational boating infrastructure, such as swing moorings. The small body of research that does exist has primarily focused on seagrass communities (Walker et al., 1989; Hastings et al., 1995; Demers et al., 2013). Studies have shown that moorings negatively impact seagrass beds through scouring, consequently increasing meadow susceptibility to erosion and reducing productivity (Walker et al., 1989; Hastings et al., 1995; Demers et al., 2013). As a result, seagrass-friendly moorings have been used in some areas to reduce the scour and ecological impacts associated with conventional "swing" moorings (Demers et al., 2013), yet their installation is still relatively rare. Moorings have also been linked to changes in grain sizes, organic matter and contamination (Hedge et al., 2017 in a parallel study to this one), however, few studies have assessed the effect of marine infrastructure (reviewed by Heery et al., 2017) or specifically boat moorings on soft-sedimentary communities (but see Herbert et al., 2009; Collins et al., 2010). This is surprising given the large number of dense mooring fields, situated on soft-sedimentary habitats (e.g., >26,000 moorings in New South Wales, Transport for New South Wales, 2014).

In this study we analyze the fine (m) and large (km) scale spatial patterns in soft-sediment assemblages associated with boat moorings. We contrast the sensitivity of sampling at different spatial scales to detect ecological change associated with recreational boating infrastructure. Thus, we simultaneously develop our understanding of boating infrastructure impacts on coastal environments; and evaluate sampling designs for future impact assessments in soft-sedimentary systems.

MATERIALS AND METHODS

Study Area

Sediments were sampled from Sydney Harbour, located on the SE coast of New South Wales, Australia (**Figure 1**). Recent data from Transport for NSW estimates registered recreational vessel numbers in Sydney Harbor at more than 20,000 with annual growth predicted at 2.9% (Transport for New South Wales, 2015). Four locations were chosen, two with large mooring fields and two reference locations containing no boating infrastructure. Clontarf (33°81′S, 151°25′E) and North Harbor (33°80′S, 151°27′E) contained a marina and extensive mooring fields. Quarantine Bay (33°81′S, 151°29′E) and East Rose Bay (33°86′S, 151°27′E) were selected as reference locations with no boating infrastructure within 500 m. All locations are protected from ocean waves and subject to diurnal tidal flushing.

Sampling Design

A Generalized Random Tessellation Stratified (GRTS) sampling design (Stevens and Olsen, 2004) was used to select samples within all locations for consistency. Sampling was stratified by distance to moorings at boating infrastructure locations or unstratified at reference sites. A GRTS design allows for spatially balanced sampling, where sampling effort is distributed evenly across both environmental (distance from shore and moorings) and geographic space (Stevens and Olsen, 2004). This coverage is important when dealing with ecological communities that exhibit variation in spatial patterning (Thrush, 1991). Divers were positioned to collect samples on the seafloor using an Ultra Short Baseline (USBL) sub-sea positioning system (Sonardyne Scout^{TM}) and diver-to-vessel communication equipment. This enabled fine-scale sampling (<1 m intervals) in relation to distance from boating infrastructure.

Thirteen samples were collected from each of the mooring fields and reference locations, for a large-scale impact assessment. Additional samples were taken at the two mooring field locations (including the original 13 samples, $n = 28$ at North Harbor and $n = 36$ at Clontarf) for a finer-scale impact assessment within

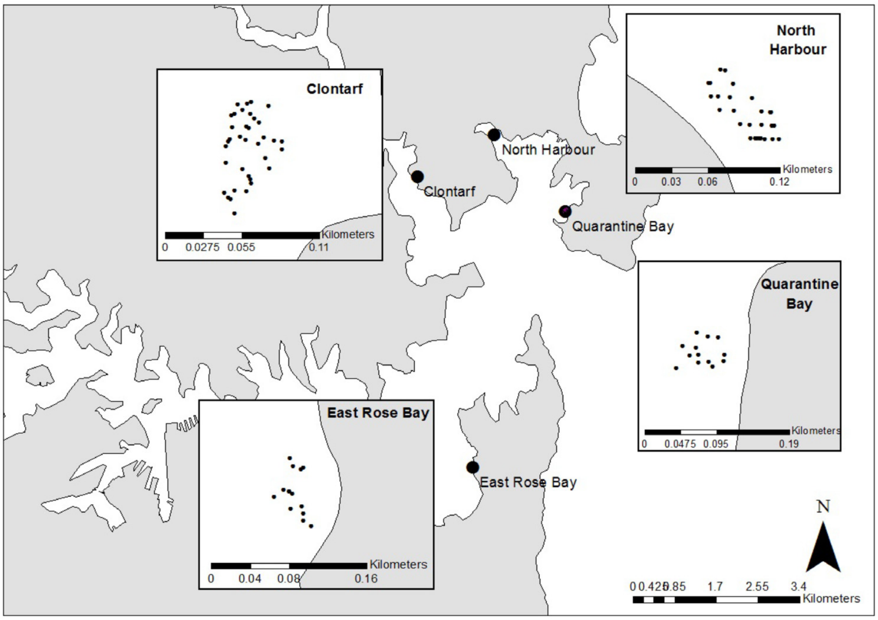

FIGURE 1 | The location of sampling locations, and sites within locations, in Sydney Harbour, Australia. Moorings were present at Clontarf (n = 36 samples taken) and North Harbour (n = 28). The reference locations lacking moorings were East Rose Bay (n = 13) and Quarantine Bay (n = 13).

locations. All sampling was subtidal at between 3 and 8 m below MLWS. Samples were collected during February and March 2015. Each sample comprised 500 mL of sediment, collected from the top 5 cm of the seafloor using a 8 cm × 5 cm corer operated by SCUBA divers.

Analysis of Infauna and Sediment Grain Size

Sediment samples were wet sieved on site through steel sieves with mesh sizes of 2 mm (to remove debris) and 500 μm (to collect biota). To aid with invertebrate identification, samples were stained with Rose Bengal and preserved in 10% formaldehyde. Half the sample was sorted for infauna under a dissecting microscope. The sieving resulted in damage to the organisms in some samples. Further, while the staining aided in picking live material from the sample, it obscured some finer morphological features, preventing classification to species-level for some specimens. Therefore, organisms were identified to order and then classified into morphospecies. Polychaetes were further identified to family as they represent ~70% of taxa within sedimentary ecosystems (Hutchings, 1998; Dafforn et al., 2013).

Subsamples (3–5 g freeze-dried sediment) were assessed using a Malvern laser particle size analyzer (Mastersizer 2000) to quantify grain size distributions (Hedge et al., 2017). The 10th percentile (D10) was used as a representative metric for grain size in further statistical analysis (Blott and Pye, 2001).

Statistical Analyses of Fine Scale Patterns

To test for fine-scale effects of moorings, we constructed generalized linear models (GLMs) to assess patterns in community structure, abundance and diversity (Baeten et al., 2014) within the two locations with mooring fields (Clontarf and North Harbor). Multivariate species composition was modeled as a function of distance to moorings, distance from shore, location, and grain size. Distance from shore was included as a proxy for depth (which was highly correlated) and for potential land-based influences, and grain size was included because of a previously described relationship between grain size and distance from moorings at these locations (Hedge et al., 2017).

Only abundant taxa (containing >10 observed individuals) were included in this analysis, effectively reducing the final taxa count from 144 to 44. This was done because taxa with less than ten individuals found throughout a study are rarely important to the multivariate analysis and may lead to zero-inflation issues during model specification (Zuur et al., 2009; Wang et al., 2012). Due to the presence of non-constant error variances in residuals, GLMs were constructed using a negative binomial distribution and log link in the mvabund package within the R Statistical Environment (Wang et al., 2012; R Core Team, 2014). Full details of this model-based multivariate approach can be found in Wang et al. (2012). Briefly, abundances of infauna taxon j at site i (Y_{ij}) were modeled as a negative binomial process where;

$$Y_{ij} \sim NB(\mu_{ij},\ k) \tag{1}$$

$$E(Y_{ij}) = var(\mathrm{Y_{ij}}) = \mu_{\mathrm{ij}} + \frac{\mu_{ij}^{2}}{k} \tag{2}$$

$$\begin{aligned} n_j = \alpha_j + \beta_1 DistMoor_j + \beta_2 DistShore_j \\ + \beta_3 Loc_j + \beta_4 GSize_j + \beta_5 LocxDistMoor_j \end{aligned} \tag{3}$$

A multivariate test statistic (Wald value) was used to test for the effects of each factor with inference from bootstrap resampling with 999 samples.

The univariate response variables of total abundance per sample, abundance of each of the dominant higher taxa (polychaetes, crustaceans, and mollusks) and diversity, were also modeled with GLMs in mvabund with the same independent variables as above, and inference from bootstrap resampling. The abundances were modeled using a negative binomial distribution with a log link. Analyses of diversity (Shannons Diversity H') used a gamma distribution and log link such that:

$$Diversity_i \sim \; Gamma \; (\mu_i, \; \tau) \tag{4}$$

$$E\left(Diversity_i\right) = \; \mu_i, \; var = \; \frac{\mu_i^2}{\tau} \tag{5}$$

$$\begin{aligned} \log(\mu_i) = \alpha_i + \beta_1 DistMoor_i + \beta_2 DistShore_i + \beta_3 Loc_i \\ + \beta_4 Gsize_i \; + \beta_4 LocxDistMoor_i \end{aligned} \tag{6}$$

For both multivariate and univariate models, model validation was done by examining both residual plots and mean-variance plots in order to confirm if the choice of the distribution family for these analyses was appropriate. A correlation plot between grain size and distance from moorings revealed no correlation between the two variables confirming the validity of our model ($P = 0.86$).

Statistical Analyses of Large-Scale Patterns

To test for larger scale impacts of boating infrastructure, we contrasted species composition, abundance and diversity among all four locations with GLMs. Multivariate species composition was modeled in mvabund with location as the predictor variable and a negative binomial distribution. The total abundance per sample and the abundance of each of the dominant higher taxa (polychaetes, crustaceans and mollusks), were also modeled with GLMs, using a negative binomial distribution. Diversity was modeled with a gamma distribution. Statistical inference and model validation were done as described above, and for all analyses, 13 replicate samples were randomly selected from the mooring field locations to ensure a balanced data set. With two locations per impact category (mooring fields vs. reference), we considered location a fixed factor and tested for differences among these two categories by pair-wise tests among all locations (using the Holm method for adjusting for the inflated error rates associated with multiple tests) (Holm, 1979). Differences in the species composition among locations were visualized with multi-dimensional scaling using the Bray-Curtis similarity coefficient and square root transformed data.

RESULTS AND DISCUSSION

Soft-sediment infaunal communities are important components of marine ecosystems, and critical to major ecosystem processes (Snelgrove, 1997; Lohrer et al., 2004; Douglas et al., 2017). Given their ecological significance, it is important that we understand how building coastal infrastructure, such as boat moorings, might affect these sediment communities and whether current impacts assessment methodologies have sufficient sensitivity to detect such effects. We found that comparisons at large scales failed to detect ecological change related to boat moorings, however, fine -scale effects of boating infrastructure on infauna were detected when considering environmental space (distance to moorings). Furthermore, while distance to moorings was found to be a predictor of ecological change, the response of each taxon varied considerably. No 'one' taxa were the main driver of this change, but rather, the effects of moorings manifested in a combination of small changes in several taxa of the communities at both locations with boat moorings present. This highlights the importance of sampling at multiple scales in impact assessments and considering taxon-specific responses.

Fine-Scale Effects of Boat Moorings

A significant multiplicative effect of distance to moorings, distance from shore, location and grain size was found on infaunal community composition (**Table 1**). While the communities nearer to the moorings were different than those found further away (**Table 1**), the direction and magnitude of this response for individual taxa varied considerably (**Figure 2** and **Supplementary Table S1**). We observed similar fine scale impact of moorings on fish communities in a parallel study (Lanham et al., 2018). No single taxon could be identified as most strongly associated with these compositional changes. Nematodes were the most abundant in close proximity to moorings (**Figure 2**) and decreased with distance from moorings independent of spatial differences (**Supplementary Table S1**). They are often used as indicator species because of their differential sensitivity to disturbance and here their response may be related to greater physical disturbance close to the swing mooring change (Bongers and Ferris, 1999; Boyd et al., 2000). Among the species with the strongest relationship with distance from moorings (**Figure 2**), there were taxa that were more abundant at sites further away from moorings, but only at Clontarf (Orbiniid polychaetes, ostracod sp. 1), while other taxa were more abundant at sites closer to moorings but, again, only at Clontarf (ostracod sp. 6, bivalve sp. 1).

One potential physical change we expected from the swing moorings was a change in grain size due to scouring (Demers et al., 2013; Hedge et al., 2017). Overall the grain sizes were similar and sandy at all sites (**Supplementary Figure S1**). Clontarf sediments had ∼4% silt, while North Harbor sediments were composed of ∼6% silt. Both Quarantine Bay and East Rose Bay had around 3% silt. While we found significant relationships between infauna and grain size changes, these were independent of the distance from the mooring. The relationships between abundance and grain size differed among taxa, and between locations within taxa (**Supplementary Figure S1** and **Table S1**). At Clontarf, some taxa, such as bivalve sp. 1 and ostracod sp. 1, decreased in abundance with increasing sediment grain size (**Supplementary Figure S1**), while Pectinariidae and Opheliidae (Polychaetes) increased in abundance with increasing sediment size (**Supplementary Table S1**). Infaunal communities

TABLE 1 | Results from generalized linear models used to test for effects of distance to moorings, distance from shore, location and grain size on the multivariate composition of infaunal species, and the univariate total abundance per sample and diversity (Shannons Diversity H').

Term	Species composition		Total abundance		Diversity	
	Estimate	***P*-value**	**Estimate**	***P*-value**	**Estimate**	***P*-value**
Distance from mooring	7.40	**0.04**	−0.0006	0.96	0.003	0.91
Distance from shore	9.53	**0.001**	−0.006	0.21	−0.003	0.89
Location	10.67	**0.001**	−0.14	0.69	−0.36	**0.002**
Grain size	8.17	**0.02**	0.001	0.35	−0.06	0.05
Distance from mooring × location	6.12	0.08	−0.008	0.75	−0.04	0.59

Statistical inference is from bootstrap resampling. P-values in bold are significant at α = 0.05.

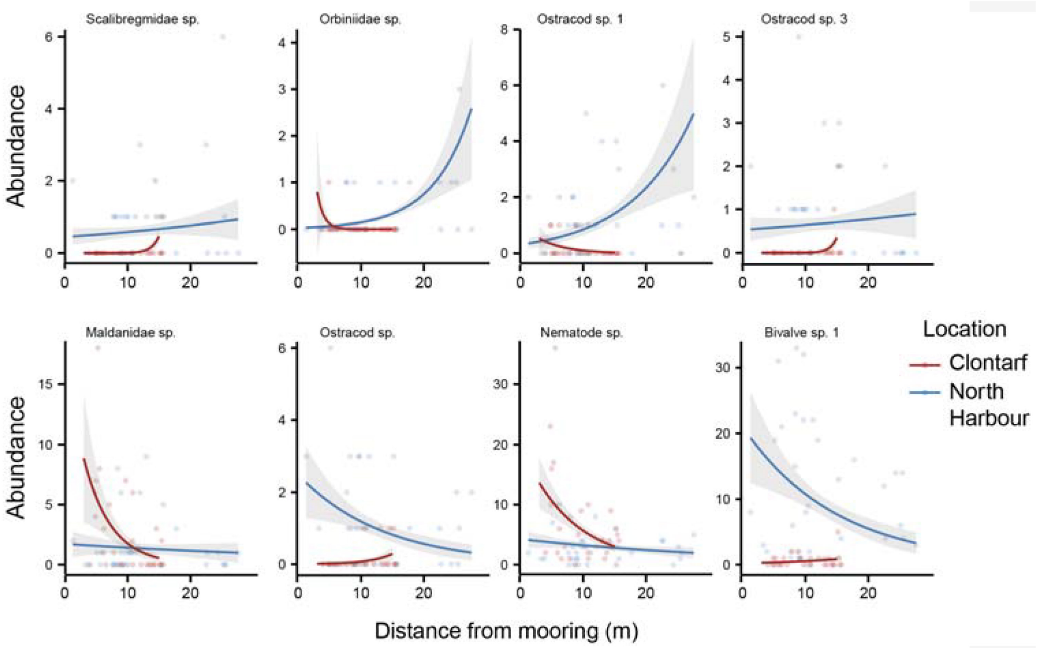

FIGURE 2 | The relationship between infaunal abundance (count per sample) and distance from moorings (m) at the two locations with boating infrastructure (Clontarf, North Harbour). The taxa plotted are the eight morphospecies with the lowest *p*-values for the distance from moorings predictor variable in the generalized linear models (GLM). Fitted lines are derived from the negative binomial GLMs and shading represents 95% CI.

are strongly influenced by sediment characteristics such as grain size (Simpson et al., 2005) and in particular, several polychaete families (Syllidae, Opheliidae, and Pectinariidae), and polychaetes as a group, increased as grain size increased, while mollusks, crustaceans and other taxa decreased overall. These changes were detected at a fine-scale, crucially allowing the effects of moorings on infauna to be distinguished from natural spatial variation in grain size.

When pooling species, the total abundance of infauna per sample did not significantly vary with distance to moorings, distance from shore, location or grain size (10th percentile, µm) (**Table 1**). Diversity per sample (Shannons Diversity H') varied among locations, but there was no effect of distance to moorings, distance from shore or grain size (**Table 1**). When the species were aggregated to higher taxonomic levels, the abundance of mollusks declined with increasing distance from moorings and increasing grain size (**Supplementary Table S2**). The abundance of crustaceans varied with location, but not with the distance from moorings or with grain size (**Supplementary Table S2**) and the abundance of polychaetes was unrelated to the distance from moorings but increased with increasing grain size (**Supplementary Table S2**).

Large Scale Effects of Boat Moorings

In this study, the large-scale analysis incorporating (1) 13 replicates at two boating infrastructures and two reference locations and (2) randomized sample sites within the locations (i.e., no consideration given to environmental space) found no effect of boat moorings on infaunal communities. Specifically, the composition of infauna varied significantly among locations (**Figure 3**, multivariate GLM, deviance 907.1, P = 0.001), but there was no evidence that the two locations with boating infrastructure (Clontarf and North Harbor) differed consistently from those lacking boating

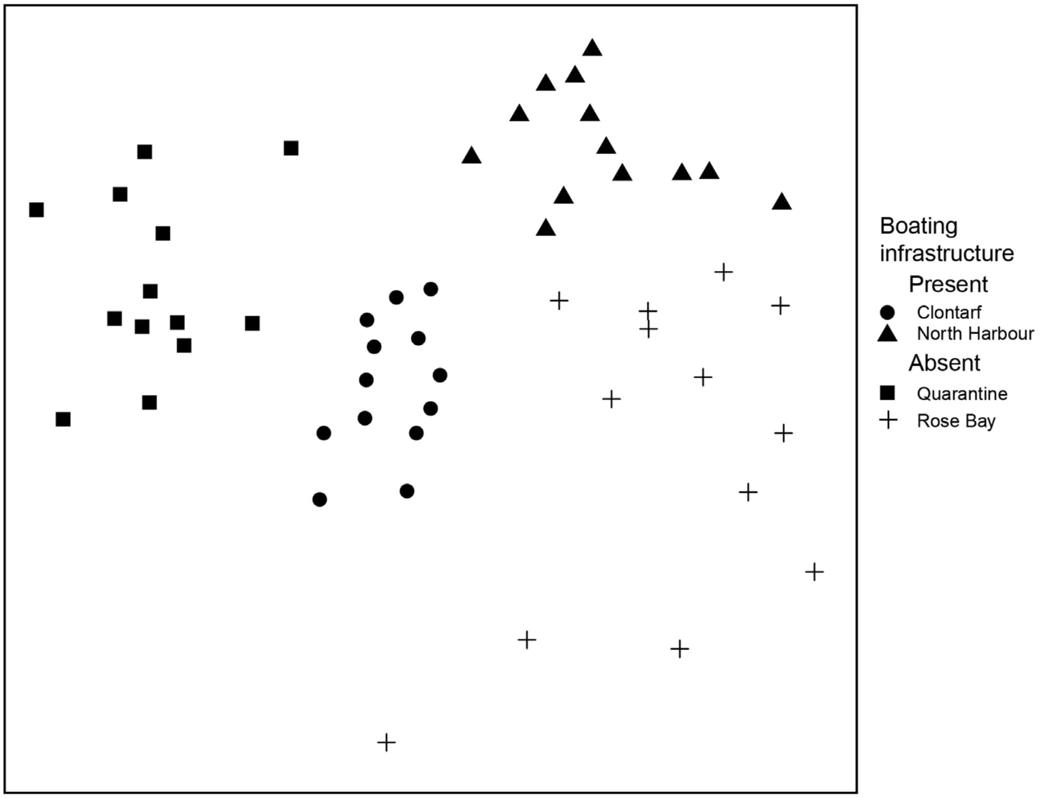

FIGURE 3 | MDS ordination visualizing the differences in composition of the infaunal community for locations with boating infrastructure (Clontarf, North Harbour) and lacking boating infrastructure (Quarantine Bay, Rose Bay). The ordination used the Bray-Curtis index of similarity and square root transformed data (stress = 0.19).

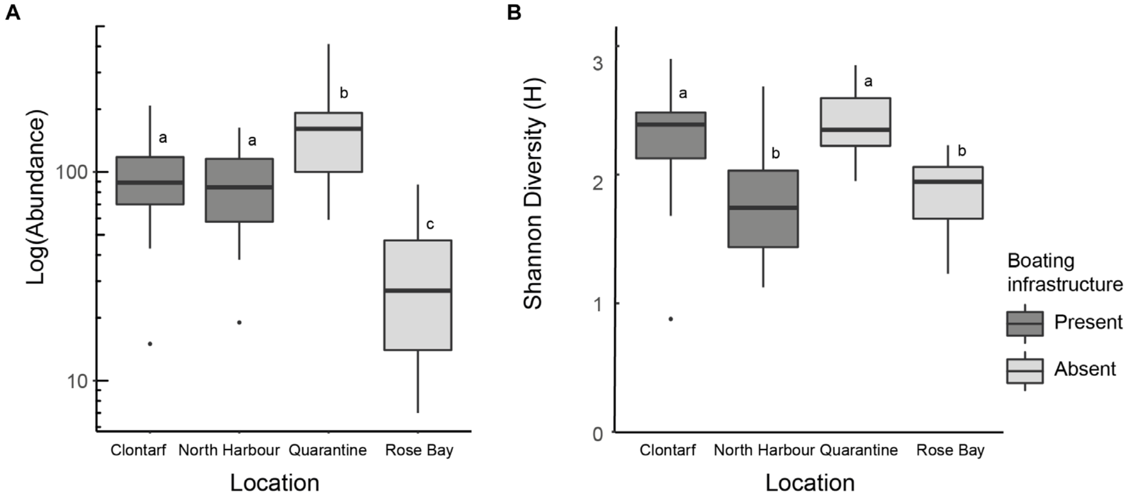

FIGURE 4 | The **(A)** abundance and **(B)** Shannons Diversity (H′) for each of the locations with boating infrastructure (Clontarf, North Harbour) and lacking boating infrastructure (Quarantine Bay, East Rose Bay). Locations sharing a letter do not differ in *post hoc* tests following **(A)** a negative binomial GLM and **(B)** a linear model contrasting all locations. Boxes indicate the upper and lower quartiles around the median, whiskers represent scores outside the middle 50%.

infrastructure (Quarantine Bay and Rose Bay). In pairwise tests, the composition of infauna at each of the locations differed significantly from all other locations (multivariate GLMs, $P = 0.006$).

The total abundance of infauna per sample varied significantly among locations (GLM, deviance = 37.32, $P = 0.001$), with no evidence for reduced abundance at the locations with boating infrastructure (**Figure 4A**). The two locations with boating infrastructure supported similar total abundances, with abundances both significantly lower (Rose Bay) and higher (Quarantine Bay) at the reference locations (**Figure 4**). Diversity (Shannons Diversity H') also varied among locations ($F_{3,48}$, $P < 0.001$), but not in the presence or absence of boating infrastructure (**Figure 4B**). Diversity was highest and similar at two sites that either had (Clontarf) or lacked (Quarantine Bay) boating infrastructure. North Harbor and Rose Bay, one of which had moorings, supported similar diversity, both significantly lower than the other two locations (**Figure 4**).

The total abundance of each of the major taxa varied among locations, but the two locations with boating infrastructure did not differ consistently from the two locations lacking boating infrastructure (**Supplementary Figure S2**; GLMs, crustaceans, deviance 55.18, $P < 0.001$; mollusks, deviance 55.18, $P < 0.001$; polychaetes, deviance 55.32, $P < 0.001$). For crustaceans, one of the reference locations (Quarantine Bay) supported the highest abundance, while the other reference location (Rose Bay) did not differ significantly from one of the locations with moorings (North Harbor; **Supplementary Figure S2A**). For mollusks, abundance was also highest at Quarantine Bay, with the other reference location supporting lower abundance than both locations with moorings (**Supplementary Figure S2B**). For polychaetes, one of the reference locations (Rose Bay) supported lower abundance than all other locations (**Supplementary Figure S2C**).

Implications for the Design of Impact Assessments

Fine scale sampling designs have been successfully incorporated into EcIA frameworks in the past. Walker et al. (2008), for example, found effects of an artificial groin on the distribution of macrofauna communities. Here they contrasted macrofauna across 11 transects with proximities of 1, 3, 5, 10, 15, 25, 50, 100, 150, 200, and 250 m from each side of the groin. Similarly, Rivero et al. (2013) and Sim et al. (2015) investigated changes in epifaunal and infaunal communities, respectively, with distance from boating structures finding patterns in larval recruitment, reduced flow, and increased contamination closest to boating structures. Yet, despite the obvious benefits of understanding the scale of impact, rather than just testing if an impact occurs, studies such as these are relatively rare.

If the effects of boating infrastructure were only assessed at larger spatial scales (locations in this study) then changes in sediment communities could be overlooked. Clearly this could be of concern for policy makers and government bodies that use the advice of analysts to guide the development of infrastructure (Morgan, 2012; Clarke and Menadue, 2016). The failure to detect an impact appears, in large part, due to the fine-scale nature of the effects of moorings, which are not well assessed by the large-scale sampling approach. For the majority of marine developments, the potential scale of ecological impacts will be difficult to predict *a priori* with any great accuracy (Dafforn et al., 2015; Bishop et al., 2017; Heery et al., 2017). In such cases, it becomes prudent to add value to impact assessment frameworks by incorporating sampling programs that use multiple scales within potentially impacted locations. Advances in the accuracy and versatility of satellite positioning systems have greatly increased the feasibility of such an approach (Dafforn et al., 2016).

Our large-scale impact assessment approach had fewer replicates per location but more samples overall for processing (13 replicate samples × $n = 4$ locations = 52) relative the within location fine-scale approaches ($n = 28$ and $n = 36$ samples, respectively, per impact location). Interestingly, 13 replicates remain high for an EcIA in a marine ecosystem [e.g., see systematic review of contamination impacts by Johnston and Roberts (2009)]. Very few analyses have employed >10 replicate samples per impact and reference locations (e.g., Anderlini and Wear, 1992; Danovaro et al., 1995; Ward and Hutchings, 1996; Stark, 1998; Lasiak et al., 2006; Skilleter et al., 2006; Fukunaga et al., 2011). Lewis et al. (2002), for example, found effects of pipeline construction on benthic invertebrate distributions in Clonakilty Bay, Ireland. The study design consisted of three sediment cores taken at an impact location and three reference locations per sampling interval. Further to this, samples were randomly allocated within sites and scale was not considered. Sampling at finer spatial scales has the potential to reduce sampling efforts without reducing the sensitivity in detecting ecological changes.

Taxonomic Resolution

Where there are limitations to the taxonomic resolution that can be attained, results show that morphospecies can be used as surrogates for taxonomic species in ecological studies (Brind'Amour et al., 2014). Further, many impact assessment studies have recommended community analysis at the level of class or family to provide a taxonomic sufficiency to detect differences among locations (Warwick, 1988; Olsgard et al., 1997; Roach et al., 2001). With limited research undertaken on the impact of boat moorings on the distribution of infaunal communities, it is unknown as to whether results from this study would differ, had a species-level of taxonomic identification been possible. Certainly, at the scale of taxonomic resolution commonly applied in impact assessments (e.g., Sydney Water, 2014), we only detected ecological changes at finer spatial scales of sampling for some taxa. Future studies in this system could investigate a species-level approach to taxonomic identification or other taxonomic and functional groupings (Bennett et al., 2014; Jansen et al., 2018). This will allow for the comparison of results using differing methods to taxonomic identification and may highlight any loss of ecological information incurred by using one approach over another.

CONCLUSION

As the world's population moves closer to the sea, those responsible for detecting impacts, approving development, and monitoring societal-environmental interactions require every tool at their disposal and must quickly adopt improved methodologies to assess potential change (Kennish, 2002; Halpern et al., 2007). There are thousands of moorings in Sydney harbor, the study area, – and in coastal ecosystems around the globe, with the potential to change soft sedimentary communities. While the impacts identified here may be fine scale – they will act to fragment infaunal assemblages, which might disrupt ecological processes at larger scales.

DATA AVAILABILITY

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

AM conducted the fieldwork, sample processing, and data analysis. EJ, KD, LH, and AP conceived the experimental design. KD and LH assisted with fieldwork, sample processing, and statistical analysis. All authors contributed to writing and editing the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00101/full#supplementary-material

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Benthic Conservation Features and Species Associated With Subsea Pipelines: Considerations for Decommissioning

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Oil and gas pipelines that lie exposed on the seabed can function as "artificial reefs," providing habitat for fish and benthic species, including some that are listed under conservation designations. As the offshore hydrocarbon industry matures, operators and national governments must decide whether decommissioned pipelines should be left *in situ* or removed for onshore disposal. In most jurisdictions, there is a requirement to evaluate the environmental consequences of different pipeline decommissioning options in a comparative assessment. To do this effectively requires an understanding of the associations between pipelines and fauna. Pipeline operators routinely collect video footage for inspection and maintenance purposes using remotely operated vehicles (ROV). This footage has the potential to provide insight into interactions between the marine environment and offshore pipelines. This study uses inspection footage from eight pipelines to quantify the presence and abundance of species and features listed under a number of EU and United Kingdom conservation designations; 12 such features and species were observed on the pipelines or neighboring sediments. The soft coral *Alcyonium digitatum* was present in the highest densities on pipelines located on mud, while *Sabellaria* sp. and *Echinus esculentus* were more common on pipelines in sand. Gadoids, anemones and hermit crabs were also frequently observed around pipelines. The study identifies the limitations to the use of industry ROV footage for ecological purposes, but shows that with consideration of taxon size, image resolution, ROV speed and altitude, this can be a valuable approach to gain additional insights into environment-infrastructure interactions. The results suggest that removal of pipelines will remove established colonies of epibenthic species, some of which have conservation value. The ecological significance of this loss, however, must be weighed against the broader considerations during pipeline decommissioning including cost, technical feasibility and impacts to other marine users.

Keywords: artificial reef, pipelines, marine spatial planning, decommissioning, benthic

INTRODUCTION

Subsea pipelines are an integral part of offshore oil and gas extraction and have been installed in all major hydrocarbon basins (Guo et al., 2005). Pipelines are generally constructed from steel, with polymer or concrete coatings, and can be installed directly on the seabed ("surface laid") or within a trench, which can be back filled naturally or artificially (Oil and Gas Uk, 2013a). Protective structures are frequently installed in association with pipelines, and can include rock armoring (referred to as "rock dump"), grout bags or articulated concrete blocks linked with wire or rope ("concrete mattresses") (Oil and Gas Uk, 2013a).

Artificial structures, including pipelines and protective structures, introduced into marine environments, provide hard substratum, which can be colonized by sessile and mobile organisms (Baine, 2001). These "artificial reef" communities can deliver ecosystem services including water filtration, carbon sequestration and the provision of commercially exploitable biomass (Moberg and Rönnbäck, 2003; Dafforn et al., 2015). The development of artificial reef communities on shipwrecks, coastal defense structures and oil and gas platforms is well-known (Pickering et al., 1999; Jørgensen et al., 2002; Gallaway et al., 2009; Firth et al., 2014), and these structures have been shown to support a locally high abundance and biomass of fauna (Boswell et al., 2010; Claisse et al., 2014). Despite the prevalence of offshore pipelines, there is a paucity of research on the extent to which pipelines support reef-associated species, and more generally on the interactions between pipelines and local ecosystems. One example of a large-scale pipeline-ecosystem interaction is the mass accumulation of jellyfish detritus along a West African pipeline (Lebrato and Jones, 2009). In part, the lack of ecological studies on pines is due to the time and cost requirements of conducting offshore environmental surveys at the necessary spatial and temporal scales (McLean et al., 2017).

Oil and gas operators routinely collect video footage of pipelines for inspection and maintenance purposes using remotely operated vehicles (ROV). This video footage has the potential to provide insight into interactions between the marine environment and offshore pipelines (Macreadie et al., 2018; McLean et al., 2018). In Australia, ROV footage was recently used to show that fish abundance on pipelines was double that of surrounding seabed, and that pipelines can provide hard substratum habitat (McLean et al., 2017; Bond et al., 2018a,b). Similarly, in California, a pipeline supported up to seven times the fish density of the adjacent seafloor (Love and York, 2005). Unlike oil and gas platforms, marine growth removal operations are not routinely carried out on pipelines, and are only performed on localized pipeline sections where there is a suspected integrity issues (Oil and Gas Uk, 2013b), potentially allowing mature hard substratum communities to develop on pipelines.

As hydrocarbon basins mature, decisions must be made on how to decommission redundant pipelines. Unlike oil and gas platforms, there are no international regulations governing pipeline decommissioning and individual nations are able to set their own legislative agenda (DECC, 2011; Oil and Gas Uk, 2013a). In North Sea states, pipelines are not covered by the OSPAR 98/3 decision that prohibits *in situ* decommissioning of platforms, and instead pipeline decommissioning is considered on a case by case basis, with all feasible methods evaluated through a comparative assessment process (Oil and Gas Uk, 2013a). Potential methods for pipeline decommissioning include full/partial removal or *in situ* decommissioning (Oil and Gas Uk, 2013a). The environmental implications of pipeline decommissioning will depend on the pipeline location, the pipeline properties and the decommissioning methods employed. The decommissioning process has the potential to affect marine organisms through physical impacts, noise disturbances, localized sediment disturbances, release of contaminants and the addition/removal of artificial hard substratum that provides habitat and/or protection from trawling damage (Burdon et al., 2018).

To predict the environmental impacts of pipeline decommissioning, and develop an evidence base to support the comparative assessment process, interactions between marine organisms and pipelines need to be quantified. This is particularly critical where pipelines intersect a designated conservation zone or interact with protected species. The United Kingdom Continental Shelf contains a number of internationally important "Features of Conservation Importance" (FOCI), which have been identified as threatened, rare, or declining (JNCC, 2010). Such FOCI include those listed in the EU Habitats Directive (Council Directive 92/43/Eec., 1992), e.g., cold-water corals, and those within designated marine protected areas. Many of the species within the FOCI group are epibenthic, and may colonize offshore pipelines (Benson et al., 2013). Additionally, some habitats, which are recognized within the FOCI group (e.g., *Sabellaria* reefs), might receive protection from fishing activities from the physical presence of pipelines. As with natural hard substratum habitats, it would be expected that the benthic community, and presence of FOCI associated with the pipelines will vary according to water depth and seabed substratum. Additionally, the intensity of mobile demersal fishing in the vicinity of the pipeline is likely to influence the abundance, diversity and biomass of benthic species and FOCI (Buhl-Mortensen et al., 2016).

In instances where pipeline decommissioning has the potential to interact with FOCI, operators must demonstrate compliance with relevant national and international environmental legislation. The operator must show that their decommissioning activities "will have no likely 'significant' effects on the designated species or feature" (Burdon et al., 2018). Significance in this context is poorly defined in terms of thresholds of change or relevant spatial scales, posing problems for interpretation (Wilding et al., 2017). In addition to serving as an evidence base for individual decommissioning decisions, quantitative data on interactions between marine species and oil and gas infrastructure can contribute toward efforts to define thresholds of change and determine meaningful spatial and temporal scales for assessments (Wilding et al., 2017).

The aim of this study was to repurpose industry ROV inspection footage to quantify associations between pipelines and observed marine fauna, and to discuss the extent to which pipelines may interact with benthic conservation features and species in the North Sea.

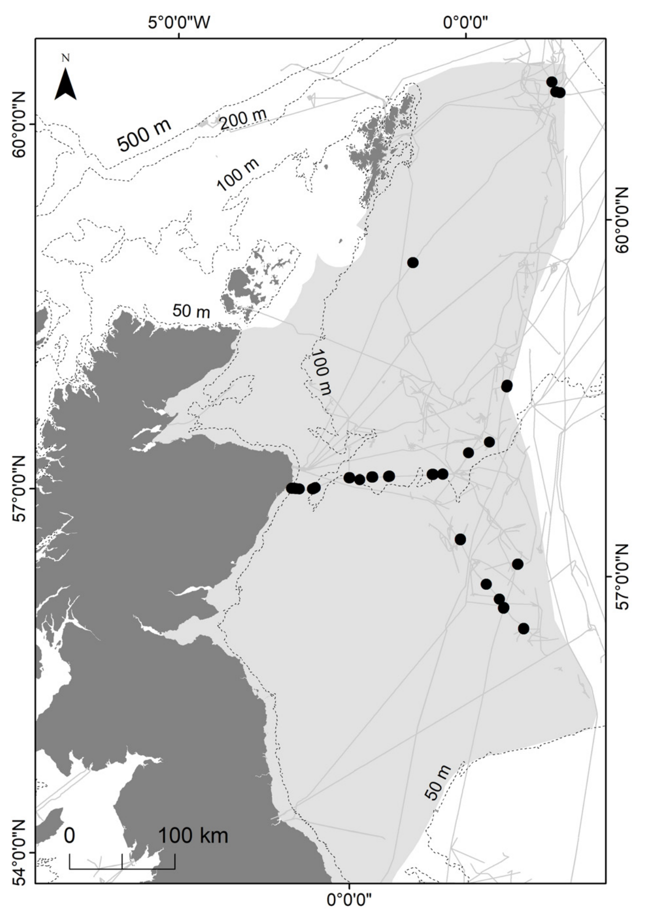

FIGURE 1 | Oil and gas pipelines (gray lines) in the northern North Sea region (shaded area). Black circles represent the location samples of ROV footages used in this study.

MATERIALS AND METHODS

Site Description

This study focuses on pipelines in the northern North Sea region (**Figure 1**). The region consists of shallow (50 m) mixed water in the south and deep (200 m) stratified water in the north (De Wilde et al., 1992). The substratum is predominately sand, with areas of coarse gravel, finer sediments and muds (Breuer et al., 2004). Within the region, there is significant human activity including commercial fishing and oil and gas production (ICES, 2016).

Features of Conservation Importance

A list of United Kingdom FOCI was obtained from the Joint Nature Conservation Council [species[1], habitats: (JNCC, 2010)]. The lists contained designations from thirteen conservations listings (**Supplementary Table 1**). From this list, benthic features were selected that could feasibly interact with subsea pipelines (i.e., hard-substratum associated species) and that could be identified from the survey footage. Species were selected if their known distributions corresponded to near- and offshore regions hosting subsea pipelines and if their body-size was sufficiently large to enable identification (typically > 5 cm diameter) or if they were present in high enough abundances to allow visualization (e.g., cup corals). The final list of target FOCI is shown in **Supplementary Table 1**. Each FOCI was categorized as a protected "species," where each individual of a species is recognized as of conservation interest, or a "feature" where the habitat formed by the species is recognized as of conservation interest (**Supplementary Table 1**). "Habitat" is defined by individuals occurring at specific densities, e.g., individuals occurring with at least 30% coverage in an area of seabed that is at least 25 m^2 (Irving, 2009; Morris, 2015).

ROV Footage and Sample Selection

The available ROV footage was recorded in standard definition throughout the day and night between 2013 and 2016. Footage was analyzed from eight pipelines greater than 16″ (∼406 mm) in diameter (**Figure 1**). All pipelines were installed prior to 2001, however data on the exact age of the pipelines were unavailable. Footage of pipelines that were fully buried, or covered by protective structures, was excluded. Pipelines that were covered by a thin sediment layer, crowning (approximately top 25% of the pipeline exposed) or fully exposed were included in the analysis. The footage comprised three concurrent views of the pipeline from starboard, port and central (top–down) cameras.

Each pipeline was divided into 1 km "sample sections." The length of each sample section was determined from the footage metadata, which specified the location of the ROV in terms of kilometers along the pipeline. The kilometer point location of the ROV, which was used to determine the start and end of each 1 km sample section, was linked to the view in the central camera. Sample sections were assigned as occurring on sand or mud, in shallow (up to 50 m), medium (50–120 m) or deep (120 m or more) water by overlaying the pipeline locations with bathymetry and EMODnet modeled substratum layers [Folk (1980) classification] in ArcGIS (Stevenson, 2012). For each of the six depth-substratum combinations, five 1 km sections were randomly selected. For some combinations, fewer than five 1 km sections were available, in which case the maximum number of sections was selected. The first, central and final 100 m lengths within each 1 km section were selected for analysis. In instances where only shorter sections of suitable pipeline were available, the footage was combined to equal 100 m where possible. If it was not possible to obtain 100 m from the randomly selected section, a new 1 km section was selected. For each sample section, an estimate of fishing intensity in the vicinity of the pipeline was obtained by spatially overlaying the center point of the sample section with standardized data layers of fishing intensity (fishing hours in 2015) for vessels operating mobile demersal gear (ICES, 2017). The location of sample sections is shown in **Figure 1**.

Analysis

All footage was viewed in VLC Media Player version 2.2.3. Central, starboard and port camera views were analyzed successively. Footage was viewed at 0.33× speed and all visible

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[1] http://jncc.defra.gov.uk/page-3408

fauna were recorded. The area of pipeline/seabed within the field of view could not be recorded and identified since the ROV footage did not contain a scale. It was not possible to use the diameter of pipeline for scale, since several pipelines were partially buried and the size of the exposed section was unknown. The number of individuals per section was counted for the target FOCI "species." The length of coverage (in meters) along the pipeline was estimated for target FOCI "features" and colonial/encrusting species. All non-target species were recorded as presence/absence. Taxon location was recorded as either pipeline or seafloor. When the same taxon was present on both the pipeline and seafloor, both instances were recorded. Taxa were identified to the lowest level of classification possible based on visual observation and known geographic and depth distribution. Species-level identifications were only assigned when image quality was sufficient that there was high confidence in the identification. Additional expert opinions were sought to confirm species-level identification for rarer species. Poor quality images were excluded from the analysis, and in instances where that was lower confidence in the species identification, a higher taxonomic designation was assigned to the observation. Observations were made on the ability to resolve taxa from ROV footage according to the speed, altitude and lighting of videos.

Data Analysis

For each taxa, the number of individuals was divided by the total length of the sample sections to provide a density estimate of individuals per unit length (linear density). The individual density estimates were combined with the coverage per unit length (linear density) estimates for the FOCI features and colonial species. Linear density, rather than areal density, were the only metrics that could be extracted from the ROV footage, given the lack of a suitable scale. Data were explored for apparent trends between density, pipeline depth and substratum type for taxa occurring on the pipelines and the adjacent seabed. Pearson's correlation coefficients with 95% confidence intervals were used to assess the magnitude of correlations between FOCI density and three pipeline location variables: water depth, latitude and fishing intensity (Pearson, 1901). A correlation coefficient could not be calculated between *Sabellaria* sp. density and fishing because all four observations were from samples with a uniform fishing effort.

RESULTS

Video footage was analyzed from eight pipelines, totaling 5.18 km and derived from 154 sample sections. The majority of sample sections (n = 150, length = 4.86 km) were on sand, with only four sections (0.325 km) available on mud (**Table 1**). The pipeline sections on mud occurred between 123–158 m depth, whereas the sand sections ranged between 43–160 m depth. Approximately 55% of the footage was recorded between 7 am and 7 pm. The fishing intensity associated with sample sections ranged from zero hours to 205 h per year.

A total of 57 taxa were identified from the video footage comprising sessile and mobile invertebrates and fish (**Supplementary Table 2**). Twenty-seven of the taxa could be identified to species level, while 17 could only be identified to order or above. The most frequently observed taxa were sea anemones, including the Deeplet anemones *Bolocera tuediae* (Johnston, 1832), Plumose anemones *Metridium dianthus* (Ellis, 1768) and Dahlia anemones, *Urticina* sp., and hermit crabs (Paguridae sp.). The majority of fish observed (and that could be identified to at least Order) were Gadoids, which were schooling around the pipelines. Ling [*Molva molva* (Linnaeus, 1758)] were observed under and around sections of the pipeline that lay unsupported above the substratum ("free spans"). Flatfish (Pleuronectiformes) were present on sediment adjacent to pipelines and three individual Rajiformes (Batoidea, skates/rays) were also observed on the adjacent sediment. There were no observations of non-indigenous species in any of the sample sections.

Features of Conservation Importance (FOCI)

Twelve FOCI were observed on the pipelines and adjacent seabed. Four FOCI were only observed on sediments, seven on sediments and pipelines and one (Crinoids) observed only on pipelines (**Table 2**). Of the FOCI observed on pipelines, the soft coral *Alcyonium digitatum* was present in the highest densities on pipelines in mud substrata, while *Sabellaria* sp. and *Echinus esculentus* were present in the greatest density on pipelines in sand substrata. Of the FOCI recorded on the sediment, burrowed communities and *Virgularia mirabilis* showed the highest densities in areas classified as mud, while *Sabellaria* sp. dominated sand areas (**Table 2**). The correlations between FOCI density and water depth varied from positive to negative (**Table 2**). The strongest correlations (>0.5) indicated an increase in density with water depth for the following four FOCI: the anemone *Actinauge richardi* and Crinoids on pipelines, and burrowed communities and Sabellidae worms on adjacent seabeds. However, the 95% confidence intervals for these correlation coefficients were large, suggesting that no correlation or a weak correlation between density and water depth were also possible. Similarly, the confidence intervals around the correlations coefficients for FOCI density against sample latitude and fishing intensity indicated that there were no relationships

TABLE 1 | Summary of the number ROV footage samples analyzed to identify features of conservation interest and the total length of pipeline reviewed, according to depth and substratum type.

Depth	Substratum	Samples	Pipeline length (m)	Number of observations
Shallow	Mud	0	–	–
Shallow	Sand	4	333	2
Mid	Mud	0	–	–
Mid	Sand	119	4320	168
Deep	Mud	4	325	29
Deep	Sand	27	203	18
Total		154	5181	217

The number of observations of FOCI taxa are shown.

between these variables. The density of the common urchin *E. esculentus* on adjacent seabeds was the only FOCI to show a moderate to strong positive correlation with latitude (r = 0.70, 95% CI: 0.36–0.88).

ROV Footage Characteristics

The ability to identify and quantify fauna associated with pipelines from video footage was influenced by the camera angles, ROV altitude and speed and illumination. Frequently, a section of pipeline could be observed in the port and starboard views up to 5 s in advance of the central view because of the angle of the lateral cameras. The angle of lateral cameras also influenced the amount of seabed visible either side of the pipeline. Typically, up to 1 m of seabed was visible either side of the pipeline, but this was substantially reduced when the lateral cameras were angled toward the pipeline.

The altitude of the ROV used to obtain the video footage affected the field of view. Altitude ranged from 0.36 to 2.75 m above the seabed (**Figure 2A**). Only two samples had an altitude of > 2.5 m. By comparing the cumulative length of pipeline analyzed with the cumulative number of observation for different ROV altitudes, it can be seen that there is a sharp increase in the number of observations between 0.5 and 0.6 m above the seabed, despite a steady increase in the length of pipeline analyzed (**Figure 2**). The majority of FOCI and other taxa records were obtained from samples sections with ROV altitudes of 1–1.5 m, with approximately half the number of observations obtained for sample sections with ROV altitudes of < 0.5 m or between 1.5 and 2.0 m. The clarity of the video footage was strongly affected by the speed of the ROV and the associated lighting. The speed of the ROV determined the extent of motion blur in the resultant footage. Speed data were only available for video footage from one of the operators and ranged from 0.03 to 1.48 km h^{-1}. Generally, the footage was evenly and adequately illuminated, however, on occasion one of the lateral camera lights was switched off to allow better visualization of gaps between the pipeline and seafloor (free spans) for structural assessment purposes.

DISCUSSION

Quantitative data on the associations between marine fauna and subsea pipelines are essential to understand the environmental consequences of installing and removing pipelines, and to support the sustainable management of anthropogenic activities in the marine environment. This study provides one of the first quantitative assessments of fish and benthic species associated with North Sea pipelines. The observation of approximately 60 taxa demonstrates that pipelines do provide habitat for a variety of marine fauna within the North Sea. The species observed were typical of North Sea hard substratum communities (Reiss et al., 2010), and are similar to the

TABLE 2 | Linear density (individuals/coverage per 100 m) of FOCI observed on pipelines and seabed, according to substratum type.

		Linear density				
FOCI	***n***	**Mud**	**Sand**	***r* (depth)**	***r* (latitude)**	***r* (Fishing effort)**
Pipeline						
Porifera spp.	17	2.15	0.71	0.17 (−0.34, 0.60)	−0.15 (−0.59, 0.36)	−0.18 (−0.61, 0.33)
Actinauge richardi (Anemone)	6	2.15	0.35	0.82 (0.04, 0.98)	−0.15 (−0.86, 0.75)	−0.30 (−0.89, 0.68)
Alcyonium digitatum (Dead Man's Fingers)	26	36.00	0.80	0.44 (0.07, 0.71)	0.11 (−0.29, 0.48)	0.34 (−0.37, 0.80)
Buccinidae (Gastropod)	6	1.54	0.96	−0.25 (−0.88, 0.71)	−0.33 (−0.90, 0.66)	−0.05 (−0.83, 0.80)
Sabellaria sp. (Polychaete)	4	NA	7.86	−0.21 (−0.97, 0.94)	−0.27 (−0.98, 0.93)	–
Sabellidae (Polychaete)	3	NA	0.10	–	–	–
Crinoid	7	0.31	0.42	0.65 (−0.20, 0.94)	0.58 (−0.30, 0.93)	−0.42 (−0.90, 0.49)
Echinus esculentus (Common Sea Urchin)	51	0.92	9.91	0.08 (−0.20, 0.35)	−0.11 (−0.38, 0.17)	−0.08 (−0.35, 0.20)
Sediment						
Porifera spp.	2	NA	0.06	–	–	–
Actinauge richardi (Anemone)	3	NA	0.07	–	–	–
Pennatula phosphorea (Phosphorescent Sea Pen)	12	9.23	1.50	−0.24 (−0.72, 0.38)	−0.03 (−0.56, 0.60)	−0.28 (−0.74, 0.35)
Virgularia mirabilis (Slender Sea Pen)	18	63.4	0.96	0.46 (−0.02, 0.76)	0.37 (−0.12, 0.71)	0.30 (−0.19, 0.67)
Alcyonium digitatum (Dead Man's Fingers)	10	NA	0.77	0.013 (−0.62, 0.64)	−0.08 (−0.68, 0.57)	0.34 (−0.37, 0.80)
Buccinidae (Gastropod)	5	NA	0.16	0.27 (−0.81, 0.93)	−0.36 (−0.94, 0.77)	−0.38 (−0.95, 0.76
Sabellaria sp. (Polychaete)	3	NA	2.33	–	–	–
Sabellidae (Polychaete)	15	NA	1.34	0.54 (−0.01, 0.84)	−0.49 (−0.09, 0.82)	−0.33 (−0.74, 0.27)
Echinus esculentus (Common Sea Urchin)	19	NA	1.12	0.49 (0.05, 0.77)	0.70 (0.36, 0.88)	−0.31 (−0.67, 0.16)
Batoidea (Rays)	1	NA	0.10	–	–	–
Burrowed communities	9	98.50	0.10	0.59 (−0.11, 0.90)	0.58 (−0.13, 0.90)	0.58 (−0.14, 0.90)

n indicates number of observations for entire survey, NA indicates no observations of the FOCI. r represents the Pearson's correlation coefficient between FOCI density and depth, latitude or fishing effort. 95% confidence intervals for correlation coefficients are shown in brackets. Correlations coefficients were only calculated for FOCI where n > 3. Only quantified (FOCI) taxa are shown.

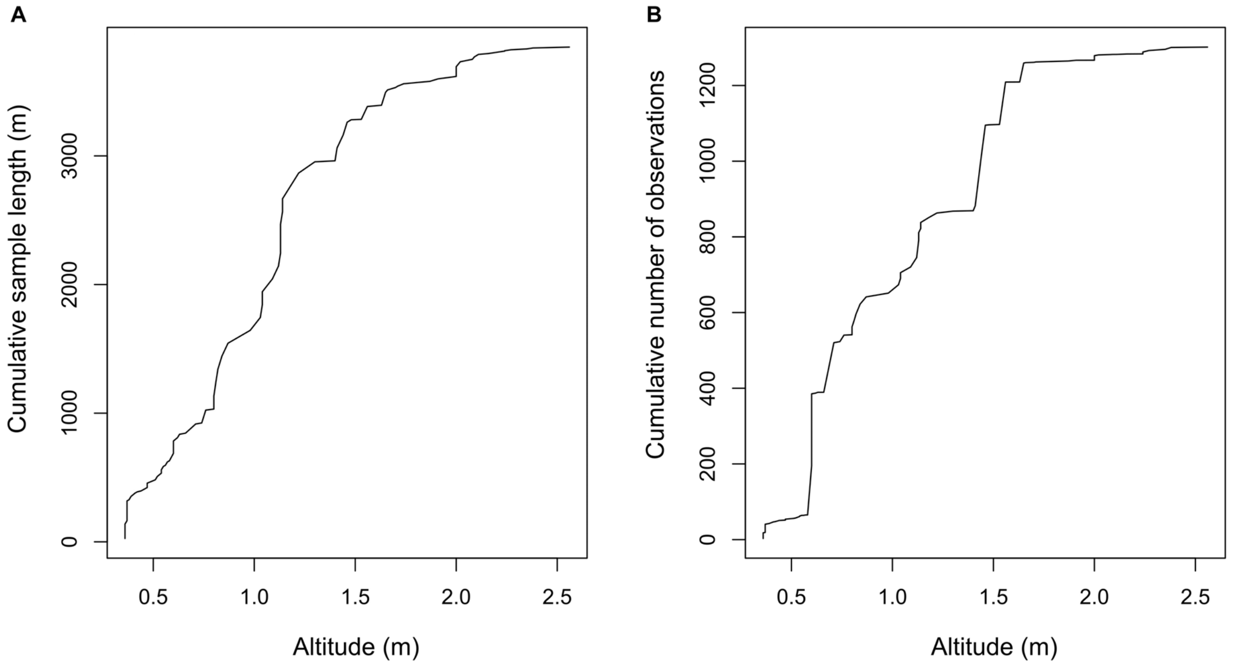

FIGURE 2 | Cumulative length of pipeline analyzed in samples **(A)** and cumulative number of observations for all taxa and features of conservation interest **(B)** according to the altitude of the ROV used to obtain video footage of pipelines.

epifaunal assemblages documented on North Sea oil and gas platforms, offshore wind turbines and shipwrecks (Zintzen et al., 2008; De Mesel et al., 2015; van der Stap et al., 2016). The video footage used in this study was restricted to pipelines located in the northern North Sea. As with other artificial and natural substrata, it would be expected that the epifaunal assemblages associated with pipelines will exhibit spatial differences due to variations in substratum type, fishing intensity, current regime and stability of subsea pipelines between the southern and northern North Sea (Angus and Moore, 1982; Gass and Roberts, 2006; Reiss et al., 2010). The positive correlation observed between the common urchin *E. esculentus* and latitude suggests that for some species, spatial differences in pipeline assemblages can operate over small spatial scales. However, for the majority of observed FOCI there were no apparent relationships between density and pipeline depth, fishing intensity or latitude. However, the lack of apparent relationships obtained in this study may be a result of the small sample size, limited number of observations and the intrinsic limits to opportunistic reuse of ROV inspection footage in this study. Access to additional pipeline footage, and a targeted sampling approach would be required to further quantify the effect of pipeline location on FOCI presence and abundance. In addition to pipeline location, pipeline properties, including size, construction material, and burial status are likely to influence the community composition and abundance of epifauna. The data obtained in the current study were insufficient to quantify relationships between individual pipeline properties and the abundance of epifauna.

A number of fish species were observed in the vicinity of the subsea pipelines. The presence of ling underneath North Sea pipeline free spans (sections of pipeline unsupported by the substratum) is consistent with observations of fish aggregations under free spanning pipelines sections in Australian and Californian waters (Love and York, 2005; McLean et al., 2017; Bond et al., 2018c). Pipeline free spans are likely to offer shelter or refuge for the fish species observed in these habitats. Ling are generally solitary fish that move between crevices (e.g., free spans) and open-water habitats within a defined "home range" (Løkkeborg et al., 2000). Ling predate on other fish and invertebrates, including crustaceans and starfish (Husebø et al., 2002). The creation of crevices (through free spans) that lie in close proximity to a food supply, in the form of epifauna and other fish on the pipelines, is likely to drive the observed aggregations of ling. The presence of other gadoid fish around pipelines supports previous suggestions that aggregations of commercial fishing activity around subsea pipelines are a result of artificial reef effects of pipelines (Rouse et al., 2017).

Six of the taxa observed on the North Sea pipelines are listed under conservation designations, with an additional six taxa that would [if present at densities specified by the EU habitats Directive (Mcleod et al., 2009)] form habitat types that are protected under conservation legislation. It is likely that the true number of FOCI associated with pipelines is greater than reported in this study. Several FOCI (e.g., the amphipod *Arrhis phyllonyx*) are too small to be identified from video footage, and thus will have been excluded. Additionally, there are likely to be regional differences in FOCI presence and those that are restricted to the southern and central North Sea will not have been observed in the present study. The level of legal protection afforded to each FOCI varies according to the legislation/scheme under which it is designated. For those FOCI listed under the EU Habitats Directive, the United Kingdom Government is obliged to maintain or restore them at a favorable conservation status. This means that the range of FOCI must be stable or increasing and that the population is maintaining itself on a long-term basis. The EU Habitats Directive specifically excludes the use of non-natural substratum as candidate sites for the creation of marine protected areas, regardless of whether the feature or species are present at the necessary density to qualify as a habitat under Annex I (Mcleod et al., 2009).

The role of pipelines in maintaining FOCI ranges and/or stable populations will depend, partly, on the connectivity between pipelines and other suitable FOCI habitats. Artificial structures in the North Sea, including oil platforms and wind turbines, are thought to provide a network of highly connected hard substrata (Hyder et al., 2017; Henry et al., 2018). Pipelines have traditionally been excluded from North Sea connectivity/larval dispersal models (Hyder et al., 2017), but our results, documenting the presence of marine fauna on pipelines, suggest that pipelines will, to some extent, contribute to the connected network of some taxa. The extended linear presence of pipelines over the seabed, connecting larger areas of artificial hard substrate (i.e., platforms), could mean that pipelines facilitate dispersal of epifauna, particularly those characterized by short-lived larval or non-larvae based reproduction (Mineur et al., 2012).

The decommissioning of pipelines poses particular challenges in the North Sea because of the wide-spread spatial distribution of pipelines, the potential hazard that pipelines pose for commercial trawlers, and the costs and technical challenges of removing pipelines (Side, 1999; Oil and Gas Uk, 2013a; Rouse et al., 2018). Operators and regulators must ensure that pipeline decommissioning practices and/or policies balance these potentially conflicting challenges, as well as accounting for environmental concerns. The results from this study suggest that removal of decommissioned pipelines will eliminate established habitat for a number of epibenthic species, some of which have conservation value. Other pipeline decommissioning methods, including trenching or rock dumping would also be expected to eliminate this established habitat, however, rock dumping would provide additional substrate which could be colonized by epibenthic species. *In situ* decommissioning of pipelines, with minimal intervention, would be expected to cause the lowest level of disturbance for established epifauna and fish. However, where pipelines are removed or trenched, it may be expected that soft-sediment habitats and species would recover (Dernie et al., 2003). The exact impacts of pipeline decommissioning will depend on the methods used and the sensitivity of FOCI and other marine species to disturbance (Burdon et al., 2018). Furthermore, the ecological significance of local disturbances to marine fauna and FOCI through pipeline decommissioning must be considered within the context of the total footprint of pipelines in the North Sea, and the extent to which pipelines contribute to connectivity between hard substrata. On the United Kingdom Continental Shelf, pipelines occupy approximately 12 km^2, equating to $<0.01\%$ of the total area (Rouse et al., 2018). The total contribution of pipelines to hard substrata habitats in the North Sea is, therefore, expected to be extremely small. However, footprints of human activities do not necessarily have a linear relationship with pressures, and small habitat fragments can contribute substantially to ecosystem functions (Jules and Shahani, 2003; Halpern et al., 2008). The inclusion of pipelines, with associated data on faunal abundance, in ecosystem models could determine whether or not pipelines make any meaningful contribution to regional ecosystem process and populations of FOCI.

Remotely operated vehicles video footage was successfully used in this study to quantify FOCI and record the presence of other marine fauna associated with pipelines. The repurposing of video footage, originally obtained for integrity assessments, for ecological analysis presented a number of challenges as have been documented previously by Gormley et al. (2018) and Macreadie et al. (2018). The lack of a suitable scale in the footage meant that the area within the field of view could not be calculated and faunal counts could not be scaled by area. The discrepancy between the central and lateral camera views led to inconsistencies in the section of pipeline assessed for each sample according to each camera view. The changing altitude of the ROV also limited the ability to identify taxa. Whilst the ROV being close to the pipeline potentially improved views of individuals and allowed for greater taxonomic resolution, it also limited the field of view either side of the pipeline and thus any individuals present there. Additionally, the altitude denoted the height above the seafloor, not above the pipeline. Therefore, footage over larger structures results in a higher recorded altitude but not necessarily a greater distance between the ROV and the structure. There are a number of small, relatively inexpensive, modifications that could be made to ROV survey design to improve the value for ROV footage for ecological analysis. With sufficient data on location, altitude and rotational parameters of the camera, it is possible to scale flat surface within images (Durden et al., 2016), however, the addition of scaling lasers to ROV would provide a more efficient methods of scaling images. Additionally, maintaining a constant ROV speed over pipelines and the collection of video/still photographs in high definition would increase the value of the footage for ecological analysis. These improvements, and access to additional pipeline ROV footage, would allow for the relationships between different pipeline properties and FOCI abundance, according to different regions of the North Sea to be quantified and provide a further evidence base to support decommissioning practices and policies.

AUTHOR CONTRIBUTIONS

SR conceived the project and drafted the manuscript. NL performed video analysis and data collection, and contributed to writing methods. PH obtained the data, provided project oversight and guidance, and commented on draft manuscript. TW conceived the project, provided technical advice and oversight, and reviewed the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00200/full#supplementary-material

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REVIEW
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Offshore Oil and Gas Platforms as Novel Ecosystems: A Global Perspective

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Offshore oil and gas platforms are found on continental shelves throughout the world's oceans. Over the course of their decades-long life-spans, these platforms become ecologically important artificial reefs, supporting a variety of marine life. When offshore platforms are no longer active they are decommissioned, which usually requires the removal of the entire platform from the marine environment, destroying the artificial reef that has been created and potentially resulting in the loss of important ecosystem services. While some countries allow for these platforms to be converted into artificial reefs under Rigs-to-Reefs programs, they face significant resistance from various stakeholders. The presence of offshore platforms and the associated marine life alters the ecosystem from that which existed prior to the installation of the platform, and there may be factors which make restoration of the ecosystem unfeasible or even detrimental to the environment. In these cases, a novel ecosystem has emerged with potentially significant ecological value. In restoration ecology, ecosystems altered in this way can be classified and managed using the novel ecosystems concept, which recognizes the value of the new ecosystem functions and services and allows for the ecosystem to be managed in its novel state, instead of being restored. Offshore platforms can be assessed under the novel ecosystems concept using existing decommissioning decision analysis models as a base. With thousands of platforms to be decommissioned around the world in coming decades, the novel ecosystems concept provides a mechanism for recognizing the ecological role played by offshore platforms.

Keywords: novel ecosystems, Rigs-to-Reefs, decommissioning, artificial reefs, environmental impacts, offshore oil and gas

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INTRODUCTION

Since 1947, when Ship Shoal Block 32 in the Gulf of Mexico became the world's first offshore oil drilling platform (Aagard and Besse, 1973), the offshore energy industry expanded rapidly to currently number over 12,000 offshore installations globally (Ars and Rios, 2017). Offshore platforms are situated on the continental shelves of 53 countries, making offshore oil and gas production a major global industry (Parente et al., 2006). Significant advances in engineering over the last 70 years have not only increased the number of rigs, but also the environmental conditions which they can withstand: offshore platforms are now larger and found in deeper waters, further from shore. These technological advances have implications for decommissioning, which occurs when hydrocarbon production ceases or the lease ends and the platform is shut down. The decommissioning process now takes longer, requires more specialized equipment and, by extension, has become more costly (Kaiser and Liu, 2014).

A 2016 study by the IHS Markit forecast the global decommissioning of over 600 offshore structures between 2017 and 2021, with a further 2,000 projects by 2040, resulting in a total cost between 2010 and 2040 of US $210 billion (IHS Markit, 2016). In countries where total removal is the legal requirement, decommissioning involves the plugging of wells, cleaning, capping and possibly removal of pipelines, removal of production equipment and removal of the structure (Hakam and Thornton, 2000). In United Kingdom waters alone, decommissioning expenditure is forecast to amount to £17 billion between 2017 and 2025 (Oil and Gas UK, 2017). Even a nation with comparatively low oil and gas production, such as Australia (0.9% of global production), has a future decommissioning liability of US $21 billion over the next 50 years (NERA, 2016). The process of decommissioning is far from straightforward in many cases, and is often complicated by the process of transferability, whereby an existing platform is sold to a company which can continue production at lower profit margins (Parente et al., 2006).

From a biological viewpoint, increasing evidence suggests that offshore oil and gas platforms provide significant ecosystem services while active. The installation of these platforms creates hard substrate in open waters which is colonized by a variety of sessile organisms and results in the formation of artificial reefs (Shinn, 1974; Scarborough-Bull, 1989). Because they may exclude commercial fishing, particularly trawling, and in some cases recreational fishing, these platforms can also act as important refuges for a variety of taxa (Frumkes, 2002; Claisse et al., 2014). The potential ecological value of offshore platforms raises the question of whether there may be alternatives to the standard decommissioning process that might have important positive ecological outcomes, and ecological factors are more recently being included in decommissioning assessments (Fowler et al., 2014; Henrion et al., 2015; Sommer et al., 2019). The successes of various Rigs-to-Reefs projects, particularly in the Gulf of Mexico, have demonstrated that these structures can be effectively repurposed as artificial reefs (Frumkes, 2002; Kaiser and Pulsipher, 2005; Sammarco et al., 2014). However, to date only a few countries around the world have successfully implemented Rigs-to-Reefs programs (summarized in Bull and Love, 2019).

Evaluating offshore platforms as novel ecosystems would provide a mechanism for considering the ecological importance of these platforms in the decommissioning process. Novel ecosystems is a relatively recent ecological concept, brought into focus by Hobbs et al. (2006), where human activity has altered ecosystems to a point where restoration may not be feasible. In a world that is increasingly being altered by human activity, the concept of novel ecosystems recognizes that in some cases, ecosystems changed from their historical state by human intervention may not feasibly be able to be restored (Hobbs et al., 2006). With many case studies throughout a variety of ecosystems around the world (Hobbs et al., 2013b), novel ecosystems provide an approach for recognizing value in altered ecosystems, rather than implementing restoration for restoration's sake. In the cases of both active and decommissioned platforms, it is possible that the concept of novel ecosystems can be applied as a way to describe the ecosystems created by the presence of the platforms. The aim of this review is to evaluate the ecological role of offshore oil and gas platforms, and to assess these platforms against the criteria of the novel ecosystems concept.

DECOMMISSIONING

Decommissioning, the end of life stage for offshore infrastructure, is a process which is regulated internationally, regionally and nationally. The 1996 Protocol to the London Dumping Convention (London Protocol) aimed to protect the marine environment from all sources of pollution, and regulates against the dumping of "... platforms or other man-made structures at sea; and any abandonment or toppling at site of platforms or other man-made structures at sea, for the sole purpose of deliberate disposal." (Elizabeth, 1996). However, the London Protocol does not expressly prohibit decommissioning of structures *in situ* (Techera and Chandler, 2015), stating that dumping does not include "placement of matter for a purpose other than disposal thereof, provided that such placement is not contrary to the aims of this Protocol (Elizabeth, 1996)." There are four alternatives to complete removal: (1) leave wholly in place with appropriate navigational aids; (2) partial removal, usually of the superstructure); (3) tow-and-place by moving the structure to a new location; and (4) toppling by laying the structure on its side (Schroeder and Love, 2004; Macreadie et al., 2011; Fowler et al., 2014).

Decommissioning regulations and options in various countries and regions have been reported on and assessed extensively in the literature. While decommissioning in the North Sea and the United States (US) has been well studied (e.g., Reggio, 1987; Löfstedt and Renn, 1997; Dauterive, 2000; Cripps and Aabel, 2002; Schroeder and Love, 2004; Kaiser and Pulsipher, 2005; Jørgensen, 2012; Claisse et al., 2015), there has been more recent focus on decommissioning policy in relatively "new" oil and gas producing regions, such as south-east Asia (Zawawi et al., 2012; Al-Ghuribi et al., 2016; Fam et al., 2018; Laister and Jagerroos, 2018), Australia (Fowler et al., 2015; Techera and Chandler, 2015; Chandler et al., 2017), and Brazil (Barros et al., 2017; Mimmi et al., 2017). Two recent reviews (Bull and Love, 2019; Sommer et al., 2019) provide comprehensive assessments of the literature on the decommissioning process, options, and regulations around the world. These two reviews complement each other by focusing on somewhat different aspects of decommissioning. Sommer et al. (2019) focuses on the ecosystem functions and services provided by platforms, and suggests a more ecosystems-based approach to decommissioning. Bull and Love (2019) provides the most in-depth review to date of the literature on offshore oil and gas platforms, including platform installation, decommissioning, relevant legislation, and platform ecology. While this review is mainly focused on the United States, it does briefly review Rigs-to-Reefs programs in other regions around the world.

RIGS-TO-REEFS

Rigs-to-Reefs is a potential decommissioning outcome for offshore oil and gas structures whereby obsolete infrastructure is re-purposed as artificial reefs instead of being brought back to shore for disposal (Kaiser and Pulsipher, 2005). The first examples of Rigs-to-Reefs occurred in the 1980s, when platforms were removed from production in Louisiana and transported to Florida where they were repurposed as artificial reefs (Kaiser, 2006; Jørgensen, 2009).

By April 2018, approximately 532 offshore platforms have been re-purposed as artificial reefs in the Gulf of Mexico, mostly in Louisiana and Texas (Ajemian et al., 2015; Bureau of Safety and Environmental Enforcement, 2018). This represents just over 11% of the total number of platforms decommissioned in the Gulf of Mexico (Bull and Love, 2019).

Offshore oil and gas platforms are spatially complex structures and their value as artificial reefs has been discussed in numerous studies (Shinn, 1974; Dugas et al., 1979; Bohnsack and Sutherland, 1985; Guerin et al., 2007). Offshore platforms have not only been shown to have a higher fish biomass than sandy bottom areas but even natural reefs (Claisse et al., 2014). This results in offshore platforms having an "enhanced fishing zone" of 200–300 m for pelagic species and 1–100 m for demersal species (Bohnsack and Sutherland, 1985). Fishing and diving around offshore rigs, in countries where it is allowed, is a major component of the local tourism industries (Stanley and Wilson, 1989). In Louisiana, recreational fishing is centered around offshore platforms – over 70% of recreational fishing trips into the EEZ are in direct association with offshore platforms, where pelagic fish densities are 20–50 times higher than surrounding areas (Dugas et al., 1979; Reggio, 1987; Dauterive, 2000). As such, sport fishers and recreational divers generally support Rigs-to-Reefs programs (Frumkes, 2002).

Both active and decommissioned offshore platforms can have a negative impact on commercial trawl fishing, and the prevention of trawling is a common criticism of Rigs-to-Reefs programs (Macdonald, 1994; Hamzah, 2003). The issue of allowing fishing around platforms is one that is still uncertain and needs to be handled carefully. In some cases where platforms have become key habitat for threatened or economically important species, it may be prudent to continue to exclude all fishing from these areas if they are converted into artificial reefs, as they can then be used to bolster populations at surrounding natural reefs where fishing occurs in the same way that marine protected areas (MPAs) do (Mcclanahan and Mangi, 2000).

In sandy, flat-bottom areas with generally limited physical structure, such as the north-west shelf of Australia, the Adriatic Sea and parts of the North Sea, offshore platforms present some of the only obstacles to trawl nets (Rijnsdorp et al., 1998; Wassenberg et al., 2002; Fabi et al., 2004). While the prevention of trawling is detrimental to commercial fisheries, it is ecologically beneficial in offering protection to benthic habitats; in a study to determine the effect of trawling on sponge communities of the north-west shelf of Australia, sponges were caught in 85% of trawls, with a mean catch of 87.2 kg per half-hour (Wassenberg et al., 2002).

Evidence on the success of Rigs-to-Reefs programs and the suitability of oil platforms as artificial reef habitat suggests that these structures can provide significantly more ecological value than other cases of "dumping" (Ajemian et al., 2015). However, it is important to note that just because Rigs-to-Reefs has been successful in a certain area (e.g., the Gulf of Mexico), it does not mean it would automatically be an ecologically beneficial exercise in the North Sea, California or Australia. Every ecosystem is different and needs to be evaluated as such; creating a reef, simply because there is a platform that needs to be decommissioned, is indeed little more than waste disposal (Macdonald, 1994; Salcido, 2005).

A major obstacle in the path of Rigs-to-Reefs legislation is the relative lack of ecological research on offshore structures. For example, despite the presence of over 40 offshore oil and gas installations on the continental shelf of north-west Australia, there has been a limited number of published studies on the ecology of the structures in this region (e.g., Fowler and Booth, 2012; Pradella et al., 2014; McLean et al., 2017, 2018; Bond et al., 2018). Macreadie et al. (2012) concluded that environmental research must be part of the development of Rigs-to-Reefs policy, pointing to the case of California, where a Rigs-to-Reefs bill was vetoed in 2001 based on a lack of evidence that reefed platforms produce net environmental benefits. Macreadie et al. (2012) argue that the subsequent successful passing of a Rigs-to-Reefs bill in 2010 was due in large part to the years of subsequent research by Dr. Milton Love and colleagues (Schroeder and Love, 2002, 2004; Love et al., 2006).

ECOLOGY OF OFFSHORE PLATFORMS

Offshore oil and gas platforms can play important ecological roles for various taxa (Friedlander et al., 2014). They provide substrate for sessile organisms such as sponges and corals and act as a refuge for fish and megafauna such as seals and whales (Forteath et al., 1982; Todd et al., 2016). When a platform is installed, the establishment of a faunal community occurs quickly, with fish appearing within hours (Bohnsack, 1989), and ecological succession results in a complex reef-type habitat within 5–6 years (Driessen, 1986). Offshore platforms can be an important source of habitat not only for fish, but also for sessile invertebrates where hard substrate is limited. Where offshore platforms are isolated from natural reefs, the free-swimming larval stages of invertebrates that settle on offshore platforms would otherwise not likely survive due to a lack of "hospitable" substrate (Driessen, 1986; Thomson et al., 2003; Macreadie et al., 2011). However, the addition of hard substrate means that offshore platforms can also provide habitat for invasive species (Page et al., 2006; Pajuelo et al., 2016).

There is considerable debate as to whether fish associated with artificial structures are actually being produced there for a net gain, or are simply being attracted from nearby natural reefs. Attraction is thought to be detrimental to fish populations, especially those which are targeted by fisheries, as previously sparsely distributed populations become concentrated, making them vulnerable to exploitation (Bohnsack, 1989). However, in the case of offshore platforms, attraction could be beneficial to pelagic species in some regions, where the platforms can act as a temporary refuge from fishing pressure. Macreadie et al. (2011) discuss the importance of habitat limitation as a factor in the attraction vs. production debate; specifically that a habitat-limited fish population would see an increase in regional biomass due to the addition of suitable habitat via artificial structures. Fowler and Booth (2012) found that offshore platforms in north-west Australia could sustain complete size- and age-structured populations of the Serranidae *Pseudanthias rubrizonatus*, with a presumed age range in sampled individuals of 22 days to 5 years. However, production of fish varied among individual platforms. The relative scales of "attraction vs. production" therefore may vary between offshore oil and gas platforms, as biotic and abiotic conditions vary from platform to platform. The presence of larval fish may not be enough to assume production, based on the proximity of other reefs (Bohnsack, 1989; Macreadie et al., 2011). In addition, production is more important in the case of demersal species, which are more dependent on benthic habitat than highly mobile pelagic species (Bohnsack, 1989).

The ecosystem created by offshore platforms means, like natural reefs, they provide economic benefits. In regions where recreational fishing is permitted, these platforms have been highly popular locations for decades (Dugas et al., 1979). "Fishing the rigs" is a major portion of the recreational fishing activity in the Gulf of Mexico, particularly Louisiana, where species caught at the platforms include sharks, billfish, and barracuda (Driessen, 1986). While recreational fishing occurs around offshore platforms, a number of commercial gear types such as trawl and longline are generally excluded from the waters around these structures due to the risk of damage to both fishing gear and subsea infrastructure such as pipelines (de Groot, 1982; Demestre et al., 2008).

In some regions, the exclusion of all vessels, including recreational and commercial fishers, can be legally mandated, and these "exclusion zones" vary in size between countries. In the North Sea, the exclusion from fishing around offshore oil platforms that have been in place for decades, has resulted in a network of *de facto* MPAs (de Groot, 1982; Fujii and Jamieson, 2016). In Australia, the "petroleum safety zones" surrounding offshore platforms extend up to 500 m from the outer edge of any well or structure (Commonwealth of Australia, 2010), while the exclusion zone around a drilling platform in the Jubilee Field in Ghana is five nautical miles (Chalfin, 2018). In 2003, Mexico created an "area of exclusion" of 5,794 km^2 around oil platforms in the Campeche region of the Gulf of Mexico (Quist and Nygren, 2015).

Various studies have described oil platforms around the world as *de facto* MPAs. Because of the exclusion of trawl fishing at all platforms in Gabon, and the exclusion of all types of recreational fishing at some platforms due to security restrictions, Friedlander et al. (2014) concluded that these platforms are functioning as *de facto* MPAs. In California, offshore oil platforms provide a significant refuge for commercially important rockfish species (Frumkes, 2002; Claisse et al., 2014; Fowler et al., 2015). Marine vessels are discouraged from entering the 150 m buffer zone surrounding platforms, meaning that fishing activity is limited, and Schroeder and Love (2002) found that rockfish surrounding an oil platform were larger and greater in density compared with the populations at recreationally and commercially fished sites. In addition, eight offshore oil and gas platforms off southern California supported 430,000 juveniles of the highly overfished and IUCN Critically Endangered Bocaccio rockfish *Sebastes paucispinis*, accounting for 20% of the average annual number of surviving juveniles of this species. In these instances, the refuges provide much higher recruitment and survival rates than natural but fished nursery grounds (Love et al., 2006).

NOVEL ECOSYSTEMS

Human activities are transforming ecosystems on a global scale (Foley et al., 2005; Mccauley et al., 2015; Laurance and Watson, 2016). Many studies and conservation efforts focus on restoring altered ecosystems to their historical states (Sanchez-Cuervo et al., 2012; Graham and Mcclanahan, 2013), but over the last two decades, the term "novel ecosystems" has emerged as a way of defining ecosystems altered by human activity, where restoration is at best unlikely (Hobbs et al., 2013a). There has been criticism that the concept may exclude restoration and may provide companies a license to trash ecosystems (Aronson et al., 2014; Murcia et al., 2014). However, the novel ecosystem concept is not intended to replace ecological restoration, but is meant to provide a management option for ecosystems where restoration is not feasible or may actually result in the loss of ecosystem value (Hobbs et al., 2014). In some cases, the novel ecosystem may provide ecosystem services that are more beneficial than those provided by the historical state. Backstrom et al. (2018) have suggested that the novel ecosystems concept is most useful in a decision or management context and in terms of meeting social, ecological and economic objectives.

The term novel ecosystems was first used in 1997 (Chapin and Starfield, 1997) but was introduced into terrestrial conservation and restoration ecology fields in 2006 (Hobbs et al., 2006). The concept has more recently been adopted by some marine ecologists, where studies on marine novel ecosystems have generally focused on coral reefs which have been altered by direct human activity, disease, climate change or introduced species (Graham et al., 2013, 2015; Yakob and Mumby, 2013; Hehre and Meeuwig, 2015). However, the concept has not yet gained significant traction amongst marine ecologists. Schläppy and Hobbs (2019) provide a comprehensive decision-making framework for applying the novel ecosystems concept to altered marine ecosystems. This framework creates a mechanism for the novel ecosystems concept to be more widely applied to marine ecosystems in future. While Schläppy and Hobbs only briefly discuss offshore platforms, Sommer et al. (2019) suggest that the ecosystem-level shifts occurring around offshore platforms are "consistent with the science on. . . novel ecosystems." However, while drawing parallels between offshore platforms and novel ecosystems, the authors do not explore the concept further, nor do they discuss the application of the concept to some or all offshore platforms.

The degree to which offshore platforms can usefully be considered a novel ecosystem may assist in assessing decommissioning options. Offshore platforms can be broadly assessed in a novel ecosystems context by evaluating these platforms against the criteria outlined in the most recent novel ecosystems definition from Hobbs et al. (2013b):

> Criterion 1: *The abiotic, biotic and social components of the system "differ from those that prevailed historically."* In the case of offshore oil and gas platforms, the abiotic and biotic states of the target ecosystem have clearly been altered due to anthropogenic forcing, specifically due to the installation of a large artificial structure and the associated disturbance of the ecosystem. Examples of this include the growth of cold-water corals on platforms in the North Sea (Gass and Roberts, 2006) and the aggregation of whale sharks around platforms in Qatar (Robinson et al., 2013) both of which are novel qualities not previously present in the historical state of the ecosystem.
>
> Criterion 2: *The ecosystems have a "tendency to self-organize and manifest novel qualities without intensive human management."* In the case of offshore oil and gas platforms, the marine life associated with offshore platforms is not managed in any way, apart from limited maintenance cleaning to remove sessile invertebrates. These ecosystems persist over the lifespan of the platform, with reports of thousands of tons of invertebrate growth on the subsea structures of platforms (Foster and Willan, 1979; Culwell, 1997). Novel qualities manifested by platforms include higher productivity of algae and invertebrates (Chou et al., 1992) and higher fish biomass (Love et al., 2006).
>
> Criterion 3: *Novel ecosystems are prevented from returning to their historical states by practical limitations, in the form of ecological, environmental and social considerations.* In the context of offshore platforms, these considerations can include many of the factors evaluated by stakeholders during the decommissioning process (**Table 1**). However, some considerations may be context specific rather than absolute, and vary among regions. For example, in California where there are relatively few platforms, their role in providing habitat for economically important species such as rockfish makes individual platforms ecologically important, particularly as some platforms produce more of these species than others (Schroeder and Love, 2002). Conversely, in an area such as the Gulf of Mexico with thousands of platforms, the ecological value of an individual platform within a regional context is not necessarily as high and therefore may not be an important ecological consideration (Schroeder and Love, 2004).

Environmental limitations could prevent the removal of offshore platforms, which means that the ecosystem cannot be returned to its historical state. Complete removal decommissioning is a potentially hazardous process both to the environment and personnel, and particularly in regions with harsh weather conditions, decommissioning could be more of a risk than leaving structures in place (Löfstedt and Renn, 1997; OGP Decommissioning Committee, 2012; Ars and Rios, 2017). Additionally, offshore platforms are known as vectors for invasive species, as they are transported long distances at low speed (Page et al., 2006; Pajuelo et al., 2016). The potential transport and spread of the many sponge, algae, coral, and even fish species associated with platforms, could be a factor preventing platform removal, and therefore restoration to historical state.

Perhaps the most significant consideration in the case of offshore platforms is the social aspect. Social factors could prohibit removal of platforms, due to prohibitive costs or platform design making removal unfeasible (Faber et al., 2001; OGP Decommissioning Committee, 2012). The social benefits derived from a platform, in the form of an artificial reef utilized by recreational divers and fishers, could be lost if the platform is removed. Conversely, social opposition to the presence of offshore platforms, as is the case in California (Pietri et al., 2011), or legislation prescribing complete removal, as is the case in Australia (Techera and Chandler, 2015) could lead to the complete removal of platforms, thereby possibly returning the ecosystem to its historical state.

It is important to avoid a blanket classification of all offshore platforms as novel ecosystems. Offshore platforms always result in the creation of habitat, but this does not by default mean that they result in novel ecosystems. For example, a platform placed near a natural reef may not significantly alter the abiotic or biotic components of the ecosystem, and may rather act simply as an "extension" of the existing reef. However, a platform placed in an area with little natural hard substrate significantly alters the abiotic nature of the ecosystem by increasing the hard substrate available, leading to changes in the community of species within the ecosystem, thereby transforming the ecosystem from its historical state.

The novel ecosystems concept can be applied to offshore platforms, so long as it is applied on a case-by-case basis. This is particularly important if the concept is used as part of the decommissioning process, as there may be incentive for energy companies to suggest platforms are novel ecosystems to avoid the costs associated with complete removal. The concept should therefore be applied conservatively and with robust evidence from ecological studies. Various studies have proposed decision analysis frameworks which assess different decommissioning alternatives based on multiple attributes (e.g., Fowler et al., 2014; Bernstein, 2015; Henrion et al., 2015). Some of these attributes can be placed within the novel ecosystems criteria as demonstrated in **Table 1**. Therefore, an assessment can be made of whether an offshore platform is a novel ecosystem simply by using existing decommissioning analysis tools. From an ecological perspective, decommissioning of offshore platforms is an ecological restoration issue. Novel ecosystems provides a tool for recognizing and retaining ecological value created through human activity, as an alternative to ecological restoration. In the same way, Rigs-to-Reefs provides the same tool, as an alternative to complete platform removal.

The decision framework for managing altered marine systems proposed by Schläppy and Hobbs (2019) would

TABLE 1 | Examples from the literature of practical considerations preventing offshore platform sites from being returned to their historical state.

Practical limitations	Example	References
Ecological considerations	Refuge for endangered and/or economically important species	Love et al., 2006
	Proportion of regional hard substrate provided by the platform	Love et al., 2003
	Attraction of fish from natural habitats, making them more vulnerable to fishing	Cowan and Ingram, 1999
	Risk of environmental contamination during removal	OGP Decommissioning Committee, 2012
	Highly productive ecosystem	Claisse et al., 2014
Environmental considerations	Spread of invasive species during removal/transport	Page et al., 2006
	Environmental damage caused by use of explosives during removal process	Kaiser and Pulsipher, 2003
	Disturbance of shell mounds and remobilization of toxic chemical contaminants	Phillips et al., 2006
	Cost of decommissioning	OGP Decommissioning Committee, 2012
Social considerations	Platform design making removal unfeasible	Parente et al., 2006
	Public support for Rigs-to-Reefs programs	Kaiser and Pulsipher, 2005
	Legal frameworks prescribing complete removal	Techera and Chandler, 2015
	Public opposition to the presence of platforms	Frumkes, 2002
	Obstruction to commercial fishing	Fabi et al., 2004

be a useful starting point for broadly classifying offshore platforms as novel ecosystems – however, because of the suite of complex, and in some cases contentious, issues surrounding oil and gas platforms, there are more factors that need to be taken into account. In this regard, the decommissioning decision analysis frameworks cited above could be used to assess a platform as a novel ecosystems even if decommissioning isn't yet being considered. For example, using the PLATFORM computer model for decommissioning analysis, Henrion et al. (2015) evaluated the impact of decommissioning options on attributes such as cost, benthic impacts, fish productivity, and water quality, all of which can be considered under novel ecosystems criterion 3 in this review.

CONCLUSION

Offshore oil and gas platforms play an ecological role for a wide variety of marine life, from corals and sponges (Gass and Roberts, 2006; Friedlander et al., 2014), to fish and sharks (Dugas et al., 1979; Schroeder and Love, 2002; Pradella et al., 2014), to marine megafauna (Robinson et al., 2013; Todd et al., 2016). At the end of their productive life, these platforms are generally removed completely and disposed of onshore, effectively removing the hard substrate and associated marine growth from an ecosystem that has developed over upward of 30–40 years (Driessen, 1986; Ferreira and Suslick, 2001). There is strong opposition to offshore drilling, and the negative perceptions of oil companies and their intentions is a big obstacle in the path of Rigs-to-Reefs programs (Löfstedt and Renn, 1997; Pietri et al., 2011). The costs of decommissioning offshore oil and gas infrastructure over the next 20–30 years run into the tens of billions of US dollars, with thousands of structures set to reach their end-of-life in this period (IHS Markit, 2016; Oil and Gas UK, 2017). In some countries, governments (and therefore taxpayers) cover some of the decommissioning costs; in the North Sea alone, this government expenditure could reach US $6.3 billion (Parente et al., 2006). Conversely, the ecosystems created by these offshore platforms have an intrinsic value in terms of fisheries, tourism, and conservation that cannot be ignored. As such, the ecological cost of decommissioning in the form of the destruction of these ecosystems must be an integral part of the decommissioning debate.

Based on the analysis of the novel ecosystems concept, many offshore oil and gas platforms can be defined as novel ecosystems, depending on a variety of factors. These platforms warrant further study, on a case-by-case basis, within the framework of novel ecosystems. This does not mean that restoration of these ecosystems should no longer be considered, as restoration may be feasible in many cases and therefore should be an option when a particular platform is to be decommissioned. However, classifying suitable offshore platforms as novel ecosystems allows for the recognition of the established, yet underappreciated, ecological value that these platforms provide.

The novel ecosystems concept can contribute to the consideration of decommissioning options using existing decommissioning decision analysis tools. Hobbs et al. (2017) proposed implementing a portfolio of approaches whereby management goals are based on the relative values of ecosystems. This approach recognizes the importance of altered ecosystems, while still allowing for conservation of high-value unaltered ecosystems. Applying this approach to decommissioning would involve identifying ecologically important platforms to be left in place for the ecosystem services they provide, while focusing decommissioning resources and effort on less ecologically valuable platforms.

One of the key arguments against novel ecosystems is that they give companies a "'license to trash' or 'get out of jail' card" (Murcia et al., 2014). This echoes the core opposition to Rigs-to-Reefs; namely that it is simply an excuse for dumping at sea (Macdonald, 1994). This argument, in both cases, ignores the potential ecological value of anthropogenically altered ecosystems. While it is undeniable that companies benefit financially from

Rigs-to-Reefs programs, this does not automatically mean that these programs are environmentally detrimental. It should be possible to ensure that any Rigs-to-Reefs policy is robust and comprehensive enough to ensure that any reefing of offshore platforms will benefit the environment.

AUTHOR CONTRIBUTIONS

SE and JM conceived the study. SE wrote the first draft of the manuscript. All authors contributed to the manuscript revision, read, and approved the submitted version.

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Ecological Role of an Offshore Industry Artificial Structure

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Decommissioning of oil and gas infrastructure globally has focused attention on its importance as hard substratum on continental shelf and slope habitats. Observational studies are needed to improve understanding of faunal assemblages supported by offshore infrastructure and better predict the effect of removal. Here, we present results from visual inspection and physical sampling of a small oil and gas industry structure decommissioned from an oil field in the North East Atlantic. This is supported by observations of similar structures nearby and by photographs of the surrounding seabed from environmental baseline surveys. The structure supported a reasonably high biomass and diversity of invertebrates (>10 kg and >39 macrofaunal and 17 megafaunal species) and fishes (>20 kg biomass and >4 species). The invertebrate megafaunal species present on the structure were a sub-set of the hard substratum fauna observed on surrounding seabed. Porifera were absent from the structure. Biological succession in the first 2 years occurred as follows. Sparse colonies of the hydroid *Obelia* sp. *stet* were early colonisers then subsequent development of thick hydroid turf (*Obelia* sp. *stet.* and *Halecium* sp. *stet.*) supported an invertebrate assemblage (2654 individuals kg wet mass^{-1}) dominated by saddle oysters [*Pododesmus squama* (Gmelin, 1791) and *Heteranomia* sp. *stet*.)] and scale worms (*Harmothoe* spp.). Percentage cover of hydroid turf varied significantly over the structure, with most growth on sections exposed to strongest currents. Commercially important fish species present around the structure included *Gadus morhua* (Atlantic cod), *Pollachius virens* (saithe) and *Lophius piscatorius* (monkfish). Studies of artificial structures such as this provide much needed data to understand their role in the ecology of seafloor habitats and inform environmental decision making on all stages of industry from exploration to decommissioning. We show that the ecological role of the decommissioned three-dimensional structures was to enhance the biomass of a sub-set of epifaunal invertebrates found in the area. This supported diverse associated macrofaunal organisms, providing a food source for motile invertebrates and fishes in an area where background hard substratum can be lost through the impacts of drilling.

Keywords: ecosystem restoration, rigs to reef, *Gadus morhua* (Teleostei), artifical reef, oil and gas activity, decommissioning

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INTRODUCTION

Artificial structures in the marine environment alter ecological structure and functioning. They provide habitat for threatened species (Bell and Smith, 1999), contribute reef habitat (Fowler et al., 2018), enhance recruitment of overfished species (Love et al., 2006), increase connectivity (Henry et al., 2018), often produce considerable fish biomass (Claisse et al., 2014) and provide foraging areas for large predators (Todd et al., 2016). These factors may vary over time, relating to environmental conditions and stage of ecological succession (Fujii, 2015). Consequently, artificial structures have a potential role in restoring degraded marine ecosystems such as coral reefs (Rinkevich, 2014), mollusc reefs (Walles et al., 2016), algal forests (Gianni et al., 2013), and have been proposed for restoration of disturbed deep-sea habitats (Cuvelier et al., 2018).

Oil and gas industry infrastructure is an important source of artificial hard substratum on continental shelf and slope habitats globally. Detailed descriptive studies of marine growth (biofouling) on oil and gas structures in the United Kingdom sector of the North Sea were carried out through early inspection and monitoring by oil and gas operators (e.g., Forteath et al., 1982). Oil and gas industry structures are rapidly colonised (Bell and Smith, 1999) and typically develop a highly productive ecosystem, e.g., ∼2700 tons of marine life have been estimated to live on the Shell Brent Alpha platform in the North Sea (Shell UK Ltd., 2017), including conservation priority species such as the reef-forming cold-water coral *Desmophyllum pertusum* (Linnaeus, 1758) (formerly *Lophelia pertusa*) (Bell and Smith, 1999). However, research on successional dynamics of organisms living on offshore infrastructure and the impacts to the surrounding benthos is surprisingly rare and generally limited to inaccessible consultancy reports (Gormley et al., 2018).

The imminent decommissioning of oil and gas infrastructure in all major basins has increased attention on their importance as hard substratum in the marine environment. This focus has led to an increase in observational studies on the role of oil and gas infrastructure in local ecosystems (Fowler et al., 2018) demonstrating that they support species of conservation importance (Rouse et al., 2019) including cold-water corals (Gass and Roberts, 2006). Oil and gas infrastructure likely increases, or at least focuses, fish production (Claisse et al., 2014), surrounding benthic biomass, diversity, and connectivity (Macreadie et al., 2011) so their removal may reduce secondary production (Pondella et al., 2015). The particular assemblages supported by these structures varies with structure age, water depth and height on the structure (McLean et al., 2018) and on different timescales (Fujii, 2015; Bond et al., 2018) so there is potentially variation within and between different basins. It is therefore important to develop a better understanding of faunal assemblages supported by offshore infrastructures in order to understand the effect of their removal. This is particularly important as environmental monitoring requirements for decommissioned oil fields are still to be established or are decided on a case-by-case basis (Jones et al., 2019).

In addition to the role of structures created by the oil and gas industry, the industry activities themselves change the surrounding environment. The oil drilling process discharges drill cuttings and drilling mud, which are released into the water and settle to the seafloor, smothering the natural sediments and associated meiofaunal, macrofaunal and megafaunal assemblages (Cordes et al., 2016). The accumulation of this material can lead to direct reductions of faunal standing stocks and biodiversity as well as secondary impacts, such as reducing habitat heterogeneity, further reducing diversity (Jones et al., 2007; Gates and Jones, 2012). Although both physical and biological recovery processes are evident, even in deep-sea ecosystems, there is evidence of persistence of disturbance for at least 10 years (Jones et al., 2012). Faunal assemblages associated with artificial structures may help mitigate or restore biomass lost to other drilling impacts but the net effects of positive and negative impacts to the environment from these anthropogenic activities are not clear.

Data from Remotely Operated Vehicle (ROV) video surveys are increasingly being made available by industry (Macreadie et al., 2018) to assess the faunal assemblages associated with subsea structures to understand the effects of decommissioning (e.g., van der Stap et al., 2016) but it is rarely possible to sample such structures. Here, we present results from visual inspection and physical sampling of an oil and gas industry subsea structure that was decommissioned from the Lancaster oil field, west of Shetland in the North East Atlantic in 2016. We aim to identify faunal assemblages associated with the structure and quantify biomass supported by it.

MATERIALS AND METHODS

Study Area and Drilling History

The observations reported here were made at wells drilled as part of a fractured basement exploration (Trice, 2014) of the Lancaster field, west of Shetland in the North East Atlantic (**Figure 1**). The Lancaster field is now undergoing development (Belaidi et al., 2018). The protective structure was in an area where the majority of the seabed consists of coarse sandy sediment with areas of hard substratum (boulders) to the north and west (**Figure 2**). The Lancaster field is within the area of an important commercial monkfish (*Lophius piscatorius* Linnaeus, 1758 and to a lesser extent, *L. budegassa* Spinola, 1807) fishery (Laurenson et al., 2008). In deeper water to the north west of Lancaster lies the Faroe-Shetland Sponge Belt Nature Conservation MPA and the west Shetland Shelf MPA lies to the south west (**Figure 1**). During field visits to the site (**Table 1**) the seabed water temperature ranged from 9 to 11°C. Seabed current direction is tidally reversing (approximately 80–100° to 270–300°, **Figure 3C**).

The *Transocean Sedco-712* semi-submersible drilling rig began drilling the Lancaster-205/21a-6 well in May 2014. On 1st July 2014 the well was suspended and a subsea protective structure (Structure A) was placed on top of the wellhead to prevent damage from trawlers. The protective structure was four sided, each side comprising three large panels of steel grating in two rows below on singe large panel, all panels were approximately 1 m^2) (**Figures 3a,b**). It was approximately a pyramid in shape (**Figure 4a**). Its seabed footprint was 3.6 × 3.6 m (12.96 m^2). In July 2016 the Lancaster-205/21a-7 well was drilled, 25 m from

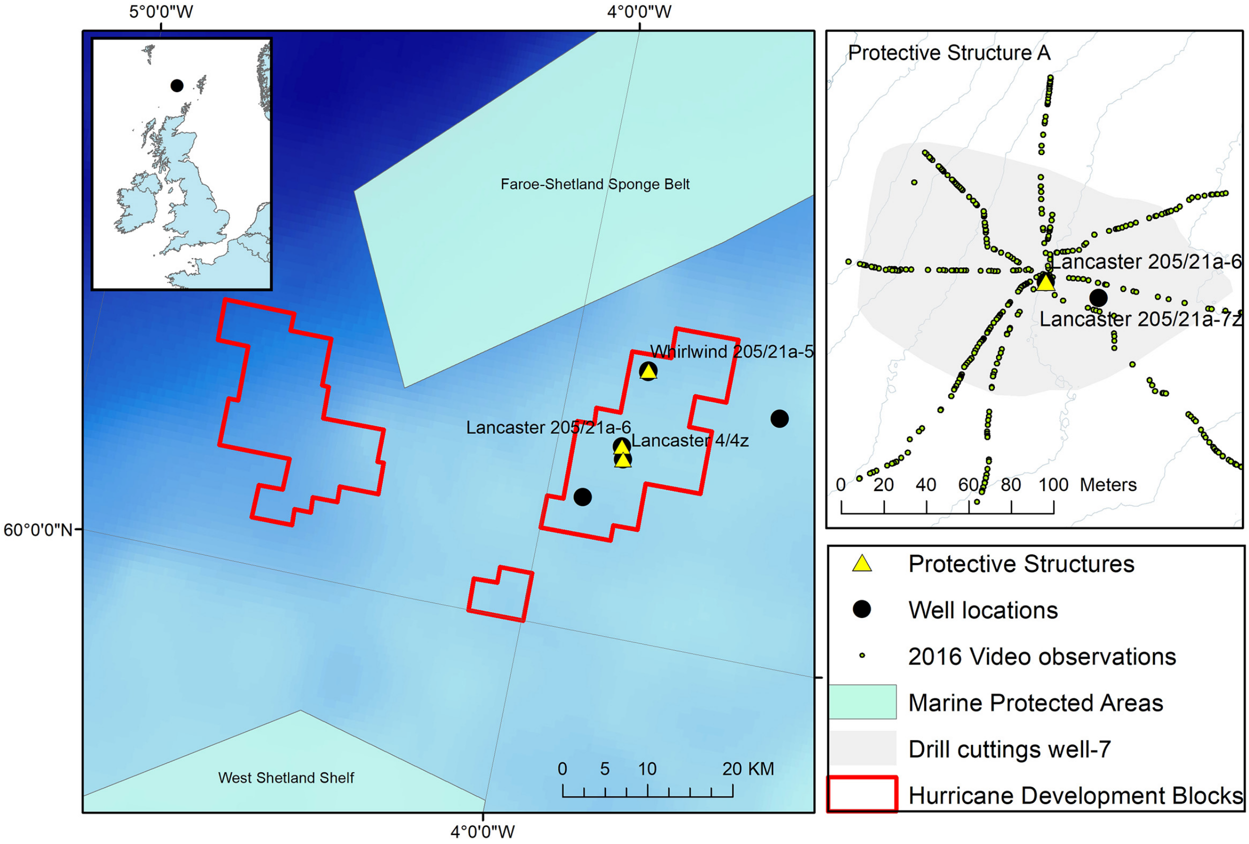

FIGURE 1 | Location of the protective structure (Lancaster 205/21a-6) investigated here. Other observations at the location are also shown including extent of drill cuttings around well after drilling Lancaster 205-21a-7z and the location of ROV video observations. Inset: Hurricane Energy well locations west of Shetland and Marine Protected areas including the Faroe-Shetland Sponge belt Nature Conservation Marine Protected Area (NCMPA) http://jncc.defra.gov.uk/page-6479.

the protective structure, by the *Transocean Spitsbergen* semi-submersible drilling rig. At this time, inspection of the protective structure revealed significant scouring around its base so the decision was taken to remove it. It had been in place for 860 days before it was decommissioned and recovered to the surface using a riser pipe on 2nd November 2016. Two other almost identical protective structures were observed at nearby well sites. The focus of this work is Structure A and we use observations from structures B and C to provide additional context. The sequence of activities reported in this study are shown in **Table 2** and details of the structures are shown in **Table 3**.

Fieldwork

Access to the *Transocean Spitsbergen* was gained through Hurricane Energy's participation in the SERPENT Project (Gates et al., 2017). *In situ* observations were made using an Oceaneering Magnum work class Remotely Operated Vehicle (ROV) equipped with standard definition video and a digital stills camera (Kongsberg OE14-208) and strobe (Kongsberg OE11-242). On 2nd October 2016 *in situ* still images of each of the large individual panels of the protective structure were taken to quantify the sessile invertebrate communities and fish assemblages. Additional close-up images of individual organisms on and around the structure were taken during that visit to the oil rig (26th September to 10th October 2016). To coincide with the recovery of Structure A (2nd November 2016) a further visit was made to the *Transocean Spitsbergen* (28th October to 3rd November 2016). Additional ROV observations were made of the structure *in situ*, although time constraints prevented a complete survey. Physical samples were collected when the protective structure arrived on deck. Samples of representative specimens of epifauna were taken for identification and biomass estimation. Five quantitative samples of the epifauna were also taken using either 250 × 250 mm quadrats or areas of the structure of measured dimensions. The time available for sampling was limited by a requirement to remove the structure from the drilling rig. During an earlier visit to the rig, a short seafloor current meter deployment was carried out using an Aanderaa SeaGuard single-point Recording Current Meter from 1st October 2016 10:40:00 to 3rd October 2016 09:30:00.

Structure B was observed in 2010 at the Lancaster-205/21-4C well after it had been on the seabed for 226 days. The Lancaster-205/21-4C well is 1.5 km from the well at Lancaster-205/21a-6. Water depth here was 155 m. It was only observed on a single occasion during the SERPENT project visit to the *Borgsten Dolphin* drilling rig and although a quantitative survey

FIGURE 2 | Seabed environment and anthropogenic impacts surrounding Protective Structure A at Lancaster-205/21a-6; **(a)** undisturbed seabed with boulders, **(b)** boulders impacted by drill cuttings, **(c)** undisturbed coarse sand, **(d)** sand covered by drill cuttings, **(e)** base of the protective structure in contact with the seabed showing no scouring, **(f)** scoured sediment on the north west corner of the protective structure, **(g–i)** examples of background hard substratum from the 2011 environmental baseline survey, note Porifera in all images.

TABLE 1 | Location of well sites used in this study.

Location	Lat	Lon	Depth (m)
Lancaster-205/21a-4	60.2	−3.9	155
Whirlwind-205/21a-5	60.3	−3.8	185
Lancaster-205/21a-6	60.2	−3.9	147
Lancaster-205/21a-7	60.2	−3.9	145

was not carried out, these observations do support the primary observation reported in this study.

Structure C was deployed at Whirlwind-205/21-A5, 9 km north of Lancaster-205/21a-6 on 16th October 2010 and provides further supporting information. On return to the Whirlwind site in 2011 Structure C was removed from the wellhead and moved 20 m away and placed at the seabed. ROV images were collected of that structure on visits to the *Wilphoenix* in 2011. These images were opportunistic and neither quantitative survey nor specimen collection were carried out.

Image Analysis

Quantitative image analysis was carried out on Structure A only. Seven individual panels of metal grating on each side of the structure were considered samples (**Figure 3A**). The area of coverage by hydroid turf was quantified using the software ImageJ (Schneider et al., 2012), using known dimensions of the panels. All animals visible in the photographs and in contact with the metal grating were counted. Animals on the painted yellow parts of the structure were not included in the quantitative analysis but are considered in the species list and their counts were used to inform total biomass estimate of epifauna on the structure. Separate colonies of colonial organisms such as *Parazoanthus* sp. *stet.* and *Filograna implexa* Berkeley, 1835 were recorded as individuals. For identification purposes, representative specimens of most taxa encountered in ROV photography were available. In most cases it was not possible to determine whether an individual in a photograph was the same animal as a specimen in the laboratory. Specimen identification from the ROV survey was also aided by using close up *in situ* images. Organisms were classified to lowest taxonomic level and named according to WoRMS (Horton et al., 2019). Open nomenclature identifiers were used to indicate identifier confidence following the protocol set out for physical specimens (Sigovini et al., 2016). The same approach was used for animals observed in photographs.

To investigate the role of height on the structure and orientation of the side of the structure two-way analysis of variance on logit transformed percentage cover of the panels by hydroid turf was done using the R programing environment (R Development Core Team, 2010). To ensure a balanced design the top panel was excluded from this test (6 panels per side of structure, 3 upper and 3 lower). Two-way analysis of variance was carried out on epibenthic megafaunal invertebrate density on the same panels.

The maximum number of fish (*maxN*) of each species in a single image was used to estimate total abundance of fishes associated with the structure. Accurate measurements of the total length (TL) of fish were possible because of the well-defined grid pattern of the metal lattice of the protective structure. TL of all individual fish that could be seen in images were taken using ImageJ. The metal lattice of the panels of the structure provided a grid of known scale against which measurements of the fishes were made (the small rectangular sections were 37 × 92 mm, measured at sea by the authors). To reduce measurement error, each fish was measured three times and a mean was taken. Where the same fish was observed in multiple sequential images (<10 s apart) measurements were taken from each image and a mean value presented. Only fishes in close proximity to the panels, whose TL was visible, were measured.

Biomass Estimation

Hydroid biomass was calculated based on the relationship between known area of panel and wet mass of hydroid from material collected on Structure A. This was calculated for one full panel from which all material was removed by scraping the hydroid turf into containers and four 0.25 × 0.25 m quadrats of material removed from the structure (**Table 4**).

Wet mass to body dimension conversions were calculated for the asteroids *Porania (Porania) pulvillus* (Müller, 1776) and *Stichastrella rosea* (Müller, 1776) (radial length (R), radius from center of disk to the end of arm) and the hormathiid anemone *Actinauge richardi* (Marion, 1882) (column diameter), to estimate biomass of these more common organisms on the structure (**Table 4**), following the method

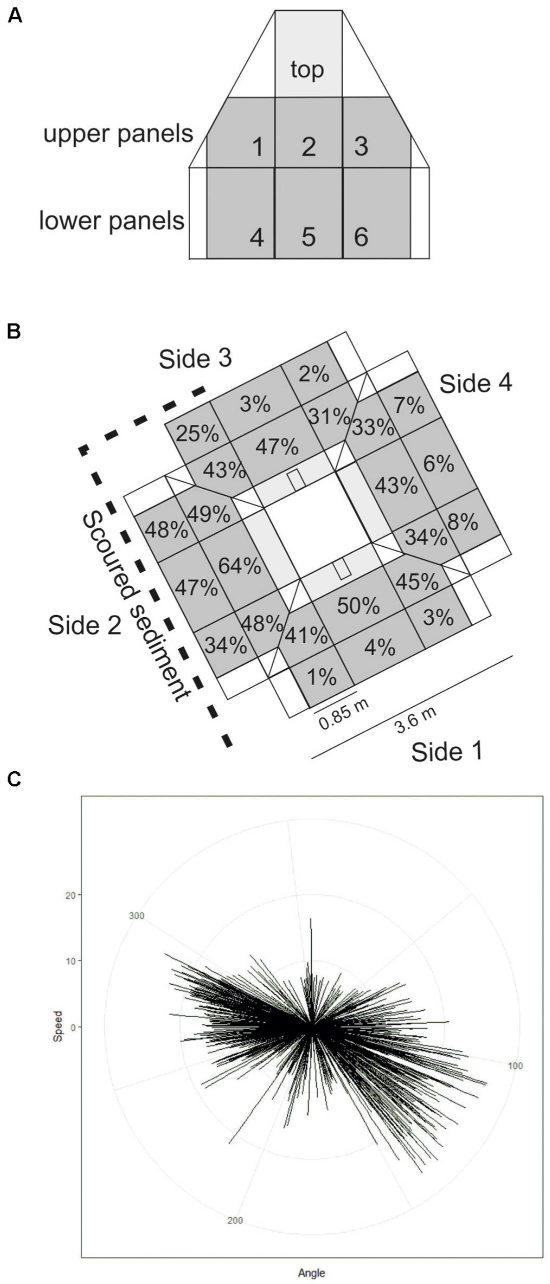

FIGURE 3 | (A) Representation of one side of Protective Structure A (not to scale). Panel numbers are labelled. Panels quantified in this study are grey in colour, those included in statistical analysis are darker grey. **(B)** Representation of the protective structure as it was orientated on the seafloor. Percentage cover by hydroid turf is indicated for each panel included in statistical analysis (diagram not to scale). **(C)** short-term seafloor current meter deployment with each line representing current speed and direction within a 5-min interval.

of Durden et al. (2016). For less common species, insufficient specimens were collected to calculate body dimension to wet weight ratios. Biomass was therefore estimated as the weight of individual representative specimens, or the mean weight if more than one individual was collected.

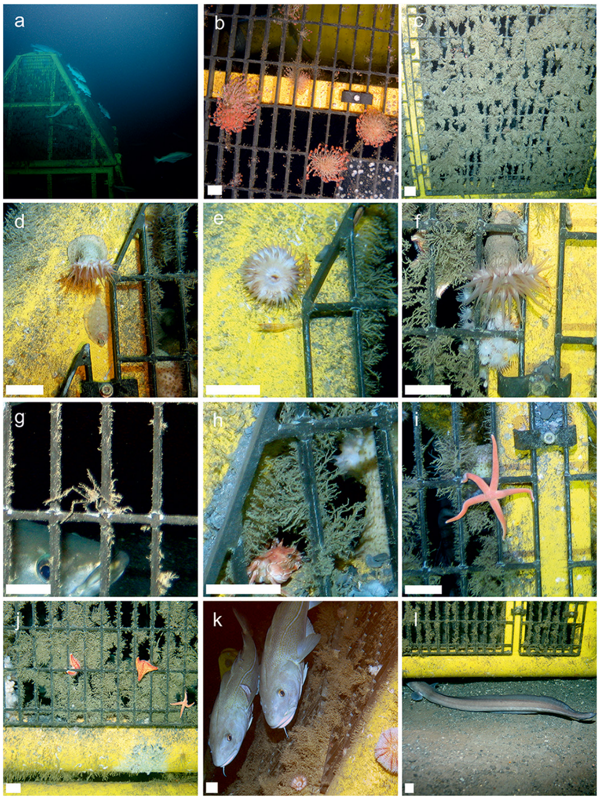

FIGURE 4 | *In situ* observations of protective structures. Scale bar on images b-l = 35 mm. **(a)** Structure A and fishes viewed from distance, **(b)** hydroid growth photographed on structure B, **(c)** hydroid turf covering a panel on the protective structure, **(d)** *Actinauge richardi* (Hormathiidae) and an unidentified flatfish, **(e)** *A. richardi* and two unidentified decapods, **(f)** *A. richardi*, hydroid turf and *Parazoanthus* sp., **(g)** *Macropodia tenuispinis* and *Pollachius virens*, **(h)** hydroid turf and the crab *Bathynectes maravigna*, **(i)** *Stichastrella rosea*, **(j)** *Porania pulvillus, S. rosea* and *Parazoanthus* sp., **(k)** Atlantic Cod, *Gadus morhua* and echinoid, **(l)** European Conger (*Conger conger*) entering structure through scoured sediment.

Biomass of fish was estimated from Length-Weight ratios in Fishbase (Froese and Pauly, 2019), which were based on measurements in Froese et al. (2014). In the case of *Pollachius virens* (Linnaeus, 1758), abundance is expected to be considerably higher than estimated by *maxN*. All individuals for which TL was visible in an image were measured and the mean value multiplied by *maxN* to estimate total biomass of that species.

Fauna Associated With Hydroid Turf

Preserved hydroid turf samples (4% borax buffered formaldehyde) were sieved to 250 μm, examined initially in white trays and then by inspection under a stereomicroscope. All specimens were removed, identified and counted. Following quality control of the sample processing it was clear that small bivalves were frequently missed from the sample picking. Therefore, a further 64.0 g (12% of the total) of hydroid turf was re-examined and the small bivalves and some additional polychaetes were enumerated and their total abundance in the overall sample estimated and included in the final count. Owing

TABLE 2 | Time-line of key events relevant to this study.

Date	Location	Time since drilling commenced (days)	Vessel	Water temperature 10 m off bottom (°C)	Activity
21–28 August 2009	Lancaster-205/21a-4	12	Byford dolphin	10.19	Seabed ROV video survey
∼13 October 2009	Lancaster-205/21a-4	∼60	Byford dolphin	–	Installation of structure B
24 May–1 June 2010	Lancaster-205/21a-4	288	Borgsten dolphin	9.33	Seabed ROV video survey Observation of structure B
16–25 October 2011	Whirlwind-205/21a-5	420	Wilphoenix	10.04	Observation of structure C
22–25 April 2014	Lancaster-205/21a-6	Pre-drill	Sedco 712	9.21	Seabed ROV video survey
28 May – 05 June 2014	Lancaster-205/21a-6	33	Sedco 712	9.50	Seabed ROV video survey
∼26 June 2014	Lancaster-205/21a-6	∼62	Sedco 712	–	Installation of Structure A
1 – 8 July 2016	Lancaster-205/21a-6	798	Transocean Spitsbergen	–	Seabed ROV video survey
26 September – 10 October 2016	Lancaster-205/21a-7	∼30	Transocean Spitsbergen	10.32	Seabed ROV video survey Observation of structure A
21 October – 02 November 2016	Lancaster-205/21a-7	∼60	Transocean Spitsbergen	10.40	Recovery and sampling of structure A
16 June 2018	Lancaster-205/21a-7	∼660	Paul B. Loyd Jr.	–	Seabed ROV video survey

to insufficient data for meaningful statistical analysis, the five hydroid turf samples were combined to provide one quantitative species list for fauna inhabiting known wet biomass of hydroid turf. Specimens were identified using literature for the north-east Atlantic and all material has been curated and housed in the Discovery Collections at the National Oceanography Centre for future scientific use.

Background Hard Substratum

In 2011, an environmental survey contractor surveyed the benthic environment at the Lancaster field. Images of the seafloor were acquired using a drop-down camera to assess the seafloor type. These images were not used for megabenthic image analysis at the time of the survey but have since been analyzed by the authors of this study. A sub-set of 189 images that contain hard substratum were used to assess the fauna associated with naturally occurring hard substratum at Lancaster for comparison with the megafaunal assemblage associated with the protective structure.

Surrounding Seabed and Drilling Impact

Vertical accumulation of drill cuttings at the well was assessed using graduated sediment marker buoys deployed before drilling operations began. These were placed at 5 and 15 m distance from the proposed well location. Spatial coverage of drilling disturbance was assessed by visual observation of the seafloor in quantitative video surveys (outlined below). Physical disturbance of the sediment by smothering with drill cuttings was classified as "complete," "partial" coverage and "undisturbed sediment" following the methods of Jones et al. (2006). Visual observations were validated using sediment samples and sediment barium concentration used to indicate sediment disturbed by drill cuttings.

On seven occasions, at different phases of the drilling operations, quantitative seafloor ROV video transect surveys were carried out at the Lancaster field (**Table 2**). Each of the seven surveys comprised eight video transects of approximately 100 m in length, radiating from the well. Video of the seafloor was recorded with standard definition colour video camera on a range of different ROVs depending on the drilling rig. In the transects, every individual whole animal that passed out of shot via the bottom of the frame was counted and their position in relation to the well was recorded. These surveys and subsequent analyses were carried out following the methods of Jones et al. (2006). In this study we present the mean abundance of megafaunal organisms and fishes (individuals m^{-2}) in each of the surveys, where each transect was treated as an individual sample in order to show the shared taxa between the protective structure and surrounding sediments and to highlight a change that occurred during the study period.

RESULTS

Structure A

Habitat Description and Anthropogenic Impact

After drilling Lancaster-205/21a-6, the surrounding seabed was covered by a layer of drill cuttings (**Figures 2b,d**) extending to 65 m from the well, after which the sediment resembled the background environment of sandy sediment (**Figure 2c**) with areas of boulders (**Figure 2a**). Following drilling of Lancaster-205/21a-7 drill cuttings extended >90 m from the well as indicated in **Figure 1**. Eight metres north of the Lancaster-205/21a-6 well, drill cuttings had accumulated 55 cm vertically. There was approximately 25 cm of cuttings accumulation at 12 m east of the well. Further from the well at 18 m, accumulation was lower and not measurable on the marker buoys but still visible in photography and in sediment samples. The base of most of the protective structure was in contact with the seabed (**Figure 2e**). On the southwest of the structure, sediment/drill cuttings had scoured, leaving a gap between the bottom of the structure on Side 2, Panels 4–6, around the corner of the structure to Panel 4 of the northwest facing side (Side 3, Panel 6) (**Figure 2h**).

Colonisation of Structure A

After 2 years 4 months (860 days), large areas of the individual panels of Structure A were covered by hydroid turf (**Figures 4c,j**). The turf comprised the hydroids *Obelia* sp. *stet*. and *Halecium*

TABLE 3 | Details of the three protective structures considered here.

Structure	Location	Date of observation	Days since installation	Description	Representative image
A	Lancaster-205/21a-6	31st October 2016	860	Coverage with hydroid turf, invertebrate epifauna and fish species in attendance	
B	Lancaster-205/21a-4	24th May 2010	226	Sparse coverage by *Obelia* sp. hydroid. No development of hydroid turf. No visible epifaunal invertebrates	
C	Whirlwind 205/21a-5	20th October 2011	364	Coverage with hydroid turf, invertebrate epifauna and fish species in attendance	

TABLE 4 | Dimension to fresh wet weight relationships from specimens used to calculate biomass on structure.

Organism	Dimension measured	*n*	Relationship to fresh biomass	R^2
Hydroid turf	Percentage cover	5	$y = 872.86x$	0.99
Actinauge richardi	Column diameter (mm)	22	$y = 0.8721x^{2.9698}$	0.82
Porania pulvillus	R (arm length to center of disk) (mm)	11	$y = 0.4277x^{2.91}$	0.91
Stichastrella rosea	R (arm length to center of disk) (mm)	3	$y = 0.0675x^{3.2201}$	0.98

sp. *stet*. The growth of hydroid turf varied over the structure. On the upper panels the percentage cover ranged from 30.9 to 63.7% (mean = 44.2%). The lower panels ranged from 1.3 to 47.8% (mean = 15.9%). On all lower panels, other than those on Side 2 (southwest facing), percentage cover was lower than 25% (with 8 of the 9 lower than 9%). On Side 2, the percentage cover of the lower panels ranged from 34–47.8% (**Figure 3B**). There was a significant difference in percentage cover between the upper and lower panels ($F = 21.1$, df = 1,20, $p < 0.001$) but no significant difference ($F = 0.1$, df = 1,20, $p = 0.75$) between the percentage cover on each side (once the height was taken into account) and there was no interaction.

TABLE 5 | Hydroid turf percentage cover and epibenthic megafaunal abundance on the visually surveyed sample panels of the protective structure.

	Side	**1 – South east**			**2 – South West**			**3 – North West**			**4 – North East**			**Surrounding seabed**
	Height	**Top**	**Upper**	**lower**	**Top**	**upper**	**lower**	**Top**	**upper**	**lower**	**Top**	**upper**	**lower**	
	Mean% cover hydroid turf	21.4	45.84	3.171	23.8	53.5	43	8	40.3	10.3	10.8	37	6.89	
	Mean estimated hydroid turf biomass (g)	186.8	349.1	245.9	180	463	367	70	349	88.2	81.2	320	58.8	
Cnidaria	*Actinauge richardi* (Marion, 1882)	1	22	1	4	10	4	2	9	1	4	9	5	X
	Parazoanthus anguicomus (Norman, 1868)		1			1			1			1		X
Annelida	*Filograna implexa* Berkeley, 1835		1						1		1	1		X
	Serpulidae *indet.*			1		2	2					1	2	X
	Terebellidae/Sabellidae *indet.*				6	4			4	1	2	8		X
Echinodermata	Comatulida *indet.*						1							X
	Echinoidea *indet.*							1						X
	Ophiopholis aculeata (Linnaeus, 1767)		1	1									1	X
	Asterias rubens Linnaeus, 1758	1		3						1	1	2	1	X
	Porania (Porania) pulvillus (Müller, 1776)	2	6	3	6	4	1	2	5	8	1	3	5	X
	Stichastrella rosea (O.F. Müller, 1776)		1	2	1	2		1	2	1		1	1	X
Arthropoda	*Galathea dispersa* Bate, 1859		4	2			11	1	2	3		2	2	X
	Macropodia tenuirostris (Leach, 1814)		2	5		1	4		1	11		1	6	X
	Decapod (Caridea) *indet.*		4	5			3		1	6	3	5	8	X
	Paguridae *indet.*			1						1				X
	Bathynectes maravigna (Prestandrea, 1839)					1						2		
Mollusca	*Calliostoma* sp. *indet.*	1		1									1	X
	Total megafauna density (ind. m^{-2})	5	16.06	8.525	19.8	8.42	8.87	7	8.75	11.3	14	12.1	10.9	

The estimated biomass of hydroid turf on the panels of the protective structure was 6.59 kg. This biomass supported smaller organisms at a density of 2654 individuals kg wet mass^{-1} of hydroid turf (**Table 6**). The associated faunal assemblage was dominated by small saddle oysters (*Pododesmus squama* (Gmelin, 1791) and *Heteranomia* sp. *stet*.) and polynoid polychaetes (*Harmothoe fraserthomsoni* McIntosh, 1897). The majority of *Pododesmus squama* and *Heteranomia* sp. were small and attached to the hydroids. Other molluscs included the nudibranch *Doto fragilis* (Forbes, 1838). The hydroid turf supported suspension feeders such as the terebellid *Pista cristata* (Müller, 1776) and the serpulids *Hydroides norvegica* Gunnerus, 1768 and *Serpula vermicularis* Linnaeus, 1767. Crustaceans included the amphipods *Stenothoe marina* (Spence Bate, 1857) and *Stenopleustes latipes* (Sars, 1858) as well as a copepod fish parasite (*Caligus* sp. *stet*.). Some of the larger associated fauna were also visible in the photographic surveys reported below.

Epifaunal Invertebrates, Fishes and Their Biomass

The quantitative photographic survey revealed the protective structure supported 17 epifaunal invertebrate megafaunal species (**Table 5**). Of these, all except the gastropod (*Calliostoma* sp. indet.) and the comatulid crinoid were identified from specimens collected from the structure. There was no significant difference in their density between the upper and lower panels, or the orientation (side) of the structure. In addition to the quantitative analysis of the panels, qualitative observations showed that the yellow painted structural lengths supported polychaetes (Family Serpulidae), larger saddle oysters attached to the surface (*Pododesmus squama* and *Heteranomia* sp. *stet*.) and hormathiid anemones (*Actinauge richardi*, **Figures 4d–f**). The serpulid polychaete *Filograna implexa* Berkeley, 1835 grew to greatest size and abundance on the upright section of the inner part of the structure, but this was not quantified here. The asteroids *Porania pulvillus* and *Stichastrella rosea* (**Figure 4i**) were common on the structure. Bare patches of grating close to these asteroids suggest they were feeding on the hydroid turf (e.g., **Figure 4j**). Galatheid squat lobsters (*Galathea dispersa* Bate, 1859) and the inachid crab *Macropodia tenuirostris* (Leach, 1814) were also observed in images.

In addition to the 6.59 kg of hydroid turf and associated invertebrates on the structure panels, there were a further 1.75 kg of epifaunal invertebrates. These were dominated by *Actinauge richardi* (0.95 kg) and *Porania pulvillus* (0.49 kg). Estimated biomass on the panels of the protective structure was therefore 8.34 kg. Addition of the estimated biomass from measurements of 133 large organisms observed on the structure outside of the sample panels (1.76 kg) results in a total estimate of 10.09 kg of epifaunal invertebrate biomass on the protective structure.

There were five species of commercially targeted fish in close association with the protective structure. In greatest abundance were *Pollachius virens* (saithe), which were observed both outside and within the structure (e.g., **Figure 4g**). They ranged from 0.50–0.57 m in length; estimated mean biomass was 1.32 kg ind^{-1} and *maxN* was eight (total biomass estimate of 10.83 kg). Two (*maxN*) *Molva molva* (Linnaeus, 1758) (ling) of 0.62 m and 0.51 m in length were associated with the structure (one inside one just outside). A single *Conger conger* (Linnaeus, 1758) (European conger) was

TABLE 6 | Species list: fauna associated with hydroid turf.

Phylum	Taxon	Density kg hydroid turf^{-1}	Representative specimens	Observed in structure photo survey	Observed on surrounding seabed
Cnidaria	*Obelia* sp. *stet.*			X	
	Halecium sp. *stet.*			X	
	Actinauge richardi (Marion, 1882)	19.70	22	X	X
	Parazoanthus anguicomus (Norman, 1868)*	1.97	1	X	X
Mollusca	*Doto fragilis* (Forbes, 1838)	7.88			
	Doto sp. *stet.*	17.73			
	Heteranomia sp. *stet.*	315.21		X	X
	Pectenidae *stet.*	19.70			X
	Pododesmus squama (Gmelin, 1791)	914.11	7	X	
Annelida	*Branchiomma bombyx* (Dalyell, 1853)	5.91			
	Filograna implexa Berkeley, 1835*	1.97		X	
	Harmothoe sp. *stet.*	63.04			
	Harmothoe fraserthomsoni McIntosh, 1897	327.03			
	Harmothoe imbricata (Linnaeus, 1767)	1.97			
	Harmothoe viridis Loshamn, 1981	5.91			
	Hydroides norvegica Gunnerus, 1768	3.94	3		
	Nereimyra punctata (Müller, 1788)	265.96			
	Nicolea venustula (Montagu, 1819)	1.97			
	Pista cristata (Müller, 1776)	3.94			
	Sabellidae *stet.*	15.76		X	
	Serpula vermicularis Linnaeus, 1767	3.94	7	(X)	(X)
	Spirobranchus triqueter (Linnaeus, 1758)	5.91		(X)	(X)
	Salvatoria clavata (Claparède, 1863)	78.80		(X)	(X)
	Trichobranchus glacialis Malmgren, 1866	5.91		(X)	(X)
Arthropoda	*Bathynectes maravigna* (Prestandrea, 1839)		1	X	
	Copepoda *stet.*	315.21			
	Galathea dispersa Bate, 1859	11.82		X	X
	Macropodia tenuirostris (Leach, 1814)		2	X	X
	Scalpellum scalpellum (Linnaeus, 1767)		2		
	Stenopleustes latipes (Sars, 1858)		24		
	Stenothoe marina (Spence Bate, 1857)	7.88	9		
	Stenothoe sp. *stet.*	218.68			
	Caligus sp. *stet.*	1.97			
	Isopoda *stet.*	1.97			
Echinodermata	*Asterias rubens* Linnaeus, 1758		3	X	X
	Ophiactis balli (Thompson, 1840)	3.94			
	Ophiopholis aculeata (Linnaeus, 1767)		1	X	X
	Porania (Porania) pulvillus (Müller, 1776)	1.97	11	X	X
	Stichastrella rosea (Müller, 1776)	1.97	3	X	X
	Individuals hydroid turf kg^{-1}	2653.66			

**Colony counted as 1 individual, (X) = tube worms visible in photographs, impossible to determine which species.*

observed swimming under the scoured sediment beneath the protective structure (estimated biomass = 0.84 kg). Four *Gadus morhua* Linnaeus, 1758 (Atlantic cod, **Figure 4k**) were observed swimming around the outside of the structure (estimated biomass = 10.07 kg). A single *Lophius piscatorius* (monkfish) was lying underneath the structure. The biomass of this individual was not estimated because only the tail could be seen. The estimated total fish biomass in close association with the protective structure was 24.73 kg.

Background Hard Substratum

The background hard substratum was assessed using seafloor images classified as "cobbles dominated" or "cobbles and sand." 34 species were associated with this substratum, dominated by Porifera (50% of observations, **Figure 5**) including *Hymedesmia* sp. *stet*. and an unidentified yellow encrusting sponge (**Figures 2g–i**). Cnidaria, dominated by an indeterminate solitary coral (Caryophylliidae) comprised 20% of observations and Bryozoa 18%. Of the 17 species (hydroids excluded) observed

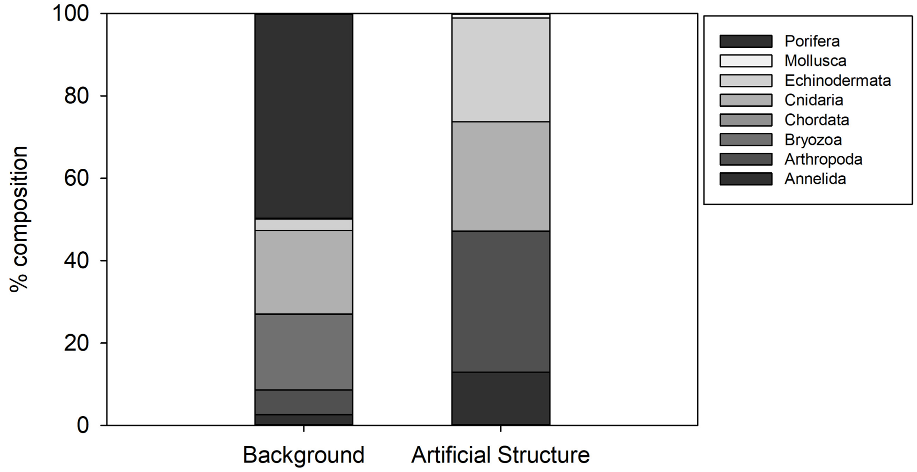

FIGURE 5 | Percentage composition of invertebrate Phyla observed on hard natural substratum at Lancaster and on the Protective Structure A.

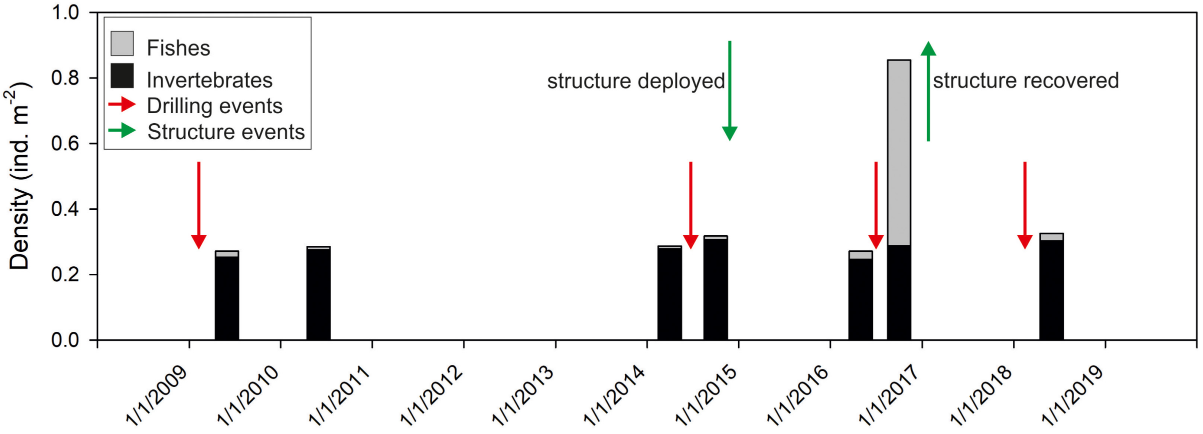

FIGURE 6 | Time series observations of mean abundance of megafaunal invertebrates and fishes in video transects around wells in the Lancaster field. Green arrows indicate deployment and recovery of Structure A. Red arrows indicate drilling events.

in the detailed image survey of Structure A, all except *Bathynectes maravigna* (Prestandrea, 1839) (**Figure 4h**) were also found in the background hard substratum. Two different crab species were encountered in the baseline survey but characters were not visible to determine if they were *B. maravigna*. Porifera, Bryozoa and Caryophillidae were not encountered on any of the protective structures. Hydroid turf was not observed in any seafloor images of hard substratum in the 2011 environmental baseline images, despite being distinctive and easily detected in images.

Seabed Surrounding the Structure

The ROV video transect survey shortly before recovery of Protective Structure A (27th September 2016) revealed increased abundance of small Triglidae sp. *stet.* (gurnard) and *Helicolenus* sp. *stet.* (redfish) fishes (**Figure 6**) as well as large numbers of small indeterminate pelagic fishes (not enumerated). In all the video surveys at the Lancaster field prior to the installation of the structure, fish abundance was lower. In a subsequent ROV survey, 18 months after removal of the structure, fish abundance had reduced to earlier levels (**Figure 6**). In addition to the increased abundance of small fishes, there were also increased numbers of predators. Eighty-four *Lophius piscatorius* were counted in the 27th September 2016 ROV survey (density = 0.07 individuals m^{-2}). In contrast only seven *L. piscatorius* were observed in the six other video surveys at Lancaster (0.001 individuals m^{-2}). Video sequences showed both *L. piscatorius* and *G. morhua* feeding on the small pelagic fish (**Supplementary Material 1**). The increased fish abundance was not recorded in the video surveys carried out in July 2016, despite the presence of the structure.

Supporting Observations – Structure B and C

On Structure B, small colonies of the hydroid *Obelia* sp. *stet.* had appeared after 9 months (**Table 3** and **Figure 4b**). On Structure

C, in the period between the end of operations in 2010 and their restart in 2011 the protective structure had been colonised by a wide variety of organisms. Hydroid turf coverage was patchy on the painted surfaces and thick growth covered the metal. Images suggest that species present were similar to Structure A, but *C. conger* was more abundant (*maxN* = 10).

DISCUSSION

In the 2 years since deployment the structure at Lancaster supported a reasonably high biomass and diversity of invertebrates (>10 kg and >39 macrofaunal and 17 megafaunal species) on a small area of seabed. This in turn supported >4 species of fish (>20 kg biomass). Data obtained from longer-established offshore structures off California, suggested they were some of the most productive habitats in the oceans (Claisse et al., 2014). This productivity is associated with the three dimensional structure of the habitat. It emphasizes the possibly important role that structures, even artificial structures, can play in marine communities.

Biological succession on the structures observed appears similar to that recorded in the Northern North Sea. With sparse colonies of *Obelia* sp. *stet.* (as seen on Structure B after >200 days, **Table 3**) being early colonisers (Forteath et al., 1982). The subsequent development of, in places, thick hydroid turf, is also typical (Boero, 1984). There were differences in the community development associated with location on Structure A. Coverage of the upper panels was greater than the lower panels and may reflect increased current velocity 2 m above the seabed. The SW facing lower panels (Side 2, 34–48% coverage) and the first panel on Side 3 (25% coverage) were unusual, in having high coverage near the seabed. In these cases, there was notable scouring of the sediment directly underneath the panels indicating that these panels were exposed to greater current velocity. The short-term seafloor current data support greater current velocity on the SW side (**Figure 3C**) but the currents were tidally reversing so do not explain why there was not greater hydroid coverage on the NE facing side of the structure. Increased current may support the colonisation of the structure through the greater exposure to suspended food particles or through the greater encounters with larvae passing the hard substratum upon which to settle (Wildish and Kristmanson, 1997).

The thick hydroid turf that developed on Structures A and C (**Table 3**) was unlike other habitat observed at Lancaster or Whirlwind (**Figure 5**) and it represents an important stage in the ecological succession of the artificial structure. It provides three-dimensional microstructure creating habitat that supports increased biological diversity (Di Camillo et al., 2017). Saddle oysters, amphipods and polynoids (*Harmothoe fraserthomsoni*) dominated the assemblage associated with the hydroid turf, similar to wind turbine structures and oil rigs in shallower water (Coolen et al., 2018). The genus *Harmothoe* has previously been reported associated with hydroid turf on artificial structures in the North Sea (Spierings et al., 2017). Juvenile and gravid female amphipods (*Stenothoe marina*) confirm the hydroid turf supporting production by other species. Specialist hydroid-consuming species such as the nudibranch *Doto fragilis,* reported to feed on *Halecium* sp. (Miller, 1961), were also present in the samples.

With the exception of the hydroid turf, the epibenthic megafaunal assemblage associated with Structure A was a subset of the fauna observed in broader surveys in the area. The absence of the most abundant groups on the background hard substratum (Porifera, Bryozoa and Caryophylliidae solitary corals) from Structure A was notable (**Figure 5**). Sponges contributed 50% of individuals on background hard substratum but were not present on the structure. Their distribution is influenced by various factors (Ramiro-Sánchez et al., 2019) such as current speed (Rice et al., 1990) and larval settlement on appropriate sediment type (Knudby et al., 2013) but it is not clear why they are not present on Structure A.

Actinauge richardi was the most abundant of the larger epifaunal species on Structure A and is common on offshore infrastructure in the North Sea (Rouse et al., 2019) and on a variety of substratum west of Shetland (Jones and Gates, 2010). *Filograna implexa* was present on the panels of Structure A, but colonies were largest and most abundant on the inner column of the structure, away from potential disturbance. *Filograna implexa* is part of the climax community of artificial structures (Forteath et al., 1982).

The complex habitat of the hydroid turf supported larger megafaunal invertebrate species. Bare patches of metal around the asteroids *Stichastrella rosea* and *Porania pulvillus* suggest they were feeding on the hydroid turf, as previously inferred for *P. pulvillus* from images taken elsewhere in the Lancaster field (Mah and Foltz, 2014). The faunal assemblage supported by the hydroid turf potentially provide a food source for the fishes observed at the site. This is supported by the fact that the organisms recorded in the hydroid turf in this study are equivalent to the polychaete worms, peracarid crustaceans and anomuran and brachyuran crabs that characterized the stomach contents of fish examined in a study of the feeding ecology around the Miller platform in the northern North Sea (Fujii, 2016).

It is most likely that the fish biomass recorded here was not produced on the structure but rather the commercially important fishes were attracted to it. *Gadus morhua* of 0.6 m in length are likely around 3–4 years old (Palakovich Carr and Kaufman, 2009) and *Pollachius virens* of 1 kg are likely 5–6 years old (Mathers et al., 1992). *Pollachius virens* is reported to aggregate around offshore oil infrastructure (Mathers et al., 1992; Fujii and Jamieson, 2016) and other structures such as fish farms (Otterå and Skilbrei, 2016) where they typically carry out diurnal vertical migrations. *Conger conger* are well known inhabitants of wrecks and other artificial structures (Steimle and Zetlin, 2000). Nonetheless, the protective structures observed here are clearly providing a service of some importance to the fishes. Many fish species have a tendency to aggregate (Fréon and Dagorn, 2000) as demonstrated by the role of fish aggregating devices in fisheries. In the North Sea, colonisation of a new structure by fish is rapid, with the first individuals arriving within 4 days (Todd et al., 2019). The increased fish abundance around the protective structure at Lancaster may have caused the increase in abundance of "sit and wait" predator *L. piscatorius* on the sediments up to 100 m from the structure. It is not clear how far the increased abundance of monkfish reached as the surveys were limited by the tether length of the ROVs launched from the drilling rigs. However, this may represent an example of the "ecological halo effect" in which artificial reefs support increased abundance and diversity over a considerably larger area than the reef itself (Reeds et al., 2018). There may also be a refuge effect because the areas around the drilling activity are protected from fishing by vessel exclusion zones.

Structures in the marine environment have long been considered for a role in ecological restoration (Grabowski and Peterson, 2007). Our results certainly suggest that they have a potentially useful role in aggregating or enhancing biomass of both fishes and invertebrates. In this area of the Faroe-Shetland Channel, natural hard substratum is common (Masson, 2001) and many of the same species naturally occurring in the area are attracted to the structures. Hydrocarbon drilling results in smothering of the seabed by drill cuttings and drilling mud, smothering hard substratum as well as causing a reduction in abundance and diversity of a range of size classes of benthic fauna (Netto et al., 2010; Gates and Jones, 2012). The loss of benthic fauna and associated ecosystem functioning from these changes in seabed type may be partially mitigated by the introduction of artificial hard substratum. However, our results suggest, on a short time-scale at least, that several of the most abundant background hard substratum species were absent from the artificial structure.

The structures evaluated here are relatively small, but commonly used in the region. In addition, much larger oil industry structures, including platforms and pipelines are present in large numbers in the North Sea area (Fowler et al., 2018). The effect of the size and type of the structure in controlling colonisation dynamics is unknown. However, the results from fine-scale detailed assessments may be valuable for providing details on the types of assemblage likely to develop on artificial structures and to improve understanding of the role of invasive species.

Environmental monitoring of offshore industries, including the oil and gas industry, generates large quantities of different types of data every year (Bean et al., 2017). Access to these data for scientific study can provide insights into the longer-term effects of industry activity (Henry et al., 2017) but are often inaccessible because they are presented as commercial consultancy reports. Opportunistic studies of seafloor infrastructure like this, and similar observations associated with industry structures such as well heads (Pradella et al., 2014) and pipelines (McLean et al., 2017) have revealed insights into how fishes and invertebrates utilize these structures, including the presence of species and features of conservation importance (Rouse et al., 2019). They provide much needed data to understand the role of these structures in the ecology of poorly studied habitats and inform environmental decision making on all stages of industry from exploration to decommissioning. Beyond understanding the impacts of industry activities in deep water, there is a global need for increased ocean observation to obtain the data needed to address challenges of societal concern (Ruhl et al., 2011). Access to industry datasets may be a way to increase the data available to the ocean observation community in order to drive a better understanding of the changing ocean in areas impacted by anthropogenic activity in addition to well-studied observatory sites (Levin et al., 2019).

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study because the research used opportunistic video and photographic observations of fish in their habitat.

AUTHOR CONTRIBUTIONS

AG conceived the idea, secured access to offshore infrastructure, and carried out the fieldwork. AS-S, TH, CC, AB, and AG analyzed the samples and images. DJ, AG, CC, AB, and KR analyzed the data and prepared the figures. AG, LG, CC, TH, AS-S, AB, KR, and DJ conceived the ideas, prepared tables and figures, and wrote the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2019.00675/full#supplementary-material

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frontiers in Marine Science

METHODS
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Enhancing the Scientific Value of Industry Remotely Operated Vehicles (ROVs) in Our Oceans

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Remotely operated vehicles (ROVs) are used extensively by the offshore oil and gas and renewables industries for inspection, maintenance, and repair of their infrastructure. With thousands of subsea structures monitored across the world's oceans from the shallows to depths greater than 1,000 m, there is a great and underutilized opportunity for their scientific use. Through slight modifications of ROV operations, and by augmenting industry workclass ROVs with a range of scientific equipment, industry can fuel scientific discoveries, contribute to an understanding of the impact of artificial structures in our oceans, and collect biotic and abiotic data to support our understanding of how oceans and marine life are changing. Here, we identify and describe operationally feasible methods to adjust the way in which industry ROVs are operated to enhance the scientific value of data that they collect, without significantly impacting scheduling or adding to deployment costs. These include: rapid marine life survey protocols, imaging improvements, the addition of a range of scientific sensors, and collection of biological samples. By partnering with qualified and experienced research scientists, industry can improve the quality of their ROV-derived data, allowing the data to be analyzed robustly. Small changes by industry now could provide substantial benefits to scientific research in the long-term and improve the quality of scientific data in existence once the structures require decommissioning. Such changes also have the potential to enhance industry's environmental stewardship by improving their environmental management and facilitating more informed engagement with a range of external stakeholders, including regulators and the public.

Keywords: subsea infrastructure, biodiversity, ocean observation, underwater technology, science-industry partnerships

INTRODUCTION

Advances in underwater technology have promoted increasingly comprehensive studies of our oceans and exploration to depths previously out of reach of humans (Marsh et al., 2013; Katija et al., 2017; Robert et al., 2017). A vast array of different underwater technologies now exists and are used, often for different purposes, by marine scientists and industries. However, oceanic scientific research, especially that conducted in the deep sea, remains costly in terms of logistics, personnel, and hardware, and restrictive spatially in terms of access to remote locations (e.g., Jones et al., 2013). The health of our oceans and sustainability of human activities therefore increasingly relies on the development of new technologies and cross-sector partnerships (Visbeck, 2018).

The offshore petroleum (oil and gas; O&G) and renewable energy industries routinely use underwater technology for observation, control, and maintenance of infrastructure in locations otherwise unavailable to marine scientists (Jones, 2009). These activities increasingly use remotely operated vehicles (ROVs), with over 550 industry workclass ROVs in operation globally (IMCA, 2015; **Figure 1**). The numbers of heavy-duty ROVs available to the scientific community is dwarfed by those utilized by industry, particularly the O&G sector. Moreover, industrial ROVs often remain at the same location for months to years whereas scientific systems deployed from a research vessel typically spend only a brief time at each study site. Annually, offshore industries invest billions of dollars collecting thousands of terabytes of data from ROVs, with much of these data archived within the companies involved and findings sequestered in commercially sensitive reports. These data remain an untapped resource, to investigate questions regarding the structure and function of offshore ecosystems, well beyond the purposes of their initial collection (Gates et al., 2017a; Macreadie et al., 2018). Consequently, industry-science partnerships that enable scientists to utilize and augment industry ROVs, and access collected data, have very high intrinsic value.

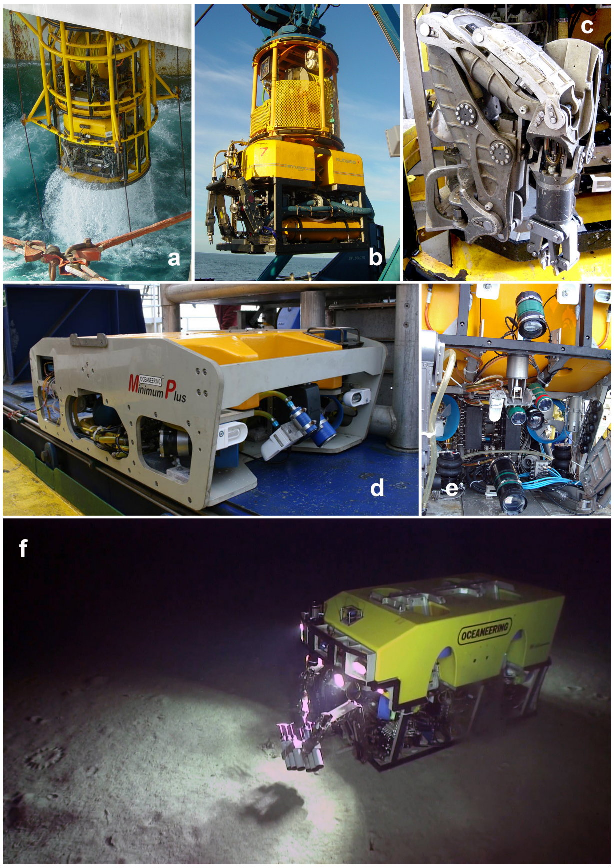

FIGURE 1 | Images of industrial ROVs photographed by the authors. **(a)** Subsea 7 work-class ROV (~2 m tall) being deployed from the *Transocean Jack Bates* semi-submersible drilling rig. **(b)** Subsea 7 *Centurion* work-class ROV (~2 m tall) on the MV *Nordica*. **(c)** Close up of Schilling Robotics *Conan* 7-function ROV manipulator (~1 m tall as imaged). **(d)** Oceaneering *Minimum Plus* observation-class ROV (~50 cm tall). **(e)** Detail of Oceaneering *Magnum* ROV (~2 m tall) showing video and still cameras on pan-and-tilt unit (center) and lights. **(f)** Underwater image of Oceaneering *Millennium* ROV (~2 m tall) taking push core samples of the seabed at around 1,700 m deep offshore Tanzania (taken with another ROV).

Industry is facilitating scientific research by providing ROV video that was originally collected purely for asset-maintenance purposes. Scientists utilize these historical industry ROV data to better understand the impact (both positive and negative) that the large numbers of static structures in our seas (offshore installations, including jackets, pipelines, wells, mattresses, etc.) have on marine ecology (Macreadie et al., 2018; McLean et al., 2018; Todd et al., 2018, 2019) and how this changes with time (McLean et al., 2017; Thomson et al., 2018). However, the extent and nature of industrial ROV operations is often not ideal for scientific analysis. For example, video resolution is frequently too poor, and illumination often too low, for species-level identification. As a result, data obtained from images are typically qualitative and descriptive, with identification to imprecise levels well above species. Even if raw video is of high-definition (HD) resolution, it is often downgraded to low resolution via down-sampling or compression to save archival space. Further, the distance of the ROV from the structure can limit its usefulness for identification of species. Close-up imagery of epibenthic organisms that often form complex "marine growth" habitats on offshore structures is required, whereas for assessments of fish populations greater setback from the structure is necessary for abundance counts. Industry ROVs usually operate on a 24-h schedule, and on an "as-required" basis and as such their timing may bias observations, particularly abundance estimates for species that exhibit diel and/or seasonal changes in behavior and population dynamics (Barker and Cowan, 2018; Bond et al., 2018a). Once imagery is obtained from industry, further challenges can be faced in video formatting for photogrammetric analysis and in spatially linking video to specific infrastructure components. For example, many industrial ROVs operate without ultra-short baseline (USBL) navigation systems that allow the ROV's position to be recorded in time and space and such information needs to be permanently tied to the video records if maximum value is to be obtained from the latter. There is an opportunity for these issues to be managed and for industry to obtain ROV video and associated metadata for their operational needs, while also providing useful data for research. This requires industry to understand what scientists need, and how scientifically useful their ROV video can be. Conversely, marine scientists need to understand better the operational practices of offshore industries if sensible proposals are to be made about changes to ROV operations. Such a bidirectional understanding will enable industry to adapt future inspection campaigns to enhance the collection of both industry-relevant and scientifically useful video.

Offshore O&G structures used by industry have finite operational lives dictated by factors such as the size of underlying hydrocarbon reserves and mechanical lifetimes. Thousands of structures will require decommissioning globally over coming decades, with decommissioning activity already increasing rapidly in some regions (e.g., North Sea; Fowler et al., 2018). Data provided by ROVs will be essential for timely investigation of the ecological role of offshore infrastructure and to predict the environmental effects of their removal/abandonment. Further, improved understanding of how marine communities are utilizing artificial structures can, and should, influence future structural design and installation to maximize environmental, social, and economic benefits. For example, the midwater sections of platforms off California provide important habitat for the recruitment of fishery-important rockfishes (*Sebastes* spp.), which then move to deeper platform sections as they mature (Love et al., 2006). Increasing the surface area and complexity of midwater structure in the design of future offshore energy installations would likely enhance fish production at such sites (Claisse et al., 2014). In this regard, ROV imagery of infrastructure is required to quantitatively measure structural features, associated marine life and to reveal relationships between them. Understanding the drivers of fish populations and fish biomass "production" would also assist Operators "design for decommissioning."

This paper identifies and describes operationally feasible ways to adapt and augment routinely performed industry ROV operations to improve the use of industry-obtained data for science (**Figure 2**). Use of ROVs by scientists independent of industry is not discussed here; however, lessons learnt from the use of science ROVs are relevant and have informed methods described herein. This paper is prepared by research scientists with the view to ensuring that suggested future modifications to industry ROV operations are operationally feasible to implement and unlikely to significantly impact on scheduling and costs.

ALIGNING EXPECTATIONS FOR SCIENCE-INDUSTRY COLLABORATION

Strong partnerships between industry and academic scientists: i.e., relationships between project managers, engineers, industry scientists, and ROV operators are essential for successful collaborative projects. This is in part because a suite of specific, costly training courses and requirements (e.g., helicopter escape safety training and offshore medicals, Federal security clearances), coupled with prior offshore experience and strict client Non-Disclosure Agreements (NDAs) are often prerequisites for accessing offshore O&G installations. Moreover, 500 m shipping-exclusion zones around many installations prevent outsider access. As a result, academic scientists are often precluded from accessing offshore assets, necessitating use of industry scientists and/or contractors to collect data. In these instances, academics can work with industry to develop succinct equipment operation instruction manuals or standard operating procedures and can perform trials to determine where difficulties in execution may occur. Consideration must be given to the expectations of ROV operators and industry partners if scientific data collection is to succeed and, in this regard, initial planning meetings and ongoing communication are essential.

A key role for ROVs in offshore drill-support operations is subsea asset integrity inspection. Such inspections provide an opportunity to document marine life growing on or living in close association with structures. Dives are performed regularly to inspect riser pipes, blow-out preventer (BOPs) and wells. Following such routine dives, ROVs are often on "standby," which presents an opportunity to perform scientific observations, at little to no extra cost to the client. Conversely, when ROVs perform "as found" seabed surveys on arrival at a new location, collection of scientific data during this period, while desirable, can cause delays to the start of operations, with unacceptable financial implications for the operator. However, where operators can look ahead at ROV schedules and include budget and time for dedicated science data collection into ROV campaigns, the resultant higher quality data can better inform environmental impact studies, decommissioning decisions and improve scientific understanding – a win-win for science and industry. For example, by collecting quantitative data on the conditions around a site prior to, and during drilling, an operator and regulatory agencies will have a much better baseline from which to assess changes to the environment due to routine activities or serious accidents.

In circumstances where ROV data collection cannot be completed independently by industry, researchers and industry could benefit from recent improvements in interactive video conferencing, which may allow researchers to interact with ROV pilots and guide ROV surveys remotely, thereby avoiding the cost and safety risks associated with visiting offshore installations. This model has been used effectively by scientific dives from the NOAA ship *Okeanos Explorer* where telepresence is used to enhance a small on-ship science team by communication with dozens of onshore experts during each ROV dive (Hoeberechts et al., 2015). The Inner Space Center is another example where telepresence is used to facilitate ocean exploration[1]. Internet access and bandwidth offshore are often limiting factors, and in the case of rig-moves, completely absent (i.e., no communications when satellite dishes are removed temporarily), so this level of communication is often not possible. If bandwidth is limited, latency may also present a challenge for remotely directed surveys because organisms of interest may not appear at the offshore and remote sites simultaneously.

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[1] http://innerspacecenter.org/

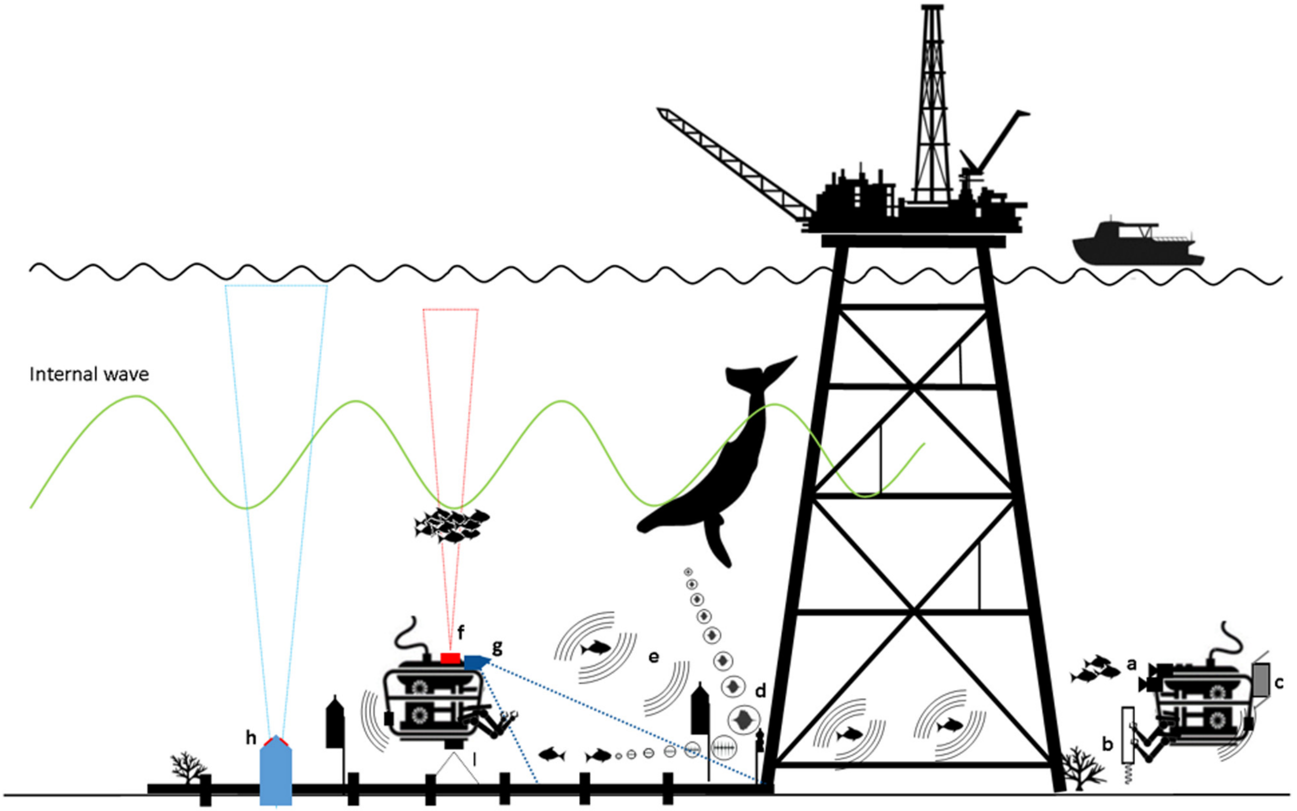

FIGURE 2 | Industrial remotely operated vehicles (ROVs) can be enhanced to collect scientific information through the addition of **(a)** high definition video and still photograph cameras; **(b)** sediment core samplers; **(c)** Niskin bottle water sampler; **(d)** passive acoustic monitoring; **(e)** acoustic telemetry tags and receivers; **(f)** echosounder; **(g)** forward-looking multibeam; **(h)** Acoustic Doppler current profilers (ADCP); **(i)** mapping multibeam. Diagram not to scale.

AUGMENTING INDUSTRIAL ROVs FOR SCIENCE

Industry ROV inspection activities include visual surveys but may also involve the use of manipulators to operate valves, changeover components, conduct cutting operations, and clear marine growth and debris (for example discarded fishing nets) (**Figure 1** and **Table 1**). Inspections are often undertaken annually but can be more or less frequent depending on the risk level and maintenance requirements. There is an enormous opportunity to use and augment these capabilities to facilitate ocean research on both temporal-spatial scales, without compromising routine O&G operations (**Figure 2**). A summary of augmentation approaches, the data they collect, and their feasibility are presented in **Table 1**.

All offshore industry instrumentation is subjected to rigorous risk assessment and must meet additional safety standards (e.g., explosion-proof). Any additional scientific instrument payload must be assessed for snagging hazard and stability testing. Further, if a standard industry ROV destined for a gas production platform survey is carrying any form of modification, such as additional cameras, all instruments must be pre-approved as “gas safe,” i.e., a non-source of ignition; consequently, the O&G industry utilizes custom-made technology that is designed specifically for use in hydrocarbon-rich locations. These costly types of modifications are made at the manufacturing stage, with a consequence of longer manufacturing lead-times, and good forward planning. Pressure housings must be rated to the maximum depth of the offshore site with a safety factor. Such considerations are hitherto unforeseen hurdles to overcome for the scientific community, who are not subject to these types of environments and associated restrictions. Consequently, good communication regarding additional scientific requirements is required long before any actual modified ROV reaches its final deployment destination. Nevertheless, the scope for augmentation is considerable and such factors can be accommodated via close liaison with offshore operators (e.g., McLean et al., 2019).

Enhanced Imaging Techniques

To facilitate accurate identification of fish, marine growth and other fauna, ROVs should collect at least HD video. Traditionally, industry ROV operations do not require collection of HD imagery and, as a result, analysis of historical imagery for science is hampered by difficulties in species identification and counting due to low image resolution (e.g., Bond et al., 2018a; McLean et al., 2018; Thomson et al., 2018). Most modern ROVs possess the ability to record HD imagery and should do so as

TABLE 1 | Key environmental indicators for assessment and the approach for obtaining them using industry ROV.

Indicator	Approach for measurement using industry ROV	Feasibility	References to support industry use
Bathymetry, seabed fabric and structural mapping	ROV mounted sonar	Low: Sector scanning sonar used as standard for navigation but not routine logging of data. ROV-mounted multibeam used in some scientific applications. Typically done using alternative approaches by industry.	Orange et al., 2002; Elvander and Hawkes, 2012; Allotta et al., 2015; Williams et al., 2016; Baran et al., 2017
Seabed appearance	ROV photography / video	High: Regular use of imagery for real-time visual characterization during routine inspection. Imagery regularly recorded. Standard approach for industry. Requirement for HD imagery.	Gates and Jones, 2012; Jones et al., 2014
Sediment physicochemistry e.g., particle size, sediment chemistry	ROV sample collection (push cores / Ekman grabs)	Medium: Use of sediment sampling equipment is straightforward but not standard. Depends on industry operations. Requires scientist to direct and process samples.	Gates and Jones, 2012
Seabed currents	Use ROV to deploy and recover current meters	High: Instrument is deployed at the seabed using the ROV and recovered subsequently. Duration of deployment depends on time occupied at the station for ROV operations and could be up to 1 year.	Salim et al., 2018
Water column properties	Water samplers (Niskin bottles) attached to ROV	Low: Water column sampling is straightforward and routine on scientific ROVs. It is rare on industry operations and equipment is not standard. Requires scientist to direct and process samples.	–
	CTD attached to ROV	High: Instrument attached to ROV collects data whilst submerged. Data collected internally and only requires an operator switch on/off before and after each dive. Some industry ROVs have built in CTD's but data extraction can be difficult. Additional sensors for chlorophyll, turbidity, dissolved organic carbon and dissolved oxygen are possible.	Gallager et al., 2004
Water column fauna	*In situ* video/still observations, including stereo-camera	High: Quantitative mid-water video transects. Often serendipitous encounters with larger animals. Benefits from additional oceanographic sensor data collection.	Benfield et al., 2009, 2013; Benfield and Graham, 2010; Hoving et al., 2013; Phillips et al., 2015; Jones and Pugh, 2018; Kupchik et al., 2018
	Downward or horizontal-looking echosounders (e.g., mounted on structure)	Medium. Use of scientific-rate echosounder equipment is straightforward, but not standard. *In situ* calibration is an issue. Data can be post-processed.	Stanley and Wilson, 2000; Rose et al., 2005; Wilson et al., 2006; Fujii and Jamieson, 2016
	Deployed ADCP/Upward-looking echosounder	Medium: Drilling operations often deploy ADCP for hydrographic assessment. Data rarely shared with scientists. Challenges with appropriate calibration for scientific requirements. Additional sensors can be added. Deployment of echosounder is similar to that of ADCP. Calibration is a potential issue. ADCP for current measurements are more common.	Osborne et al., 1977; Scotti and Pineda, 2004; Jones et al., 2006; Li et al., 2009
	Acoustic telemetry	High: Potential for ROV to carry receiver unit and opportunistically detect tagged fauna or carry and an acoustic tag and provide opportunistic performance testing. ROV can deploy acoustic receivers along maintenance routes or in dedicated trips to produce an array.	Heupel et al., 2015; Hussey et al., 2015; Thums et al., 2016; Wilson et al., 2018
Epifauna (epigrowth) abundance and composition	ROV photography/video	High: Subsea structure inspection is routinely carried out using industry ROV. Many examples of scientific study of routine pipeline inspection video. This can provide information about background faunal assemblages, oceanographic processes and the role of artificial structures.	Guerin et al., 2007; Lebrato and Jones, 2009; McLean et al., 2017, 2018; Bond et al., 2018a,d; Thomson et al., 2018

(Continued)

TABLE 1 | Continued

Indicator	Approach for measurement using industry ROV	Feasibility	References to support industry use
	Specimen collection	Low: Collection of specimens from structures can enhance quality of studies of artificial structures by improving identification of organisms, identifying associated organisms and measurement of biomass.	Gates et al., 2019
Epibenthic megafauna abundance and composition	ROV photography / video	High: Modification of routine as-found surveys can provide information on the pre-drilling seabed appearance. Scientist-directed ROV video transect surveys provide information about background faunal assemblages and how they change following anthropogenic impacts	Gates and Jones, 2012; Jones et al., 2013
	Specimen collection	Low: Collection of specimens provides improved identification of poorly known fauna. This is particularly valuable in deep or other poorly explored areas where many species may be undescribed species.	Horton and Thurston, 2015
Microbial/Infauna abundance and composition	ROV sample collection (push cores / Ekman grabs)	Low: Use of ROV sediment sampling equipment is straightforward but not standard. Sample collection is time-consuming and there is a requirement for processing at sea so needs scientist on board. Ability to collect these data depends on industry operations.	Nguyen et al., 2018
Marine mammal abundance and composition	Video assessments	Medium: Real-time ROV observations are unlikely to be a good way of observing marine mammals, owing to behavioral modifications.	Todd et al., 2016
	ROV-deployed hydrophones/ PAM	Medium: Installation or carrying of PAM is feasible but may be complicated by industrial noise in active areas and recovery of long-term moorings in inactive areas. Linking between industry and science can alleviate recovery of long-term moorings through the use of acoustic release systems.	Todd et al., 2009, 2016; Erbe et al., 2015
	Multibeam/imaging sonar	Low: Imaging sonar systems can detect "targets" at greater ranges than visual techniques, reducing effect of ROV presence and noise; however, target identification is limited.	Becker et al., 2013; Parsons et al., 2013, 2014; Lieber et al., 2015
Fish abundance, behavior and composition	Archive (historical) video of structures	High: Use of historical video imagery of structures to quantify associated fish.	McLean et al., 2017, 2018; Bond et al., 2018a; Thomson et al., 2018
	ROV video in routine operations or drill support standby time	High: Fish surrounding drilling operations can be documented during chance encounters or targeted video transect surveys.	Laurenson et al., 2004; Pradella et al., 2014
	Mid-water photography/video assessment	Medium: Deployment of bait to attract necrophagous species.	Benfield et al., 2008; Phillips et al., 2015
	Deployment of standalone camera	Low: Standalone camera for time-series observations.	Gates and Jones, 2012; Gates et al., 2017b
	ROV deployed hydrophones/ Passive acoustic monitoring	Medium: *In situ* observations of fish are complicated by ROV and industrial noise. Behavioral effects from ROV presence and noise are similar to that of video and sonar observations. Installation or carrying of PAM is feasible but may be complicated by industrial noise in active areas and recovery of long-term moorings in inactive areas. Linking between industry and science can alleviate recovery of long-term moorings through the use of acoustic release systems.	Rountree et al., 2002, 2003; Erbe et al., 2015; McCauley and Cato, 2016; Parsons et al., 2016; Todd et al., 2016
	Acoustic telemetry	High: Potential for ROV to carry receiver unit and opportunistically detect tagged fauna or carry and acoustic tag and provide opportunistic performance testing. ROV can deploy acoustic receivers along maintenance routes or in dedicated trips to produce an array.	Topping and Szedlmayer, 2011; Piraino and Szedlmayer, 2014; Froehlich et al., 2019

For each approach, feasibility is assessed and references to existing studies provided.

standard practice. Although the acquisition of HD imagery is relatively straightforward, storage of HD data and its handling can add a significant indirect overhead and cost (see section "Data Management").

Additional video and digital still cameras can be mounted onto workclass ROVs for the specific purpose of collecting HD imagery that facilitates scientific research (McLean et al., 2019; **Figures 2**, **3G,H** and **Table 1**). For example, McLean et al. (2019) recently added a compact stereo-video system onto a workclass ROV to collect HD imagery and enable accurate measurement of organisms' size. An ability to accurately measure size is particularly useful for assessing biomass, life-history stages of organisms and their ontogenetic shifts in habitat usage (Elliott et al., 2017), as well as the potential value of recreationally or commercially important fish species associated with offshore structures (Bond et al., 2018b,c). Stereo imagery also enables three-dimensional (3D) modeling of the environment which is important for describing both structures and marine growth, quantifying biological volumes, volumetric changes, and biological growth rates (Harvey and Shortis, 1995; Shortis and Harvey, 1998; Abdo et al., 2006). 3D HD video cameras have been developed (e.g., Oceaneering Ocean ProHD 3D imaging system) to provide the pilot with depth perception; however, these cameras also allow precise measurements of marine life to be performed. The addition of parallel lasers separated by a known distance (either point or line generators) is a simple and relatively inexpensive means of adding measurement capabilities to industrial ROVs. Such enhancements provide utility to the operator as well as for science.

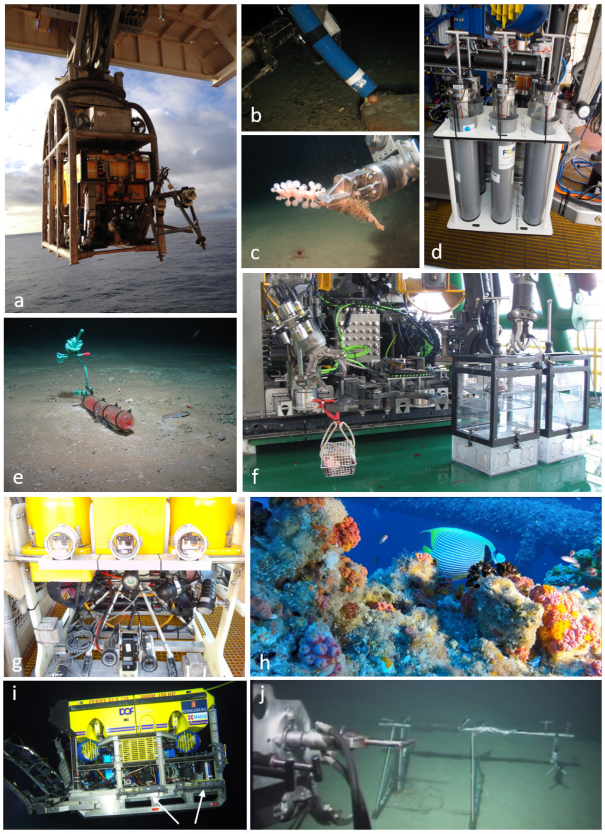

FIGURE 3 | Examples of how ROVs can be utilized to collect scientific information; **(a)** deployment of Oceaneering Magnum ROV holding a time-lapse camera system (to the right of the image) in 5-function manipulator arm, **(b)** modified zip pump for collecting fragile organisms (sea urchin pictured) suction tube attached to 5-function manipulator arm, **(c)** use of the 7-function ROV manipulator arm to collect specimens; **(d)** sediment core samplers, **(e)** deploying small traps to collect specimens (red scavenging isopods pictured), **(f)** bait cage and light traps to attract and capture organisms, **(g)** mounted stereo-video cameras to collect high definition imagery **(h)**. **(i)** DOF Subsea ROV fitted with Niskin Bottles (indicated by arrows) from the Sydney-Kormoran Project Expedition 2015 provided courtesy of Curtin University and WA Museum©. WA Museum **(j)** ROV recovery of a current meter from the seabed at 375 m, after Salim et al. (2018).

The bright lights and loud sound produced by electro-hydraulic ROVs may bias some surveys in favor of those taxa that do not actively avoid the vehicle. Under such circumstances, it may be useful to dim lights or equip the ROV with red-filtered lights. Widder et al. (2005) showed that an ROV equipped with red light was much less disruptive than white light for observations of deep sea sablefish (*Anoplopoma fimbria*). By shutting down the hydraulic pumps on an industrial ROV and using red-filtered lights, Benfield et al. (2019) were able to document the presence of additional fish species around a Gulf of Mexico rig that had not been detected during surveys with white lights while the ROV was running. Although additional lights and specialist cameras have proven value, careful consideration must be given to the location of such instruments added to an ROV to avoid interference with ROV operations, impacts of the ROV on the recordings made by added instruments (e.g., sound/lighting interference) and equipment damage.

Acoustic Transmitters and Receivers

Industry ROVs are well suited to carry and/or deploy certain types of acoustic sensors, which are extremely useful tools to census marine life on multiple spatial scales (Rountree et al., 2003; Koslow, 2009; Richardson et al., 2013; Hussey et al., 2015; **Figure 2**). Data acquisition is often autonomous, long-term, and non-interactive. Sensors can be mounted on ROVs or deployed to infrastructure or the seafloor and retrieved/redeployed on up to an annual basis (dependent on battery size, storage capacity, and recording schedule), to collect near-continuous temporal datasets at minimal effort (**Table 1**).

Passive acoustic monitoring (PAM) sensors (**Figure 4A**) can be carried or deployed by ROVs to monitor environmental noise around O&G activities (Todd et al., 2009; Todd, 2016), to detect vocalizing marine fauna (see section "ROV Acoustic Surveys of Infrastructure and Surrounding Environments") or to evaluate ecosystem changes evident in the soundscape (Gordon et al., 2018) but the ROV's acoustics signature may sometimes interfere with such uses. ROVs can more easily be used to carry acoustic telemetry receivers (**Figure 4B**) to detect marine fauna with attached or implanted acoustic tags (**Figure 4C**) and help understand how these animals (e.g., whale sharks) may use infrastructure (Topping and Szedlmayer, 2011; Piraino and Szedlmayer, 2014; Froehlich et al., 2019; **Table 1**), which may have consequences for decommissioning. The previously

FIGURE 4 | Examples of acoustic instruments that can be attached to, or deployed by an industry ROV to collect scientific information including **(a)** OceanInstruments SoundTrap ST300 (PAM recorder), **(b)** Vemco acoustic telemetry receiver, **(c)** acoustic telemetry tags (Images courtesy of VEMCO, part of Innovasea), **(d)** Blueview M900-2250 imaging sonar (Images courtesy of Teledyne BlueView), **(e)** Simrad WBAT echosounder, **(f)** Norbit Subsea WBMS multibeam echosounder (for seafloor and water column), **(g)** Teledyne RDI ADCP (acoustic Doppler current profiler).

mentioned caveat regarding depth ratings of pressure housings is relevant here because most passive acoustic receivers are designed to operate in relatively shallow depths. Imaging sonars (**Figure 4D**) can be mounted on ROVs to provide seafloor textural information or to count and size targets in front of the ROV (see section "ROV Acoustic Surveys of Infrastructure and Surrounding Environments"). ROVs could also deploy single-/splitbeam echosounders (**Figure 4E**) to evaluate biota in the water column and these have previously been mounted on and near O&G platforms to estimate numbers of fish around the structures (e.g., Wilson et al., 2006). Alternatively, multibeam systems or sector side-scan sonars (**Figure 4F**) can acquire data from the water column above or below the ROV to map fauna targets (e.g., schools of fish or megafauna) or natural/anthropogenic structures (Baran et al., 2017). Long-term datasets from ROV-deployed acoustic Doppler current profilers (ADCPs; **Figure 4G**) can provide oceanographic information pertinent to O&G structures, such as currents, temperatures, acoustic backscatter, or the detection of internal waves (Li et al., 2009).

Such ROV-acoustic programs are not without limitations. There is currently limited understanding of the potential performance and range-limiting effects of noise around structures and from ROVs themselves (Rountree et al., 2002; Giacalone et al., 2005; Dziak et al., 2018), and little data on potential behavioral bias their presence may induce (Spanier et al., 1994). Some acoustic-related activities may require dedicated ROV time and the presence of a specialist to initiate these activities (e.g., range-testing telemetry arrays, programming recording schedules, calibrating echosounders) and in extreme cases (multibeam hydrography or water column) a specialist operator for data acquisition.

ROVs as *in situ* Samplers

Industry ROVs are well equipped to take samples of sediments and organisms from structures, the water column and the seafloor (**Figure 1C**) but may require additional equipment to store collected samples. Many workclass ROVs are equipped with a 7-function manipulator arm, which is dexterous and capable of precision sampling. Science-class ROVs (e.g., ROV Jason, ISIS, Holland) and human occupied vehicles (e.g., the Alvin submersible) typically use the same or similar 7-function manipulator arms (**Figure 3C**). Industry ROVs tend to also be equipped with a 5-function manipulator (**Figures 3A,B**), which is of less value in precision sampling, but can be used to hold sampling equipment. These arms can be used to either directly pick up robust objects or to hold specific tools for precision sampling, particularly for the collection of sediment or marine biological specimen samples.

The collection of marine specimens can aid identification of organisms and provide specimens for other biological studies. While some species are possible to identify from video, many require detailed examination of features that are not visible from images, for example because of orientation or resolution (Macreadie et al., 2018). In many areas, the species present are poorly characterized and may include undescribed species, making collections more important for identification. Specimen collection can be done using a wide range of methods, including suction samplers (**Figure 3B**), small manipulator-operated nets or core samplers (**Figure 3D**), and direct collections using the manipulators, micromanipulators (**Figures 1C**, **3C**; Galloway et al., 2016) or traps (**Figures 3E,F**). In addition to making collections, it is also possible to directly introduce animals into experimental equipment to carry out *in situ* manipulative experiments using industrial ROVs (e.g., Hughes et al., 2010). Adding additional functionality to ROVs to collect multiple biological samples in a controlled manner (minimizing cross contamination) is an area requiring some engineering solutions.

In marine ecosystems, an emerging technology is the collection of "bulk" environmental samples (e.g., seawater, marine growth, or sediment) for genetic analysis and long-term biobanking (Jarman et al., 2018), of which ROVs will play an increasingly important role. For example, **Figure 3I** illustrates the use of ROV-mounted Niskin bottles for the collection of seawater. Environmental DNA (eDNA) is defined as all the genetic material that can be recovered from an environmental sample, which can originate from multiple species and from a variety of biological sources including: whole organisms, biological secretions, reproductive propagules, shed skin/hair, degrading tissue, or can exist as free DNA molecules (Taberlet et al., 2018). Once collected, information embedded within the DNA provides a lens through which to study the organisms that were present within that environment. Accordingly, the dependence of traditional biological surveys on retrieving whole organisms and complexities associated with morphological identification (i.e., time-consuming microscopy, difficulties identifying different life stages, sexes, and cryptic species) is largely overcome.

ROVs as Oceanographic Samplers

Oceanographic sensors can be mounted onto industry ROVs to collect data as the ROV travels through the water column or can be deployed by the ROV onto the seabed to collect data on near-bed conditions. Although many industry ROVs contain either CTD (Conductivity, Temperature, Depth) instruments or at least temperature sensors, extracting these data from the whole suite of data collected by the ROV has been problematic in the past. Alternatively, dedicated CTD instruments that may contain optional sensors to measure dissolved oxygen, chlorophyll, turbidity and dissolved organic matter can be mounted on the ROV and collect data autonomously (e.g., Gallager et al., 2004) and independent of ROV central systems. Additional sensors can be used, particularly low-power, cost-effective, high-precision sensors capable of deployment to depth and in rough conditions while still be able to relay information in real time. Such sensors can measure a whole range of oceanographic conditions including light, temperature, sound, mass, chemicals (including synthetic compounds and organic molecules), pH, Eh, fluorescence, chlorophyll, etc. ROVs could also be used as a vehicle to deploy and recover oceanographic instruments on the seabed. Here, the instruments may be placed on the seafloor to collect data that include ocean currents and properties (e.g., CTD and optional sensors as described above). The duration of data collection is dependent on ROV activity and can extend from a few hours to months. Salim et al. (2018) described such a deployment on the north-west shelf of Australia to study sediment transport processes (**Figure 3J**). Such data from widely dispersed locations can yield meaningful datasets as evidenced by programs such as ARGO and ships of opportunity[2] whilst also bridging knowledge gaps between oceanographic research and monitoring platforms elsewhere along coastlines.

SCIENTIFIC SURVEYS OF INFRASTRUCTURE AND SURROUNDING ENVIRONMENTS USING AUGMENTED INDUSTRIAL ROVs

In general, there are several different environments accessible to industry ROVs: the infrastructure and surrounding water column, the seafloor beneath and immediately surrounding infrastructure and undisturbed areas subject to O&G exploration (**Figure 2**). Scientific evaluations in each of these environments are valuable and the areas impacted by industry activities are often inaccessible to any other approach.

For the seafloor, many locations accessible by industry ROVs have been subject to disturbance by industry or in some cases by natural activities (Lebrato and Jones, 2009; Jones and Brewer, 2012; Jones et al., 2014). These areas include those disturbed by sediment deposition (e.g., cuttings piles), physically altered by industry (e.g., covered in concrete during well construction or anchor scars in the sediment), and potentially chemical pollution (e.g., hydrocarbon release). Disturbed locations are generally small in spatial extent and difficult to access for scientific study, often being close to industry operations (Jones et al., 2007). As such, scientific information on these areas is limited and reliant on working with industry to better understand impacts from disturbance and recovery (Jones, 2009; Gates and Jones, 2012; Jones et al., 2012).

Offshore infrastructure and associated ROVs provide a rare opportunity to study reef ecosystems in deep water. Considerably less is known about reefs beyond depths accessible to divers, compared to those in shallow water, owing to logistical challenges and costs associated with their investigation (Brokovich et al., 2008). Vulnerability of deep reefs to anthropogenic impacts and environmental change also make them a priority for research (Roberts et al., 2006). Although unplanned, offshore installations can behave like artificial reefs and develop entire reef ecosystems during their operational life (Schroeder and Love, 2004; Macreadie et al., 2011). While they may not completely mimic natural reefs, these infrastructure ecosystems facilitate investigation of various aspects of community structure and function, including settlement processes, species composition, trophic dynamics and behavioral interactions (Macreadie et al., 2018).

A scientifically dedicated methodical survey of infrastructure-associated marine life can yield quantitative data which is more accurate and useful for informing development proposals, impact assessments and, at end of field life, decommissioning comparative assessments (CAs) or Net Environmental Benefit Analysis assessments (NEBAs) (Nicolette et al., 2013; Sommer et al., 2019). Such programs can be completed with efficiency (e.g., McLean et al., 2019), adding little cost to field campaigns. There are a variety of different ways that dedicated surveys of marine communities associated with infrastructure can occur, with the chosen technique largely dictated by ROV operational logistics. For example, vessel-deployed ROVs may have more maneuverability and therefore capacity to survey a greater proportion of a platform jacket than platform-deployed ROVs. Water currents, structural design/elements and depth of the infrastructure all also influence survey design.

The seafloor and water column surrounding infrastructure can be surveyed using ROVs typically through observation and collection of unknown species (e.g., Pugh et al., 2018), behavioral studies (Haddock et al., 2018), or quantitative evaluation of community structure (Bamstedt et al., 2003). These rely primarily on good quality imaging, precision specimen collection, and video transect surveys (**Table 1**). These approaches are possible, and many are routine operations for industrial ROVs. In the case of exploration O&G vessels, the only structure between the rig or ship and the seabed is a small riser. Thus, observations of the water column typically reflect the ambient plankton and nekton with little influence from the riser. Because of the unusual locations of many industry ROV operations, faunal

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[2] http://imos.org.au/

observations from industry ROVs have proved valuable in quantifying new distributions (Benfield et al., 2008; Hirai and Jones, 2012; Jones and Pugh, 2018), describing new species (Vecchione et al., 2001), observing new behaviors (Laurenson et al., 2004; Benfield et al., 2013), evidence of reproduction (Todd et al., 2018), and quantifying baseline environmental conditions (Macreadie et al., 2018).

ROV Video Surveys

ROV Video Surveys of Infrastructure and the Surrounding Water Column

Figure 5 illustrates examples of how dedicated ROV visual surveys can be carried out on jackets (**Figures 5A–D**) and wells (**Figures 5E,F**) to survey marine growth and fish communities. Surveys of infrastructure communities across depth zones can be obtained by facing the ROV cameras toward the structure during descent and ascent. Offset distances should be carefully chosen for effective survey of the species of interest and this distance maintained (as well as consistent ROV speed). For example, McLean et al. (2019) augmented an industry ROV with stereo-video cameras (**Figure 3G**) and instructed ROV operators in the collection of imagery from vertical transects of a platform jacket, with surveys conducted independent of scientists at the site of operations. Four surface-to-seabed and return transects took a total of 1.25 h to complete (∼15 min per vertical transect) within a scheduled 2-week ROV field campaign and produced quantitative information on fish diversity, abundance, size structure, and marine growth extent and complexity (McLean et al., 2019).

Further surveys of infrastructure marine communities could be obtained opportunistically by simply facing ROV cameras toward the structure when ROVs are transiting between locations, resting on the seabed, or in the tether management system (TMS) between operations. ROV pilots can be trained to conduct belt transects around the base of infrastructure during ROV down time. Belt transects involve "circling" the structure at a constant speed while maintaining a constant distance and field of view (Love et al., 2000; **Figure 5F**). Such transects can be quick, taking approximately 90 s to complete, and can assist the training of less experienced pilots in ROV operations. Resulting opportunistic video has already been used to quantify the abundance and diversity of tropical reef fishes associated with wells and platforms in north-west Australia (Pradella et al., 2014; McLean et al., 2018; Thomson et al., 2018).

HD imagery collected during these surveys provides information on fish diversity, abundance and size (particularly where stereo-cameras are used), of marine growth type, extent and complexity, and of behavioral associations between fauna and structures (McLean et al., 2019). An emerging image processing technique called "Structure from Motion Photogrammetry (SfMP)" can be used to produce spatially referenced 3D models of structures and marine growth using sequences of 2D images (e.g., video frames) acquired from multiple perspectives by a single moving camera (**Figure 6**). Recurring features within a sequence of overlapping images are used to align images and estimate angle and orientation of the camera for each image. Camera positions are then used to assign *x*, *y*, *z* coordinates to features within images and generate a 3D point cloud (Westoby et al., 2012). A mesh, textured with original source images, can be interpolated between the points to produce the final 3D model (Figueira et al., 2015; **Figure 6**). If an object(s) with known dimensions and/or location in a real-word coordinate system are included in the images, the 3D model can be scaled and orientated, and then used to derive the area, volume, and surface roughness of objects or landscapes (Bennecke et al., 2016; Ferrari et al., 2016). For objects (e.g., marine growth species) that have a known density, the volume estimates can be converted into mass (Westoby et al., 2012). 2D to 3D reconstruction can be problematic for moving objects (such as fish, or marine mammals, and also moving marine growth), when based on single camera image acquisition, though stereo-camera (**Figure 3G**) or multi-bank cameras may provide solutions to this problem. In the context of surveys of marine growth on offshore infrastructure, it offers considerable scope for quantification and description of biofouling organisms. An understanding of marine growth type and mass is an urgent priority for research with multisector, multidisciplinary, and global relevance. This is because establishment and colonization of marine growth on subsea infrastructure over time has significant consequences for engineering, both in terms of maintenance and design, while the marine communities themselves deliver ecosystem services including water filtration, carbon sequestration, and fisheries.

ROV Video Surveys of the Seafloor Surrounding Infrastructure

Seafloor ROV video-transect surveys have proved valuable to quantify benthic disturbance (e.g., Gates and Jones, 2012; Jones et al., 2012). Repeat ROV transect surveys carried out at different phases in drilling operations over time reveal changes to seabed habitats and their potential for recovery. Feasibility of such a time series depends on a plethora of factors such as: cost, project duration, collaboration with industry and wider industry partners (if a consortium of operators). Surveys could occur: (1) before operations (e.g., during baseline surveys before arrival of a drilling rig, or pre-laid pipeline route survey), (2) survey immediately prior to operations ("as found" surveys are often carried from the drilling rig to check the site is safe to drill), (3) at various operational stages throughout drilling (typically 2–3 months), and (4) during subsequent operations at the same site or return visit by ship (Jones et al., 2012).

An ROV launched from stationary oil industry infrastructure such as a production platform (e.g., those in **Figure 1**), semi-submersible drilling rig, drill ship or jack-up rig will be restricted by tether length to around 100–500 m of horizontal movement over the seafloor. As a result, the most practical survey strategy is radial transects. Such transects radiate along a disturbance gradient from the well in the center. Video (ideally HD color video) and still images are typically collected continually or at regular intervals along the transect (e.g., as detailed in Jones et al., 2007; Benfield et al., 2019). If not present, augmenting an industry ROV with a high

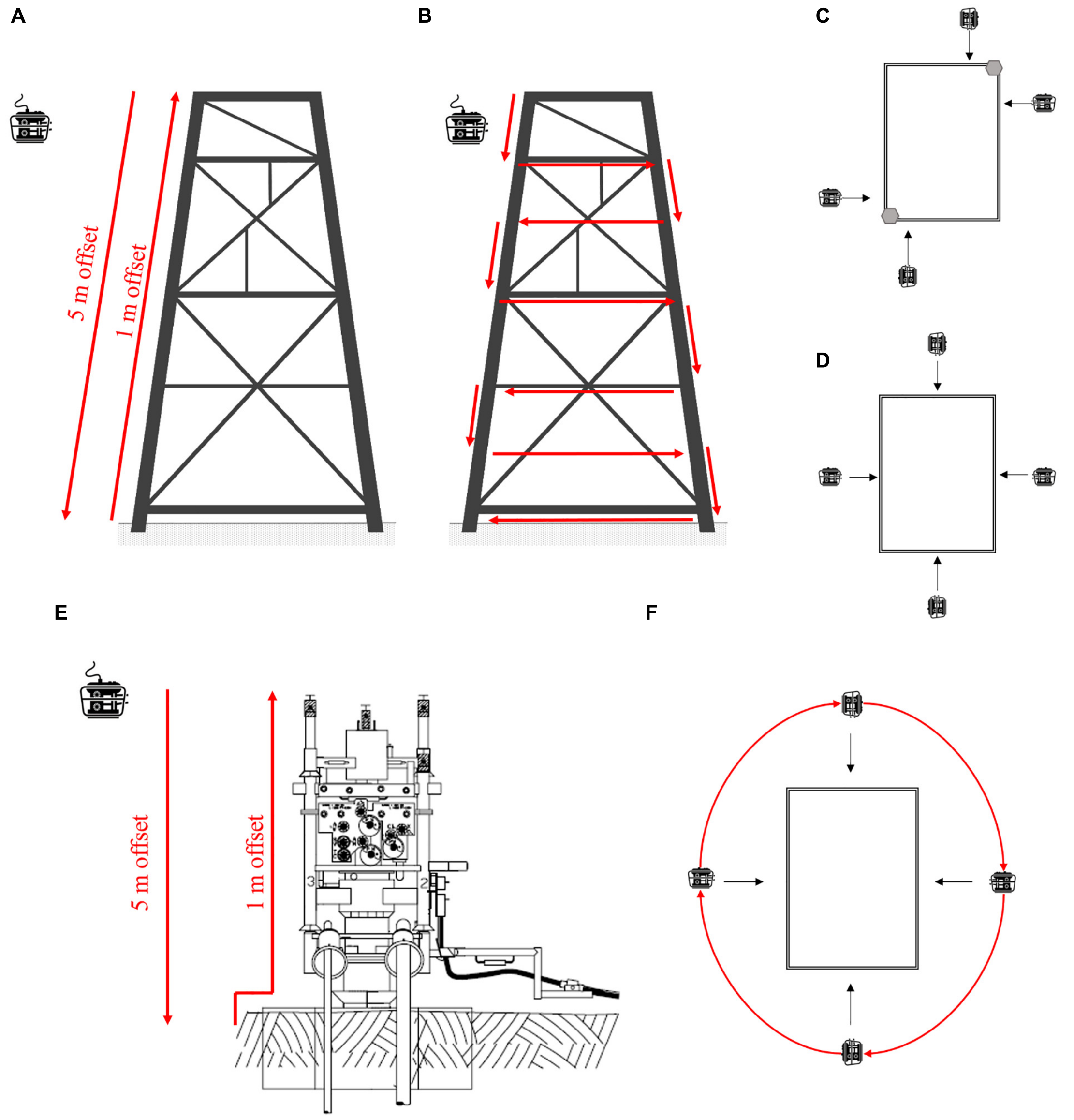

FIGURE 5 | (A) Vertical transects of a platform jacket, **(B)** horizontal and vertical surveys of a platform jacket, **(C)** birds eye view of a platform-deployed ROV limited to certain areas of a jacket and **(D)** birds eye view of vessel-deployed ROV survey of a jacket. **(E)** Wells are typically <9 m in height and surveyed by performing a vertical survey close to and further away and **(F)** by moving the ROV around the well structure.

resolution photographic camera (and flash/strobe) considerably enhances the quality of observations in these transect surveys. The addition of laser scalers, preferably line projectors, is essential to quantify the field of view. A subsea navigation system (USBL) is desirable to provide precise positioning information. This is particularly important if transects are to be duplicated at a later date. Transects need to be carried out at low altitude, slow speed and with good navigational accuracy to be most useful. With good quality transects, sediment can be classified based on observations of level of coverage of the seafloor by drill cuttings. Quantification of fish and megabenthic invertebrate assemblages (definition >1 cm, practical definition – visible in photography) is also regularly done (e.g., Jones et al., 2012). Extreme care to avoid use of downward-facing thrusters is essential to prevent disturbance of the sediments which may obscure visibility for an extended period. Consideration of the survey design is required to balance data collection and time constraints. More transects (radial surveys at different headings) enable higher resolution mapping of habitats and disturbance, while replicating headings (with a small offset) enables greater statistical power in assessing impact on fauna. Addition of sediment sampling (**Figures 1E, 2B, 3D**) can validate visual inspection through measurement of parameters such as hydrocarbon concentration or sediment barium concentration, or by improving taxonomic resolution of faunal identifications.

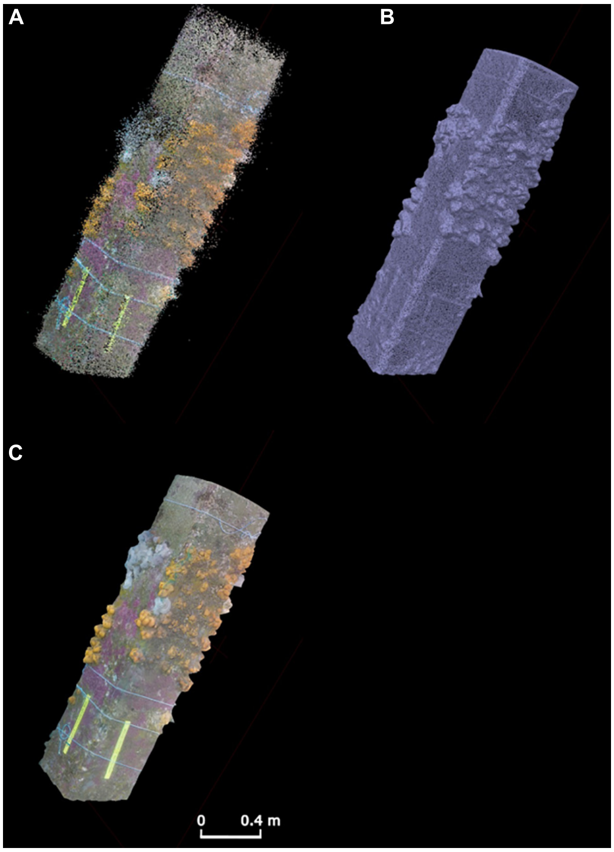

FIGURE 6 | The three stages of 3D model development underpinned by video/photographic imagery including **(A)** generation of point cloud **(B)** interpolation into mesh and **(C)** texturing with source images to produce a final model. Pictured here is a 2-m section of pier (pillar) with soft corals and anemones.

ROV Acoustic Surveys of Infrastructure and Surrounding Environments

Passive Acoustic Monitoring

Using ROVs to record sound around infrastructure will help industry understand their impacts in the marine environment. In an often dark, deep or turbid ocean, acoustic cues are vital for many life functions of marine fauna. The importance of sound has been shown in its recognition as an Essential Ocean Variable (EOV) by the Global Ocean Observing System (Goos BioEco, 2018). ROV recordings of sound around O&G structures provides information on soniferous (sound producing) species present, anthropogenic noise and how the latter impacts the former (Williams et al., 2015; Todd, 2016). Moreover, underwater noise pollution is a serious issue, and the introduction of further anthropogenic noise, such as that generated by O&G activities (during exploration, construction or operation), often requires permits and use of Marine Mammal Observers (MMOs) and PAM to determine potential effects on marine fauna (Todd et al., 2015, 2016). Such activities add significant cost and operational constraints on offshore projects and are increasingly important components of Environmental Impact Assessments (EIAs), whether for an individual activity or under shared sector and cross sector "noise budgets." Short-term PAM recordings could be conducted for the duration of an ROV survey, by mounting a recorder on the vehicle, while longer term recordings could be made by using the ROV to deploy a recorder at a designated site.

There is potential for the operational noise of industry ROVs and AUVs to mask signals of interest, particularly where recording systems are mounted on the vehicle. Few reports, however, characterize the noise from ROVs, AUVs or even human-occupied vehicles (HOVs) (Cai et al., 2010; Cai and Bingham, 2011; Stimperta et al., 2019). Communication noise between support crew and the subsea unit may also swamp recordings (Stimperta et al., 2019), while the dominant frequencies of electric vehicles appear to be from the electric motor in the tens to hundreds of Hz, with the only available reported source level (root mean squared) at 146 dB re 1 μPa (Cai et al., 2010; Cai and Bingham, 2011). Nevertheless, ROV-mounted PAM recorders have previously been used to detect various marine fauna (Ura et al., 2006; Mann et al., 2008; Rountree and Juanes, 2010; Rountree et al., 2018). The extent of signal masking by noise from industry ROVs, however, requires investigation. Once an instrument is deployed, a departed ROV would lessen or no longer affect the recording.

Long-term PAM, deployed by an ROV, provides near-continuous data on soniferous species around the infrastructure, revealing temporal patterns in their abundance, behavior, migration patterns, and responses to stressors (**Figure 4A** and **Table 1**). Taxa that can be investigated using PAM include fish (Erbe et al., 2015; McCauley and Cato, 2016; Parsons et al., 2016), mammals (Todd et al., 2009, 2016), and invertebrates (Radford et al., 2008; McWilliam et al., 2017). Characteristics of the soundscape (the combined natural and anthropogenic sounds in the environment) can help detect changes in ecosystem health and between certain types of habitats (Radford et al., 2010; Kaplan et al., 2015; Gordon et al., 2018).

Acoustic Telemetry

Tracking the movement and interactions of acoustically tagged marine fauna with subsea infrastructure will help industry understand how their structures may act as artificial reefs (**Table 1**). Recently, underwater acoustic telemetry tracking of animals has provided scientists with a panoramic view into the marine world and is now an important tool used worldwide in understanding animal movement (Hussey et al., 2015). Transmitters or tags with unique identification codes are either attached or implanted in target species such as fish, turtles and sharks (**Figure 4C**) and their high-frequency transmissions are detected when within range of acoustic receivers, which can be deployed on infrastructure using ROVs (**Figure 4B**). Acoustic telemetry can address questions on both fine and large-scale movement of marine fauna and has already shown potential around infrastructure and the surrounding water column (Heupel et al., 2015; Thums et al., 2016; Wilson et al., 2018). Telemetry tracking has been conducted directly from autonomous underwater vehicles (AUVs; White C.F. et al., 2016) and tracking fish and invertebrate movements around artificial structures and habitat has provided valuable information on how these habitats are used (Jensen et al., 2000; Abecasis et al., 2013). Gulf red snapper (*Lutjanus campechanus*), is one of the most economically important reef fish in the Gulf of Mexico (Curtis et al., 2015) and a successful example of how telemetry networks have integrated artificial structures such as O&G platforms, to better understand species movement, residency, and potentially benefit a major fishery (Topping and Szedlmayer, 2011; Piraino and Szedlmayer, 2014; Froehlich et al., 2019).

Acoustic telemetry currently appears to be the best approach to understanding megafauna interactions with offshore installations (**Figure 2E** and **Table 1**). On a broad scale, deploying acoustic receivers (**Figure 3B**) along continental coastlines will greatly extend telemetry networks around the world (Hussey et al., 2015; Abecasis et al., 2018; Hoenner et al., 2018), which is especially important in remote areas. In these areas, there is substantial potential for numerous O&G structures, such as platforms, wells and pipelines to become telemetry nodes and fill in the gaps between larger networks. Such an opportunistic grid would not only broaden the coverage of these acoustic telemetry networks, but provide valuable insights into how artificial structures are utilized individually or together, by resident and transient species, respectively.

Active Acoustics

Single- or split-beam echosounders (**Figure 3E** and **Table 1**) can be used to estimate fish biomass around the infrastructure and how distance from the structure influences the assemblage (Stanley and Wilson, 1996, 1998; Wilson et al., 2006). ROVs with mounted echosounders could perform short-term surveys of the water column around infrastructure (Baran et al., 2017; **Figure 2F**). The detection and quantification of internal waves using upward-looking echosounders (Li et al., 2009) provides valuable information for structural design and maintenance (Osborne et al., 1977), but also the transport and vertical distribution and density of plankton (Benoit-Bird et al., 2009), a fundamental food source in the ocean. While internal waves (gravity waves) are best detected by stationary systems, they can also be seen in data captured from mobile platforms (such as ROVs) and can be quantified once the motion of the platform has been accounted for.

Multibeam echosounders are sometimes used aboard ROVs to assist with ROV localization and positioning, mapping seafloor habitat, or checking maintenance issues (Petillot et al., 2002; Biffard et al., 2007; Vralstad et al., 2011). Multibeam can provide detailed 3D information on the seafloor, such as substrate type or sediment deposition and scouring around seafloor structures, and therefore assist in identifying potential fatigue points in structures or quantifying changes in habitat. Textural representations (2D images) of this type of data have been collected aboard ROVs in the past, using sector scanning or sidescan sonar (Orange et al., 2002; Elvander and Hawkes, 2012). Relatively new developments in multibeam echosounders have brought simple-to-use, forward-looking imaging sonars that can provide the same information in front of the ROV and expand that seafloor image as the ROV moves forward (Allotta et al., 2015; **Figure 2G**). Mapping the changing habitats around new, long-existing or decommissioned (removed) O&G structures can provide valuable information on how they enhance local habitat (e.g., defining the extent around the platform that is influenced by biota on the structure) and where the substrate changes to an environment more typical of far-field habitat.

Multibeam echosounders (**Figure 3F**) have also more recently been re-tasked to include water column data (Colbo et al., 2014). Whether positioned on a vessel or on an ROV in the water column, multibeam water column backscatter can be used to provide 3-dimensional maps of subsea structures (Williams et al., 2016), bubble plumes from gas seeps or leaks (Weber et al., 2012), or schools of plankton or individual fish (Parsons et al., 2013; Colbo et al., 2014; **Table 1**). Structure and marine growth data can be combined with photogrammetry techniques to provide picture quality visualizations and volumetric assessments of structure and marine growth, similar to those of **Figure 5**. Imaging sonars (**Figure 4D**) improve the resolution of the acoustic data (at the expense of range), providing near-picture quality images of targets in front of the sonar. Many of these imaging sonars have been simplified for non-technical use and can collect data autonomously. In ecological research, this has provided potential alternative or complementary data to optical techniques (Harvey et al., 2018), whether assessing the abundance, length distribution or behavior of fauna (Becker et al., 2013; Parsons et al., 2014; Lieber et al., 2015). This concept is transferrable to assessments of marine fauna around O&G infrastructure if mounted on or deployed by ROVs (Rose et al., 2005). ADCPs(**Figures 2H**, **4G**) are commonly used to collect oceanographic water column data and are, on occasion, deployed by ROVs for O&G purposes (Jones et al., 2006). Such oceanographic data can be of great use for scientific purposes whether providing complementary environmental data for biological studies, or detection of geophysical phenomena, such as internal waves (Scotti and Pineda, 2004), as well as biologically derived backscatter (Bozzano et al., 2014).

ROV Collections of Specimens and Samples

Industry ROVs with manipulator arms have been used successfully in the field to scrape marine growth, collect water, and sample sediment for eDNA analyses from around offshore installations and pipelines (MB, EH, unpublished data; **Figures 1C**, **2C**, **3C**). The most relevant applications of analyses from these samples to the offshore O&G sector is: (i) detection of potential alien invasive species (AIS), (ii) baseline monitoring of biota before and after infrastructure is commissioned and (iii) understanding what biota has recruited to infrastructure that is scheduled for decommissioning. Beyond industry applications, these samples also offer valuable ecological data on fishes, coral, sponges, plankton, which provide better scientific understanding of the organisms living in proximity to industry operations.

Sediment sampling push cores (sampling ∼0.002 m^2 of sediment; **Figures 1E**, **2B**, **3D**) and grabs (∼0.0625 m^2) can be used effectively by industry ROV (Jones, 2009) and such samples used to characterize sediment chemistry, physical properties and biology (quantitative samples of microbes to macrofauna;

eDNA samples). Samples are important for understanding the characteristics of the range of habitat types created by industry operations (areas impacted by cuttings, potential pollution, scours etc.) as well as natural seafloor conditions. The high temporal-spatial precision in ROV sampling can allow scientific questions (e.g., on the nature of disturbance) to be addressed in a way not possible with other approaches. Typically, replicated sediment samples are taken in each habitat or area of interest, or along a gradient of disturbance. As well as quantitative sediment sampling, it is also possible to obtain good-quality samples of objects of interest, which may be natural (e.g., megafaunal individuals, rocks; **Figures 3B,C**) or anthropogenic (e.g., garbage, cement, etc.).

ROV Surveys of Areas Undisturbed by O&G Activity

Some industry operations take place in areas undisturbed by industrial activity, or at least areas with low levels of disturbance, which could be considered representative of "natural" conditions. These baseline observations may be performed with industry ROVs during surveys of conditions prior to development (exploration drilling, pipelines, etc.), immediately prior to operations (e.g., drilling, platform construction) or during ROV operations distant from more disturbed areas. These surveys are often on the critical path of operations and tend to be focussed on quantitative assessment of seafloor conditions, rather than more time-consuming single point observations. These opportunities can also be used for deployment of equipment for longer term monitoring, such as fixed-point moorings. Results from industry surveys like this have already been important in describing new areas (Jones et al., 2013, 2014) prior to offshore platform placement and subsequent colonization (Todd et al., 2019), and in quantifying natural processes that are poorly known (e.g., carbon fluxes to the seafloor: Lebrato and Jones, 2009; Higgs et al., 2014).

USING HISTORICAL ROV IMAGERY FOR SCIENCE

The images and video already collected by industry ROVs represent one of the most substantial visual datasets available from our oceans. The global offshore energy industry holds a vast archive comprising millions of hours of underwater video and millions of still images, often collected from locations, habitats and depths rarely encountered by independent researchers (Macreadie et al., 2018). Although not collected for the purposes of scientific research, historical ROV imagery represents a rare and ready-made resource for ocean observation (Macreadie et al., 2018; Levin et al., 2019), with the potential to facilitate investigation of a broad range of ecological, biological, behavioral and oceanographic questions, once adequately screened for quality. Historical ROV imagery has already been used to characterize fauna communities that develop on O&G infrastructure (Pradella et al., 2014; Rouse et al., 2018; Thomson et al., 2018; McLean et al., 2019) and assess the commercial fishery value of pipelines and wells (McLean et al., 2017, 2018; Bond et al., 2018d). Some archival collections span decades and may even provide insights into the effects of longer-term environmental change on offshore ecosystems.

Despite the potential utility of historical ROV imagery, considerable quality control is required before it can be used reliably for scientific investigation. There are generally a small number of factors that limit the scientific value of ROV imagery; (1) the ROV being too far away (for marine growth) or too close (for fish) to the structure, (2) particles in the water column limiting visibility, (3) inconsistency in speed or position of the ROV that makes imagery jerky/blurry or limits field of view, and (4) poor video resolution. Further, ROVs are often equipped with multiple simultaneously filming cameras with each providing a slightly different view of a structure. In many instances, not all cameras are functional. To assist industry and scientists in the refinement of the amount of ROV imagery that is to be used for assessment of fish and marine growth habitats associated with infrastructure, we provide detail on a simple three-step process that can be used to select appropriate imagery (**Supplementary Material**). In summary, the steps are:

1. scoring the extent of usable imagery of a structure (0 = unusable, 1 = sections usable, 2 = most usable; with descriptions of the imagery within each category);
2. scoring data resolution that can be obtained from ROV imagery from 0 (unusable) through to 5 (excellent), with descriptions of the imagery within each category and examples of ecological metrics that could be collected for each; and
3. estimating the scientific value of historical ROV imagery. This step combines the scores from Step 1 and 2 into a matrix table that clearly identifies scientific value.

Until recently, most historical ROV imagery has been collected in standard definition. Even if video records are HD resolution, and scenes are well (evenly) lit, with appropriate color balance, images are often inherently low contrast (especially of more distant objects) due to the scattering of light between object and camera, and further degraded by back-scatter and veiling light from suspended particles of diverse sizes. As computing power has increased, however, options are emerging for video image enhancement (e.g., Tang et al., 2019), either processing individual frames, or using information from sequences of frames, to "defog" or otherwise contrast enhance video records. In the future it is likely that such methods will become standard in ROV video analytics.

DATA MANAGEMENT

Access to and the use of historical ROV imagery and metadata and undertaking new scientific ventures in partnership with the O&G industry inevitably requires strong data management protocols to be implemented and followed. Scientists often encounter a number of hurdles when attempting to access and use data provided by industry. Many companies restrict the use of external portable hard drives for transferring data to scientists external to their organization. Industry often also have software programs designed specifically (sometimes in-house) to view and access ROV metadata with these programs sometimes not available or provided to scientists. Further, industry utilize different ROV contractors, often with these contractors using different data storage and handling platforms. Ease of data management is therefore reliant ultimately on the willingness of industry to share data that are accessible and interpretable by scientists.

Data size can also pose challenges. For example, 1 h of uncompressed HD video at 1920 × 1080 resolution (24 Hz) has a file size of approx. 500 GB, compared to approx. 10 GB when filmed in standard definition (720 × 576 at 24 Hz). A single O&G operator in north-west Australia can collect up to 5,500 h of underwater imagery in a year equating to ∼2,750 terabytes of data (uncompressed). For passive acoustic datasets, size is dependent on recording schedule and sampling rate, but processed data from a single logger, recording a 50% duty cycle can exceed 1 TB within a month. Meanwhile, high-resolution data from multi-beam echosounder systems collecting water column data can exceed 1 TB within days, dependent on the "ping rate" employed. Due to the resolution and lower sampling volume of single-beam echosounders and ADCPs, these datasets can be an order of magnitude lower than those of multibeam systems. Datasets from acoustic telemetry receivers are significantly smaller still, but like those of PAM this is multiplied by the number of sensors deployed in any multi-sensor arrays. Clearly more storage will be needed to facilitate an industry-wide switch to HD and collection of additional scientific data such as that collected using PAM. Improvements and protocols surrounding the exchange of large data from operators to researchers, as well as data manipulation comprising the end-to-end workflow around the analyses of data all require IT systems capable of handling large datasets.

The costs of data management and ownership of the data are also very important considerations. Historically, scientific data were often regarded by scientists as their property. More recently ownership is recognized to lie with an employer. Regardless, strict data ownerships agreements would need to be established and understood at the onset of a project to avoid conflict. Data sharing agreements are also important for science-industry partnerships where any restrictions or sensitivities of data use are understood clearly by scientists. Industry must also recognize that scientists require the ability to publish research results in journals and to present them at conferences. In the future it will increasingly be a requirement of funding agencies that scientific data are not only properly managed and stored, but that data are available for others to verify scientific findings or to reuse for other purposes. These activities should be actively encouraged by industry with benefits including an improved understanding of the marine environment that is accessible to all. Superficially, this emerging perspective on data ownership poses a challenge for industry, either because industry has paid for its acquisition, or there is a commercial or intellectual property reason for confidentiality. In practice, however, industry is increasingly realizing the value in data sharing, even with competitors, and the value of sharing data, albeit at an appropriate time, or with caveats, with third parties including scientific researchers. For this sharing to work, industry will find the same data management challenges as academic researchers and will need to collaborate to find effective solutions.

Overall, a grand view of data management would envisage regional and/or international depositories of metadata from science related, industry ROV missions. Such a scheme would make at least metadata freely accessible to enable international, scientific collaborations on broad scale science questions. Data banks already exist for ocean monitoring on both regional and international levels [e.g., Australia's Integrated Marine Observing System (IMOS) and The Global Ocean Observing System (GOOS)] and their expertise and infrastructure could be easily utilized for this purpose.

FROM ROVs TO MARINE AUTONOMOUS SYSTEMS

Recent advances in marine autonomous systems (MAS) and sensor technology offer the prospect of substantial improvements in spatial resolution of offshore surveys. There may be efficiency gains over current practice, leading to potentially reduced ship time for equivalent data collection, resulting in potentially significant cost savings, reduced carbon emissions and improved safety (fewer personnel days at sea). Generally, less complex vessels are required for MAS operations, ensuing a larger range of available vessels and cheaper voyages. MAS are therefore used increasingly by industry to support some areas of the offshore O&G industry, such as geophysical data acquisition (Wynn et al., 2014). Other emerging applications in the oil industry include pipeline inspection (Fernandes et al., 2016), environmental data collection (Pai, 2015), environmental monitoring of decommissioned fields (Jones et al., 2019) and oil spill monitoring (White H. et al., 2016).

Uptake may be increasing, but limitations in key areas of MAS technology – fine scale navigation and battery life – currently prevent MAS completely replacing ROVs. Many of the scientific benefits of ROVs are also applicable to MAS. However, as autonomy is slowly introduced into routine operations, scientists must understand any differences in data collection, and how switching vehicles may impact comparisons to historical datasets using alternative methods. For example, typical ROV visual inspections of pipeline use multiple cameras to obtain a central, port and starboard view of the pipeline while an AUV usually uses a single, central camera. In this instance, the ability to see, identify and count fish that most often reside underneath or in the crevice between the seabed and pipeline (Bond et al., 2018a; McLean et al., 2018) is removed and the value of the survey for documenting fish populations significantly reduced. Potential limitations in data comparison should also be outlined to industry and rigorous method comparisons could be undertaken. Particular attention should be paid to biological data collected from visual observations. Vehicle speed, noise and vibration, and altitude as well as the number of cameras, their positions and quality, and lighting are all likely to change from ROV to MAS; each potentially impacting animal behavior in different, species-specific ways.

CONCLUSION

Here we describe a range of sensors, samplers and imaging equipment that can be attached to or used by ROV's during routine inspection, maintenance and control campaigns – to collect more robust, quantitative scientific data from the marine environment (**Figure 2** and **Table 1**). The feasibility and importance of augmentation and performing scientific survey work is highlighted in addition to the underutilized yet high valuable historical ROV data held by industry. The enhancements and surveys suggested, while often simple, need industry project managers, engineers and ROV operators to work in partnership with research scientists. Such collaborations improve understanding of needs, priorities and restrictions held by all parties. The projects must also have sufficient science personnel to analyze imagery, specimens and measurements. For video analysis, this can equate to significant hours however, machine learning tools to increase the speed of annotation are advancing. Where possible, industry should look ahead at future ROV schedules to best integrate science, ideally allocating time and budget to promote a successful mission. Pilot studies are particularly beneficial for forging industry-science partnerships, trialing ROV enhancements and determining science value.

For the offshore energy industry, the benefits of enhancing industry ROVs are clear: accurate data to underpin environmental impact/condition assessments and decommissioning comparative assessments, enhanced environmental stewardship and more informed engagement with regulators and the public. The benefits of enhancing ROVs for the scientific community are vast: data can be obtained from the most unexplored regions of our oceans leading to new discoveries, the impacts of infrastructure in our oceans can be assessed and understood, and, importantly, we may better understand how marine life in our oceans is changing. With this knowledge comes improved management and conservation strategies for the sustainability of ocean life.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

The research presented within the manuscript is original and was contributed to by each of the authors. All authors contributed to the writing and reviewing while DM, MP, AG, CP, DJ, SR, and EH each additionally contributed to visualization (graphics). All authors agreed with the contents of the manuscript and approved it for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00220/full#supplementary-material

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Underwater Visual Records of Marine Megafauna Around Offshore Anthropogenic Structures

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In oceans and seas worldwide, an increasing number of end-of-life anthropogenic offshore structures (e.g., platforms, pipelines, manifolds, windfarms, etc.) are facing full or partial removal. As part of the decommissioning process, studies on potential importance of subsea infrastructure to marine megafauna (defined as: cetaceans, pinnipeds, sirenians, large fish – such as sharks, rays, billfishes, and tuna, as well as marine reptiles, and seabirds) are lacking. Dedicated scientific Remotely Operated Vehicle (ROV) surveys around offshore installations are rare, but there is a wealth of archived industrial data and noteworthy species sightings posted publicly on various social media platforms. This study used routine, incidentally collected ROV (*n* = 73) and commercial diver (*n* = 9) video recordings spanning 1998–2019 globally. Data were gathered directly from industrial partners (*n* = 36) and the public domain (*YouTube*; *n* = 46) to provide an account of marine megafauna presence and potential feeding behavior in the near-visible vicinity of subsea anthropogenic structures. A total of 79 video clips and 3 still images of marine megafauna near offshore structures were examined, resulting in 67 individual sightings and 16 sub-sightings (in which an individual was recorded within the same day). At least 178 individuals were identified to a minimum of 17 species of marine megafauna, amounting to a total (combined) sighting duration of 01:09:35 (hh:mm:ss). Results demonstrated proximate presence of marine megafauna (many of which are threatened species) to anthropogenic structures, with most animals displaying foraging or interaction behaviors with the structures. Observations included the deepest (2,779 m) confirmed record of a sleeper shark (*Somniosus* spp.) and the first confirmed visual evidence of seals following pipelines. These ROV observations demonstrate a latent source of easily accessible information that can expand understanding of marine megafauna interactions with offshore anthropogenic infrastructure. Consequently, other workers in this field should be encouraged to re-analyze archived datasets, commence further collaborative research projects with industrial partners, and/or expand Internet search terms to additional species assemblages, in a bid to quantitatively elucidate relationships between offshore infrastructure and marine species.

Keywords: marine megafauna, Remotely Operated Vehicle (ROV), Oil & Gas (O&G), mammal, shark, platform, pipeline

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INTRODUCTION

Since the industrial revolution, many species have been struggling to cope with habitat modifications, climate change, and other detrimental anthropogenic pressures (e.g., Cheung et al., 2009; Dirzo et al., 2014; IPBES, 2019). Despite increasing interest in the renewables sector, enduring investment in Oil & Gas (O&G) Exploration and Production to satisfy global demand has resulted in the ongoing operation of thousands of offshore structures and the continued establishment of new infrastructure in the marine environment (e.g., Todd et al., 2019). Of these, 7,500 marine offshore O&G installations across jurisdictions of 53 countries are expected to become obsolete in the near future (Parente et al., 2006; Fowler et al., 2018). Windfarm decommissioning is also on the horizon, but as a comparably newer industry, is less impending (Topham and McMillan, 2017).

Subsea-anthropogenic infrastructure provides structurally complex hard substrata on an often otherwise featureless sedimentary seafloor (Larcom et al., 2014). These structures can accommodate diverse sessile-invertebrate communities comprising anemones, hydroids, bryozoans, sponges, mussels, barnacles, soft, and even hard corals that can attract motile invertebrates (Guerin, 2009; Langhamer and Wilhelmsson, 2009; McLean et al., 2017; Meyer et al., 2018; Todd et al., 2018, 2019). With time, subsea structures encourage benthic complexity, which supports fish populations through a strong foraging base (Cowan and Rose, 2016), increasing trophic-level ecological succession. Consequently, anthropogenic infrastructure has large-scale positive and negative social, economic, and ecological impacts. These structures sustain trophic connectivity through nutrient and resource flow, pelagic-larval dispersal, pipeline linkage, and movement of mobile predators (Macreadie et al., 2011; Henry et al., 2018; van der Molen et al., 2018).

A total of 14.7% of the North Sea has been designated as Marine Protected Areas, MPAs (OSPAR, 2017), and an additional 0.1% comprises the 500-m shipping (including fishing) exclusion zones of offshore O&G structures (UK Public General Acts, 1987; UK National Data Repository, 2019). This safety exclusion zone is reduced to 50 m around windfarm turbine bases, and only increases to 500 m around renewable construction sites (FLOWW, 2014). Some offshore installations could potentially be acting as mini MPAs, which has galvanized a number of both academic and industry scientists to jointly investigate possibility that there may be positive environmental outcomes of industrial usage of the marine environment. Consequently, this has sparked debate about whether to leave some, well-placed offshore structures, particularly O&G platforms, *in situ* as artificial Rigs-to-Reefs, RTR (Kaiser and Pulsipher, 2005) or remove them completely from the seabed. Removal of infrastructure at the end of its field life is legislated currently in Europe under the Oslo and Paris Convention (OSPAR) Decision 98/3, on the Disposal of Disused Offshore Installations (OSPAR, 1998).

In the North Sea, there is likely a high level of ecological interconnectedness between the network of local ecosystems comprising 1,350 offshore O&G installations, including 545 fixed steel platforms, 81 offshore windfarms with 3,589 turbines (North SEE, 2016), and a plethora of wrecks (Coolen and Jak, 2018). There is no doubt that increasing numbers of artificial structures in the marine environment – adversely termed 'ocean sprawl' by Bishop et al. (2017) – modifies the local marine ecosystem. In a similar vein, commercial fishing (especially dredging and trawling) has also altered large tranches of the ocean seabed beyond recognition. Re-conversion to the original ecosystem is unlikely to be achievable in the long term, as the North Sea continues to be exploited by industrial stakeholders.

The RTR concept began as early as 1975, when in the South China Sea (Malaysia) the storm-damaged *Baram-8* platform was toppled to form the base of an artificial reef (Zawawi et al., 2012). Since then, RTR schemes have been legislated only in Brunei's sector of the South China Sea (Twomey, 2012), and in the United States' (US) Gulf of Mexico (GoM) in Louisiana and Texas, and in Pacific California (Jørgensen, 2009). Successful RTR implementation, however, has occurred only in Brunei and the GoM, despite increasing support from both academic and industry scientists, for the concept of leaving well-studied and potentially environmentally beneficial offshore structures *in situ* at the end of operational lifetimes (Jørgensen, 2012; Macreadie et al., 2012; McLean et al., 2017; Fowler et al., 2018; Todd et al., 2018, 2019; van Elden et al., 2019).

To determine environmental impacts of future decommissioning projects on a case-by-case basis, it is important to consider potential interactions between prey species congregating around structures with organisms at higher trophic levels. For example, because of the general tendency of any object placed in the water to attract fish and other marine life, marine megafauna, such as mammals, large fish (including sharks), reptiles and seabirds, might be attracted to these locations for temporo-spatially predictable foraging opportunities (McClellan et al., 2014; McLean et al., 2019b).

To date, only a handful of studies have been conducted on marine mammal interactions with offshore anthropogenic infrastructure, mostly in the North Sea. The first acoustic study was carried out in the German North Sea by Todd et al. (2009) using analog autonomous underwater passive cetacean echolocation-click detectors (T-PODs) deployed directly from an offshore O&G exploration jack-up drilling rig at an established production platform. Harbor porpoises (*Phocoena phocoena*) were detected, predominantly at night, in very close proximity (<300 m) to installations with a pronounced diel pattern (Todd et al., 2009). Another North and Irish Sea acoustic and visual study on various offshore O&G installations, reported multiple marine mammal species, including harbor porpoise, Atlantic white-sided (*Lagenorhynchus acutus*), white-beaked (*Lagenorhynchus albirostris*), and common (*Delphinus delphis*) dolphin, minke whale (*Balaenoptera acutorostrata*), and both common (*Phoca vitulina*) and gray (*Halichoerus grypus*) seal (Todd et al., 2016). Similarly, Delefosse et al. (2018) analyzed visual sightings from 25 O&G installations in the Dutch sector of the North Sea, recording 131 sightings of 288 individual marine mammals over a 3-year period. Harbor porpoise was the most commonly sighted species (41% of sightings), followed by minke whale (31%). Regarding offshore renewables, Russell et al. (2014) observed tagged common and gray seal targeting windfarm turbine piles, presumably to feed. In other oceans worldwide, a visual study in the Adriatic sea reported bottlenose dolphin (*Tursiops truncatus*) within 750 m of O&G platforms (Triossi et al., 2013), and in the Pacific USA, California sea lion (*Zalophus californianus*) were found to haul out and nurse young regularly on O&G installations (Orr et al., 2017). In Australia, Arnould et al. (2015) reported that 25% of 36 tagged Australian fur seals (*Arctocephalus pusillus*) exhibited foraging behavior near offshore anthropogenic structures, potentially targeting areas around O&G pipelines and cable routes.

Little information is available on associations of elasmobranchs (cartilaginous fish, such as sharks) with anthropogenic structures. In the North Sea, a basking shark has been sighted within the 500-m zone of an offshore installation (Todd et al., 2016). Whale sharks (*Rhincodon typus*) aggregate in high densities around offshore Arabian Gulf O&G platforms (Robinson et al., 2013). Four whale sharks were observed at a platform in Australia (McLean et al., 2019a). ROV imagery of wells in Australia has also led to observations of a gray nurse shark (*Carcharias taurus*), spinner shark (*Carcharhinus brevipinna*), ribbontail ray (*Taeniura meyeni*), and speckled swellsharks, *Cephaloscyllium speccum* (McLean et al., 2018b). Most investigations have focussed on impacts of Electro-Magnetic Fields (EMFs), since sharks can sense very weak, bioelectric fields to detect prey (Kalmijn, 1982; Walker et al., 1992; Kajiura and Holland, 2002; Kajiura, 2003), predators (Sisneros et al., 1998; Kempster et al., 2013), mates (Tricas et al., 1995; Sisneros and Tricas, 2002), and potentially navigate on large global scales (Klimley, 1993; Klimley et al., 2002; Meyer et al., 2005). Consequently, interactions with anthropogenic structures are likely commonplace, but underreported.

Studies of large bony fish species that can grow to 2 m or bigger near offshore O&G platforms are rare. Martin and Lowe (2010) reported 53 fish species near O&G platforms off coastal California, United States, including two sightings of the ocean sunfish (*Mola mola*), the largest bony fish in the world (Wood, 1983; Pan et al., 2016). Most studies have focused predominantly on commercially important fish observed in association with subsea infrastructure (Jørgensen et al., 2002; Løkkeborg et al., 2002; Soldal et al., 2002; Guerin, 2009; Friedlander et al., 2014; Fujii et al., 2014). Indeed, many of the fish in these studies were juveniles, preferentially selecting structurally complex habitats (Sayer et al., 2005).

Impacts of O&G operations on marine reptiles (e.g., marine turtles and sea snakes) are unknown (Maxwell et al., 2019). A review of underwater sightings in the GoM found that marine turtles were observed visiting anthropogenic structures for short periods (1–5 h), particularly at night (Rosman et al., 1987). Marine turtles have also been found to associate more frequently with smaller, unmanned platforms that are closer to shore (Lohoefener et al., 1990). A recent Australian ROV study observed a flatback turtle (*Natator depressus*) and an unidentified sea snake (Hydrophiidae) immediately adjacent to a pipeline (McLean et al., 2019b).

Acoustic and surface-based detections have been analyzed previously (as reviewed here); however, there are few reports of megafauna from underwater visual sources. The objective of this study was to determine feasibility of using industry-supplied or publicly sourced visual data to assess presence and behavior of marine megafauna in close vicinity to offshore anthropogenic structures. These data were collected during routine offshore O&G and renewable energy infrastructure ROV General Visual Inspection (GVI) and commercial-diver Inspection, Maintenance and Repair (IMR) surveys. These industrial data are beginning to be used more regularly for assessments of fish and benthic species (e.g., McLean et al., 2017, 2020), but use in investigations of megafauna presence is understudied.

MATERIALS AND METHODS

Data Acquisition

Industrial recordings of marine megafauna sightings were sourced from routine, non-scientific inspection and maintenance operations of subsea offshore infrastructure using inspection-class ROVs and commercial diver video recording systems of unknown specifications. A mass data request email was sent to the contact email address (e.g., info@) of >50 companies in these sectors, outlining purpose of the research, a re-assurance of data confidentiality, and an invitation to submit marine megafauna imagery of interest. Marine megafauna were defined as: cetaceans, pinnipeds, sirenians, large fish (species known to exceed 2 m), such as sharks, rays, billfishes and tuna, as well as marine reptiles, such as marine turtles, sea snakes, and, seabirds (adapted from McClellan et al., 2014). Imagery was either provided anonymously by industry (e.g., ROV pilots) or obtained through dedicated Internet searches of public domain *YouTube*[1] online archives. Use of data from *Twitter* and *Facebook* was investigated, but *YouTube* was found to be the richest source of data for these purposes, and was therefore the focus.

Analysis was performed between the months of February to October 2019, applying a similar methodological approach used for medical reviews (e.g., by Ache and Wallace, 2008; Duncan et al., 2013). Search strategy consisted of entering a combination of marine megafauna and anthropogenic structure search terms (**Table 1**), followed by either '*ROV*' or '*commercial diver*' into *YouTube* using *Google Chrome* (version 75.0.3770.142) or *Firefox* (v68.0.1). Anthropogenic terms were derived from the O&G (Schlumberger, 2019) and Wind energy glossaries (Nordman, 2010; Wind Energy Association, 2020). These term combinations were used to search iteratively, e.g., '*Aquatic mammal blowout preventer ROV,*' followed by '*Aquatic mammal blowout preventer commercial diver,*' then *'big fish blowout preventer ROV,'* and so on. Since there is considerable variation across industry in use/non-use of spacing between terms (e.g., spudcan vs. spud can, windfarm vs. wind farm, etc.), both were used where applicable. Moreover, since the O&G industry (and to a lesser extent, the renewable sector) rely predominantly on acronyms, these were also applied, where appropriate. Each listed result was reviewed individually until no search terms remained in the video title. Only videos originating from an ROV camera or a commercial

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[1] www.YouTube.com

dive recording were considered. The online video converter 'YouTube Converter & Downloader' (v3.0) was used to download and convert imagery from YouTube to Moving Pictures Expert Group (MPEG) – 4 Part 14 (.mp4) format.

Imagery Processing

Each video was quality assessed and analyzed using *Windows Media Player 12* (Microsoft) or *VLC media player* (VideoLAN, v3.0.6). Imagery was first screened for visible presence of marine megafauna and a permanent anthropogenic structure. Videos lacking clear images, as a result of very low quality, or non-commercial imagery (e.g., recreational dive recordings), were excluded from further analysis. All available metadata (e.g., time, date, depth, coordinates, operation type, project, equipment, and structure type) were recorded. Stills of imagery were taken, with permission from industry providers, using the built-in *VLC media player* snapshot tool or *Windows Snipping tool.* Permission to present stills from YouTube videos was challenging; consequently, only Uniform Resource Locator (URL) hyperlinks to footage have been presented. Authors are not accountable for URL content and have endeavored to provide correct links; however, uploaded videos are occasionally removed or may be country-restrictive.

Audible ROV pilot/commercial diver commentaries, and annotations found in video descriptions (e.g., location) were recorded if they augmented video metadata. Geographic coordinates, including Universal Transverse Mercator (UTM) zone – or at least an approximate location – were applied to species identification (ID) analysis. The sighting map was created in *QGIS v3.2.3* using WGS '84. Diel variations were not accounted for, due to varying conditions (artificial lights, depth, time zone, lack of metadata). Time-stamp format (local or in Greenwich Mean Time Zone, GMT) was not available, and hence assumed to be in local time.

TABLE 1 | Internet search terms used in every possible permutation, followed by either the term '*ROV*' or '*commercial diver*' to source video imagery related to marine megafauna sightings from the public domain (listed alphabetically).

Marine megafauna terms	Anthropogenic structure terms	
Aquatic mammal	Blowout preventer	Pipeline
Big fish	BOP	Platform
Cetacean	Cable	Production
Dolphin	Caissons	Renewable
Dugong	Conductor	Rig
Elasmobranch	Construction	Spudcan
Manatee	Drilling	Structure
Marine mammal	Foundation	Subsea
Megafauna	Gas	Tower
Pinniped	Industry	Tripod
Porpoise	Inspection	Turbine
Ray	Installation	Umbilical
Reptile	Jacket	Underwater
Seabird	Jack-up	Valve
Seal	Leg	Welding
Sea lion	Manifold	Well
Sea monster	Maritime	Wind
Sea otter	Monopile	Windfarm
Sea snake	O&G	Windpark
Shark	Offshore	
Sirenia	Oil	
Turtle	Operation	
Walrus	Pile	
Whale	Piling	

BOP, BlowOut Preventer; O&G, Oil & Gas.

Several videos were available only as truncated segments of what would have clearly been a full-length sighting. Consequently, imagery that was obviously recorded on the same day of an operation that contained multiple sightings of the same individual from the same installation (i.e., re-identified due to unique recognizable markings, such as white pigmentation on a seal) were combined as '*sub-sightings*' of the same '*sighting.*' Video clips taken by multiple cameras on the same ROV (duplicates), but from a different angle within a 5-s period, were noted but excluded from further analysis to avoid replication. All commercially sensitive information (such as client, vessel, project, ROV operator, etc.) sourced from videos submitted by clients under Non-Disclosure Agreement (NDA) were redacted (**Figure 1**).

Megafauna were categorized into groups (e.g., *whale, dolphin, seal, shark, ray, fish, turtle, bird,* etc.) and identified to lowest taxonomic rank using appropriate identification guides (e.g., Hvass, 1978; Compagno et al., 2004; Shirihai and Jarrett, 2006; Jefferson et al., 2015; Nelson et al., 2016). Identifying the sex of an individual was not considered a classifiable criterion, because of a high margin of error. Threat status of all identified species was sourced from the International Union for Conservation

FIGURE 1 | ROV 'still' image of a bottlenose dolphin approaching a cable-laying operation. Highlighted sections (red boxes) indicate additional information of interest for data analysis (e.g., time, date, type of operation, location and depth in m). Confidential client data (black boxes) were redacted for Non-Disclosure Agreement (NDA) purposes.

of Nature (IUCN) Red List of Threatened Species[2]. Additional species considered to be interacting with megafauna (e.g., prey) were also recorded and identified, where possible, as was sighting duration (time in which individual was visible on-screen). Quantification of size of individual or distance to anthropogenic structure at closest approach was not possible and was reliant on camera Field of View (FOV). Any fish species that is known to grow to 2 m or more was included in analysis; however, only a qualitative estimation of size (big vs. small) can be made, as morphometrics/scarring/colouration and even sex of individuals, especially fish, change visibly with age. For example, most groupers (Epinephelidae) are protogynous hermaphrodites, which means that as they age, body proportions change in appearance, and notable color fading/scarring can be visible (Zhou and Gui, 2010; Chen et al., 2019).

Imagery was grouped by distance of approach between animal and structure. Categories included 'direct' physical contact with structure, 'approaching' structure to within 1–2 m (based on FOV and normal size of adult animals), and 'no approach,' e.g., swimming past structure in the distance. Imagery duration (effort) and sighting duration were recorded in minutes:seconds (mm:ss).

Categorization of behavioral activity was somewhat difficult to quantify for such a wide range of species; however, for certain megafauna, such as seals, and perhaps some sharks, an attempt was made to classify behavior into five categories. These were based on predominantly observed behavior, adapted from Wauters et al. (1992): *searching*, *foraging*, *traveling*, *interaction*, or *other*. *Searching* was defined as head moving up and down and/or from side to side in what was perceived as a method to identify prey. *Foraging* attempts were defined as individuals deviating from travel path sharply to attempt to and/or catch and feed on prey. *Traveling* was assigned when an individual swam in a consistent direction with no deviation associated with *foraging* or *searching*. *Interaction* was defined as individuals that were affected (i.e., change in behavior) through contact with anthropogenic equipment, structure, or diver. Behaviors were classified as *other* when they did not fall into any previous categories, e.g., no movement.

RESULTS

Recording Source

A total of $n = 79$ video clips and $n = 3$ still images were analyzed (total $n = 82$) with a total effort duration of 1:46:30 hh:mm:ss. The info@ call for new industrial footage elicited zero response from industry; consequently, most videos ($n = 46$) were sourced from the public domain (*YouTube*), and $n = 36$ were from archived or obtained directly from colleagues working in the O&G industry. Imagery was recorded mainly by ROVs of unknown specification ($n = 73$), and a handful ($n = 9$) recorded by commercial dive cameras of unknown specification. Some video clips ($n = 5$) were collected from more than one date (e.g., a compilation of several clips). Of videos sourced from industry, $n = 8$ duplicates were excluded from further analysis, resulting in a total of $n = 74$ useable recordings.

Data were sourced across 21 years from May 1998 to July 2019, originating from 62 separate days. Most imagery was recorded in 2018 ($n = 17$ recordings, all in August), followed by 2006 ($n = 10$), and 2009 ($n = 9$). Imagery was collected throughout the year, with a seasonal trend toward August ($n = 23$), followed by September and October ($n = 5$) in the northern hemisphere ($n = 54$). In the southern hemisphere ($n = 6$), more data were collected in October ($n = 3$). Remaining recordings without location data ($n = 22$) were recorded equally throughout the year.

Marine Megafauna Sightings

From the $n = 74$ useable recordings, $n = 67$ individual marine megafauna sightings were identified (**Table 2** and **Supplementary Table 1**) totalling 01:09:35 hh:mm:ss. Each sighting ranged from 1 s to 4 min 29 s with a mean ± standard deviation (SD) of 00:50 ± 01:02. Sightings were classified in $n = 7$ marine megafauna groups: whale ($n = 3$), dolphin ($n = 1$), seal ($n = 16$), shark ($n = 29$), ray ($n = 4$), fish ($n = 11$), and turtle ($n = 3$; **Table 2** and **Figure 2**). No birds or other marine reptiles were identified in any imagery.

Of these $n = 67$ sightings, $n = 178$ individuals were recorded, including a school of approximately 100 requiem sharks (*Carcharhinus* sp.) and $n = 1$ unidentified shark egg case. Most taxa ($n = 17$) were identified to species level, while $n = 3$ recordings could be identified only to genus level. The remaining $n = 9$ individuals could be identified only to family level or above, mainly due to lack of location metadata. Sharks ($n = 134$), fish ($n = 17$), and seals ($n = 16$) comprised the majority of individuals. The most sighted species was gray seal ($n = 11$), followed by porbeagle shark (*Lamna nasus,* $n = 10$), and bluntnose sixgill shark (*Hexanchus griseus,* $n = 6$).

One gray seal, identified due to a distinct white mark on its lower back (**Figure 3**) was re-sighted 17 times across three days (28/08/2018, 30/08/2018, and 31/08/2018), resulting in $n = 14$ sub-sightings. Additionally, $n = 2$ still images of a bottlenose dolphin were classified as sub-sightings, since they were recorded within a 2-min period of the same operation. In total, these additional $n = 16$ sub-sighting data were only included in sighting duration and behavioral analysis (total $n = 83$).

Location metadata or an approximate region were supplied for $n = 44$ sightings. These originated from six main geographical regions (**Figure 4**), the majority of which were recorded in the North-East Atlantic ($n = 24$). These included locations west of Ireland ($n = 7$), the North Sea, within maritime borders of England, Scotland, Denmark, and Norway ($n = 15$), and the Irish sea ($n = 1$, see UK inset in **Figure 4**). Further recordings originated from the GoM ($n = 9$), Australia ($n = 4$), south-east Asia ($n = 3$), Brazil ($n = 2$), and the Persian Gulf ($n = 2$).

Recording depths ranged from 13.8 to 3,000 m (**Table 2**). The deepest three recordings were of sleeper sharks at 2,650 m, 2,779 m and an approximate depth of 3,000 m. The shallowest recordings (13.8–16.4 m) were of common seals, a porbeagle shark, round ribbontail ray, and a hawksbill turtle. The most variable depth-range was observed for gray seals over a vertical

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[2] www.iucnredlist.org

TABLE 2 | Marine megafauna sightings ($n = 67$) compiled into groups, and then identified alphabetically to lowest taxonomic rank, as per Shirihai and Jarrett (2006) and Nelson et al. (2016), including number of animals, sighting duration, location, depth, IUCN Red List threat status for identified species (https://www.iucnredlist.org/), and YouTube URL hyperlinks (blank cells representing footage sourced solely through industry).

Marine megafauna group	Scientific name	Common name/type	Sightings/ Animals (n)	Location	Sighting depth (m)	IUCN threat status	YouTube URLs
Whale	*Physeter macrocephalus*[‡]	Sperm whale	3/3	GoM	900	Vulnerable	https://www.youtube.com/watch?v=CHdJIlxpTnU https://www.youtube.com/watch?v=4S6zzd9H5D4 https://www.youtube.com/watch?v=pb-hujQs2dY
Dolphin	*Tursiops truncatus*	Bottlenose dolphin	1*/1-3	North Sea	26	Least concern	
Seal	*Halichoerus grypus*	Gray seal	11**/10	North-East Atlantic	71–353	Least concern	https://www.youtube.com/watch?v=bVWooelhhvY https://www.youtube.com/watch?v=G4d1Nd9VBYE https://www.youtube.com/watch?v=41zUzbOvAxk https://www.youtube.com/watch?v=f6I6MDZM9Pc https://www.youtube.com/watch?v=tuKQ7P9Kwig
	Phoca vitulina	Common seal	4/4	North-East Atlantic	14–135	Least concern	https://www.youtube.com/watch?v=jH3ce4xVpYI https://www.youtube.com/watch?v=F9kxs-Z62hc https://www.youtube.com/watch?v=ngP0Rlh2EDM
	Otariidae[‡]	Sea lion	1/1		50		https://www.youtube.com/watch?v=4cpBFG6SBks
Shark	*Carcharhinus melanopterus*	Blacktip reef shark	1/1	Persian Gulf	45	Near threatened	
	Carcharhinus sp.	Requiem shark	1/100	Western Australia	28		https://www.youtube.com/watch?v=LamD8ezGuXg
	Carcharias taurus[‡]	Gray nurse shark	2/2	GoM	91	Vulnerable	https://www.youtube.com/watch?v=rrD57i8qTqo https://www.youtube.com/watch?v=Qbf0zh2HK_k
	Cetorhinus maximus	Basking shark	2/3	North Sea	30–335	Vulnerable	https://www.youtube.com/watch?v=UrTzfKJUKyE&t=0s&list=PLEFA6A4C2B63D71FC&index=3
	Chondrichthyes[‡]	Shark egg case	1/1				https://www.youtube.com/watch?v=wsgTxgw9h8I
	Hexanchus griseus	Bluntnose sixgill shark	6/6	GoM & East of Brazil	518	Near threatened	https://www.youtube.com/watch?v=NWI9pp7oNuQ https://www.youtube.com/watch?v=1ex7uTQf4bQ https://www.youtube.com/watch?v=N137nlx5fss https://www.youtube.com/watch?v=R_2zUC23jss https://www.youtube.com/watch?v=8c6EQvPwQy8 https://www.youtube.com/watch?v=eux4oDo-vGU
	Isurus paucus[‡]	Longfin mako shark	1/1		40	Endangered	https://www.youtube.com/watch?v=dar3ELxBkb0
	Lamna nasus	Porbeagle shark	6/6-9	North Sea	15–70	Vulnerable	https://www.youtube.com/watch?v=mvltmqWZ36c https://www.youtube.com/watch?v=vfQcKztCj4c https://www.youtube.com/watch?v=ugcNC719v3A https://www.youtube.com/watch?v=AxAtz_2BtHs https://www.youtube.com/watch?v=CDLlfifasi4 https://www.youtube.com/watch?v=BpEyOqFUheY
	Rhincodon typus	Whale shark	5/6	Gulf of Thailand, South China Sea, and Western Australia	140	Endangered	https://www.youtube.com/watch?v=RsbvJviW1rs https://www.youtube.com/watch?v=4S6zzd9H5D4 https://www.youtube.com/watch?v=sEirbV7VaXE https://www.youtube.com/watch?v=4S6zzd9H5D6 https://www.youtube.com/watch?v=XF3zUGnKzS0

(Continued)

TABLE 2 | Continued

Marine megafauna group	Scientific name	Common name/type	Sightings/ Animals (*n*)	Location	Sighting depth (m)	IUCN threat status	YouTube URLs
	Somniosus sp.	Sleeper shark	3/3	GoM & East of Brazil	2,650–3,000[†]		https://www.youtube.com/watch?v=B-whabKr_r0 https://www.youtube.com/watch?v=P__-YVYdNmE https://www.youtube.com/watch?v=kPS9Vifm3rk
	Sphyrnidae[‡]	Hammerhead shark	1/1		512		https://www.youtube.com/watch?v=N137nlx5fss
Ray	Mobulidae[‡]	Mobula ray	1/1		22		https://www.youtube.com/watch?v=ItbuUoOYsv4
	Myliobatoidei[‡]	Stingray	1/1		44		https://www.youtube.com/watch?v=xJHZ1exaz5o
	Rhinobatidae[‡]	Guitar fish	1/1		150		https://www.youtube.com/watch?v=xJHZ1exaz5o
	Taeniurops meyeni	Round ribbontail ray	1/1	Persian Gulf	16	Vulnerable	
Fish	Muraenidae [‡]	Moray eel	2/2	GoM	150–1,500[†]		https://www.youtube.com/watch?v=xJHZ1exaz5o https://www.youtube.com/watch?v=eux4oDo-vGU
	Serranidae[‡]	Grouper	3/6		77		https://www.youtube.com/watch?v=xJHZ1exaz5o https://www.youtube.com/watch?v=jIGH7nivlEM https://www.youtube.com/watch?v=Qbf0zh2HK_k
	Mola mola	Ocean sunfish	1/1	Western Australia	125[†]	Vulnerable	https://www.youtube.com/watch?v=DTY240NHF0U
	Molidae	Sunfish	1/4	West of Ireland	27		
	Thunnus obesus	Bigeye tuna	1/1	GoM	1,070	Vulnerable	https://www.youtube.com/watch?v=yT1EL9kpel4
	Xiphias gladius[‡]	Broadbill swordfish	3/3	Western Australia	119–730	Least concern	https://www.youtube.com/watch?v=N137nlx5fss https://www.youtube.com/watch?v=Bn22jvxfHOE https://www.youtube.com/watch?v=FEGeSWoB8Bg
Turtle	*Eretmochelys imbricata*	Hawksbill turtle	2/2	Gulf of Thailand	16	Critically endangered	https://www.youtube.com/watch?v=o0ZnrV0VWuY https://www.youtube.com/watch?v=UyK_9rsd-p4
	Lepidochelys olivacea[‡]	Olive ridley turtle	1/1			Vulnerable	https://www.youtube.com/watch?v=ISvMEPAxy0k

Records marked with symbols have been divided into further sub-sightings: *n = 3, **n = 24. [†]Estimated depth by ROV surveyor who uploaded video. [‡]Included sightings without geographical location (blank cells representing missing information). GoM, Gulf of Mexico. Authors of this paper have endeavored to provide correct links; however, uploaded videos are occasionally removed, and some links may be country-restrictive.

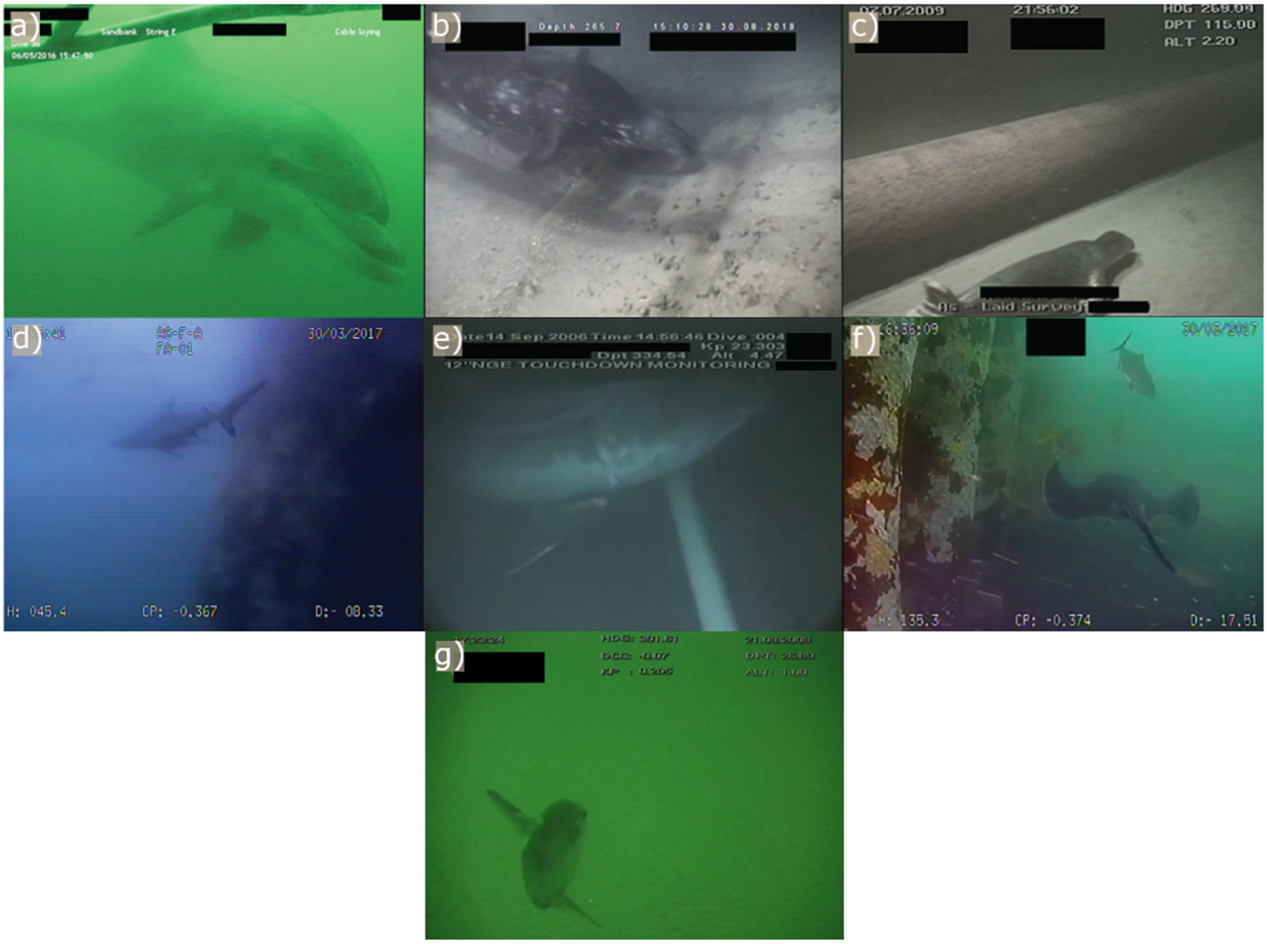

FIGURE 2 | Examples of still images taken from video imagery sourced directly from industry partners showing marine megafauna species close to anthropogenic structures: **(a)** bottlenose dolphin (*Tursiops truncatus*), **(b)** gray seal (*Halichoerus grypus*), **(c)** common seal (*Phoca vitulina*); **(d)** blacktip reef shark (*Carcharhinus melanopterus*), **(e)** basking shark (*Cetorhinus maximus*), **(f)** round ribbontail ray (*Taeniurops meyeni*), and **(g)** sunfish (Molidae).

depth of 282 m from 71 to 353 m. No depth information was available in *n* = 20 sightings.

Seven types of anthropogenic O&G structures were identified visibly, or through surveyor comments (**Table 3**). No data were available from renewable infrastructure. Most sightings were taken during installation or maintenance operations of platforms, followed by pipelines and umbilicals. Megafauna were observed mostly 'approaching' structure (*n* = 29; **Figure 2**), while direct physical contact was recorded on *n* = 20 occasions, and 'no-contact' was observed at *n* = 18 sightings. On one occasion, whilst crossing over a pipeline, a seal touched the surface, and disturbed the sediment with its flipper. Most commonly observed behavior was *interaction* followed by *searching*, and *traveling*, while *foraging* and *other* were observed least (**Figure 5**). Food-related behaviors (*searching* and *foraging*; **Figure 6**) were observed for seals, sharks and fish, while an *interaction* between organism and structure/diver was observed in all groups except ray, and *traveling* was recorded for whale, seal, shark, ray and fish.

A 3-s gray seal foraging attempt was recorded near a concrete pipeline in which the seal chased an unidentified fish, which sought refuge under the pipeline (**Figure 6d**). Foraging success could not be determined, as camera FOV did not record the outcome. A common seal was also filmed following a pipeline during an inspection in the Irish Sea (**Figure 2c**). While prey were present in close proximity to both the seal and the pipeline, foraging interactions were not observed. Basking sharks were recorded swimming along a pipeline (<1 m from the structure, two clips totaling 39 s), and a turtle was recorded swimming past a diver and platform, and came into direct physical contact with both (duration 03:02).

DISCUSSION

This review evaluated presence and potential behavior of marine megafauna around offshore anthropogenic structures through video sub-surface sightings from opportunistic industry-supplied or openly-available ROV and commercial-diver data collected during GVI or IMR operations. In total, 79 video clips and three still images were collected resulting in 67

FIGURE 3 | Still image of gray seal (*Halichoerus grypus*) identified due to white mark on lower back, as a re-occurring individual in subsequent sub-sightings.

sightings of 178 individuals from at least 17 species, 3 genera, and 9 family (or taxonomically above) groups of marine megafauna. This work demonstrated occurrence and potential foraging of marine megafauna in vicinity of anthropogenic structures on a global scale, and the first confirmed visual verification of seals following pipelines. Due to the nature of opportunistically-collected data, only qualitative descriptions were possible. Statistical comparisons would have only been achievable using dedicated-survey data. While data presented here are a 'snapshot,' given the thousands of hours of ROV data taken each month, it is conceivable that these observations could magnify quickly into full global data sets within their own right. Consequently, we urge other workers to follow suit, in a bid to quantify the frequency and nature of these events.

Over the last 30+ years, ROVs have become a dominant tool for subsea operations in the O&G sector; however, no imagery was sourced from the renewable energy sector. This could be for a number of reasons. Firstly, the renewable industry is newer, and there has been less time for ROV data to have accumulated and reach the various social medial platforms. Secondly, ROV surveys in this sector are likely less common, potentially because renewable energy structures are located in more exacting site conditions (shallow waters, high wind/wave/swell/currents, etc.), making standardized ROV surveys unfeasible operationally.

Structure Association

Previous acoustic and surface-based visual studies that have detected marine mammals around O&G structures (e.g., Cremer et al., 2009; Todd et al., 2009, 2016; Thompson et al., 2010; Delefosse et al., 2018), have now been corroborated by the commercial subsea imagery in this study. Presence of other megafauna species such as sharks (Haugen and Papastamatiou, 2019), rays, and turtles (McLean et al., 2017, 2019b) confirm that marine mammals are not unique in their exploitation of resources on/around anthropogenic structures. Individuals approached structures in close proximity, occasionally resulting in direct contact, which is not a behavior associated typically with avoidance (Mackintosh, 1974). Visual evidence of foraging behavior and successful prey capture by seals, sharks and large fish in vicinity of submerged pipelines, umbilicals, and platforms support previous evidence from Russell et al. (2014); Arnould et al. (2015), and Todd et al. (2016) that marine megafauna may target anthropogenic structures as foraging locations. Some of the searching behavior observed in the imagery may well have been simple 'curiosity.' Arnould et al. (2015) also suggested that animals not only take advantage of the actual hard structures themselves, but also benefit potentially from their impact on the surrounding environment through induced currents and subsequent scour on microhabitat.

Diving to New Depths

To our knowledge, the *YouTube* imagery of the sleeper shark recorded about 400 km off the coast of Brazil at a depth of 2,779 m (**Supplementary Table 1**) could potentially be the deepest recording of a large sleeper shark, *Somniosus* spp. (ANON, 2012). While the sighting of the sleeper shark at 3,000 m could suggest an even deeper observation, estimated depth could not be confirmed, since it was only mentioned in the video title, and not on-screen, or in video commentary. Morphological similarities make it impossible to distinguish *Somniosus* spp., which are '*virtually impossible to distinguish based solely on imagery*' (Gallant, 2012), and require molecular analysis (Benz et al., 2007). While smaller species of sleeper sharks (*Centroscymnus* spp.), including Portuguese dogfish, *Centroscymnus coelolepis*, are known to inhabit deeper waters up to 3,675 m (Compagno, 1984), only a handful of sightings of *Somniosus* spp. have been published in the western Atlantic. These included the ROV recording of the first report of a somniosid shark in the southern Colombian Caribbean (1,820 m; Acero et al., 2018) and northern GoM (1,423 m; Benfield et al., 2008). The maximum confirmed depth of a large sleeper shark (which was also the earliest sighting for this review), was recorded at 2,647 m (Benz et al., 2007). This was also the first reliable sighting of *Somniosus* spp. in the GoM and one of the deepest records for any shark species. These data, based purely on industrial ROV imagery demonstrates significant value of these industrial recordings.

In this study, recorded depths of one of the sperm whales (∼900 m) and gray seals (up to 353 m) were also near the limits of each species' known capacity. Tracking studies have recorded sperm whale modal dives at around 600 m (Watwood et al., 2006), while their foraging dives have been tracked to 1,185 m and possibly 2,000 m (Watkins et al., 1993; Watwood et al., 2006; Whitehead, 2018). Gray seals have been tracked diving to at least 412 m (Beck et al., 2003). Overall, tagging data require substantial financial and logistical investment, can be invasive and deeply traumatic for captured animals (especially marine mammals, consequently altering behavior and biasing results), whereas recordings used for this review were opportunistic.

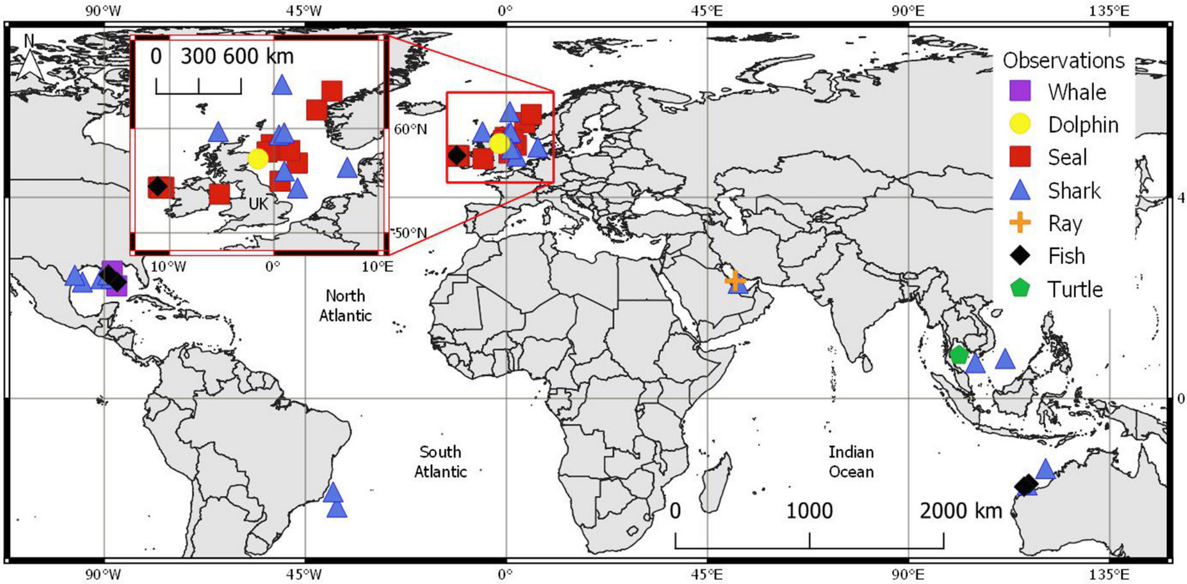

FIGURE 4 | Approximate marine megafauna sighting locations (*n* = 44) per group with UK map inset: whale (■*n* = 2), dolphin (●*n* = 1), seal (■*n* = 15), shark (▲*n* = 19), ray (+*n* = 1), fish (◆*n* = 5) and, turtle (⬟ *n* = 1). Coordinates in WGS '84.

TABLE 3 | Types of anthropogenic O&G structure, including descriptions based on Schlumberger (2019), that were identified during marine megafauna encounters, followed by number of recordings (*n* = 67).

Structure	Description	Sightings (*n*)
BlowOut Preventer (BOP)	Series of hydraulic pumps to close well off in emergency	6
Jumper	Flexible jumper linking pipes to various structures	1
Pipeline	System of pipes to transport oil or gas from production site (on seabed or submerged)	13
Platform	O&G production platform, including parts of platform visible (e.g., caissons)	26
Riser	Pipe connecting BOP to rig	1
Umbilical	(Buried) cables containing different cores used to carry electrical power, chemicals and control fluids	10
Wellhead	Surface termination of a wellbore utilized for drilling operations	10

Utilization of ROV Data

Gates et al. (2017) and Macreadie et al. (2018) highlighted the value of collaboration between scientists and industry, specifically in relation to the large volume of ROV datasets that could increase scientific discovery and enhance likelihood of encountering large enigmatic marine organisms. The Scientific and Environmental ROV Partnership using Existing Industrial Technology (SERPENT) Project produced 40 peer-reviewed publications from over 120 visits to offshore infrastructure (Gates et al., 2017). These publications include deepest recorded scalloped hammerhead (*Sphyrna lewini*) and southern sunfish (*Mola ramsayi*), as well as the first *in situ* observation of an oarfish, *Regalecus glesne* Gates et al. (2017). Results such as these, and those summarized in Macreadie et al. (2018), highlight benefits of industry and scientific cooperation and collaboration.

Increased availability of industry/research collaboration consortia [e.g., Influence of man-made Structures in the Ecosystem (INSITE) in the North Sea, and National Decommissioning Research Institute (NDRI) in Australia] could potentially facilitate data sharing across institutions. In addition, commercial industrial/environmental monitoring suppliers providing protected research time to staff would allow a vast amount of research to proceed; however, the reality is that industry has its 'preferred' users, so access to these proprietary data is often merely knowing the 'right' people, being in the 'right place at the right time', and having the drive (and the time) to complete 'unremunerated' analysis of data in a bid to further the field.

ROV data come generally in two forms (1) dedicated, scientifically designed surveys, often collected by instrumentally augmented (non-standard) ROVs for research purposes (e.g., McLean et al., 2019a), and (2) non-scientific surveys, collected by standard, off-the-shelf ROVs for industry purposes, where the analyst has no control over survey design/sensor configuration, distance to objects, scale, FOV, or even knowledge on make and model of ROV (e.g., Todd et al., 2018, 2019). Consequently, analysis options for these types of data differ. Dedicated scientific surveys can make use of techniques such as machine learning (e.g., Piechaud et al., 2019), whereas applicability of

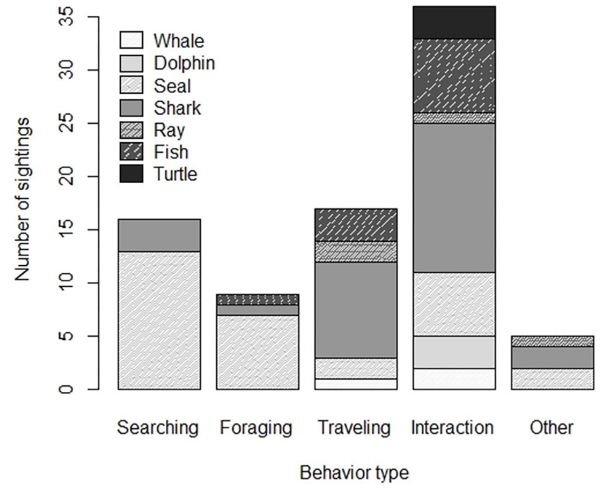

FIGURE 5 | Summary of behavior types observed across recordings (including sub-sightings, $n = 83$).

FIGURE 6 | Examples of foraging attempts and successful prey capture (behavior classification 2) in the vicinity of anthropogenic O&G structures by a gray seal (*Halichoerus grypus*). **(a)** Possible foraging attempt on gadoid fish (Gadidae), highlighted by red ellipse, **(b)** foraging attempt on unidentified fish, highlighted by red circle, **(c)** possible foraging attempt on subsurface prey, and, **(d)** successful fish capture.

these techniques to industry data are, to the best of our knowledge, untested.

Making use of ROV imagery and commercial-diver recordings, which are regularly collected *en masse* for the O&G industry worldwide (at the cost of billions of pounds each year) has multiple advantages. Collaborations can be formed between industry and scientists, allowing studies on habitats that are otherwise inaccessible, improving our knowledge of poorly-understood species (Gates et al., 2017; McLean et al., 2018a, 2020). Imagery collected routinely for industrial purposes can be made available to researchers for free, avoiding additional funding and effort. Industrial ROVs are often equipped with other useful sensors, such as hydrophones, Sound Navigation And Ranging (SONAR), or motion sensors (Muyzert et al., 2015; Macreadie et al., 2018), data from which can potentially be analyzed without requiring added cost or license approvals. Tapping into these surveys can reduce environmental disturbance from conducting duplicate surveys (Jones, 2009). To address major knowledge gaps and conservation challenges, it has been suggested recently to take advantage of ever-increasing supply of publicly available, open-source data on social media platforms to supplement conventional research protocols (Klemann-Junior et al., 2017; McKinley et al., 2017; Mancini et al., 2019; Pace et al., 2019; Parton et al., 2019). This has been demonstrated to be a reliable tool for obtaining cetacean-distribution data (Hann et al., 2018; Pace et al., 2019); therefore, collecting ROV data from international public-domain platforms can aid research purposes on a global scale. There are additional caveats on public data, e.g., only interesting/exciting sections are posted which makes effort impossible to determined.

Industrial ROV surveys have several disadvantages, most obvious of which is a complete lack of ecological survey design, making robust statistical analysis difficult (e.g., lack of pre-installation 'control' data, short temporal scale, variable seasonality, etc.). While time format for scientific data collection is standardized to GMT, time stamps on industrial imagery were supplied without format clarifications, necessitating assumption of local time stamps. Since data were also collected across different time zones, seasonality of ROV recordings was difficult to assess. While most data were collected between August and October for the northern hemisphere, a seasonal trend toward the summer period could be suggested. This could be a consequence of increase in effort and visibility during periods of clement weather, but could also suggest seasonality of megafauna presence in the vicinity of anthropogenic structures for e.g., feeding, breeding. This inference, however, remains unquantifiable from these data. While again, it would be interesting to confer this trend with the southern hemisphere, not enough data were collected there to allow for such a comparison. Additionally, presence of the ROV itself may alter behavior or presence of animals being surveyed (Hudson et al., 2005; Andaloro et al., 2013). For example, previous research has shown artificial light alters predator-prey relationships (reviewed in Longcore and Rich, 2004). This is particularly the case for seals, which have shown significantly increased feeding behavior with artificial lights (Yurk and Trites, 2000). No information was available as to presence/absence of lights on the ROV, diver, or offshore installation, meaning any inferences on periodicity were not possible.

Overall, quality of video imagery on industrial ROV surveys is often low priority, since surveys are designed mainly to use other technologies, such as pipe tracker and multibeam, lacking provision of a distance-range estimator (Macreadie et al., 2018). Additionally, several recordings were not provided with location or depth information, necessary for a reliable species identification. Improving design of ROV surveys could be achieved through collection of High Definition (HD) imagery, a laser range-distance estimator, and inclusion of necessary metadata as discussed in Todd et al. (2019) and McLean et al. (2020).

Dates of available data in this study were a consequence of various factors, as opposed to a lack of available ROV data. While the first ROVs were available commercially to the US Navy in the late 1960s (Hudson et al., 2005), the majority of imagery in this study originated post-2006. This may have been for a number of reasons. Firstly, data availability has increased with access to the Internet, accessible by the public since August 1991 and the popularity of YouTube (founded in 2005) over the last few years. Additionally, the greatest issue of industrial ROV data is the restricted access to commercially sensitive imagery, often protected by NDAs; therefore, partnerships between industry and scientists are invaluable to gain access to otherwise inaccessible data and researchers should be encouraged to consult archived data (Sward et al., 2019; Todd et al., 2019; McLean et al., 2020).

Marine Megafauna Sightings in the Face of Decommissioning

Of 17 taxa identified here to species level, 13 are listed on the IUCN Red List as threatened species, one of which, the hawksbill turtle, is critically endangered. Other taxa that could not be identified to species level include many that are listed as data deficient. Overall, presence of marine megafauna at offshore industrial sites warrants further investigation and consideration of potential interactions between megafauna and O&G installations.

Marine mammals have been detected visually and acoustically around these structures (e.g., platforms and pipelines), and it has been suggested that they target these sites for foraging (Todd et al., 2009, 2016; Russell et al., 2014; Orr et al., 2017; Delefosse et al., 2018). If echolocating cetaceans cluster regularly around installations within the 500-m exclusion zones, then it is highly likely they have been omitted unintentionally from historical visual and acoustic population surveys. This is because High-Frequency (HF) sound attenuates rapidly underwater (Urick, 1983), especially for odontocetes that emit Narrow Band HF (NBHF) echolocation clicks, such as harbor porpoises (Au, 1999; Teilmann et al., 2002; Villadsgaard et al., 2007). This limits survey detection to 250–300 m of the hydrophone (on-axis) at best, in ideal-propagation conditions (Todd et al., 2015, 2016), resulting in potentially significant underestimations of their true population status, e.g., by Small Cetacean Abundance in the North Sea or SCANS (Hammond et al., 2002, 2013, 2017). Future studies utilizing previously collected and archived data to study potential associations of marine mammals with offshore anthropogenic infrastructure, must therefore account for the fact that data likely do not include animals in close proximity (i.e., within the typical 500-m shipping exclusion zone) of offshore structures.

Fowler et al. (2014, 2018) and van Elden et al. (2019) highlighted key concerns with complete decommissioning and how it could affect marine ecosystems. The consensus was that a case-by-case approach is required, in which careful consideration of environmental impacts of removal of each installation would be required, as opposed to a blanket policy. The role these structures play for megafauna on a population level, acting as a network of protected areas with further potential to provide spatially predictable foraging locations, is still unknown. In addition, the notion that some offshore installations can act potentially as 'ecological mortality traps' warrants further investigation. For example, there are confirmed industry records (Victoria L. G. Todd, *unpublished obs.*) that whale sharks targeting offshore O&G installations suffer fatal interactions with support vessel propeller shafts, and vessel collisions. Clearly, continued research is necessary to address these major knowledge gaps.

CONCLUSION AND RECOMMENDATIONS

This manuscript provides a subsea account of marine megafauna presence and initial accounts of feeding behavior in vicinity of anthropogenic structures using visual ROV and commercial-diver data collected for industrial purposes. It also highlights the deepest recording of a sleeper shark in published literature and the first confirmed visual evidence of seals following pipelines. Future research on this topic could focus on non-commercial, recreational imagery from divers around offshore installations, and include Internet searches using non-technical terms, such as '*cool deep-sea fish gets eaten by shark near oil rig.*'

In the face of imminent decommissioning, our understanding of the role these structures play in the lives of marine megafauna is incomplete, at best. Investigations into use of ROV imagery demonstrate potential for collaborations between researchers and industry, that have capacity to unlock a vast wealth of data, which can be used to make more informed decisions regarding future management of these structures.

DATA AVAILABILITY STATEMENT

Links to publicly available YouTube videos are provided in **Table 2** and **Supplementary Table 1**. Other imagery is protected by NDA from industrial partners.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study because observations of animals were sourced from YouTube videos. Written informed consent was not obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

VT conceived the idea, sourced imagery, and contributed to drafting of the manuscript. LL sourced imagery and contributed to drafting of the manuscript. LW oversaw analysis and contributed to drafting of the manuscript. IP sourced imagery.

AH edited a previous draft. SC administered the analysis. IT sourced imagery. PM and DM contributed to manuscript drafts.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00230/full#supplementary-material

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