



# Effects of man-made structures on sedimentary oxygenation: Extent, seasonality and implications for offshore renewables



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## ABSTRACT

The number of man-made structures to be placed in the marine environment is set to increase massively in the near future as a consequence of the wide-scale adoption and commercialisation of offshore electricity generation. Marine renewable energy devices (MREDS) interact with their receiving environment and are *de-facto* artificial reefs. The Loch Linnhe Artificial Reef (LLR) complex is a large-scale experimental facility, with the main matrix consisting of 30 separate reef modules deployed in 10–30 m depth and over a gradient of hydrographic and sedimentological conditions. The LLR offers potential to examine impacts that are analogous to those likely to occur around MREDS. The extent of the impact of the LLR modules on the receiving environment was assessed by determining their effect on sedimentary redox potential, as a function of distance from the reef-edge, and season, using an innovative, hand-held, underwater redox probe. The results are commensurate with the reef-proximal baffling of water flow resulting in the entrapment of drifting phytodetritus. At the least current-exposed reef-group the expected decrease in mean redox, at 80 mm sediment depth, was 80 mV (95% CI 40, 120 mV) but this effect was not observed more than 1 m from the reef edge and only occurred during summer months (water temperature > 10 °C). Redox at the reef edge, particularly during summer months, was more variable compared to redox taken at 1 m and 4 m reef-distance and was associated with the patchy distribution of phytodetrital accumulations. At all reef groups, there was no discernible difference in modelled mean redox between observations taken at 1 m and 4 m-reef distance. Artificial structures, including MREDS, may cause quite major sedimentary changes but this evidence suggests that these effects will be of limited spatial scale and, where phytodetrital accumulations occur, are only likely to be detrimental in oxygen-deficient sediments. Where these changes occur at well-flushed sites they are likely to be associated with increased infaunal biomass typical of moderate organic enrichment.

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## 1. Introduction

Man is increasingly intervening in the near-shore marine environment through activities including coastal protection/reclamation, marine-aquaculture, marina-development and the deployment of marine renewable energy devices (MREDS) (Alexander et al., 2012). The scale of the potential MRED development is considerable, for example, the UK is projecting a 46 GW offshore wind capacity in its territorial waters (Anon, 2012) which equates to approximately one third of Europe's projected capacity of 150 GW by 2030 (EWEA, 2013). One hundred and fifty GW is equivalent to a staggering 30,000–50,000 wind-turbines based on a standard 3–5 MW per device (the London Array wind turbines

are 3.6 MW per device; Anon, 2014). In addition to offshore wind developments there is interest in deploying wave- and tidal-devices and all such developments will be supported by infrastructure that includes sub-stations, meteorological masts and cabling. MREDS, and their supporting infrastructure, will be deployed over a wide range of water depths and sediment types including clays, muds, silts and fine sands (Table 6 in Linley et al., 2007). There is likely to be greater future overlap between offshore renewables and fine muddy sediments as the wind-industry moves further offshore and into deeper water (e.g. UK 'Round 3' sites; The Crown Estate, 2013).

MREDS will act as *de-facto* artificial reefs by providing attachment points for encrusting fauna and flora and shelter from tidal flows (Miller et al., 2013). Whilst MREDS are not classified as artificial reefs, because their primary function is not to emulate a natural reef in some way (Anon, 1997), much artificial-reef impact research is directly relevant to their likely impacts. Once placed on

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the seabed man-made structures, of any type, interact immediately with the local current regime. This hydrographic interaction may result in the acceleration or baffling of flow around the structures, the formation of various types of vortices and the generation of turbulence and wave breaking (Ali-Albourae, 2013; Sumer et al., 2001). Such hydrographic interactions potentially affect both the particulate transport around reefs and the associated epibenthic and infaunal assemblages (see below). Research into the broader effects of artificial reefs on their surrounding sediment is limited and contradictory: Fabi et al. (2002), Guiral et al. (1995) and Wilding (2006) all report an increase in fine- and/or organic-material content associated with reef-proximity whilst Barros et al. (2001), Ambrose and Anderson (1990) and Davis et al. (1982) report increased scour and a reduction of fine material at the reef edge. Relatively fine sediments are frequently associated with higher organic contents and greater macrobenthic diversity and biomass compared with coarser sediments (Snelgrove and Butman, 1994) but this changes when the organic load becomes excessive (see below).

Water flow is critical to benthic assemblages as it supplies both food and oxygen and removes waste-products (Gray et al., 2002; Jumars and Nowell, 1984; Pearson and Rosenberg, 1978; Vogel, 1994). Sedimentary hypoxia can occur naturally, for example where water exchange is limited (Karlson et al., 2002), but it is often linked to the deposition of organic matter from anthropogenic activities such as aquaculture (Black, 1998; Diaz and Rosenberg, 1995, 2008) and wood processing (Pearson and Rosenberg, 1978). The effect of organic enrichment on benthic fauna is gradual and, initially, is frequently associated with an increase in biodiversity and/or biomass until such a point where bacterial respiratory oxygen demand exceeds supply and the sediment becomes hypoxic (Hargrave et al., 2008; Pearson and Rosenberg, 1978). In conditions where oxygen is effectively absent, indicated by an electric potential (Eh, redox potential, henceforth redox) of  $< \sim 0$  mV on the hydrogen scale (Hargrave et al., 2008; Zobell, 1946) benthic anaerobic bacteria reduce a series of proton receptors (consisting of various oxides and sulphates) during respiration (Christensen et al., 2000). The reduction of sulphates produces hydrogen sulphide (Diaz and Rosenberg, 1995; Pearson and Rosenberg, 1978; Snelgrove and Butman, 1994) that is toxic to all but relatively few adapted species and, consequently, anoxic sediments are characteristically species poor (Diaz and Rosenberg, 1995; Pearson and Rosenberg, 1978). The oxygenation status of muddy sediments, measured using a redox probe, is a widely used and cost-effective, real-time indicator of the ability of that sediment to support a diverse and productive benthic infauna (Pearson and Stanley, 1979; Wilding, 2006, 2012; Wildish et al., 2001). Redox is used as a proxy of the status of sediments around putative point-impact sources such as pulp-mills and aquaculture sites (Pearson and Stanley, 1979; Wilding, 2006, 2012; Wildish et al., 2001). Wildish et al. (2001) defined four zones, in relation to likely macrofaunal response, according to the measured redox (mV):  $>+100$  = normal,  $+100-0$  = transitory,  $0$  to  $-100$  = polluted,  $<-100$  = grossly polluted. These broad categories of redox v. macrofaunal response will be used here.

Redox decreases rapidly with depth into the sediment, with the greatest change occurring in the surficial layers and minimal changes occurring at sediment-depths exceeding 40 mm (Pearson and Stanley, 1979). For this reason, redox at  $>40$  mm sediment-depth can be used as a single-point metric of the “overall [redox] level down the sediment column” thus allowing between-sample comparisons (Pearson and Stanley, 1979) within a linear modelling framework.

Redox is normally measured remotely *in situ* (e.g. using a benthic lander) or in sediment cores that have been collected remotely, or by hand, and returned to the surface for analysis. *In situ* measurements have the advantage that they do not disturb the sediment compared with coring (Viollier et al., 2003) but are disadvantaged in heterogeneous (stony) sediments where the delicate probes are vulnerable to breakage, and where very high spatial accuracy is required. Taking cores, using a remotely deployed coring device, is both time consuming and of limited spatial accuracy ( $\sim 1$  m) but this latter disadvantage can be overcome using divers. However, using divers to collect and return cores to the surface for redox analysis, is relatively time-consuming and, consequently, costly.

Over the last ten years there has been increasing concern about the likely impacts of the development of the marine renewables industry with urgent calls for additional research (reviewed in Boehlert and Gill, 2010; Gill, 2005; Inger et al., 2009; Lin and Yu, 2012; Shields et al., 2011; Wilhelmsson et al., 2010) particularly in relation to likely the biodiversity consequences of such a major alteration of the marine environment. In addition, within the European Community and under the Marine Strategy Framework Directive (MSFD) Descriptor 7.1 and 7.2, there is a requirement for member states to achieve and maintain ‘good environmental status’ and to ensure that their marine activities (e.g. offshore construction) does not adversely affect marine ecosystems by altering hydrographic conditions (European Commission, 2008). There is also interest in the potential positive benefits of offshore structures, in relation to crustacean fisheries, through habitat creation (Langhamer et al., 2010; Linley et al., 2007). Crevice obligate species, such as lobsters, often show a preference for the interface between hard substrata and soft sediments as this allows the construction of bespoke burrows that are protected from above (Howard and Bennett, 1979). Understanding the mechanisms behind change occurring within this boundary area is, therefore, crucial in predicting the likely fishery consequences of the expanding marine renewable energy sector.

This research was conducted on the Loch Linnhe Artificial Reef (LLR) complex which is one of the largest of its kind in Europe (6230 t in total). The LLR is a purpose-built research facility, designed to address how man-made structures perform across a gradient of marine environments. The Loch Linnhe Reef most closely resembles the scour protection material (‘rip-rap’) that may be placed around the bases of turbines or along cable runs (Miller et al., 2013). Previous research on the LLR has shown that the sediments over the reef site area, prior to reef deployment, were predominantly oxic (Wilding and Sayer, 2002). Furthermore, Wilding (2006) observed that, following reef construction, substantial phytodetrital accumulations occurred at the perimeters of two reef modules (research conducted during July 2002 on reef-modules that are not part of the current study) and that this was associated with a reduction in redox and macrobenthic changes. The macrobenthic changes included an increase in opportunistic bivalves (Wilding, 2006). However, the seasonal variability in redox, and the spatial extent of measurable change, was not investigated at that time (Wilding, 2006).

The purpose of this research was not to test hypotheses of no change or impact (a logical fallacy; Anderson et al., 2000; Gigerenzer, 2004; Johnson, 1999) but rather to (1) give an estimate, with confidence intervals, of the spatial and temporal patterns in sedimentary redox in close proximity to the reef modules, (2) to use the redox proxy to infer to the broader consequences of reef-proximity to macrobenthic assemblages and (3) make

recommendations with regard the likely benthic consequences of the burgeoning offshore renewables industry.

## 2. Methods

The main part of the Loch Linnhe Reef is made of five reef-groups, each reef-group consisting of six individual modules giving a total of 30 modules (Fig. 1).

### 2.1. Site and dates

The three reef-groups (18 modules) that were used in this study (termed A, B and D) were deployed during May, August and September 2003, respectively, and were selected on the basis of their age-similarity and their location in contrasting current/sedimentary regimes. Each reef module, around which the measurements were made, consists of approximately 4000 concrete blocks, each block having external dimensions of  $200 \times 200 \times 400$  mm. Each module is between 3 and 4.5 m high, roughly conical in shape, with a diameter of 15–20 m. For the purpose of this study the reef-module's 'edge' consisted of the concrete block that was lying on the sediment corresponding to the random distance located by the diver. Reef groups A, B and D lie in approximately 18, 15 and 14 m (chart datum) of water respectively (Fig. 1). The redox monitoring work, which spanned 19 months, was conducted in March, May, July and November of 2004 and February, March, May, July, August, September and October in 2005. The reefs had, therefore, been in place at least six months prior to the initiation of sampling.

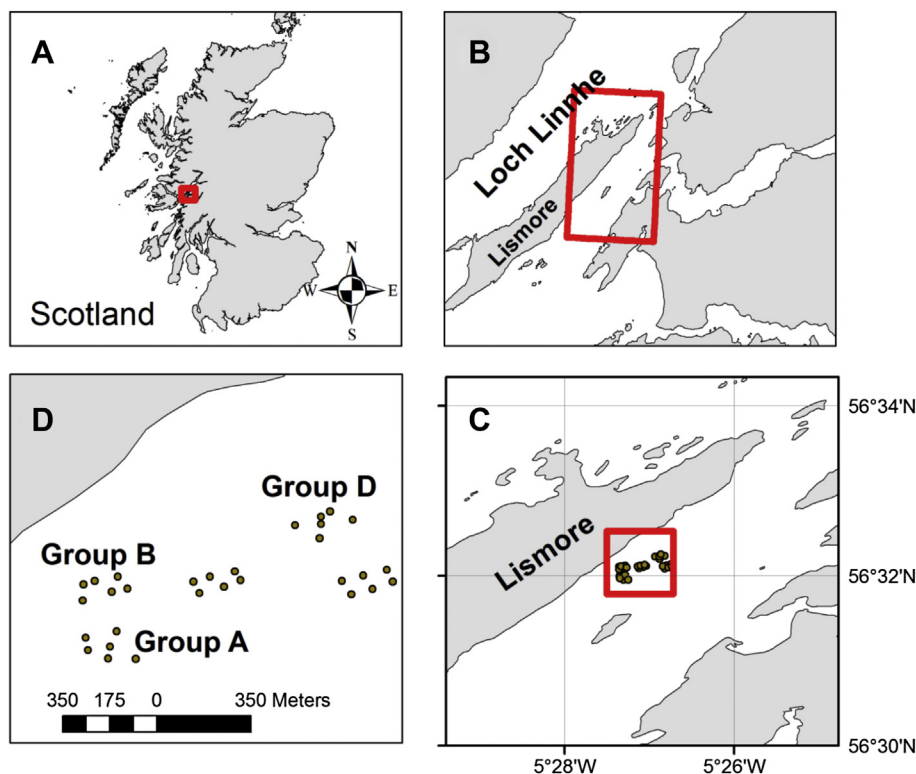
### 2.2. Measuring redox

Redox shows considerable heterogeneity over small-scales (Pearson and Stanley, 1979) and, consequently, there was a

requirement to take as many measurements as possible around the reef to assess this variability and increase the precision of any main-effect estimates. The requirement for both sufficient replication and a high degree of spatial accuracy ( $\pm 10$  cm) meant that an *in situ* method of measuring redox was required. In order to achieve this, a waterproof redox probe (Russel pH Ltd., Auchtermuchty, Scotland, Model CMPtR106/150 mm) connected to a waterproof-housed meter (Testo-term 2303) was used, by diver, *in situ*. Immediately prior to use, whilst on the surface, the probe was checked by reference to a proprietary standard solution (redox potential of 125 mV, Russel pH Ltd). Redox measurements were taken by inserting the probe into the sediment to a depth of 80 mm. The sediment depth of 80 mm was chosen for four reasons: (1) previous research had indicated that the pre-deployment (baseline) sediment at the reef-site was oxitic at this sediment-depth (Wilding and Sayer, 2002), (2) that achieving very accurate depth penetration by the probe was difficult underwater meaning the errors were proportionately less the greater the sediment-penetration-depth, (3) that at 80 mm the probe could be left standing, unassisted, in the sediment until the reading had stabilised thus eliminating diver-caused probe shake and (4) as per the recommendation given in Pearson and Stanley (1979) for between-station comparisons. Between measurements, on the same dive, the probe was cleaned by shaking it in the surrounding seawater until a highly positive reading was observed. Where necessary any phytodetrital material was moved to one side prior to inserting the probe. Reported probe readings were adjusted to the hydrogen scale by the addition of 198 mV (Zobell, 1946) and adjusted for temperature (SEPA, 2005).

### 2.3. Physical parameters

Water current speed data were generated over the entire reef site during August 2004 (spring tides, 4.0 m range) using a research



**Fig. 1.** Location of the Loch Linnhe Artificial Reef with increasing detail (A–D). A: Scotland (excluding Shetland Isles). B: detail from A showing part of Loch Linnhe and the island of Lismore. C – detail from B showing reef location in Loch Linnhe. D: detail from C showing the 30 reef units in the main reef-matrix with reef groups, each of six individual modules, marked A–D. The co-ordinates shown in C are in degrees and minutes, WGS 84 datum. A is shown in the British National Grid projection.

vessel-mounted acoustic Doppler current profiler (RD Instruments, Mariner, 300 kHz) set to record at 60 Hz. The survey vessel's course ran approximately NE–SW, parallel to the shore of Lismore, at a speed of 6–8 knots. The survey consisted of four survey tracks, each approximately 150 m apart. Each survey track ran over, or in close proximity to, the reef groups and each was surveyed 9 times during the 12.5 h survey period (one complete tidal cycle). The current speed data from within 75 m of the centre of each reef group was extracted. ADCP measure current speeds throughout the water column, however, in this case only the current data for the lowest measurable depth (10% of water depth above the seabed) were used to more closely reflect the current environment around the reef modules on the seabed. Outliers were removed by excluding the highest 1% of recorded currents prior to the calculation of median values and the first and third quartiles.

#### 2.4. Statistical analyses

The response variable was redox. The distance effect was the main factor of interest. Distances of 0, 1 and 4 m from the reef edge were chosen on the basis of prior observations (Wilding, 2006) and Distance was, therefore, considered fixed. The effect of location (Reef Group) on the distance effect was also of interest. The reef groups were chosen on the basis of their differing characteristics (current exposed or unexposed) and were, therefore, considered fixed. The reefs were sampled over time in order to estimate temporal effects. Of primary interest were major seasonal differences in the effect of location and distance. Two seasons were considered, nominally referred to here as winter and summer reflecting water temperature (less than 10 °C and more than 10 °C respectively). Season was, therefore, also considered fixed.

For each sampling time (Month) two individual reef modules from each group of six (Group) were randomly selected. At each reef-distance 10 redox measurements were taken, the locations of which were randomly allocated by the diver swimming for a pre-selected random time of between 1 and 15 s around the reef perimeter (0 m stations) or guided to 1 and 4 m stations using a marked rope. The objective was to take 180 measurements per time interval (3 groups, 2 modules from each group, 30 readings per module). However, during periods of poor weather this sampling programme was not completed and the following numbers of modules per group (A, B and D) were measured on the following dates: March 2005 A: zero, B: one and D: two; September 2005 A: zero, B: one and D: one and October 2005 A: two, B: one and D: one. At all other occasions the full sampling programme was achieved.

Two dives were permissible per day resulting in a minimum of three consecutive days to visit the six modules (two modules on each of three groups). During poor weather the period over which a single time-period's data were collected was extended up to seven days. These data were considered to represent one time period. Visual assessments of the reefs and the surrounding environment were made, particularly in reference to any accumulations of organic material and the nature of the sediment. The bottom-water temperature was recorded using an integral depth gauge and thermometer during each dive. The mean temperature for each month is reported.

##### 2.4.1. Data analysis

Pre-analysis data exploration (checking outliers, homogeneity, normality) followed the protocol of Zuur et al. (2010). Model development and selection in mixed models can be relatively complex (and iterative) and the guidance given in Zuur et al. (2009), detailed below, was followed:

1. The beyond optimal (all fixed effects and interactions) model was initially fitted using generalised least-squares regression and the residuals examined for homoscedasticity. If any residual trends were identified a range of variance structures were tested and compared on the basis of their Akaike information criteria (AIC) score (where the lowest AIC was considered the optimal model). The goal was to identify, and allow for, differences in variance as a function of either one or more categorical predictors. Residuals from the model with the lowest AIC were reassessed to check that any heteroscedasticity had been incorporated into the model.
2. The next step was to identify the optimal model containing random components (if necessary). A range of models were trialled (allowing for random differences between modules and modules nested within groups) and the optimal model chosen based on the lowest AIC. Temporal autocorrelation was checked by plotting residuals v. month and by plotting the monthly mean residual autocorrelation.
3. The significance (or otherwise) of the fixed effects was then determined based on maximum likelihood (ML) parameter estimates. The full model was fitted and model terms sequentially tested using a likelihood ratio test. Insignificant terms ( $P > 0.1$ ) were removed.
4. Model validation was then conducted based on an analysis of residual patterns. The standardised residuals were plotted against predictors and assessed for normality and homoscedasticity. Patterns in the residuals resulted in a reassessment of the model.
5. The final model was re-fitted using REML and is reported.

All model predictions, and 95% confidence intervals (shown graphically) relate only to the fixed factors. The variability accounted for by random factors is given by the standard deviation of the random effect's intercept term. The heteroscedasticity accounted for by the model is reported in the weightings and relates to the relative standard deviation, for each stratum, compared with the base level. Mixed effect models were developed in R (R, 2009) using the 'nlme' package (Pinheiro et al., 2012). Graphical representations were made using the 'effects' package (Fox, 2003).

### 3. Results

#### 3.1. Physical description

The physical environment around each of the reef groups was, visually, very different despite their close proximity. Around Group A the sediment was flat, soft and muddy while around Group B the sediment consisted of numerous cobbles and stones in a muddy matrix. The sediment surrounding Group D consisted of coarse sands and gravels intermixed with mud and overlain with large stones and occasional boulders. There was no evidence (by visual inspection) of any sediment-scouring around any reef at any time. Further site details are provided in Wilding and Sayer (2002).

The water column at the experimental site was often seen to contain drifting phytodetritus consisting of detached macroalgal fronds (mostly *Laminaria* sp). The phytodetritus was, periodically, seen to accumulate around the modules in Groups A and B but not Group D. The patches of accumulated phytodetrital material varied in extent, from simply being trapped in among the blocks at the module edge, to accumulations of approximately 0.5 m depth which were patchily distributed along the reef edge extending outwards by 1–2 m. On Groups A and B, particularly during the late summer and autumn, the phytodetrital material appeared to be relatively broken down, consisting of unrecognisable organic fragments, and was occasionally associated with colonies of the

sulphate reducing bacteria *Beggiatoa* sp (identified as a pale, fibrous mat growing atop the organic material). The sediment underneath accumulated phytodetritus was frequently very dark or black (indicative of reducing conditions) compared with non-covered sediments which were typically light brown. None of the reef modules were extensively colonised with macroalgae during the sampling period.

### 3.2. Current speed and temperature

Current speeds around the reef modules varied, with a maximal median value occurring at Group D (44 cm s<sup>-1</sup>) with current speeds around Groups A and B showing the same median value of 37 cm s<sup>-1</sup> (Table 1).

The temperatures recorded over the duration of the survey ranged between 7 and 14 °C. Mean temperatures (°C) recorded in 2004 were: March: 7.1, May: 8.9, July: 11.4, November: 11.7 and during 2005 were: February: 7.7, March: 7.3, May: 9.0, July: 11.8, August: 12.9, September: 13.5 and October: 13.2.

### 3.3. Redox probe performance and redox measurements

The housed redox probe performed well and was sufficiently robust to survive the duration of the sampling period despite being used in sediments that consisted of consolidated muddy-sands and which frequently contained stones. Each measurement took between 45 and 60 s allowing 30 measurements to be taken per dive.

Over the entire survey the redox ranged between -162 and 388 mV but was predominantly (~95%) oxid (>0 mV) with a median of 128 mV, IQR 61 mV ( $n = 1740$ ). However, there were considerable differences between Reef Groups, Distances and Seasons. At Group A, at the reef edge (0 m) and during the summer, nearly half of measurements indicated hypoxia (<0 mV). This contrasted markedly with 4 m distance, at the same reef group, where none of the stations were hypoxic and during winter where the proportion indicating hypoxia/anoxia, at the reef edge, was much lower (23%) (Table 2, Fig. 2). This trend, of increased hypoxia during summer, and as a function of reef-proximity, was also seen, but of reduced magnitude, at Group B but virtually absent at Group D (Table 2, Fig. 2). However, at Group D there was a trend of increased proportions of samples that were 'transition' (sensu Wildish et al., 2001) as a function of season and reef-proximity (Table 2, Fig. 2).

In close proximity to the reef, redox was highly variable, for example on Group A, during the summer, redox varied between -160 and +190 mV at the reef edge (Fig. 2). In terms of the random effects, within reef groups, there were differences between modules (Table 3). There was also higher variability in redox during summer months compared with winter months (standard deviation multiplier ranged between 0.50 and 1.3) and 1.6 × the variability in redox at 0 m compared with 4 m (see weightings in Table 3).

In terms of the modelled fixed effects, mean redox differed between distances but this was influenced by both the reef location and season (Fig. 3). Redox was lower in close proximity to the reef (compare zero and 1 m distance, Fig. 3), and this difference was

**Table 1**  
Current speeds (cm s<sup>-1</sup>) (1st quartile, median and 3rd quartile) at the three Groups (A, B and D). Survey details are provided in the main text.

| Reef group | N    | Q1   | Median | Q3   |
|------------|------|------|--------|------|
| A          | 1262 | 22.4 | 36.6   | 56.2 |
| B          | 1261 | 24.0 | 37.3   | 54.5 |
| D          | 1223 | 27.6 | 43.7   | 65.8 |

**Table 2**

Summary of redox (mV) at all combinations of Distance, Season and Group. The transition zones (trans, 100–0 mV) and hypoxic (0 to -100 mV) are highlighted in Fig. 2 (designations based on Wildish et al. (2001)). Only 6 readings were <-100 mV.

| Distance (m) | Season | Group | Minimum | % trans | % hypoxic | Median | Sample size |
|--------------|--------|-------|---------|---------|-----------|--------|-------------|
| 0            | Summer | A     | -160    | 39      | 43        | 20     | 100         |
| 1            | Summer | A     | -50     | 32      | 4         | 120    | 100         |
| 4            | Summer | A     | 50      | 35      | 0         | 110    | 100         |
| 0            | Winter | A     | -60     | 21      | 11        | 120    | 80          |
| 1            | Winter | A     | 40      | 13      | 0         | 150    | 80          |
| 4            | Winter | A     | 50      | 15      | 0         | 140    | 80          |
| 0            | Summer | B     | -160    | 38      | 23        | 95     | 110         |
| 1            | Summer | B     | -80     | 30      | 5         | 125    | 110         |
| 4            | Summer | B     | -20     | 35      | 1         | 110    | 110         |
| 0            | Winter | B     | -20     | 25      | 3         | 140    | 100         |
| 1            | Winter | B     | 0       | 9       | 1         | 150    | 100         |
| 4            | Winter | B     | 60      | 5       | 0         | 140    | 100         |
| 0            | Summer | D     | -20     | 60      | 2         | 90     | 90          |
| 1            | Summer | D     | 0       | 23      | 1         | 120    | 90          |
| 4            | Summer | D     | 50      | 20      | 0         | 120    | 90          |
| 0            | Winter | D     | 10      | 18      | 0         | 150    | 100         |
| 1            | Winter | D     | 0       | 4       | 1         | 150    | 100         |
| 4            | Winter | D     | 80      | 6       | 0         | 150    | 100         |

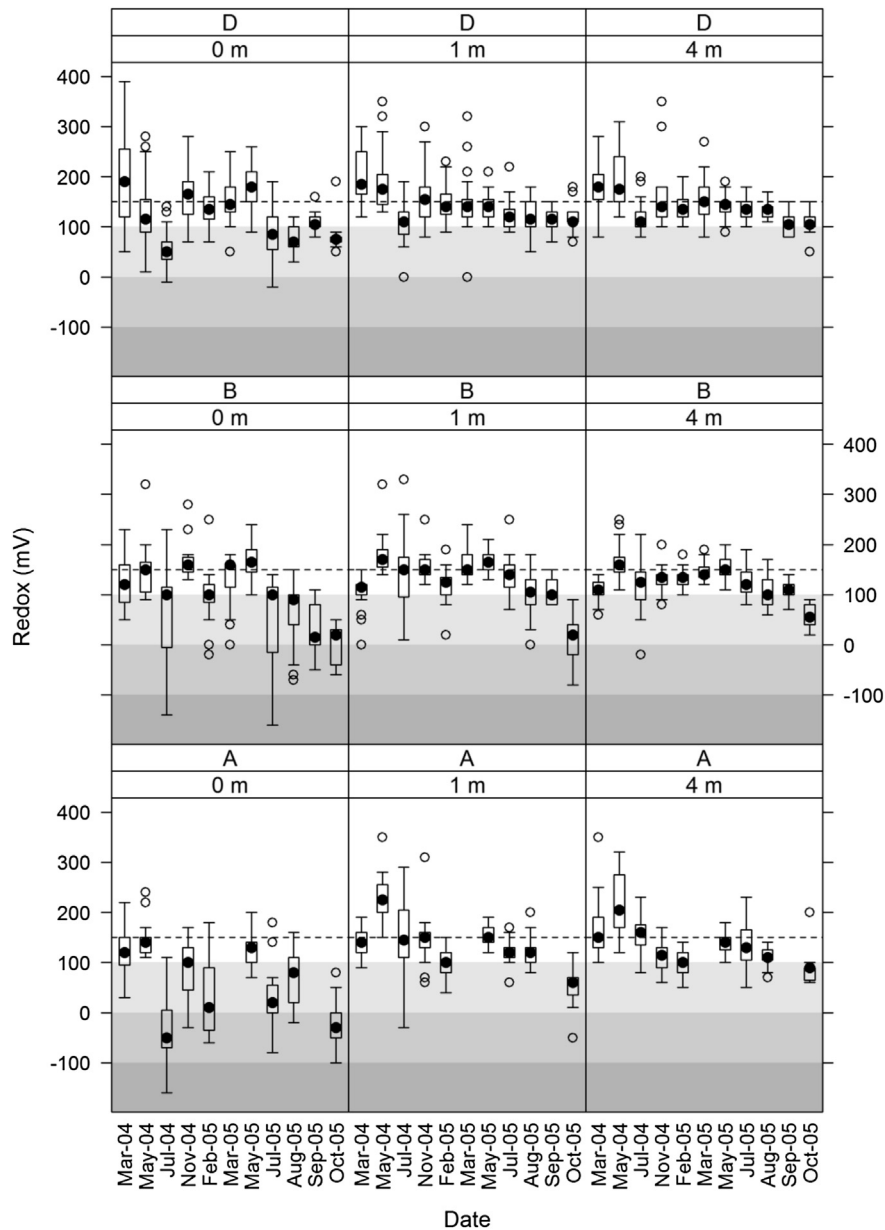
maximal during the summer, particularly at Group A, with projected means, at the reef-edge, being lower by 40–120 mV (95% CI) (Fig. 3). This affect was still discernible, but of reduced magnitude, at Group B, but only during the summer (Fig. 3). At Group D there were negligible differences in mean redox as a function of distance regardless of season (Fig. 3) but, across all Groups and Distances, there was a general trend of redox levels being lower in the summer compared to winter (Fig. 2). The exception to this seasonal trend occurred during February 2005, at Group A (0 m), where negative redox values were recorded (Fig. 2). The confidence intervals shown in Fig. 3, for distances 1 and 4 m, are entirely overlapping at all combinations of Season and Group and this is interpreted as indicating that the discernible impacts, on redox, of the reef did not extend beyond 1 m.

## 4. Discussion

The measurable impacts of the LLR, on sedimentary oxygenation status, did not extend more than 1 m from the reef edge. At the reef edge, redox levels were highly variable with a mean expected reduction of 80 mV during the summer, at Group A. At other reef groups reef-proximity had less of an effect and there was a clear trend of decreasing change in mean redox from Group A to B to D and from summer to winter. These results are discussed within the context of logistical issues and the likely extent of changes in redox and associated benthos around artificial structures such as MREDS.

### 4.1. Monitoring logistics

Environmental monitoring programmes are, inevitably, financially constrained and consideration must be given to monitoring methodologies that provide the greatest degree of sensitivity for a given cost. Minimising the cost of individual measurements allows commensurately greater temporal and/or spatial coverage and a superior assessment of the spatial and temporal nature of impacts. Redox, assessed at a single sediment depth, is a useful measure of the impact status of sediment subject to organic enrichment (Pearson and Stanley, 1979) and it has been used widely in assessing impacts from aquaculture operations including fish (Wildish et al., 2001) and shellfish (Cranford et al., 2009; Otero et al., 2006; Wilding, 2012) and in relation to artificial reefs (Wilding, 2006). The linkage between redox status, oxygen



**Fig. 2.** Box plot showing interquartile range (box), upper and lower adjacent values (serifs) and outlying observations (circles) ( $n = 30$  with some exceptions, see main text) for Redox measured at each Distance (0, 1, 4 m) and Group (A, B, D) over time. Note the anomalously low redox recorded for Feb 05, at Reef A, Distance 0 m. Three redox regions are delimited: very light grey = transition (100–0 mV), light grey = hypoxic (0 to –100 mV), dark grey = anoxic (<–100 mV) (thresholds from Wildish et al., 2001). The dotted line (150 mV) indicates the general background redox for these stations, at 8 cm sediment depth (from Wilding and Sayer, 2002).

concentration and macrobenthic community is well-established and it is fair to infer from changes in redox to changes in macrobenthic assemblages (Diaz and Rosenberg, 1995; Pearson and Rosenberg, 1978; Pearson and Stanley, 1979). Most impacts studies using redox are based on remotely or diver collected cores (e.g. Callier et al., 2007; Otero et al., 2009) or where redox is taken from the surface of grab samples (Miron et al., 2005; Wildish et al., 1999). Such approaches are time-consuming and/or of limited spatial resolution. These disadvantages were overcome in the current study through the development of a simple, robust, hand-held redox probe that could be used, by diver, *in situ*. This method allowed numerous measurements per dive, thereby reducing the per-measurement cost. The research reported here details the most comprehensive, single-sediment-depth, assessment of the fine-scale spatial variability in redox, measured over time, in the

marine environment to date. The technique described here is recommended for use in similar circumstances.

Assigning cause and effect in manipulative field experiments is, in any circumstance, made difficult by the presence of confounding factors. In the current case, the reef units were replicated (within reef-group) and these groups were characterised by quite different receiving environments: Group A and B were exposed to approximately equivalent current regimes, whilst Group D was more exposed. The substratum at Groups B and D contained more rocks and stones compared with Group A. The lack of control of potentially confounding factors in this type of field observation prevents inference to individual factors. In the present case different current exposures will be linked to differences in the background sediment conditions and either of these, or other related factors, could influence redox.

**Table 3**

Results from Linear Mixed Weighted Model. The response is Redox (mV). The random effects are Module and the Residual. Weightings indicate the degree of heterogeneity in each stratum of the factor of interest. Months are coded 3 (March 2004)–22 (Oct 2005). Fixed effect estimates (Distance, and Season and their second order interactions) are also shown. The base level in this model (indicated as the intercept) is Distance = 0 m, Season = Summer, Group = A. The fixed-effect predictions from this model, and 95% CIs, are shown in Fig. 3.

| Random effects  |          |           |           |          |       |       |       |       |       |       |       |
|---|----------|-----------|-----------|----------|-------|-------|-------|-------|-------|-------|-------|
| per Module  |          |           | Intercept | Residual |       |       |       |       |       |       |       |
| Standard deviation  |          |           | 12.4      | 76.4     |       |       |       |       |       |       |       |
| Weightings as standard deviation (Stand. dev) per stratum |          |           |           |          |       |       |       |       |       |       |       |
| Distances (m)   | 0        | 1         | 4         |          |       |       |       |       |       |       |       |
| Stand. dev  | 1.00     | 0.721     | 0.594     |          |       |       |       |       |       |       |       |
| Months  | 3        | 5         | 7         | 11       | 14    | 15    | 17    | 19    | 20    | 21    | 22    |
| Stand. dev  | 1.00     | 1.26      | 1.23      | 1.10     | 0.855 | 0.795 | 0.501 | 0.769 | 0.638 | 0.511 | 1.093 |
| Fixed effects   |          |           |           |          |       |       |       |       |       |       |       |
|   | Estimate | St. error | T value   | P value  |       |       |       |       |       |       |       |
| (Intercept)   | 29.96    | 9.19      | 3.26      | <0.01    |       |       |       |       |       |       |       |
| 1 m   | 81.52    | 6.58      | 12.40     | <0.01    |       |       |       |       |       |       |       |
| 4 m   | 81.52    | 6.21      | 13.14     | <0.01    |       |       |       |       |       |       |       |
| Winter  | 69.48    | 6.00      | 11.58     | <0.01    |       |       |       |       |       |       |       |
| B   | 36.00    | 6.95      | 5.18      | <0.01    |       |       |       |       |       |       |       |
| D   | 65.43    | 7.33      | 8.93      | <0.01    |       |       |       |       |       |       |       |
| 1 m: Winter   | -41.68   | 6.19      | -6.73     | <0.01    |       |       |       |       |       |       |       |
| 4 m: Winter   | -46.72   | 5.84      | -8.00     | <0.01    |       |       |       |       |       |       |       |
| 1 m: B  | -31.30   | 7.61      | -4.11     | <0.01    |       |       |       |       |       |       |       |
| 4 m: B  | -33.87   | 7.18      | -4.72     | <0.01    |       |       |       |       |       |       |       |
| 1 m: D  | -53.20   | 7.87      | -6.76     | <0.01    |       |       |       |       |       |       |       |
| 1 m: D  | -48.98   | 7.43      | -6.60     | <0.01    |       |       |       |       |       |       |       |
| Winter: B   | 0.99     | 5.26      | 0.19      | 0.85     |       |       |       |       |       |       |       |
| Winter: D   | 11.22    | 5.52      | 2.03      | 0.04     |       |       |       |       |       |       |       |

#### 4.2. Implications of spatial and temporal variability in redox

Redox at the reef edge was lower overall, and associated with higher variability, compared with 1 m and 4 m distance. The most likely mechanism for this change in redox is the patchy accumulation of drifting phytodetritus that was commonly observed in close proximity to the reef (see also Wilding, 2006) and which was noted as a common feature during pre-deployment survey work (Wilding and Sayer, 2002). The most plausible mechanism linking the reef-modules, drifting phytodetritus and reductions in redox is a baffling of water currents by the reef structure and the subsequent deposition of entrained material. This hypothesised mechanism is supported by hydrological modelling which has predicted a reduction in water currents in close proximity to the reef (Al-Bourae, 2013). The depositional environment at the reef edge, reported here, contrasts with that reported around other artificial structures, for example Davis et al. (1982), Ambrose and Anderson (1990) and Barros et al. (2001) (collectively referred to as DAB Reefs from here) report a coarsening of the sediment, and by inference, an increase in current speed, at the boundary of their study-reefs. The impact-differences between the DAB Reefs and the LLR reef-modules may be attributed to the adjacent substratum: the DAB reefs were located on a fine sand contrasting markedly with the LLR site which consists of a cohesive, muddy-sand (Wilding, 2006; Wilding and Sayer, 2002). In the case of the LLR, the piles of concrete blocks may offer a semi-permeable barrier to water thereby effectively acting to baffle, rather than deflect and accelerate, water flow around the perimeter. This baffling-effect is in-line with findings of Fabi et al. (2002) and Guiral et al. (1995) who both report increased fine material associated with artificial structures.

A simple reduction in current speed, over the sediment, will result in a decrease in the advective delivery of oxygenated water to the sediment surface (Diaz and Rosenberg, 1995; Ziebis et al., 1996). This may explain the findings around Group D. Group D was exposed to relatively high water flow and phytodetritus was not seen to accumulate around it at any time. The minor reductions in redox at the reef edge (Group D), which only occurred during the

summer, may represent the consequences of hydrographic interactions that are independent of the deposition of phytodetritus.

The lower sedimentary redox observed during the summer and autumn, compared with the rest of the year, were predicted as previous research had shown the accumulation of phytodetritus during that period (Wilding, 2006). The ~80 mV reduction at the reef edge reported here is commensurate with that found at the edge of Loch Linnhe mussel farms, at 20 mm sediment depth, and which was associated with an increase, by between 1.8 and 8×, in macrofaunal abundance (Wilding, 2012; Wilding and Nickell, 2013). It is likely that localised enhancement of the benthic infauna at the edges of offshore structures would enhance reef-periphery-based fisheries, including lobsters, particularly given the lobster's tolerance to hypoxia (Diaz and Rosenberg, 1995).

The positive correlation between higher water temperatures and the abundance of phytodetritus, such as that occurring during summer/autumn, makes it difficult to distinguish the relative importance of each factor, as a driver of redox, at the reef edge. However, the accumulation of phytodetritus at Group A in February 2005, followed unusually violent storms during the previous month, and was associated with a clear reduction in redox at the reef edge. This indicates the major factor determining redox around the LLR was the accumulation of phytodetritus rather than water temperature. This hypothesis is supported by the relatively small reduction in redox that was observed at the reef edge of Group D, where phytodetritus was never observed to accumulate. In the current case, at the most impacted stations (Group A, reef edge, summer), sedimentary hypoxia (redox of <0 mV) was commonly observed indicated a moderate degree of impact (as defined by Wildish et al., 2001). However, this change in sediment was rarely observed at 1 m or more and, even at the reef edge, was highly patchy. This patchy reduction in redox is in line with the impact being caused by phytodetrital accumulation and subsequent periodic isolation of the seabed from the overlying water column. The data presented here indicate that MREDs will be associated with a moderate degree of impact where located in sedimentary environments where phytodetrital accumulations can occur but that

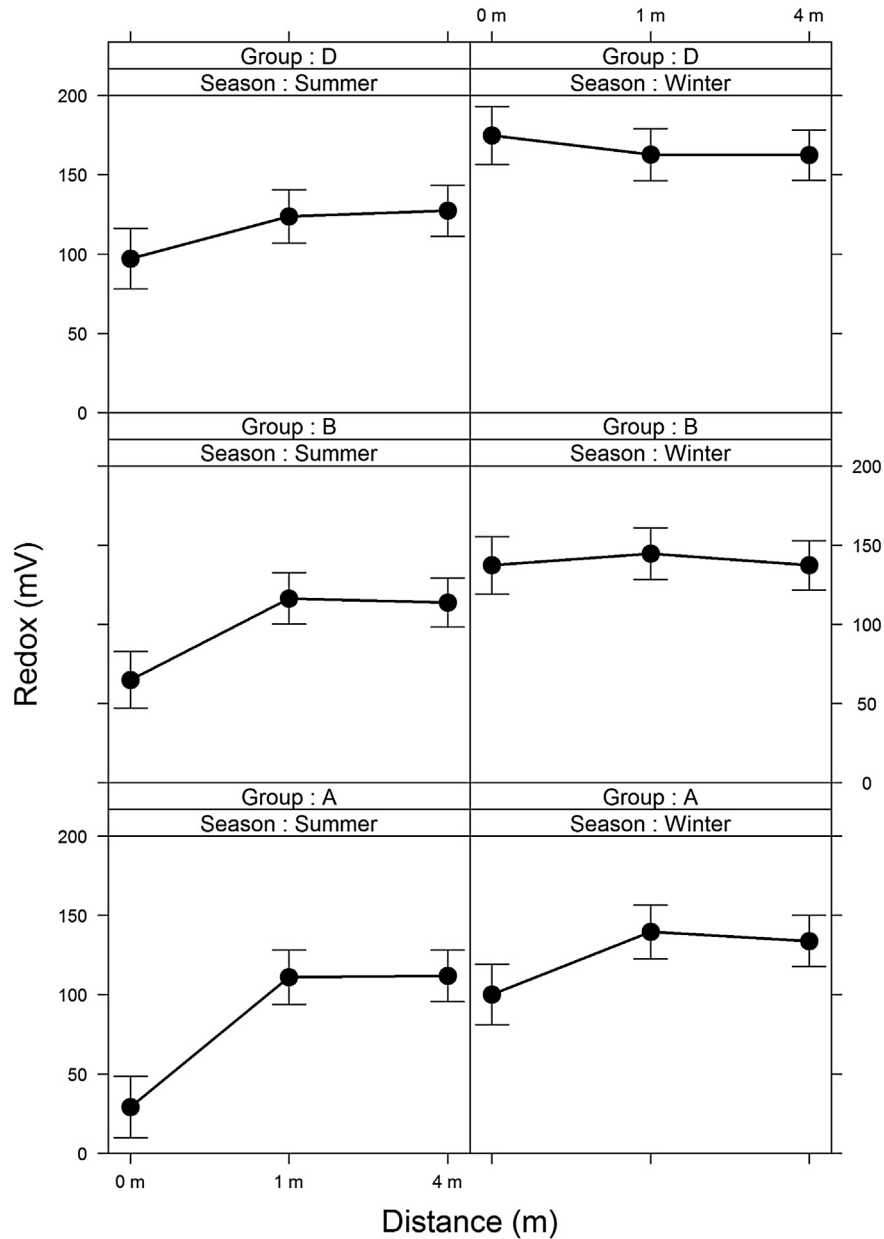


Fig. 3. Model output (mean expected values and 95% confidence interval for that mean) for each combination of Distance, Season and Group. The major change occurs during summer, Group A between Distances 0 and 1 m.

these impacts are likely to be of limited spatial extent. The MFSD itself does not specify limits or thresholds beyond which change is unacceptable (European Commission, 2008) but it seems unlikely that the spatial extent, and nature, of the change reported here would be considered problematic. The results presented here are in broad agreement with the conclusion of Wilhelmsson et al. (2010) that detectable (meaningful) benthic impacts around offshore structures are limited.

## 5. Conclusions

MREDs and associated infrastructure will become *de-facto* artificial reefs. Where located in temperate coastal waters, on cohesive sediments, the results presented here indicate that reef-proximal sediments are likely to remain relatively unchanged, in terms of oxygenation status, except in cases where significant quantities of

macroalgal detritus are trapped by the reef structure. This is likely to occur in areas subject to moderate water flows, where there is a supply of detached macroalgae (e.g. following infrastructure cleaning operations or storms) and where there is significant baffling of water currents around the structures. The consequence of moderate organic enrichment, by phytodetritus or other debris, is likely to be an increase in localised benthic productivity, potentially benefiting some fishery species. However, in areas already subjected to oxygen stress, for example the Baltic Sea, the accumulation of organic matter around offshore structures may exacerbate sedimentary hypoxia leading to a localised reduction in benthic productivity and the exclusion of some fishery species. Additional research is required to identify other factors that are likely to influence the utilisation of the proposed MRED structures by valuable commercial species and how to maximise this potential through design modification and site selection.



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