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Food for Thought

Monitoring fisheries resources at offshore wind farms: BACI vs. BAG designs

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Offshore wind farms often co-occur with biodiverse marine ecosystems with high ecological, economic, and cultural value. Yet there are many uncertainties about how wind farms affect marine organisms and their environment. The before–after–control–impact (BACI) design, an approach that compares an impact location with an unaffected control both before and after the intervention, is the most common method used to study how offshore wind farms affect finfish. Unfortunately, this design has several methodological limitations that undermine its ability to detect effects in these studies. An alternative approach, the before–after-gradient (BAG) design, would sample along a gradient with increasing distance from the turbines both before and after the intervention, and could overcome many of the limitations of BACI. The BAG design would eliminate the difficult task of finding a suitable control, allow for the assessment of the spatial scale and extent of wind farm effects, and improve statistical power by incorporating distance as an independent variable in analytical models rather than relegating it to the error term. This article explores the strengths and weaknesses of the BACI and BAG designs in the context of offshore wind development and suggests an approach to incorporating the BAG design into existing fisheries surveys and a regional monitoring framework.

Keywords: finfish, gradient, habitat, renewable energy, survey design, turbine

Introduction

Since the installation of the first wind farm in European waters in 1991 (Olsen and Dyre, 1993), offshore wind has become a burgeoning industry on continental shelves around the world. At the end of 2018, there were 23.1 GW of installed offshore wind energy production capacity worldwide, an increase of 4.5 GW since the previous year, and a fourfold increase since 2011 [Global Wind Energy Council (GWEC), 2019]. The United Kingdom (34%), China (20%), and Germany (28%) are currently the world's leaders in installed production capacity, with several other countries contributing smaller amounts [Global Wind Energy Council (GWEC), 2019]. In the United States, the production of energy from offshore wind is just getting started. Currently, there are one 30 MW operational wind farm off the coast of Rhode Island (Orsted, 2020), 15 active leases for commercial-scale operations in various early phases of development along the Atlantic coast, and several areas along the Pacific coast and near Hawaii being explored [Bureau of Ocean Energy Management (BOEM), 2019a].

Areas delineated for offshore wind development often coincide with complex bottom habitats (Guida *et al.*, 2017), productive marine communities [National Marine Fisheries Service (NMFS), 2018], and protected species (Kraus *et al.*, 2005; Ingram *et al.*, 2019). They also often overlap with economically valuable and culturally rich fishing grounds [Berkenhagen *et al.*, 2010; Gray *et al.*, 2016; National Marine Fisheries Service (NMFS), 2018]. Wind farms may affect marine species through a variety of means that act over varying spatial scales (Table 1; see reviews by Gill, 2005;

Table 1. Direct and indirect effects on fish associated with offshore wind farms and the expected spatial extent of effect ranging from local scale (within 10–100 s of metres from turbine), to moderate scale (within 1000 s of metres from turbine) and broad scale (within 10,000 s of metres from turbine).

Effect	Expected spatial scale of effect	Supporting literature
Direct effects		
Habitat provision via turbine structures	Local	Wilhelmsson *et al.* (2006), Andersson and Öhman (2010), and Langhamer (2012)
Food provision for benthivorous and piscivorous fish via species growing on/near turbines	Local	Mavraki *et al.* (2019)
Attraction to turbine structures	Local	Wilhelmsson *et al.* (2006) and Andersson and Öhman (2010)
Electromagnetic field effects on movement or behaviour	Local to moderate	Westerberg and Begout-Anras (2000), Westerberg and Lagenfelt (2008), and Gill *et al.* (2012)
Alteration of seabed habitat	Local to moderate	van Deurs *et al.* (2012)
Obstruction to fishing	Local to moderate	Gray *et al.* (2016)
Pile driving effects on behaviour or physiology	Moderate to broad	Wahlberg and Westerberg (2005) and Popper and Hawkins (2019)
Indirect effects		
Altered nutrient cycling due to benthic species growing on turbine structures via suspension feeding, excretion, biodeposition, etc.	Local	Coates *et al.* (2014)
Change in abundance and composition of benthic forage for benthivorous fish due to sediment enrichment from organisms associated with turbine structures	Local	Coates *et al.* (2014)
Change in fishing behaviour	Moderate to broad	Gray *et al.* (2016)
Altered food web dynamics due to food subsidy from organisms associated with turbine structures	Moderate to broad	Pezy *et al.* (2018) and Mavraki *et al.* (2019)
Hydrodynamic effects on primary/secondary production and particle movement	Moderate to broad	Broström (2008) and Wang *et al.* (2017)
Spillover of finfish from inside of wind farm where fishing may be reduced to outside the wind farm	Moderate to broad	Punt *et al.* (2009) and Ashley (2014)

Boehlert and Gill, 2010; Bailey *et al.*, 2014). The spatial overlap between the advancing offshore wind industry and natural resources with inherent biological, economic, and cultural value suggests an urgent need to understand how wind farms affect the distribution and abundance of demersal, pelagic, and bentho-pelagic finfish species hereafter referred to as fisheries resources in this article. This information is needed to inform stock assessment models, set fishing quotas, and inform fisheries management decisions in regions where the offshore wind industry is advancing.

The before–after–control–impact (BACI) design (Green, 1979) is a frequently used approach to study the effect of offshore wind farms on finfish (Table 2). In a basic BACI design, a single impact location (i.e. wind farm) and a single control location are sampled at random both before and after the intervention (i.e. the construction, operation, or presence of the wind farm) (Green, 1979). A change caused by the intervention is determined statistically by testing the significance of the interaction between sampling time point (before vs. after) and treatment (impact vs. control) with analysis of variance (ANOVA). A simpler approach, the control–impact (CI) design, collects data at the control and impact locations only after the intervention and is a similarly common method used to examine effects on fish at offshore wind farms (Table 2). Note that in this article, the term "effect" is used to refer to the biological response caused by the intervention and the term "impact" is used in reference to the "I" in BACI. Either term could indicate a beneficial, adverse, or neutral interaction depending on perspective. In marine ecosystems, BACI designs have successfully demonstrated effects due to disturbances such as sewage spills, fish farms, and fisheries exclusion (Smith *et al.*, 1999; Aguado-Giménez *et al.*, 2012; Moland *et al.*, 2013). In contrast, studies employing the BACI or CI design at offshore wind farms have had a more difficult time detecting changes and often report either inconsistent effects or weak effects for fish species or species groups (e.g. Vandendriessche *et al.*, 2015).

An alternative design, the before–after gradient (BAG) method, would sample along a spatial gradient with increasing distance from the turbines , both before and after the intervention. A significant change from baseline in the variables of interest is assessed using statistical methods that allow for the exploration of changes in spatial relationships over time (e.g. Brandt *et al.*, 2011, 2018). Initially applied to assess chemical spill effects on environmental and biological receptors (e.g. Wiens and Parker, 1995; Ellis and Schneider, 1997), gradient designs have often been used to evaluate the patterns of fish distributions in and around marine-protected areas and have been especially useful in elucidating spillover effects (e.g. McClanahan and Mangi, 2000). Although rarely discussed in the context of finfish at offshore wind farms [but see National Academies of Sciences, Engineering, and Medicine (NAS), 2018; Secor, 2018], the BAG design has demonstrated the effects of noise generated during offshore wind farm construction on marine mammals (Brandt *et al.*, 2011, 2018) and wind farm operation on bird distributions (Petersen *et al.*, 2004).

With the goal of improving monitoring designs for fisheries resources at offshore wind farms, the purposes of this article are to explore the strengths and weaknesses of the BACI and BAG designs in this context and to suggest an approach for incorporating the

Table 2. Summary of studies using BACI, CI, and Impact Location Only methods at offshore wind farms and whether distance from the turbines was examined.

Water body	Country	Distance from turbine studied (yes/no)	Distance effect detected (yes/no)	Response variables examined	Citation
BACI					
Baltic Region (Øresund)	Sweden	Yes; depending on the year of sampling: 130–1,350 m, 20–140 m	Yes	Species biomass, richness, and diversity; community composition of demersal fish	Bergström *et al.* (2013)[a]
Baltic Sea/ Kattegat Region	Denmark	No	NA	Abundance and density of demersal, bentho-pelagic, and pelagic fish, and movement patterns for common eel (*Anguilla anguilla*)	DONG Energy *et al.* (2006)[b]
North Sea	Belgium	Yes: sites inside wind farm at 200 m from turbine and just outside wind farm	In some instances	Species density, biomass, richness, fish length of demersal, and bentho-pelagic fish	Degraer *et al.* (2012, 2016, 2018)[b]
North Sea	Belgium	Yes: sites inside wind farm at 200 m from turbine and just outside wind farm	In some instances	Species density, biomass, richness and fish length of demersal and bentho-pelagic fish	Vandendriessche *et al.* (2015)[a]
North Sea	Belgium	No	NA	Gut content analysis of lesser weever (*Echiichthys vipera*), dab (*Limanda limanda*), and whiting (*Merlangius merlangus*)	Degraer *et al.* (2016)[b]
North Sea	Denmark	No	NA	Abundance and density of demersal, bentho-pelagic, and pelagic fish	DONG Energy *et al.* (2006)[b]
North Sea	Denmark	Yes; 1–100, 120–220, and 230–330 m	Yes	Species abundance and diversity and fish length of demersal, pelagic, and rocky habitat fish	Stenberg *et al.* (2015)[a]
North Sea	Denmark	No	NA	Density of juvenile and adult sand eel species; habitat	van Deurs *et al.* (2012)[a]
North Sea	Germany	No	NA	Species abundance and diet of mackerel (*Scomber scombrus*)	Lüdeke (2015)[b] and Krägefsky (2014)[b]
North Sea	Netherlands	No	NA	Species abundance and richness of demersal fish	Hillie Ris Lambers and ter Hofstede (2009)[b]
North Sea	Netherlands	No	NA	Abundance of demersal and pelagic fish	Lindeboom *et al.* (2011)[a]
Northwest Atlantic Ocean	United States	No	NA	Abundance, fish length, and condition for flatfish	Wilber *et al.* (2018)[a]
CI					
Baltic Sea	Sweden	Yes; sites at 0, 1–5, and 20 m, from turbine and reference	Yes	Species abundance, richness, and diversity of demersal fish	Wilhelmsson *et al.* (2006)[a]
Baltic Region (Kalmaar Strait)	Sweden	Yes; sites at 0, 1, and 20 m from turbines	Yes	Species abundance and density for adults and juveniles of all fish in the community	Andersson and Öhman (2010)[a]
Baltic Region (Øresund)	Sweden	No	NA	Fish length, weight, histosomatic index, gonadosomatic index, condition index and population size for Eelpout (*Zoarces viviparous*)	Langhamer *et al.* (2018)[a]
Irish Sea	Ireland	No; all sites within 200 m from turbine	NA	Species abundance, richness, diversity of demersal fish	Atalah *et al.* (2012)[a]
Irish Sea	United Kingdom	Yes; sites at 0 and 100 m from turbines	Yes	Species abundance and richness of all fish	Griffin *et al.* (2016)[a]
North Sea	Belgium	No	NA	Gut content analysis of lesser weever (*E. vipera*), horse mackerel (*Trachurus trachurus*), solenette (*Buglossidium luteum*), dragonet (*Callionymus* sp.), dab (*L. limanda*), and whiting (*M. merlangus*)	Degraer *et al.* (2012)[b]

Continued

Table 2. continued

Water body	Country	Distance from turbine studied (yes/no)	Distance effect detected (yes/no)	Response variables examined	Citation
North Sea	Belgium	No	Fish aggregated at turbines	Abundance for cod (*Gadus morhua*) and Pouting (*Trisopterus luscus*)	Reubens *et al.* (2013b)[a]
North Sea	Belgium	No	NA	Species abundance and density of ichthyoplankton and squid larvae	Degraer *et al.* (2016)[b]
North Sea	Belgium	No	NA	Barotrauma of European sea bass (*Dicentrarchus labrax*) exposed to pile driving	Degraer *et al.* (2016)[b]
North Sea	Denmark	No	NA	Abundance of pelgaic and semi-pelagic fish	Hvidt *et al.* (2005)[b]
North Sea	Netherlands	No	NA	Movement and behaviour of sole (*Solea vulgaris*) and cod (*G. morhua*)	Winter *et al.* (2010)[b]
North Sea	Netherlands	Yes; used modelled Didson sonar data	Yes	Species abundance, density, and fish length of demersal, bentho-demersal, and pelagic fish	van Hal *et al.* (2017)[a]
Impact Location Only					
North Sea	Belgium	No	NA	Diel feeding and movement patterns of cod (*G. morhua*)	Reubens *et al.* (2014)[a]
North Sea	Belgium	Yes; acoustic array encompassed distances from 0 to 150 m from turbines	Yes	Residency and site fidelity of cod (*G. morhua*)	Reubens *et al.* (2013a)[a]
North Sea	Belgium	No	NA	Species biomass, density, diet for pouting (*T. luscus*)	Reubens *et al.* (2011)[a]
North Sea	Belgium	Yes	Yes	Barotrauma of cod (*G. morhua*) exposed to pile driving	Degraer *et al.* (2017)[b]
North Sea	Belgium	No	NA	Reef-associated fish species presence/absence	Degraer *et al.* (2018)[b]
North Sea	Belgium	No	NA	Trophic relationships based on ^{13}C and ^{15}N isotope signatures for benthic, bentho-pelagic, and pelagic fish	Mavraki *et al.* (2019)[a]
Northwest Atlantic Ocean	United States	No	NA	Baseline species abundance and diversity of epibenthic fish	Cruz-Marrero *et al.* (2019)[a]
Northwest Atlantic Ocean	United States	No	NA	Baseline residency and movement patterns for Atlantic sturgeon (*A. oxyrhynchus oxyrhynchus*)	Ingram *et al.* (2019)[a]

Papers included were found through a comprehensive literature search conducted using common scientific databases (e.g. Science Citation Index Expanded, Biosis Citation Index). Search terms included "offshore wind", "wind farm", "demersal", "pelagic", "fish" and combinations thereof. The reference section of each paper was searched for additional peer-reviewed and non-peer-reviewed papers. Papers in the latter category were included in the table if they were available online. NA, not applicable.

[a]A peer review research paper.

[b]A research report or other types of grey literature.

BAG design into existing fisheries surveys and a broader regional monitoring framework. An open discussion among fisheries researchers regarding sampling design issues at offshore wind farms has the potential to drive methodological innovation and lead to a better understanding of how offshore wind development affects finfish at the individual, population, and ecosystem scales.

BACI: a well-established method with recognized limitations

The strengths and weaknesses of the BACI design have been widely discussed in the ecological literature, and this has led to several suggested modifications for the BACI design (Smokorowski and Randall, 2017). One of the earliest criticisms of the basic BACI design was that it was pseudoreplicated because multiple samples within a single control and a single impact location are considered to be replicates (Hurlbert, 1984). Subsampling in this manner Hurlbert (1984) argued is not true replication and only allows for comparisons between those two specific locations. Hurlbert (1984) further criticized the basic BACI design for being temporally psuedoreplicated, because repeated samples from the same location are likely to also be temporally autocorrelated. Stewart-Oaten *et al.* (1986) suggested that spatial and temporal variation could be accounted for with a before–after–control–impact paired series design in which control–impact site pairs are sampled near-simultaneously several times before and after the impact. This would allow for an assessment of the spatial and temporal variation in the mean difference between control and impact locations both within and among time points (Stewart-Oaten *et al*., 1986).

Underwood (1991, 1992, 1993, 1994) suggested that spatial variability could be addressed by using an asymmetrical BACI design in which multiple controls are selected at random among a set of appropriate control locations. Furthermore, sampling at random time intervals could help to address the issue of temporal autocorrelation (Underwood, 1991). Incorporating random spatial and temporal variability into subsequent statistical models, he argued, would make it possible to distinguish between natural variation and a change caused by the intervention. Stewart-Oaten and Bence (2001) countered the idea of using more than one control, stating that the variation among multiple controls does not matter since controls in a BACI design are not experimental controls but rather are non-randomly chosen to be similar to the impact area. BACI designs, Stewart-Oaten and Bence (2001) suggested, are useful so long as they are accompanied by appropriate statistical models and interpretations. Additional topics raised in the literature have included the need for proper power analysis to determine the appropriate sample size needed (Underwood, 1992, 1994) and the importance of understanding whether the intervention is a "pulse" or a "press" perturbation (Underwood, 1992, 1994; *sensu* Bender *et al*., 1984).

Although no definitive consensus emerged from these papers on the best way to resolve the issues associated with BACI designs, these discussions have provided ecologists with a broader perspective of BACI issues and a larger toolbox with which to modify the BACI design to answer specific questions in their systems. This is important because BACI remains one of the most popular methods for assessing environmental impacts in aquatic ecosystems (Smith *et al*., 1999; Aguado-Giménez *et al*., 2012; Moland *et al*., 2013).

BACI and CI are the most common designs used at offshore wind farms

The BACI and CI designs (Green, 1979) are the most frequent approaches used to study the effects on fisheries resources at offshore wind farms (Table 2). Table 2 lists the peer-reviewed studies and available research reports found through a comprehensive literature search (see Table 2 legend for details on the literature search method) that have conducted field studies to examine the effect of offshore wind farms on finfish. In general, these studies have sought to examine how a single wind farm affects finfish metrics such as abundance, biomass, diversity, size, distribution, or community composition. Of the 32 studies found, 12 used a BACI design, 12 used a CI design, and the remaining 8 were targeted studies that collected data only at the wind farm or prospective wind farm location (Table 2). (Note that multiple reports describing different years of the same study were only counted once.) Despite the problems and potential solutions associated with BACI discussed in the ecological literature, there has been little discussion on the methodological challenges posed by BACI for monitoring fish at offshore wind farms, the ability of these designs to detect effects in this context, and how these designs might be improved to do so.

Challenges with applying the BACI design at offshore wind farms

Three key assumptions made by BACI (and CI) designs as they are applied to offshore wind farms pose significant methodological challenges and warrant discussion (Table 3). The first assumption is that suitable control locations can be found. The importance of this assumption was emphasized by Stewart-Oaten *et al*. (1986) and is particularly problematic for offshore wind farm studies because choosing valid controls (i.e. locations that are ecologically

Table 3. Comparison of basic BACI and BAG designs.

Issue	BACI	BAG
Control site selection	Pro: Controls are intended to provide information on how the system changes over time in the absence of the intervention. It may be possible to find valid controls when effects are expected to have limited spatial and temporal extent	Pro: Control sites are not required
	Con: Difficult to find valid controls for finfish studies at offshore wind farms.	Con: Not applicable
Spatial heterogeneity	Pro: That basic BACI designs do not address spatial heterogeneity is a con. However, basic BACI designs could be improved by stratifying the study area by relevant environmental variables or by including these variables as covariates in analyses, or by using power analysis to determine the number of samples needed to detect effects	Pro: Allows for the exploration of spatial heterogeneity before and after the intervention. Attributes variance due to distance from the nearest turbine to the main effect in statistical models
	Con: Does not address spatial heterogeneity. This will likely increase the size of the error term in statistical models and result in analyses with lower statistical power	Con: Requires knowledge of the precise positions of the turbines far enough in advance to collect baseline data along the distance gradient before the intervention
Spatial scale and extent	Pro: Sampling only inside of the wind farm is a con when hypotheses include potential effects beyond the wind farm boundary, e.g. for highly migratory species, spillover effects. BACI may be appropriate for studies of effects expected to have a limited spatial and temporal extent	Pro: Allows for the study of the spatial scale and extent of effects by examining patterns along a distance gradient. Attributes variance due to distance from the nearest turbine to a main effect in statistical models
	Con: Samples "impact" sites only within the footprint of the wind farm, so is unable to explore effects beyond this boundary. Does not address the spatial scale or extent of effect	Con: Requires knowledge of the precise positions of the turbines far enough in advance to collect baseline data along the distance gradient before the intervention

and physically similar but far enough apart to be statistically independent) is difficult in the open ocean. This is evidenced by several studies that have found significant differences between control and prospective impact locations during baseline surveys (e.g. Wilber *et al.*, 2018), significant and unexplainable changes at the control location over time (e.g. Atalah *et al.*, 2012; Degraer *et al.*, 2013), and changes at control locations that differed from the changes recorded at the wind farm location (e.g.DONG Energy *et al.*, 2006; Stenberg *et al.*, 2015). The frequency of inconsistent outcomes suggests that the conditions at the control locations chosen are not representative of their respective wind farms and that statistical comparisons between control and wind farm locations are not meaningful.

That suitable controls are hard to find should not be surprising. The ocean is spatially and temporally dynamic, and finding two locations that are statistically identical to one another while also being geographically far enough apart to be statistically independent poses a clear challenge. Further complicating the act of selecting controls in the open ocean is the limited option from which researchers can choose after areas with hazards, conflicting uses (e.g. navigation, military, sanctuaries), and logistically onerous locations are excluded. This underscores that the baseline condition of the ocean is not pristine and that wind farms are developed amid numerous interacting stressors. Finding a suitable control is a difficult challenge to overcome. Even with extensive baseline sampling of candidate control locations to ensure their similarity to the impact location, controls may change over the lifespan of a wind lease (e.g. 30 years in the United States) due to natural factors (e.g. hydrodynamic effects on bottom type, changes in fish distributions) or manmade factors (e.g. changes in fishing pressure, development of additional wind farms or other industries such as aquaculture) that may be difficult to factor into a BACI data analysis framework. In addition, the lack of appropriate controls makes it difficult to apply any of the BACI design modifications suggested by Stewart-Oaten *et al.* (1986), Underwood (1991, 1992, 1993, 1994), and Stewart-Oaten and Bence (2001) because these rely upon valid controls.

The second assumption made by BACI studies is that the area within the wind farm is homogenous and all fish species respond the same way to the windfarm regardless of where inside the wind farm they are sampled. This assumption is unlikely to be met given the spatial variability inside many offshore wind farms. With BACI, sites are assigned randomly within impact and control locations, thereby disregarding the spatial variability in all potential covariates. Even in studies in which the assignment of sites is not completely random, site selection criteria are typically bounded by gear type and safety factors rather than meaningful environmental covariates (but see Degraer *et al.*, 2018). There is ample baseline evidence from potential wind energy areas showing wide variability in depth, bottom type, benthic, and epibenthic community composition (e.g. Guida *et al.*, 2017), which are basic habitat characteristics that are well known to affect fish distributions. Ignoring such variables when selecting sampling sites is likely to introduce site-to-site variability in the measurements of interest (e.g. abundance), leading to a reduction in the statistical power of the study to detect differences between wind farms and control locations. This limitation could potentially be addressed by sampling a sufficiently large number of sites (determined through a power analysis) to reduce the error term associated with the mean, thereby increasing the power of statistical analyses. In addition, this limitation could be addressed by including habitat variables (e.g. depth and bottom type) as covariates in statistical analyses or by stratifying the study area by habitat and then calculating the variables of interest by stratum within the location.

The third assumption is that the spatial scale of the effect is known. Although individual stressors related to wind farms may be generally categorized as having local-, moderate-, or broad-scale effects based on existing literature (Table 1), a comprehensive and mechanistic understanding of the scale of effects is currently lacking. The understanding of scale has been hampered by the inconsistency among studies in how far from turbines sampling occurs (Table 2). Sites that are right at the base of a turbine and those that are 100 s of metres away from the turbine have been classified as "wind farm sites" and not surprisingly have reported very different effects (e.g. Andersson and Öhman, 2010; Stenberg *et al.*, 2015). Current evidence suggests that some effects attenuate with distance from the turbine foundations (e.g. Bergström *et al.*, 2013), but there is still uncertainty about how far from turbines' effects may extend. Adding to this challenge is that often researchers are attempting to assess the integrated effect of all stressors associated with wind farms on aggregate metrics such as total demersal fish abundance (e.g.DONG Energy *et al.*, 2006). If there is in fact an effect of distance from the turbines and sites are assigned randomly with BACI, these sites may not represent the full spatial extent of effects, preventing a determination of the scale of impact.

Furthermore, assigning "wind farm" sites only inside the boundary of the wind farm precludes any exploration of effects occurring beyond that boundary. The wind farm perimeter delineated by leasing agencies has no relevance to the scale of ecological effects, and therefore, there is no expectation that the effects of the intervention either during or after construction will be confined to the leased area or a particular portion of it. Although some effects are expected to be relatively localized to the turbines (Table 1), placing sampling stations outside of the wind farm might be especially important for considering effects on mobile species (e.g. pelagics such as squid and large predatory fish) that may move across the wind farm boundary readily as well as mobile covariables such as commercial and recreational fishing pressure, which may also have transboundary effects. If, for example the abundance of a target species increases inside of the wind farm and fishing is excluded, there is a potential for spillover of fish outside of the fishing exclusion area similar to that described for some marine-protected areas (Punt *et al.*, 2009; Busch *et al.*, 2011; Ashley, 2014). Conversely, allowing fishing on aggregates of individuals within the windfarm boundary could have an adverse impact on regional populations (Bohnsack, 1989). However, even if fishing is allowed, there is the potential that fishermen may avoid the more difficult endeavour of fishing between turbines and instead fish at the edge or adjacent to the wind farm causing greater exploitation effects outside the windfarm (e.g. Gray *et al.*, 2016). Moreover, there are some effects, such as those associated with the land to sea electrical cable that may occur outside the wind farm boundary (e.g.DONG Energy *et al.*, 2006). The scale of impact may not be fully evaluated by sampling exclusively inside the perimeter of the wind farm with a BACI design. Extending the footprint of studies beyond this boundary could help to address this issue.

A frequent conclusion from BACI studies at wind farms is that natural variation is too large for effects to be detected (e.g.DONG Energy *et al.*, 2006; Hillie Ris Lambers and ter Hofstede, 2009).

The indirect implication being made is that wind farm effects are small compared with the range of natural variation and therefore are of minimal consequence. This is problematic from a marine conservation perspective in which finding a false negative effect (i.e. finding no effect when in fact one exists referred to as Type II error) is of even greater concern than finding a false positive effect (i.e. finding an effect when in fact none exists, referred to as Type I error). It is just as likely, and perhaps more so, that methodological issues and low statistical power prevented the detection of real effects. The limitations faced by BACI studies and the frequency with which their findings are equivocal (e.g. Vandendriessche *et al.*, 2015) suggest caution in applying the same approach at new wind farms in the United States and elsewhere. Nevertheless, the majority of studies at offshore wind farms in the United States and around the world continues to use this approach or plans to do so in the future [e.g. Lüdeke, 2015; Rijkswaterstaat, 2016; Degraer *et al.*, 2018; Bureau of Ocean Energy Management (BOEM), 2019b]. This may lead to a 'data-rich, information-poor' (DRIP) effect for fisheries resources similar to that described for benthic information at offshore wind farms (Wilding *et al.*, 2017).

When might BACI designs be appropriate?

Although the application of BACI designs at offshore wind farms is often problematic, BACI may be useful in answering research questions about effects that are expected to occur over a limited spatial and temporal extent (Tables 1 and 3). For example, sessile and reef-associated biota have been observed directly attached to and in the immediate vicinity of turbines ($\leq$20 m) (Wilhelmsson *et al.*, 2006). Although these artificial reefs could have indirect effects that reach much further afield, the direct effect of reef fish utilizing habitat associated with turbines occurs at or very near the structures. Similarly, BACI could be useful in short-term targeted studies of relatively slow-moving or sedentary species. In these examples, the spatial and temporal extent of the study could be constrained by the research question, which would make it more straightforward to accommodate the assumptions of the BACI design. In a sense, these studies would consider the BACI design as a special case of a gradient design in which only two locations along the gradient are sampled (impact and control) and the distance from the intervention is not expected to be important in statistical analyses. In practice, however, finfish studies at wind farms often examine response variables (e.g. a change in target species abundance) that are simultaneously affected by multiple drivers (e.g. change in prey field, fishing pressure, predators, habitat), making it difficult to disentangle individual drivers and determine what the appropriate scale of study should be at the outset.

Evidence suggests proximity of sampling to turbines matters

Nearly every study that has considered proximity to the turbines in its sampling design and data analysis, even in some limited capacity (i.e. 2–3 distance categories), has found that effects depend on how close to the turbines samples were collected (Table 2). This distance effect has been particularly evident when comparing abundances at or very near the turbines (within $\sim$20 m) with those much further away (see Methratta and Dardick, 2019 for review). Most often, when distance-based data are collected, researchers tend to focus on comparisons of data collected at each distance with a control location rather than with data collected at other distances (Table 2). Among those that have reported quantitative comparisons among distances (e.g. Wilhelmsson *et al.*, 2006; Bergström *et al.*, 2013; van Hal *et al.*, 2017), none have reported temporal changes in gradient patterns. Temporal changes could arise, for example if successional changes in the benthic species attached to the turbines (De Mesel *et al.*, 2015) attract different predator species over time, or if different finfish species migrate into or out of the area as the seasons change. Furthermore, with just 2–3 distance categories, only a coarse exploration of the spatial scale of effect has been possible so far. The distance effect remains one of the most consistently reported effects among studies, suggesting that the proximity of sampling to the turbines should receive serious consideration as a regular sampling design element for finfish studies at offshore wind farms.

BAG designs as an alternative to BACI for monitoring at offshore wind farms

The BAG design is an alternative method that can overcome the challenges posed by BACI (Ellis and Schneider, 1997) (Table 3). In the BAG design, sampling stations would be located along a gradient with increasing distance from the turbines where distance is defined as the distance to the nearest turbine for any given sampling point. Sites would be located along this gradient both within the wind farm and beyond its boundary and would be sampled both before and after the intervention. The increments along the gradient to be sampled and the spatial arrangement of sampling points should be guided by the research question(s) of interest and determined through an exploration of existing baseline data from the location of study. Statistical analysis of data collected with a BAG design could be conducted with general linear models, generalized additive models, or mixed model approaches using distance to the nearest turbine (i.e. nearest-neighbour distance) for a given wind energy array as one of the main effects in the model (for examples of analytical approaches and interpretation, see Petersen *et al.*, 2004; Brandt *et al.*, 2011, 2018). The analytical approach chosen will depend on the statistical assumptions made such as whether linear or non-linear effects are expected and whether effects are thought to be random or fixed (Zuur *et al.*, 2009). As with any sampling design, *a priori* power analyses should be used to inform the sample size needed to detect the effects of interest. Given that distance to the turbine often matters for finfish at offshore wind farms, including distance as an element in the monitoring design and as an independent variable in subsequent statistical analyses would remove this source of variability from the error term and thereby improve the precision and statistical power of the analyses.

BAG overcomes the first assumption of the BACI design regarding control locations by entirely eliminating the need to identify suitable controls. Rather than focusing sampling effort at a control location that is not truly representative of the wind farm location, the effort of the survey is instead focused on sampling multiple sites along a spatial gradient within and around the wind farm.

The BAG design overcomes the second assumption of BACI regarding spatial homogeneity by building into its design the capability to explore patterns of spatial variation in the target variables of interest. Rather than assuming that all wind farm sites are equivalent, BAG considers the possibility that distance from the nearest turbine may be an important factor in determining the magnitude of the effect. Distance intervals should be guided by the research question being asked since individual species exhibit unique patterns of spatial variability in terms of abundance, distribution, size, age class, and other important measures. Measuring relevant biological and physical covariates (e.g. depth and bottom type) at each sampling site and incorporating this information into analyses can further account for spatial heterogeneity. Including covariates in analyses is not unique to BAG designs and would be extremely useful for any design used.

BAG overcomes the third assumption of BACI regarding scale by collecting data in a manner that allows for the evaluation of the scale of effect. With sites positioned along a gradient with increasing distance from the nearest turbine, a BAG design would allow researchers to assess how far from the turbines effects begin to attenuate, the rate of attenuation across space, at what spatial distance there are no longer any detectable effects, and how these patterns vary among species. Here again, a clearly defined research question is needed because the scale of effect will be specific to individual species and specific aspects of their biology. For example, effects on reef fish abundance are likely to occur within 20 m of the turbine (e.g. Wilhelmsson *et al.*, 2006) whereas effects on soft-bottom species abundance may be found throughout the wind farm where soft-bottom habitat occurs or occurred prior to the intervention (e.g. Degraer *et al.*, 2013). Moreover, sampling both before the intervention and at several time points afterwards will allow researchers to examine changes in gradient patterns over time.

BAG designs have been extremely useful in studying offshore wind farm effects on marine mammals and birds. BAG has been particularly well suited for acoustic effect studies for marine mammals because pile driving generates a sound pressure wave that attenuates along a spatial gradient from its point of origin (Thompson *et al.*, 2010). For example, Brandt *et al.* (2011) deployed passive acoustic monitoring devices along a transect line of increasing distance from pile driving activities (2.5–21.2 km) to monitor echolocation clicks made by marine mammals. They were able to demonstrate that the period of acoustic inactivity exhibited by harbour porpoises (*Phocoena phocoena*) during pile driving decreased along a distance gradient out to 17.8 km and that, at 21.2 km, there was no detectable effect. Brandt *et al.* (2018) found similar patterns for harbour porpoises at multiple wind farms. Petersen *et al.* (2004) showed that the bird species, the common scoter (*Melanitta nigra*), avoided a wind farm area following its construction and that avoidance was greatest nearest to the turbines, which the authors suggested may be due to direct effects of the turbines or indirect effects such as increased human activity associated with turbine maintenance, or changes in food resources in the wider study area.

Challenges with applying the BAG design

Implementing a BAG design, or any design involving pre-construction data at measured distances from the turbines, requires knowing the precise placement of the turbines prior to construction with sufficient time to collect baseline gradient data during the "before" time period (Table 3). Currently, decisions about precise placement and arrangement of turbines are generally made less than a year prior to construction, leaving only a brief window of time for baseline data collection. To overcome this challenge, additional time between final wind farm design and construction would be needed to allow for at least one full year of sampling if not more.

Incorporating the BAG design into existing fisheries surveys and regional monitoring

The concept of a regional monitoring framework for offshore wind farms has been discussed in the United States and elsewhere [e.g. National Marine Fisheries Service (NMFS), 2019; RODA, 2019]. Such a framework would take a regionally standardized approach to monitoring while maintaining some flexibility in the design for individual regions to address regionally specific issues. This approach could facilitate the comparability of ecological patterns among wind farms and across regions. In many ecosystems, there are long-term surveys of fisheries resources that overlap with wind energy areas (Azarovitz, 1981; Pinnegar *et al.*, 2002; Nicholson and Jennings, 2004). For example, the NOAA bottom-trawl survey has sampled the entire Northeast US continental shelf from Nova Scotia to North Carolina for 56 years using a stratified-random method in which strata are based primarily on depth (Azarovitz, 1981). Once a wind farm is developed, the portion of the survey's sampling frame in which the turbine array occurs may be excluded from bottom-trawl sampling for logistical or other reasons. In the United States, researchers are currently evaluating how such reductions in the sampling frame will affect natural resource surveys and the information they provide [Northeast Fisheries Science Center (NEFSC), 2019]. A regional monitoring framework for offshore wind farms may help to minimize effects on existing surveys by adopting a sampling approach that can be integrated into broader, long-term monitoring schemes.

One approach would be to incorporate a BAG sampling design by delineating substrata at and around wind farms (i.e. subdivisions of each existing bottom-trawl survey strata that overlap the wind energy areas) that are defined by distance to the nearest turbine and then randomly assigning sites into these distance-based substrata. The substrata could be defined operationally across the entire turbine array with a desktop Geographic Information Systems (GIS) analysis that identifies all possible sampling points within each nearest-neighbour distance category. For example, if one of the distance categories is 0–100 m, then all of the points in the array that occur within 0–100 m of the turbines would be included in that stratum. At each sampling interval, a subset of these points would be randomly chosen for sampling. Distance-based substrata could occur both inside and outside of the wind farm. Several important questions should be considered in developing such a design, including: (i) How many distance categories are needed to detect a gradient effect and what should the numerical bounds of the categories be? (ii) Should bottom type be considered as a covariate in analyses, or is further sub-stratification needed to evaluate this level of heterogeneity? (iii) How far beyond the wind farm boundary is sampling necessary? This question should include the consideration of sampling needed along the submerged sea to land electrical cable. (iv) In addition to baseline sampling, at what temporal interval should sampling occur after the intervention? (v) What sampling modality for a given research question should be used at 10, 100, 1000, or 10 000 s of metres from turbines and how can data collected with different modalities be integrated? In addition to distance from the turbines, sampling modality will also depend on target species and the specific research question being considered. For example, small trawls and hydroacoustics have commonly been used to study fish abundance ≥100 s of metres from turbines (Degraer *et al.*, 2013; van Hal *et al.*, 2017), while a variety of methods including visual observation and video have been employed within 20 of turbines (Wilhelmsson *et al.*, 2006; Griffin *et al.*, 2016). Acoustic telemetry has also proven to be a valuable method for studying patterns finfish and marine mammal distributions near wind farms (Brandt *et al.*, 2011, 2018; Reubens *et al.*, 2013, 2014; Russell *et al.*, 2014, 2016). Such a technique might be particularly useful for species that are not well represented in traditional trawl surveys such as the endangered Atlantic sturgeon *Acipenser oxyrhynchus oxyrhynchus*. (vi) How and when should the study design be adapted as more is learned about the gradient of effect? Exploring baseline environmental and ecological assessment data from prospective wind farms and their vicinity as well as knowing the planned spatial layout of the turbines well in advance will aid in answering these questions with the expectation that some refinements are likely to be needed as the monitoring programme progresses. In the United States as well as in other countries where multiple offshore wind projects exist or are in the pipeline, such an approach could underpin a consistent regional monitoring framework, maintain the integrity of existing long-term surveys for fisheries resources, and facilitate spatial and temporal comparisons.

Conclusions

The offshore wind industry coexists with marine ecosystems and is developing rapidly, yet uncertainty remains about how wind farms affect marine fisheries resources. BACI and CI designs are common, but this article suggests that the BAG design be considered as an alternative methodological approach because it can overcome the limitations of BACI and CI and could potentially be incorporated into existing fisheries resource surveys as well as a regional monitoring framework for offshore wind. BACI and CI designs have recognized limitations but may continue to be useful to address research questions about effects with a limited spatial and temporal extent. A robust understanding of offshore wind farm effects on marine ecosystems will provide much-needed information for decision-makers and other stakeholders.

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