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Monitoring benthic biodiversity restoration in Lyme Bay Marine Protected Area: design, sampling and analysis

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Abstract

Long-standing concerns about the effects of scallop dredging and demersal trawling on high diversity mudstone reef and cobble habitats in Lyme Bay, southwest England, were addressed by the exclusion of bottom towed fishing gear from a 206 km^2 area in July 2008. A consortium led by Plymouth University Marine Institute was funded by the UK Department of Environment, Food and Rural Affairs to design and implement a study (initially funded for 3 years) to examine the effects of the closure on both nekton and epibenthos. This paper provides a detailed account of the methodology employed from survey design to data analysis to provide a protocol for future MPA monitoring programmes. Information on historical fishing effort, substrate distributions and current and previous closure boundaries was overlaid using GIS to locate suitable monitoring sites. Non-destructive and cost-effective techniques, including a towed high-definition video array and static baited video, were used to quantify changes in relative abundances of epibenthos and nekton over three years at sites previously fished but now closed to bottom towed fishing compared to both fished and un-fished reference sites. The monitoring program as described provides a model for robust, cost-effective evaluation of the efficacy of policy instruments for feedback into the adaptive management cycle.

Keywords: Marine Spatial Planning, Marine Protected Areas, Effectiveness, United Kingdom, Lyme Bay, Monitoring, Restoration

Highlights

- A survey to assess effects of a 206km^2 fishing closure in Lyme Bay, southwest England, is described.
- Design integrates historical fishing effort, substrate types, and current and previous closures.
- Non-destructive and cost-effective techniques were used to quantify changes in epibenthos and nekton.
- Provides a model for robust evaluation of policy efficacy for use in the adaptive management cycle.

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Abstract

Long-standing concerns about the effects of scallop dredging and demersal trawling on high diversity mudstone reef and cobble habitats in Lyme Bay, southwest England, were addressed by the exclusion of bottom towed fishing gear from a 206 km^2 area in July 2008. A consortium led by Plymouth University Marine Institute was funded by the UK Department of Environment, Food and Rural Affairs to design and implement a study (initially funded for 3 years) to examine the effects of the closure on both nekton and epibenthos. This paper provides a detailed account of the methodology employed from survey design to data analysis to provide a protocol for future MPA monitoring programmes. Information on historical fishing effort, substrate distributions and current and previous closure boundaries was overlaid using GIS to locate suitable monitoring sites. Non-destructive and cost-effective techniques, including a towed high-definition video array and static baited video, were used to quantify changes in relative abundances of epibenthos and nekton over three years at sites previously fished but now closed to bottom towed fishing compared to both fished and un-fished reference sites. The monitoring program as described provides a model for robust, cost-effective evaluation of the efficacy of policy instruments for feedback into the adaptive management cycle.

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1 Introduction

Marine conservation concerns have been increasingly addressed over the last two decades by means of area-based methods [1,2], including Marine Protected Areas (MPAs, however defined), rather than fishery-specific management tools [3]. This has been a general trend, but with different rates of uptake internationally according to management culture, history and established pattern of use [4]. In the UK, adoption of this so-called ecosystem approach [5,6] has been relatively recent [7]. Prior to 1981, protection of marine sites in the UK relied on Voluntary Marine Conservation Areas (VMCAs). More than 20 were established, and some are still extant, but they provided limited protection, and were not systematically selected. Legislative changes in the Wildlife and Countryside Act 1981 provided for the designation of statutory Marine Nature Reserves (MNRs); however only three were ever established: at Lundy (1986), Skomer (1990) and Strangford Lough (1995) [8]. More recently, policy shifts at the European and national level have led to the on-going designation of a network of small multiple-use MPAs designed to represent and conserve marine habitats and species. These Special Areas of Conservation (SACs) and Special Protection Areas (SPAs), collectively called European Marine Sites (EMSs) are required under European law [9,10]. However, while welcomed by conservationists, there are concerns that they do not contain no-take areas [11], and that their effectiveness may therefore be limited. Only Lundy, re-designated as the UK's first marine conservation zone (MCZ) in 2010 under the Marine and Coastal Access Act 2009, contains a no-take area.

In contrast, international adoption of MPAs has been widespread and rapid [12,13,14,15]. Common (but by no means universal) themes have been the designation of relatively large areas containing a combination of zones with varying levels of protection, clearly articulated goals and objectives for each MPA, and formalised consultative processes with significant stakeholder input [16,17,18].

This international adoption of MPAs as a management tool has been mirrored by the rapid development and adoption of quantitative methods of planning for such areas [19,20,21,22], specifically in evaluating design alternatives to minimise economic, social or ecological cost [23,24], for instance by using optimisation techniques such as the software package 'Marxan' [25].

Implementation of area-based management, however well designed, is not always followed by robust post-implementation assessment (e.g. [26]) of the efficacy of the management measures, from the points of view of stated management aims, unintended consequences, and economic impacts on users [5]. A range of studies has highlighted that this aspect of MPA management [27,28] lags markedly behind advances in planning and design. This is at odds with contemporary adaptive management approaches [5,29] which, in essence, seek to treat management as a series of experiments, so that the consequences can be quantified and fed back into the next design phase.

Lyme Bay, on the southwest coast of the UK (Figure 1), was the setting for an ongoing resource use conflict [30] between the conservation values of high biodiversity mudstone reefs [31,32] and the social and economic importance of scallop dredging to the local and regional economy [33]. Concerns about the effects of scalloping, specifically the negative impacts of dredging on iconic species such as the Pink Sea Fan *Eunicella verrucosa* (listed since 1992 under Schedule 5 of the UK Wildlife and Countryside Act 1981), and erosion of the soft mudstone substratum itself, had been raised for many years [34,35] . Small voluntary closures were implemented initially in 2001, and again in 2006 [36], but for a variety of reasons were regarded as either not successful, or of insufficient size to be effective [36]. Legislative and policy changes in Europe and the UK, in particular the European Commission Marine Strategy Framework Directive [37], laid the foundation for the Statutory Instrument (SI) [38] excluding bottom towed fishing gear from a 206 km^2 (60 nm^2) area in the northern portion of the bay. This took effect in July 2008 [38]. At the time of its designation, Lyme Bay was the largest UK MPA. It was the first declared under the regime adopting the ecosystem approach to marine nature conservation in the UK [39], and although not a no-take zone, is an important test case for the emerging UK marine spatial planning landscape [7,31], particularly as the UK moves to designation of a network of MCZs within its waters.

<<Figure 1 about here>>

Concurrent with the closure, and in keeping with the contemporary adaptive management ethos [5,40,41], the UK Department of Environment, Food, and Rural

Affairs (DEFRA) funded initially a three-year study to evaluate the effectiveness of the closure and its socio-economic consequences.

The DEFRA brief identified eight explicit aims [42], encompassing sampling design, selection of indicator species [43], quantification of "recovery" (our quotes) [42] and socio-economic impacts [44]. Throughout this paper the use of the term "recovery" is avoided, because it implies a shift towards a pristine or at least un-impacted state, which does not exist in Lyme Bay (or indeed, virtually anywhere [45]), because a range of extractive and potentially disturbing activities continue within the areas closed to bottom towed fishing gear (potting, netting, angling, diving, hand collection of scallops). The term "restoration" is used instead, to suggest improved status over time, in comparison to less impacted reference areas.

This paper reports on the development and implementation of a robust monitoring program to quantify restoration of epibenthos subsequent to the imposition of the SI, including the appropriate methods, design, and analyses, plus example data from the first two years of this important trial of marine conservation policy in the UK. Detailed analyses of the effects of the closure, including consideration of a range of ecological drivers for small-scale variation, have been reported to the funding agency [42] and are in preparation for publication (Sheehan *et al*. in prep). The aim of this paper is to:

- outline the background information used and approach taken in designing a broadly-based and low-impact monitoring program to quantify changes in benthic biodiversity after the implementation of the SI, and
- report on the practical and logistical aspects of implementation of such a program for a large MPA.

2 Survey and analytical design

2.1 Study rationale and spatial design

Tests of the effectiveness of management interventions seldom take place on a blank canvas. In the case of the Lyme Bay closure, the study design had to take into account historical patterns of use, including existing voluntary closures, distribution of fishing effort, and differences in benthic habitat types. In addition, the design required extensive spatial replication to account for landscape-scale variability.

Importantly, the study also had to be considered within the wider context of Lyme Bay ecosystems. The bay has been a productive fishery, both for static and mobile gear, for many years. It is also a valuable recreational resource [31], and some offshore areas are used for aggregate extraction [46]. Previous studies [32,34,35,46] had evaluated the relative sensitivity of areas of Lyme Bay reefs, and highlighted the contribution of the mudstone reefs to local and regional biodiversity [46]. Many of the sessile species associated with these reefs are long-lived, slow-growing and therefore particularly vulnerable to the effects of bottom towed fishing gear [47]. It was these species and their associated epifauna in reef habitats that were the focus for this study.

To select candidate sites for monitoring, spatial analysis of three types of information was conducted using ArcGIS 9.2 software. Information on patterns of historical fishing effort was derived from vessel patrol sightings from 2005 – 2008 provided by Devon Sea Fisheries Committee (DSFC) and overflight sightings from 2001 – 2007 provided by the Marine and Fisheries Agency (MFA). In each case, these point data were filtered for year and fishing types, and used to construct a density plot of relative fishing effort. While there was good agreement between the two datasets for distribution of scalloping effort in the centre and west of the closed area, there was apparent bias in the DSFC data to the east of the closed area, resulting from decreased patrol effort. The two datasets were therefore merged to construct a composite plot of relative scalloping effort (Figure 2a), by classifying both datasets into five classes of relative effort, calibrated so that where the datasets had significant overlap, the class boundaries matched.

<<Figure 2 about here>>

Data on benthic substrate and biotope distributions were provided by the Devon Biodiversity Records Centre, so that sites could be located on substrates known to support the taxa of most interest (Figure 2b). The boundaries of the SI and those areas that were previously closed under voluntary agreements (Figure 2c) were added, since they in part define current patterns of use. The three layers (boundaries, substrates, and mobile bottom fishing effort) were overlaid and merged to provide a single layer of polygons incorporating all the attributes of the source layers, enabling selection of those that met the necessary criteria (Figure 2d). All sites were located on hard or mixed substrates. Newly closed or open sites were located where scalloping effort was historically moderate to high, whereas closed sites were located where it was lower (because they were within the voluntary closures). The experimental design could therefore be constructed with four different treatment types, as follows:

The experimental treatment, termed New Closure (NC) - where bottom towed fishing was previously carried out, but not permitted after July 2008, and three control treatments, as follows:

Closed Controls (CC) - areas voluntarily closed to bottom towed fishing since 2001 or 2006, and remaining closed;

Near Open Controls (NOC) - areas open to bottom towed fishing before the closure, and remaining open within 5km of the new closure boundary, and

Far Open Controls (FOC) - areas open to bottom towed fishing before the closure, and remaining open, lying more than 10km from the new closure boundary.

Final selection of sites (Figure 3) was conducted after ground-truthing at the commencement of the first sampling period; for example, local knowledge allowed the selection of sites of suitable habitat not identified in the existing habitat classification.

Baseline surveys were conducted as soon as was feasibly possible after the closure in summer 2008, and surveys were repeated in the summers of 2009 and 2010. For the purpose of this study, a restoration scenario was considered: one in which response metrics of epibenthos (see Table 1) would change over time within the NC to more closely resemble the CC, while the CC, NOC, and FOC treatments should not change over time. There are other scenarios that could be explored with this design; for instance the previously isolated CC treatment areas may benefit from the buffering effects of the surrounding NC and the connections it provides between these relatively intact “islands” of biodiversity [48].

<<Figure 3 about here>>

2.2 Survey components

2.2.1 Towed video datasets

Information on the relative abundance and size of epibenthos was gathered using a cost-effective and low-impact towed array developed for the survey [49] carrying a high definition (HD) video sensor positioned at 45° to the seabed, over nominally 200m transects. Tow speed was limited to 0.25 $m.s^{-1}$ to prevent blurring of the video images. Data were extracted by examination of individual HD video frames and by examination of the entire transect for infrequently occurring sessile fauna (Table 1).

Density of infrequently occurring sessile fauna that may have been underestimated by examination of individual frames (see below) was assessed using a “through the gate” counting method. Two lasers mounted in parallel projected dots onto the sea floor a fixed distance (0.5m) apart. Start and end points of each tow were determined using GPS. The whole video transect was viewed, and occurrences of each taxon recorded if it passed through the “gate” formed by the two laser dots. Raw counts were converted to density by dividing by the total area sampled (laser spacing x tow distance).

Individual frames were sampled from the raw video stream at five second intervals to avoid any overlap between the images, and were overlaid with a 0.25 m^2 counting grid, allowing the extraction of density and % cover information for each taxon (Figure 4a). About 100 frames were available from each transect, but examination of every frame would have been prohibitively time-consuming. Therefore, an initial analysis of a sample data set from 12 sites was conducted to determine the number of frames that could be sampled without loss of accuracy compared to sampling all frames. This analysis used a two-stage approach within the PRIMER software package [50]. First, matrices of Bray-Curtis similarity of multivariate (70 taxa) assemblage structure between the 12 test sites were derived separately for the full (100%) dataset, and from decreasing fractions (1/2, 1/3, 1/4). The relationships between the matrices from these separate fractions, and the full dataset, were then compared using two-stage MDS and ANOSIM [42]. This showed that sampling 1/2 or 1/3 of the available frames per 200 m transect gave equivalent results to sampling all frames, but below that level there was an unacceptable loss of information for the gain in sample processing time. Mean number of frames available (i.e. 100%) in the trial was 88.75; 1/3 of this is 29.6, so future analyses were therefore conducted using 30 frames. All data extracted from frames were pooled at the transect level to avoid pseudoreplication issues and to increase the precision with which differences in epibenthic response metrics could be detected.

For each frame, the dominant substrate type was recorded, so that frames on principally soft substrates were eliminated from the analysis, as were those where visibility was poor, or the array was not in the correct orientation or distance from the seafloor. Species identifications were visual, but augmented by comparison of video images with diver-collected reference specimens obtained from a small number of sites in the first year of surveys.

Size class data were extracted for six large, frequently occurring, benthic taxa, using the counting grid. Data were only collected for those individual organisms where the axis being measured was oriented normally to the camera angle. The size of organisms was scored in calibrated (10cm) grid squares, corrected for the position of the lasers. The five size classes derived were ≤5 cm, 5-10 cm, 11-20 cm, 21-30 cm, and >30 cm.

<<Figure 4 about here>>

2.2.2 Baited video

Baited Remote Underwater Video (BRUV) sampling [51] is a non-destructive technique commonly used to sample fish populations, often with respect to closed areas [26,52]. In this study, BRUVs were used to provide additional information about the effects of the closure on reef-associated nekton not captured by flying array, such as fishes and mobile invertebrates, thus extending the relevance of the study. The same HD video sensor used for the towed video work was mounted on a static frame with 100g of fresh mackerel bait held within a cage positioned one metre in front of the camera. The frame was deployed on the sea floor, and no data extracted for a minimum of two minutes to allow disturbed sediment to settle, and nekton to become accustomed to its presence. Following this, 15 minutes of video was recorded for analysis. Relative abundance of nekton was assessed by counting the maximum number of individuals (Max N) of each species within each one-minute slice of video, and averaged to give a mean Max N for each species (Figure 4b).

<<Table 1 about here>>

2.3 Analytical methods and design

2.3.1 Towed video

Differences between treatments (CC, NC, NOC, FOC) over the three years (2008, 2009, 2010) of the study, and in particular interaction effects (Year x Treatment), could be assessed for the following metrics, all derived from the towed video datasets (Table 1). In each case, these take the form of a site/species matrix, with values in the cells representing mean density (ind.m^{-2}) or % cover for each taxon at that site.

Metrics available include total abundance (N), species richness (S), diversity indices (e.g. H', J', 1-λ), abundance of indicator species (derived in a separate desk-top study before monitoring commenced [43]),and multivariate assemblage structure based on relative abundance of each taxon per site. Univariate analyses were based on a Euclidean distance matrix and the multivariate analysis on Bray-Curtis dissimilarities. For each metric, two-way analyses (treatment x year) were conducted using a permuted analysis of variance (PERMANOVA [53,54]) within the PRIMER software package [50]. PERMANOVA allows testing of complex multifactorial designs on similarity matrices, without the constraints of conventional parametric testing. Previous studies [55,56] have demonstrated the utility of these techniques on datasets from benthic environments. PERMANOVA is most commonly applied to similarity matrices derived from multivariate (e.g. site x species tables) data. It can be just as readily applied to a similarity matrix derived from a single response variable (N, S, a diversity index, or individual species' abundance) with the appropriate distance measure. In this case, PERMANOVA yields Fischer's F-statistic (Anderson, *pers comm*.), but with p-values obtained by permutation.

Within each of the PERMANOVA analyses, pairwise tests were used to determine how similarity between treatments changed over time. Our restoration scenario predicts a shift in univariate or multivariate response metrics for NC relative to the control treatments over time, in that NC should become more similar to CC (univariate example given in Figure 5).

<<Figure 5 about here>>

Where differences were observed between years and treatments in the multivariate analyses, SIMPER (similarity percentage) analysis within the PRIMER package was used to examine the relative contribution of individual taxa to those differences.

Analyses of size class data used multinomial logit log-linear models [57] in SPSS version 18 (IBM corporation, 2010), because these data were categorical, rather than direct measurements, and the distribution of observations across years and treatments was severely unbalanced.

2.3.2 Baited video

The baited video technique was trialled in 2008, and implemented in 2009 and 2010. Because of the relative mobility of nekton, a seasonal component was included in the design, with sampling conducted in both winter and summer. Since resources were limited, the trade-off was that the Far Open Control (FOC) treatment was not included in the design.

The resulting dataset is in the form of a site/species matrix, with values in the cells being mean maximum number of individuals per one-minute slice (mean Max N). Differences in relative abundance of nekton between treatments and years could be therefore also assessed for univariate (N, S, diversity indices, or individual species) or multivariate (nekton assemblage structure) metrics using the same techniques employed for the towed video datasets. However, because the sampling techniques are different, and the resulting data matrices are in different units, it is not possible to combine epibenthos and nekton into a single analysis.

3 Survey implementation

3.1 Field effort

Fieldwork was conducted annually between July and September, with a seasonal component added for BRUV which also took place in March-April. Both the BRUV and towed video kits could be deployed safely in sea conditions up to force four; beyond this the survey was suspended until conditions were more favourable. When weather permitted it was possible to complete three sites (nine replicates) per day for the BRUV and eight sites per day for the towed video. Based on the designs presented here, and assuming suitable weather conditions, the summer sampling work required about 14 boat days to complete, with an additional six boat days for the BRUV deployments in spring. In practice, poor weather notwithstanding, the summer survey program was conducted within one month (16 days in 2008 with no BRUV component, 30 days in 2009, and 25 days in 2010). The spring BRUV deployments were carried out over periods of six days in 2009, and 16 days in 2010.

For both survey methods a minimum of three people were required in addition to the boat captain, giving a total of 60 person-days per year.

3.2 Data extraction and analyses

Data extraction commenced immediately following the completion of the field season. On average, it was possible to extract video data from four sites per day, frame data from one to two sites per day and BRUV data from two sites (six replicates) per day, requiring approximately 70 person-days per year. The extraction speed varied with the quality of the video captured, habitat type and treatment (as the CC and NC had larger numbers of taxa per frame than the open control sites). It is important to note that despite extraction speed increasing over time due to familiarity with methods and species, the greater the degree of restoration of the reef habitats, the slower the extraction process due to the addition of new species and the increasing number of taxa per frame. It is therefore likely that in such a scenario the time taken for extraction will remain consistent between years.

The key issues with extracting data from video relate to confidence in the identification of taxa, and the need for consistency. Identifications were visual, but augmented by comparison of video images with diver collected reference specimens collected at a small number of sites in the first year of surveys. Some taxa were unable to be resolved to the species level, and were therefore allocated to species complexes, or higher taxonomic levels. Where necessary, experts from Plymouth University and from the Marine Biological Association of the UK assisted with identification of unusual species. As with all studies where multiple individuals are involved with data extraction, a standard approach to the process was essential. Training and cross-checking of species identification and quantification methods prior to extraction and at regular intervals throughout the process ensured consistency was maintained.

Over the three years of the study, a total of 226 towed video transects were surveyed. Of these, 40 were discarded as unsuitable for analysis due to poor water clarity, unsuitable habitat, or incorrect placement. The dataset available for the frame-by-frame analysis therefore contained 186 transects, representing 16 CC, 16 NC, 16 NOC and 14 FOC, surveyed in each of the years 2008, 2009 and 2010. Data on the relative abundance, as density or % cover, of 96 taxa were extracted from a total of 5,580 individual video frames.

Over the two years of BRUV sampling, a total of 216 video samples were recorded, and all were suitable for analysis. The dataset represented six sites in each of the CC, NC and NOC treatments, with three replicates at each site, over summer and spring of 2009 and 2010. Although primarily aimed at quantifying nekton abundance, several highly mobile epibenthic taxa were routinely attracted to the bait and were also recorded. Relative abundance, as mean Max N, of 36 nektonic or epibenthic taxa were quantified, including 15 fish species, 15 crustaceans, four molluscs and two echinoderms, from a total of 72 hours of video.

4 Discussion

The survey design processes used, and cost-effective methods employed, provide a robust base-line for on-going monitoring, while providing some early evidence of restoration [42]. This is especially important, given the evolving policy landscape for marine conservation in the UK. In Lyme Bay, planning and consultation is currently underway for possible modification of the protected area boundaries, and management provisions [58]; the existing monitoring design has been adapted to allow for this while ensuring compatibility with the three years of data already collected.

Studies of the effects of fishing closures elsewhere have highlighted time-lags between implementation of such closures, and detectable effects [48] sometimes over decades [59]. These vary with the location and type of the reserve [60], and the biota being assessed (e.g. fishes versus benthos) [12,48,61]. All have in common the need for robust designs, and a commitment to long-term monitoring. A common theme is a lack of pre-closure data over several years or seasons, that would permit a full Before-After Control-Impact (BACI) design [62,63]. This is a reality of marine conservation policy, and Lyme Bay is no different in this respect. The solution adopted in this study is the use of multiple, replicated controls. The design deliberately included a spread of sites across the closed area. Although this makes analysis more complex, it allows examination of geographic effects that may not be detectable with a simpler design. These are critical to understanding whether changes observed are as a result of the closure, or extrinsic influences at the landscape scale [64].

Lyme Bay has been the focus of marine conservation initiatives for over 20 years, and the area has consequently been well studied prior to the closure [30,32,34,35,36,46]. It is also relatively shallow and inshore (so accessible for divers), so that detailed information on substrate distributions and patterns of use was available to aid in the survey design. In similar studies elsewhere, this is often not the case, and this information has to be collected before monitoring can commence (e.g. [28,65]).

Many studies on restoration after closures to fishing focus on changes in abundance on target species [66,67], or a small number of indicator species [68] or functional groups [69]. This study has taken a broader approach. The DEFRA brief included the derivation of an indicator species list (reported elsewhere [43]), and reports to the agency have covered these taxa. However, this study also assessed changes in the (visible) epibenthic assemblage as a whole, in order to assess more subtle changes that may not be detected by indicator or target species [61]. While this involves additional work at the data extraction phase (enumerating 110 taxa over all sampling types, rather than the 17 indicator species encountered), it provides additional perspectives that are crucial to the so-called “ecosystem” approach [5,28].

UK marine conservation policy has undergone a transformation over recent years, with the passing of landmark legislation (Marine and Coastal Access Act 2009) and restructuring of government agencies (in particular the creation of the Marine Management Organisation) reflecting policy shifts across the EC [37,70]. Central to the current policy ethos is the adoption of ecosystem-wide approaches to resource management [71,72], and the incorporation of adaptive management loops to feed back the results of monitoring [29,40]. The current study is a topical example of this practice. While its implementation has been locally controversial [33], and treads the boards of familiar conflicts between fisheries and biodiversity values [31,41,73], this study provides an objective evaluation of the “top-down” [74] policy position taken at the time. It is possible because sufficient funding to undertake robust monitoring was made available in a realistic timeframe – there are important lessons in this for future marine conservation initiatives in the UK and elsewhere. A recent review [75] identified 100 ecological issues needing information to allow them to be incorporated into UK the policy landscape. The Lyme Bay closure and subsequent monitoring address several of these, especially numbers 27 (size and siting of MPAs) and 31 (recovery of marine habitats) [75]. A separate study has assessed economic effects on fishers and fish processors [76] .

Viewed more broadly, the conduct of this study compares favourably with published criteria for good practice on the monitoring, evaluation and management of marine resources, e.g. [5,27]. In particular, previous studies have highlighted that it is important for agencies charged with planning and implementing MPA designs to: (a) follow up and objectively assess the effects, and effectiveness, of the policy instruments [28,77], and then (b) formalise mechanisms to incorporate these findings into open reviews of, and revisions to, the management regime [6,27]. While it is too soon to draw any conclusions about the latter, the approach taken by the UK government in this case with regards to the former, and including its desire to see the results presented in the peer-reviewed literature, is a positive one. Ongoing work remains to be undertaken to determine whether the initial trends continue; this presents further challenges, especially in securing funding once the initial high profile has diminished.

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CAPTIONS

Figure 1: Location map showing closure boundary and ports

Figure 2: Derivation of candidate areas for monitoring. Spatial information on historical fishing effort (a), hard substrate distribution (b) and current and past closure boundaries (c), were combined to define candidate areas (d).

Figure 3: Monitoring sites (a) Video transect locations (b) BRUV sites. CC = closed control sites; NC = new closure sites; NOC = near open control sites; FOC = far open control sites; OC = open control sites.

Figure 4: Sample video images: (a) Frame grab with counting grid from a closed control site, summer 2010. The laser markers used to calibrate the area sampled are clearly visible as green dots just outside the grid. (b) BRUV frame from a closed control site, spring 2010, showing bait box and Goldsinny wrasse *Ctenolabrus rupestris.*

Figure 5: Sample analysis: Changes in density of the bryozoan *Pentapora fascialis* (Ross Coral) between 2008 and 2009. Inset table shows p-values from pairwise PERMANOVA on Euclidean Distance similarity from 4^{th} root transformed data, illustrating changing relationships between the experimental treatment (NC) and closed and open controls. In 2008, densities of *P.fascialis* in NC and CC were significantly different, whereas in 2009 they had converged.

Table 1: Summary of Survey Elements

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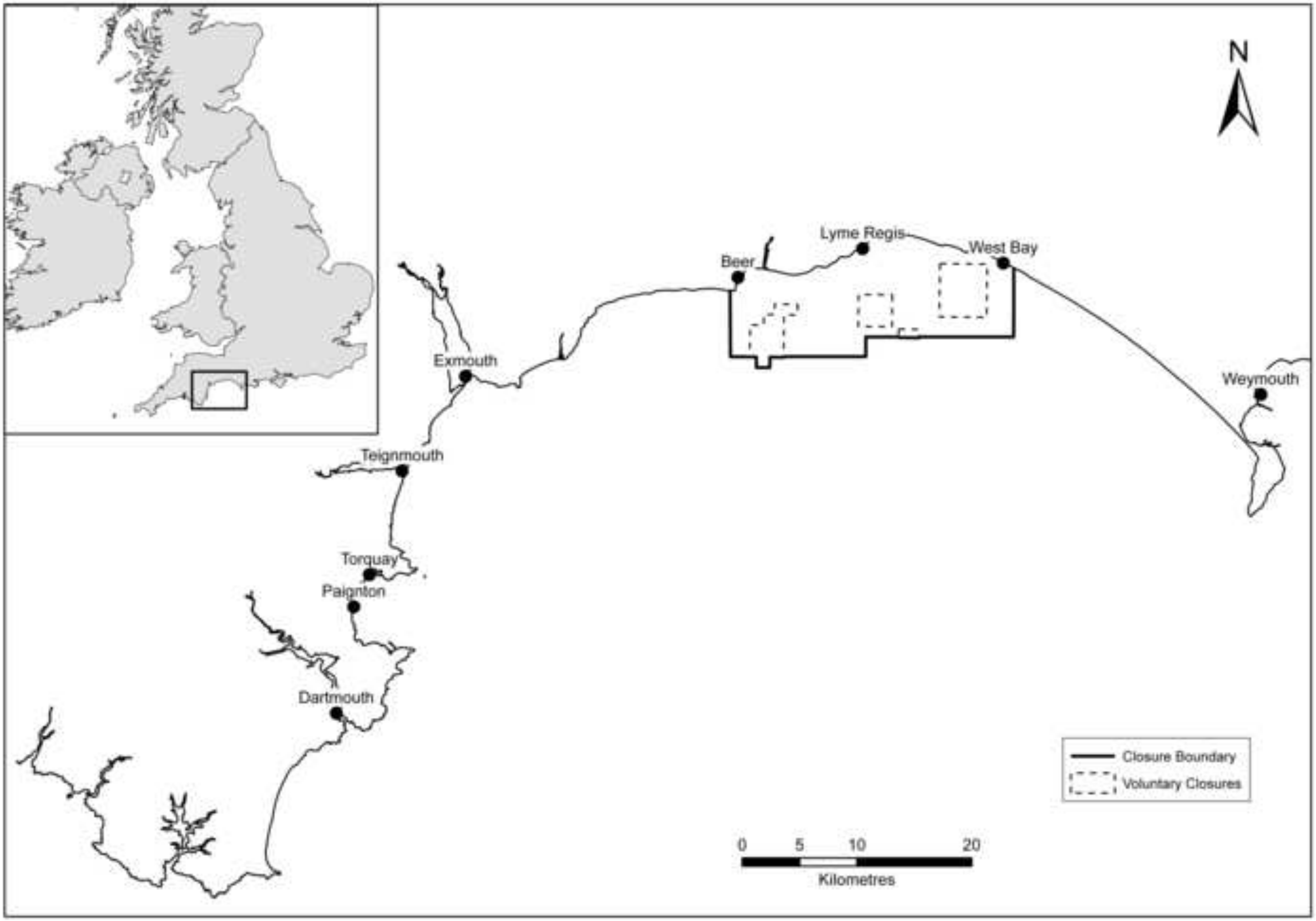

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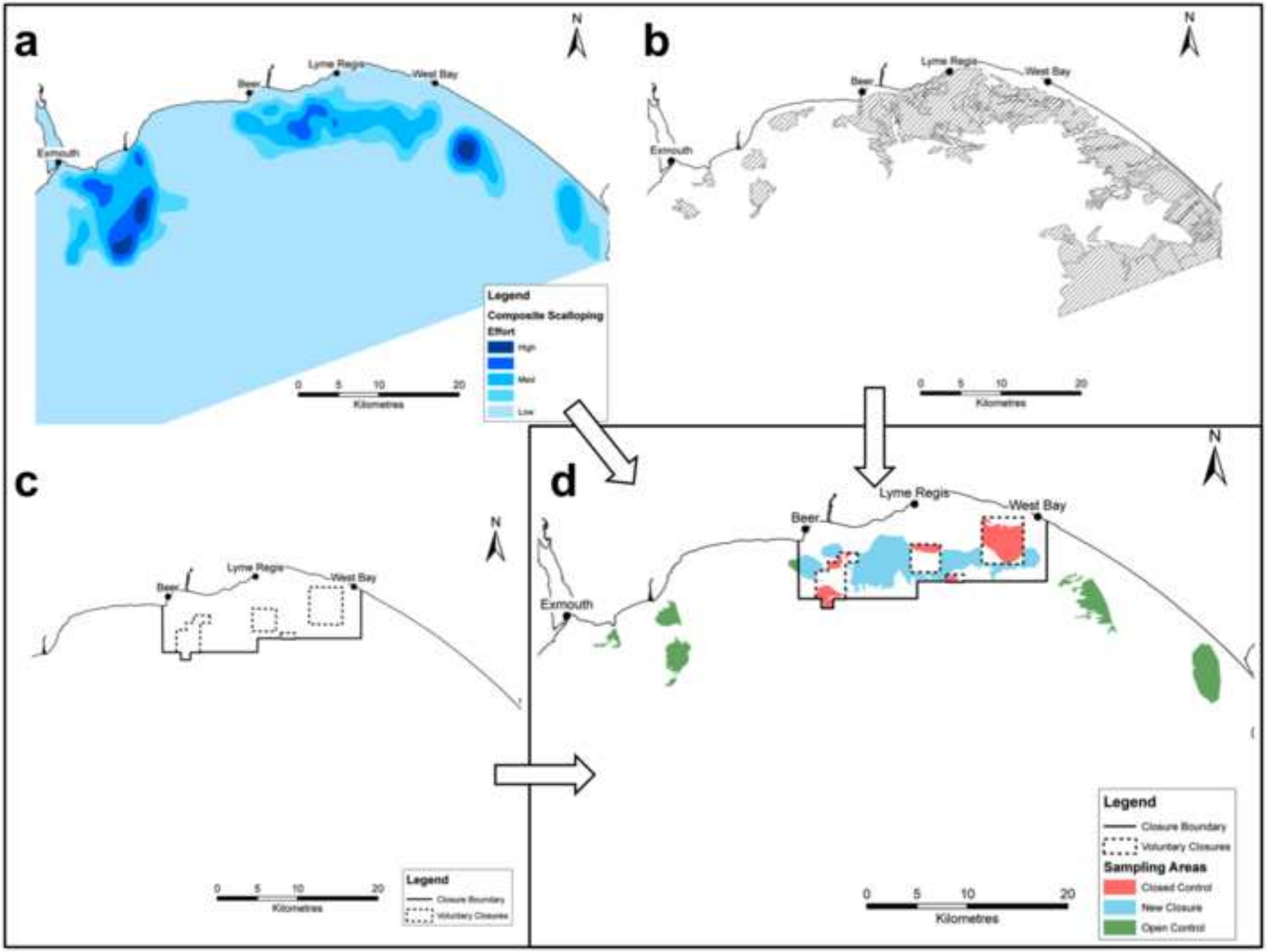

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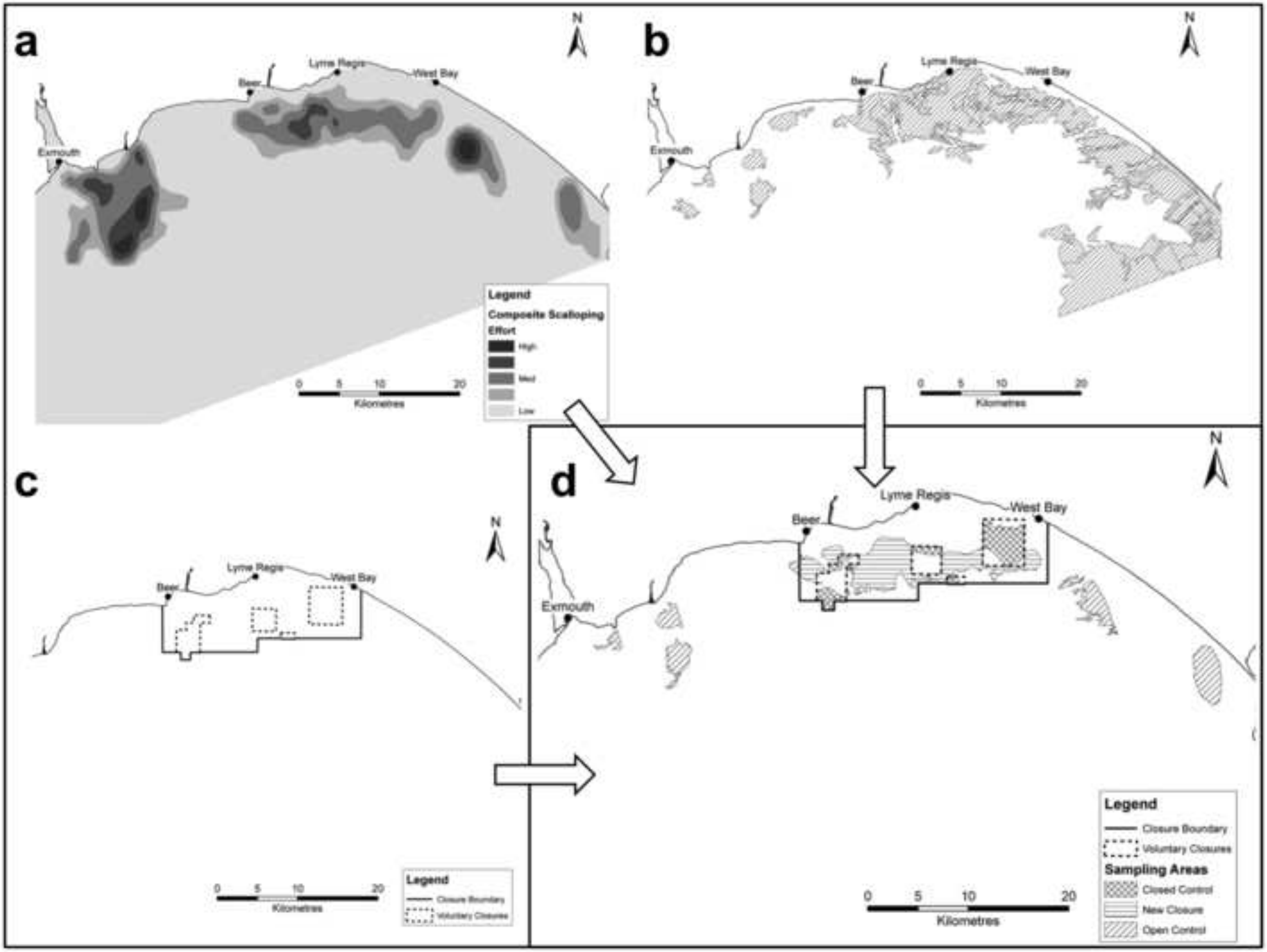

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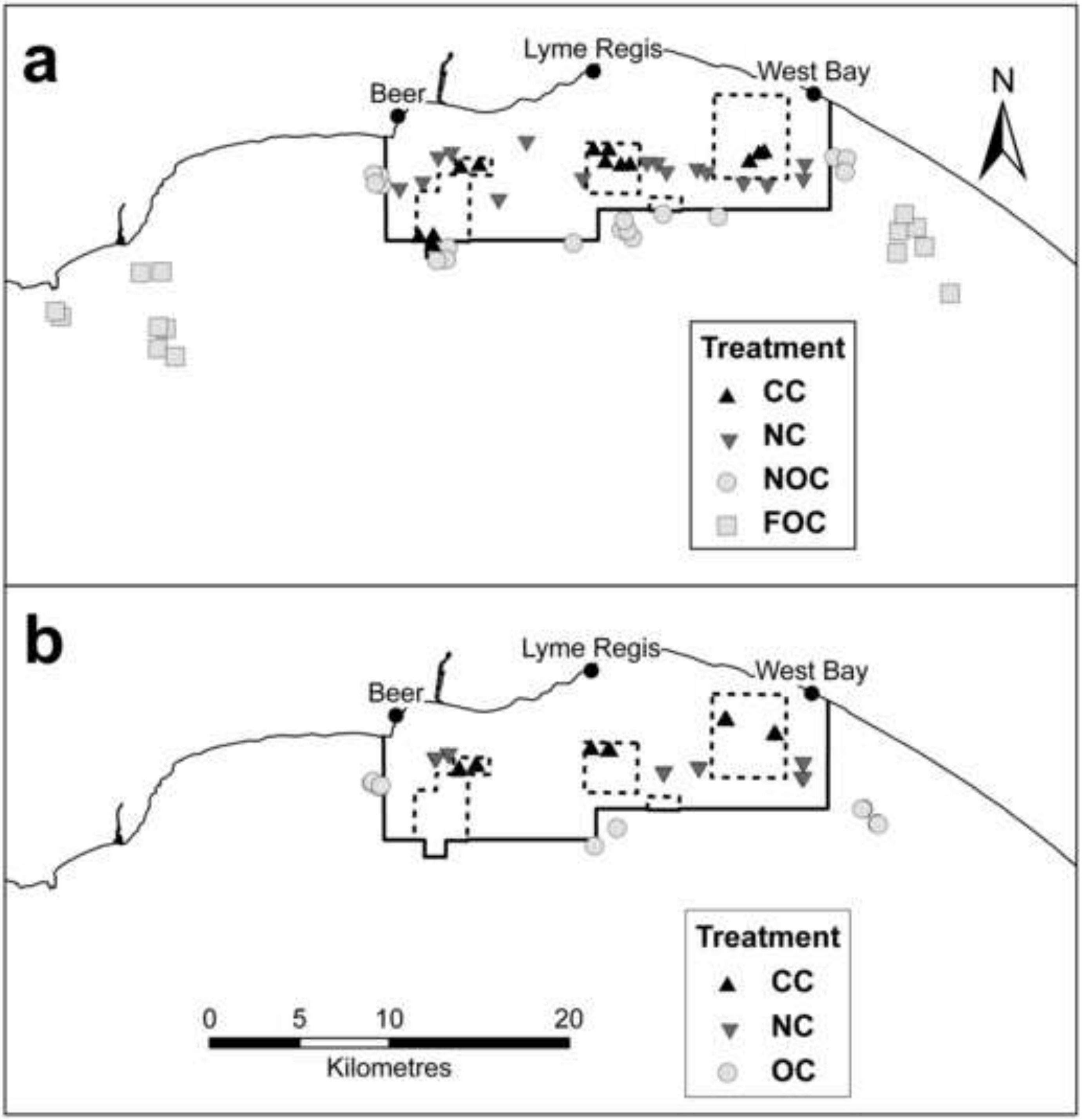

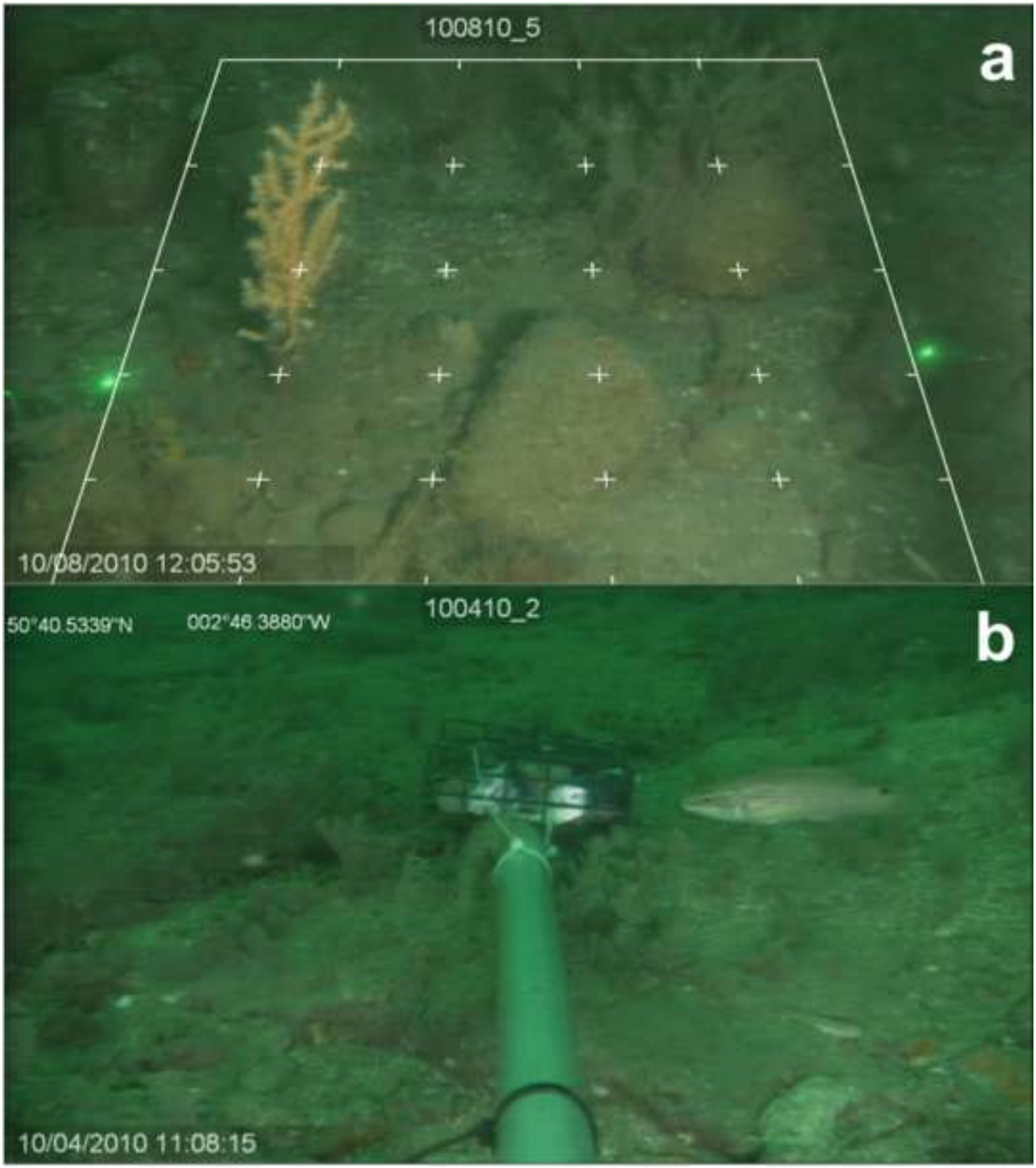
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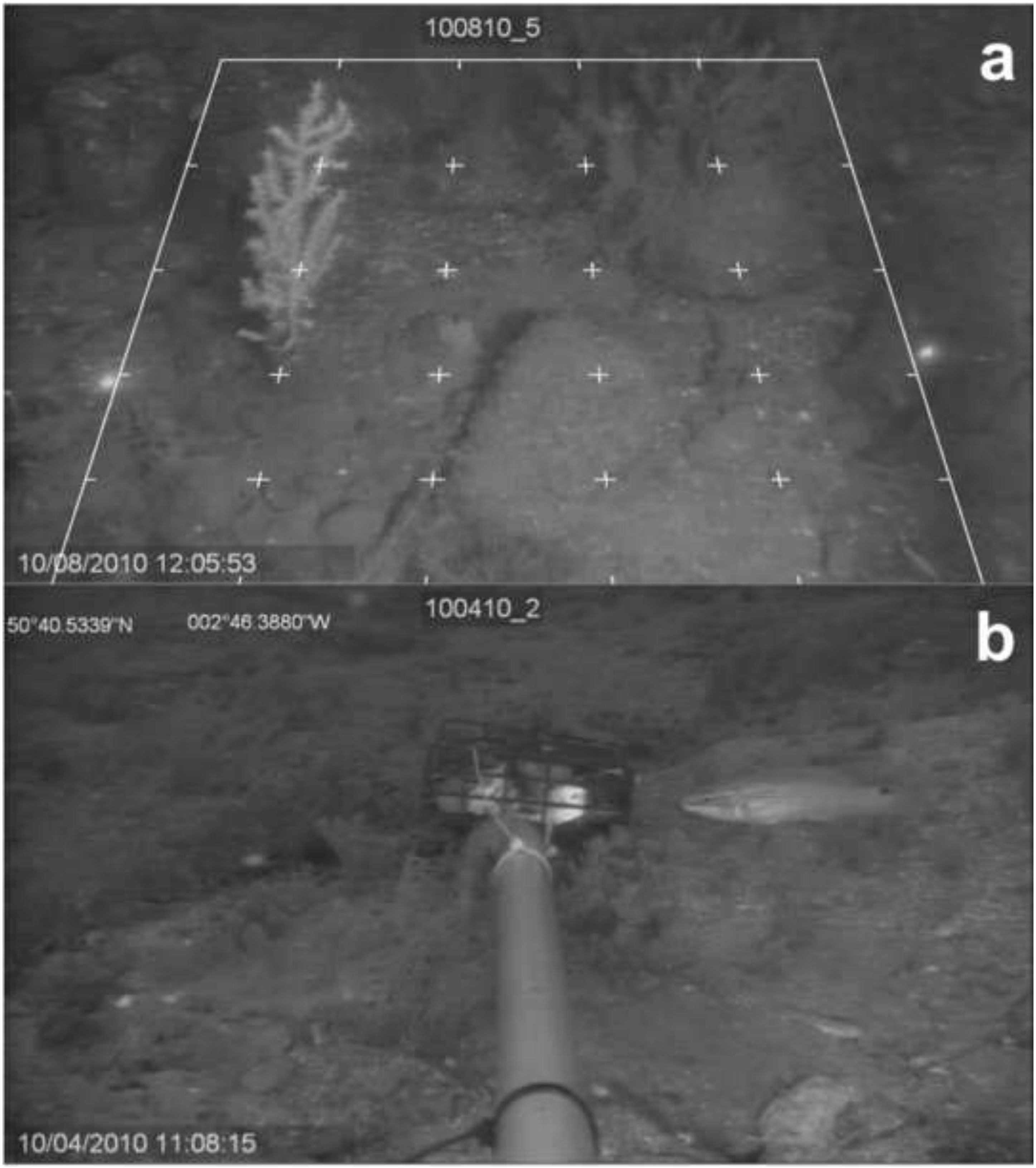

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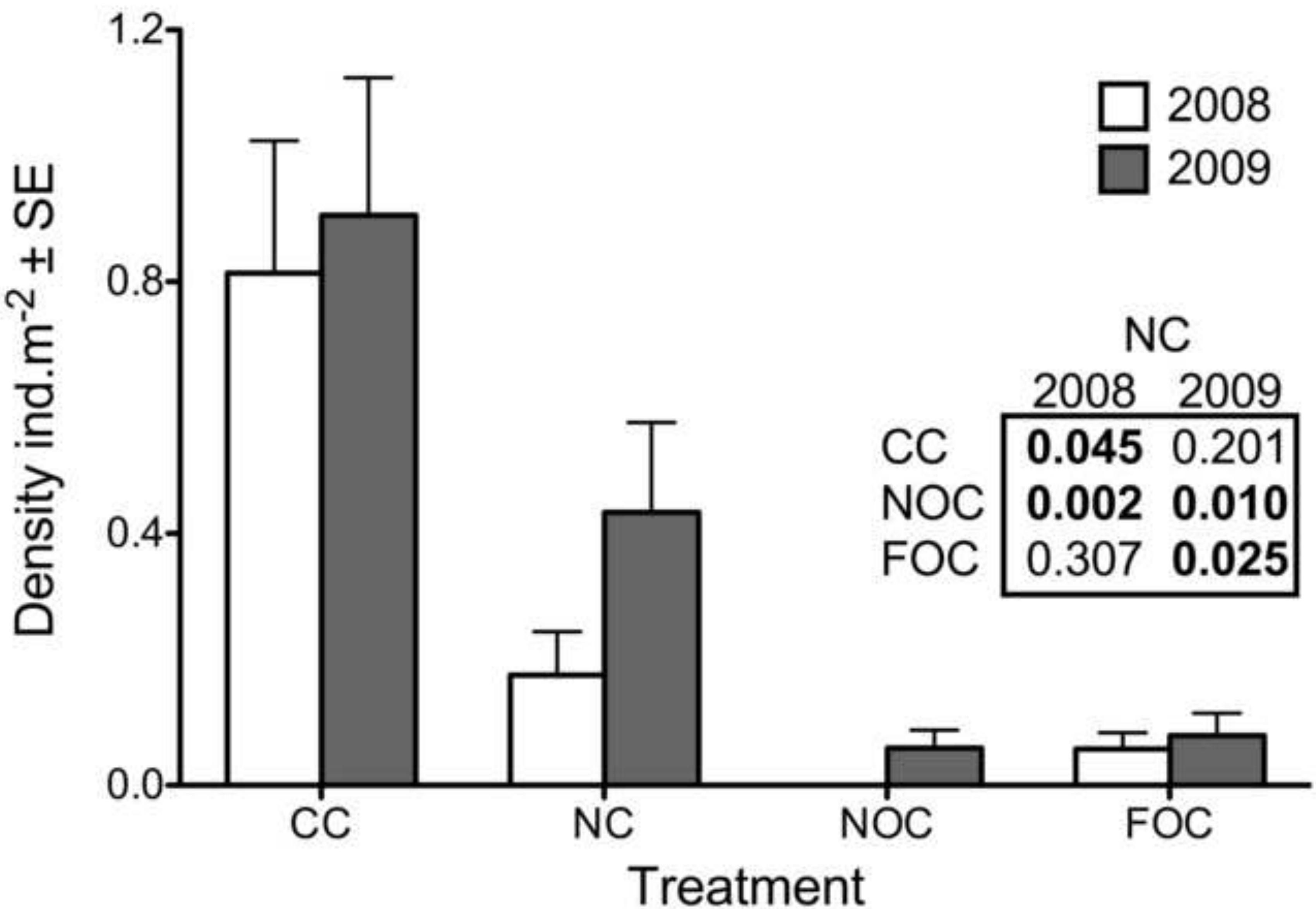

Table

Survey Type	Description	Coverage	What measured	How measured	Data type	How analysed
Towed Video	c.200 m towed HD video transects with laser markers at fixed distance apart.	c.200m by 0.5m transects at nominally 64 sites within 4 treatments by three years	Abundance of large, rare epibenthic taxa	Count of individuals passing through the laser markers for entire transect. Distance determined by GPS.	Density	Ordination with permuted MANOVA (PERMANOVA)
		30 frames at 60 sites within 4 treatments by three years	Abundance of discrete epibenthic taxa, incl indicator species	Count of individuals within 0.25 m^2 counting frames, averaged for transect	Density	Ordination with PERMANOVA
		30 frames at nominally 64 sites within 4 treatments by three years	Abundance of colonial or cover-forming epibenthos	Dots lying over taxa within 0.25 m^2 counting grid, averaged for transect	Percent cover	Ordination with PERMANOVA
		30 frames at nominally 64 sites within 4 treatments by three years	Size of selected taxa	Allocation of individuals to size classes from 0.25 m^2 counting grid	Size class distribution	Multinomial logit log-linear models
Baited Remote Underwater Video (BRUV)	15 minute BRUV deployment, baited with 100 grams of mackerel	3 replicates at 18 sites within 3 treatments (no FOC) in 2 seasons by 2 years	Abundance of reef-associated nekton (fishes and mobile invertebrates)	Maximum no. of individuals within a one-minute slice, repeated 15 times per replicate.	Max N	Ordination with PERMANOVA