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Benthic monitoring and sampling design and effort to detect spatial changes: A case study using data from offshore wind farm sites

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ABSTRACT

The exploitation of renewable energies, in particular offshore wind farms (OWFs), is an expanding sector which involves activities that may adversely affect the marine benthic ecology. Fit-for-purpose monitoring is required with sufficient statistical power to detect ecologically meaningful changes, but to date there have been no studies on the suitability of monitoring programmes applied to OWFs. The theoretical relationship of sampling effort with precision in community estimates and sensitivity of the analysis in detecting spatial changes was investigated, this latter assessed through power analysis. Benthic community monitoring strategies and descriptors applied to UK OWFs were used to interrogate real data variability in the marine environment. There was a general lack of clarity in the survey rationale and hypotheses tested within OWF monitoring programmes hence a lack of rigour in the survey design and statistical testing. Consequently the statistical properties of monitoring strategies have been rarely assessed. Precision of mean estimates of benthic community descriptors and the sensitivity in detecting differences in the means increased with sampling effort. At the average sampling effort applied in the OWF case studies (4 stations per impact type area and 3 replicates per station), the studies had sufficient power to detect a ≥50% change between areas in mean benthic species richness (S; 5 species). Due to their higher variability than S, more stations per impact type area were required to reliably detect a ≥50% change between areas in mean benthic abundance (N; 5 stations) and mean biomass (B; 10 stations). Higher sensitivity and precision of estimates of S, N and B was achieved with transformation of data. Understanding the general implications of monitoring design on the sensitivity of the detection of spatial changes is important, particularly when monitoring effort has to be adjusted due to logistic and financial constraints. Although there is no 'one-size-fits-all' approach to marine environmental data acquisition, this study guides researchers, developers and regulators in optimising benthic monitoring strategies at OWFs.

1. Introduction

All human activities in the marine environment have the potential to adversely affect the natural system (Gray and Elliott, 2009). Renewable energy generating devices lessen the depletion of non-renewable resources and have perceived lesser environmental effects (Gill, 2005). Offshore wind generating capacity in particular is the most rapidly expanding sector of the renewable energy industry (Wilson et al., 2010) and the UK is globally leading this with as much capacity already installed as the rest of the world combined (RenewableUK, 2014).

Offshore wind farms (OWFs) produce 'green energy'. Their construction, operation and decommissioning, however, may impact the composition and structure of benthic communities through loss or change of habitat and physical disturbance of the seabed in ways that are difficult to measure, minimise and mitigate (Gill, 2005; Wilson et al., 2010). Whether these effects constitute an ecologically significant impact depends on their direction, duration, extent and magnitude, and on the value and sensitivity of the receiving habitats and organisms (Boehlert and Gill, 2010; IEEM, 2010; Wilson et al., 2010; Garel et al., 2014). Monitoring the condition of the benthos is a condition of the operating license for an OWF. The developer has to prove that the OWF will not cause harm rather than the regulator having to show that harm will occur (Gray

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and Elliott, 2009). Hence environmental impact assessment (EIA) and response is urgently needed in the renewable energy sector (Inger et al., 2009; Vaissière et al., 2014) despite there being large knowledge gaps (Garel et al., 2014).

The European Directive 2011/92/EU requires that an EIA is carried out for the consent of projects having significant effects on the environment, including OWFs (CEFAS, 2004). In the resulting Environmental Statement (ES), the main stressors and receptors should be identified and the significance of potential impacts assessed. The consenting process should test impact hypotheses in construction and operation and validate predictions (Judd, 2012). The existing guidance for monitoring and assessment of potentially impacting activities in the marine environment, including OWFs (CEFAS, 2004; Judd, 2012; IEEM, 2010), inevitably can only be generic rather than a highly prescriptive methodology, largely because of site – specificity and the questions being asked regarding habitat distribution, diversity and heterogeneity (CEFAS, 2004; Judd, 2012).

Environmental monitoring usually aims to investigate changes relative to a defined baseline condition or set of parameters to quantify any impact. Changes are assessed before and after construction, during construction vs. pre-construction, inside vs. outside the wind farm array, while also accounting for temporal and spatial natural variability (Judd, 2012). Sampling programmes should allow hypothesis-testing statistical techniques usually based on a Before–After-Control-Impact (BACI) Paired-series approach or its modifications (Underwood, 1994; Ellis and Schneider, 1997). Whether the monitoring is aimed at assessing an impact or characterising spatial variability of baseline conditions, an adequate sampling effort is required (CEFAS, 2004; Judd, 2012) to quantify parameters with a certain level of precision and sufficient statistical power to detect the signal of change, minimise the risk of Type I and II errors and correctly reject the null hypothesis (Zar, 1999). Power analysis is capable of informing sampling design during the planning stage of a study (prospective power analysis; Cohen, 1988; Underwood and Chapman, 2003), and it can be applied also after the data have been collected and analysed to evaluate the adequacy of a specific design in detecting biologically meaningful patterns (retrospective power analysis; Andrew and Mapstone, 1987; Thomas, 1997).

CEFAS et al. (2010) recently reviewed UK FEPA (Food and Environment Protection Act 1985) OWF monitoring datasets to give preliminary recommendations on sampling adequacy, but to date there are no studies specifically appraising the suitability of monitoring programmes to detect variability in the status of the marine environment at OWF sites. The present study aims to integrate existing experience to guide suitable monitoring strategies of benthic communities at OWFs. Survey data and information from a selection of UK OWF monitoring studies were interrogated with the following objectives: (1) to review benthic monitoring strategies applied to OWFs in the light of existing guidance for EIA monitoring of benthic communities; (2) to assess the precision of mean estimates of benthic community descriptors in relation to the sampling effort at the station level, and (3) to apply power analysis in order to identify the overall most appropriate monitoring effort needed to detect spatial variability in benthic communities with a certain statistical power.

2. Materials and methods

2.1. Dataset, survey strategies and benthic variables

The UK offshore wind energy generating sector comprises several licensing phases co-ordinated by the Crown Estate (the landlord and owner of the seabed), with Round 1 launched in 2001, Round 2 in 2003 and Round 3 in 2010. Subtidal benthic survey data from a selection of Round 1 and Round 2 wind farms were compiled from ES and monitoring reports, the COWRIE (Collaborative Offshore Wind Research into the Environment) website (http://www.offshorewindfarms.co.uk) and also from individual developers (Table B.1).

OWF benthic sampling regimes were summarised using several parameters, including sampling method, number of surveyed stations and replicate samples collected per station. Non-parametric analysis (Mann–Whitney *U* test) assessed differences between Round 1 and Round 2 OWF monitoring programmes.

Monitoring designs at the studied OWFs located sampling stations within and around development sites, often by distinguishing areas based on the expected distribution of impacts generated by the OWF. Criteria for station allocations to sampling areas were derived from the description of sampling regimes and survey maps as provided in the monitoring reports. According to these, stations were located within the OWF area and in some cases within the near-field area of the wind turbine foundations to determine scour effects. Stations were also often sited along the OWF cable corridor, around the development site within one tidal excursion from it (e.g., within the area affected by sediment transport and deposition; BOWind, 2007) or outside the tidal excursion to represent control areas. All these areas were classified in this study respectively as DS (development site), SA (scour assessment), CC (cable corridor), SI (secondary impact) and reference/control sites (RS). Survey strategies were reviewed in the light of existing guidance for monitoring benthic communities and for EIA of OWFs. Primary benthic community descriptors (mean species richness S, total benthic abundance N and biomass B) were derived from each dataset, depending on data availability.

2.2. Power analysis and precision assessment

Power analysis was employed to investigate the theoretical relationship between the sampling effort applied in monitoring designs and the size of the detectable change in mean S, N and B (minimum detectable effect size, MDES). The sample variances used in the power analysis were derived from ANOVAs on the benthic data collected at the studied OWF sites. By using data from a wide variety of case studies our findings apply as measures of central tendency for the group as a whole.

The ANOVA model applied to the OWF study designs is a 2-level nested ANOVA (Sokal and Rohlf, 1995). It partitions the variance in the measured variables due to the main factor Area (factor *A*) and the nested factor Stations within Area (factor $B(A)$), and tests for differences among Areas. The null hypotheses tested in this study are no differences in the means of S, N and B among impact type areas (H_0: DS $\neq$ SA $\neq$ CC $\neq$ SI $\neq$ RS). The minimum effect size (i.e., the difference between mean values of the analysed variable) that can be detected by the ANOVA was calculated as (Ling and Cotter, 2003)

$$\mathrm{MDES} = \Phi^{-1}(P) \cdot \sqrt{2as_Y^2} \tag{1}$$

where *P* is the power of the statistical test, Φ^{-1} is the inverse of the normal distribution function Φ, *a* is the number of areas (groups) compared in the analysis and s_Y^2 is the sample variance of group means. The term s_Y^2 was calculated for the 2-level nested ANOVA as the ratio between the mean square for the nested term ($MS_{B(A)}$) and the product of the number of stations per area (*b*) by the number of replicate samples per station (*n*) (Ling and Cotter, 2003). After expressing $MS_{B(A)}$ as the ratio between the sum of squares for the nested term ($SS_{B(A)}$) and the associated degrees of freedom ($a(b-1)$), the resulting formula for the calculation of MDES was:

$$\mathrm{MDES} = \Phi^{-1}(P) \cdot \sqrt{\frac{2SS_{B(A)}}{nb(b-1)}}. \tag{2}$$