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# Modeling consequences of spatial closures for offshore energy: Loss of fishing grounds and fishery-independent data

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**Abstract**

Many jurisdictions are currently pursuing renewable sources of energy from the ocean, including offshore wind farms (OWFs). While these could have direct positive effects for global climate change by reducing fossil fuel consumption, there could be unintended consequences for fisheries and conservation. These include the potential loss of fishing grounds (and the consequent spatial displacement of fishing effort) and the potential loss of fishery-independent survey data in OWF areas. Because fishing and other types of vessel traffic are often limited in the OWF area, OWFs may also serve as other effective area-based conservation measures (OECMs), an important type of spatial protection in the context of the Convention on Biological Diversity 30 × 30 initiative. We used spatially explicit population models of groundfish fisheries on an idealized coastline in a management strategy evaluation to investigate the effects of OWF placement on conservation objectives (increased fish biomass) and fishery objectives (maintaining fishery yield). We simulated the loss of fishing grounds on 10% of the coastline, and the concurrent loss of 10% of fishery-independent survey data, introducing uncertainty and bias into stock status estimates. This produced two effects in the model: initial loss of fishery yield due to the closure and reductions in fishing effort when the loss of data triggered precautionary measures in the harvest control rule. Additionally, we assessed scenarios with different placements of the OWF relative to high-quality fish habitat, as OWFs could be placed without fish habitat considerations in mind. As expected (given the sustainable harvest rates we simulated), we found that placing the OWF on high-quality habitat produced the greatest negative effects of fishing grounds and fishery-independent data on fishery yields, but placing the OWF on low-quality habitat caused it to be ineffective as an OECM (in terms of increasing fish population biomass). Additionally, the loss of survey data had a greater effect for less mobile fish species. Our findings highlight the expected trade-offs between the fishery and conservation (i.e., OECM) consequences of

OWF expansion and the need to compensate for the loss of fishery-independent data by accounting for species distributions relative to habitat in survey indices.

**KEYWORDS**
fishery-independent surveys, marine spatial planning, offshore wind farms, other effective conservation area-based measure (OECM)

## INTRODUCTION

Spatial management has emerged as a pivotal component of global marine conservation endeavors. Most notably, the 2022 Conference of the Parties to the Convention on Biological Diversity produced a commitment to protecting at least 30% of the planet's land and ocean by 2030, commonly referred to as the "30 by 30" goal (UNFCCC, 2022). Ideally, achieving such ambitious objectives would be accompanied by the implementation of adaptive management strategies to effectively balance economic and environmental considerations (Grafton & Kompas, 2005; Walters & Hilborn, 1978; White et al., 2011). Along the coastlines of the United States, offshore renewable energy represents a prime new example of marine spatial management, particularly the development of spatial zones for the placement of offshore wind farms (OWFs). OWFs are widespread in Europe, where a variety of studies have investigated the effects of OWFs on fish ecology and social-ecological systems (e.g., Breton & Moe, 2009; Burkhard & Gee, 2012; Wilhelmsson et al., 2006). However, there has been little research on the consequences of OWF installation on fishery closures and fisheries data collection. The installation of an OWF and associated restrictions of fishing vessels in the area carries the potential to alter fish ecology, disrupt fishing patterns, and limit the collection of fishery-independent survey data (Allen-Jacobson et al., 2023; Püts et al., 2023). On the other hand, these OWF areas may also qualify as "other effective area-based conservation measures" (OECMs). OECMs are spatial areas that may not have been originally designed with conservation goals but can still produce conservation outcomes, such as meeting the 30 by 30 target (Garcia et al., 2022). Consequently, if OWF areas are to be considered OECMs for the purpose of meeting the 30 by 30 goals, we must assess the factors that determine OWF effectiveness in achieving conservation goals.

The expansion of offshore wind energy presents a number of potential impacts on marine ecosystems. These include stressors such as energy extraction, electromagnetic fields, physical infrastructure, device dynamics, acoustics, and chemical effects (Farr et al., 2021). However, our focus is not on those consequences but rather on understanding how the rapid growth of OWFs may affect fisheries and fishery management; this understanding is essential for informed decision-making and scientific efforts aimed at monitoring changes in our oceans and coastlines. The spatial planning surrounding OWFs could include minimizing overlap with existing fishing grounds, but even so, the eventual dynamic consequences for fishery stocks and management practices are not well known, as can be the case for marine protected area (MPA) design (Field et al., 2006; White et al., 2011). Nonetheless, the eventual fishery and conservation outcomes of OWF areas are of interest, even if those are not part of the design process (as they would not be for any OECM, by definition).

The closure of fishing areas due to offshore wind placement has dynamic implications for both fishers and managers (Agardy et al., 2011). One immediate consequence of the placement of OWFs is limitations on vessel traffic to avoid collision or entanglement with physical infrastructure, including both statutory safety zones and precautionary behavior by vessel owners and captains (e.g., Yu et al., 2020). For example, European regulations prohibit vessel access within a radius of 500 m surrounding OWF structures (European Commission, 2019), though we note there are also some efforts to colocate OWF and some fishery gear types (Bonsu et al., 2024). In general, these navigation restrictions will lead to the displacement of fishery effort from the OWF area to other locations. Additionally, at least in the United States, it is generally the case that large vessels used in fishery-independent scientific surveys are not able to operate in OWF areas. Drawing upon the established theoretical understanding of spatial fishery closures (e.g., Botsford et al., 2019; White et al., 2011), we can investigate whether the closures associated with OWFs will advance spatial conservation goals. For instance, it is plausible that the total biomass of fishery stocks might rise as a result of the de facto closure. This type of protection benefit has been reported in some types of OECMs, such as military exclusion zones where vessel traffic and fishing are restricted (Esgro et al., 2020). Conversely, the concentration of displaced fishing effort may have a net negative impact on fishery yield, contingent on the prevailing fishery management regime (Holland & Brazee, 1996; White, Botsford, Hastings, & Largier, 2010). Moreover, by reducing the spatial domain over which fishery-independent surveys can be conducted, the closure of these areas could adversely stock assessments used for fishery management purposes (Methratta et al., 2020). For example, a reduction in sampling area would affect survey design and sampling methodologies while potentially introducing unforeseen biases as well as increased uncertainty (Field et al., 2006).

The MPA literature offers valuable insights into these questions, as OWFs and MPAs both represent closure areas within the ocean, albeit with different objectives. Prior theoretical and empirical studies have thoroughly investigated the expected effects of closure area size, larval connectivity, adult fish movement, and fishery management on population dynamics within protected areas, including the temporal dynamics of changes in biomass and fishery yield (reviewed by Baskett & Barnett, 2015; Botsford et al., 2019; Grüss et al., 2011; White et al., 2011, 2025). Based on that body of literature, we could generally predict how spatial closures for OWFs should interact with fisheries. For example, if a fishery is being overharvested, a spatial closure might initially reduce fishery yield but should eventually increase it because of increased reproductive output in the newly protected part of the stock (Hopf et al., 2023). If the fishery is managed sustainably, we would anticipate a spatial closure to cause modest decline in fishery yield due to the redistribution of fishing effort. These effects would be more dramatic for more sedentary species and of lower magnitude for highly mobile species (reviewed by White et al., 2011, 2025).

Here, we address two aspects of spatial closures that have not been thoroughly investigated. The first is that strategic models of population dynamics in MPAs generally assume that habitat is homogenous, or that MPAs are intentionally placed in locations of high-quality habitat (e.g., Beger et al., 2010; Botsford et al., 2001; White, Botsford, Moffitt, & Fischer, 2010). However, OWF placement is based on constraints related to prevailing winds, bathymetry, and the proximity of coastal infrastructure. While marine spatial planning surrounding OWFs may sometimes also include information on fishery landings or fish habitat, OWFs could ultimately represent spatial closures in habitat of higher or lower quality for fish (Pınarbaşı et al., 2019). Here, we investigate the theoretical consequences of protecting areas without regard to habitat to understand the overarching conditions and species for which an OWF might function as an OECM. Second, the potential loss of fishery-independent data due to OWF closures could affect estimates of stock status and therefore fishery management via associated harvest specifications based on estimates of stock status (Field et al., 2006). We investigate how the closure of OWF areas might influence fishery management strategies for those specific species and assess the costs and benefits. While the specific details of vessel restrictions vary among OWFs, we assumed for the purpose of our modeling that both fishing and scientific survey vessels were equally excluded from OWF areas.

We used population models to simulate the fishery costs and conservation benefits associated with OWFs, from the perspective of precautionary management goals of keeping both fish biomass and fishery yield above pre-specific reference levels. To do this, we took a management strategy evaluation approach in which we modeled the process, observation, and management response to observation (Punt et al., 2016). We focused on groundfish fisheries specifically because they may be particularly affected by OWF development. Groundfish themselves may be affected by OWF development because many species are relatively sedentary compared to more migratory and pelagic species and are long-lived and slow-growing, which can cause long time lags before population responses to perturbations are fully realized. In addition, the fishing grounds used by many vessels targeting groundfish are relatively consistent over time, making the potential for displacement due to OWF potentially greater than for fisheries using a wider range of fishing grounds (Warlick et al., 2025), and the trawling gear used in many groundfish fisheries is particularly incompatible with moored OWF infrastructure (as opposed to, say, fixed gear traps for crustaceans; Bonsu et al., 2024). Therefore, the effects of OWF displacement on groundfish fisheries could include changes in habitat quality, altered fishery dynamics, and the associated challenges in maintaining effective fishery management strategies amid evolving spatial closures. Our primary goal was to identify how the closure of areas for OWFs might influence fishery management outcomes and to identify the overarching conditions and species for which an OWF might function as an OECM. To accomplish this, we employed a model representing an idealized continuous coastline habitat, and we assessed the impacts of OWF closures on fisheries for species with a range of life histories, with OWF closures in different locations relative to high-quality habitat for those species.

# METHODS

## Overview

Our overall objective was to quantify the potential responses of fisheries species to OWF-like spatial closures. We were interested in two aspects of this issue. First, we wanted to explore the anticipated impacts on fisheries yield and the accuracy of fisheries status assessments arising from the loss of fishery-independent data in OWF areas. Second, we wished to examine the expected conservation outcomes of these closures, that is, the degree to which an OWF could be considered an OECM by effectively elevating biomass relative to areas not contained within an OWF. The MPA literature has thoroughly investigated the relationship between MPA size, spacing, animal movement, fishery management, and the ecological effects of MPAs on fished populations (e.g., Rassweiler et al., 2012; Thomas et al., 2014; Vandeperre et al., 2011; White et al., 2011, 2025). These studies have consistently demonstrated that MPA size plays a crucial role in influencing not only the abundance and diversity of marine species within protected areas, but also the potential positive spillover effects which contribute to sustainable fisheries management in surrounding areas. With that in mind, we did not explore those variables.

For the purposes of this paper, we define an area of higher habitat quality as one with potential for greater population density. Most guidance for MPA design emphasizes their placement in areas of high habitat quality (Leslie, 2005), and so most model analyses of MPA effects have assumed they are in "good" habitat (e.g., Botsford et al., 2001; though see Game et al., 2008 for a notable exception). By focusing on these key issues: loss of fishing grounds, loss of data, and placement relative to high-quality habitat, we attempted to isolate the direct impact of OWF placement on fisheries, independent of other factors that are known to influence the relationship between fisheries and closure areas under specific conditions.

We used a management strategy evaluation approach to simulate population dynamics, observation, and management response to observation. Our model had three components: process, observation, and management sub-models (Figure 1; additional model details and equations are given in Appendix S1). The process sub-model describes the spatially explicit metapopulation dynamics of groundfish species on a linear coastline. The populations have age-structured growth, reproduction, and fishery selectivity; adult fish move within home ranges that can span habitat patches, and patches are also connected by larval dispersal, with density-dependent post-settlement mortality of new recruits. We represented differences in habitat "quality" between patches by varying the maximum density of recruits that can survive in each patch. The model assumes that fishing harvest mortality is from a fleet that allocates effort in response to spatial differences in fish abundance. The observation sub-model simulates the collection of fishery-independent survey data on the population's biomass at multiple locations along the coastline. Finally, the management sub-model uses a harvest control rule that sets limits on fishing effort based on the estimates of population status from the fishery-independent survey data. We now explain these components in greater detail.

## Archetypal species

We simulated population dynamics for five different archetypal species, which were simplified representations of groundfishes on the U.S. Pacific coast. Two key aspects of life history that affect the population dynamic response to spatial closures are home range movements and lifespan (i.e., natural mortality rate). In our model we did not attempt to represent more complex migratory behaviors that some groundfish species are thought to exhibit over the multiyear time scales encompassed by our simulations, including ontogenetic and spawning migrations.

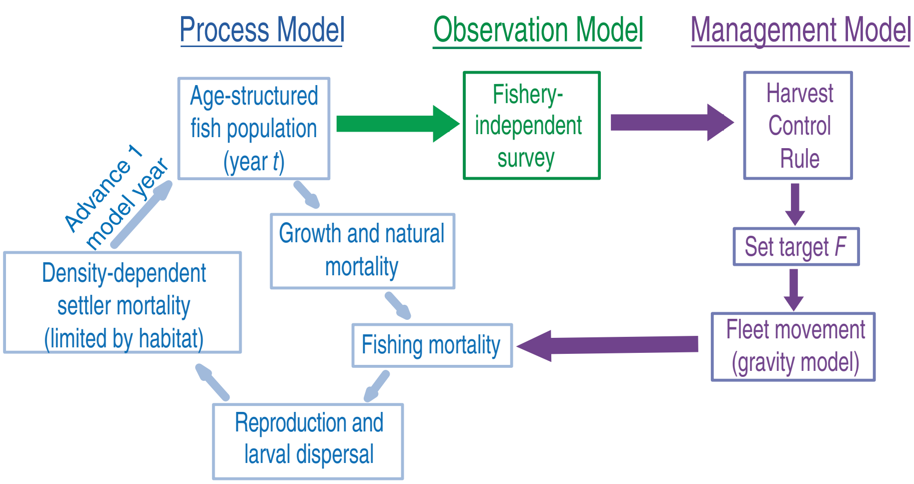


**FIGURE 1** Overview of the components of the management strategy evaluation model.

This modeling choice reflects both a lack of information on the scale and tempo of those movements, and the challenge of representing such movements within the idealized seascape of our model domain. Thus, our model results should be interpreted as reflecting archetypal groundfish species with fast/slow life histories and small/large home range movements, rather than specific species. The archetypes we modeled were inspired by ecologically and economically important fisheries that might be expected to show variable response patterns following implementation of OWFs. The five representative U.S. Pacific coast groundfish species we chose were Sablefish (*Anoplopoma fimbria*), Dover Sole (*Microstomus pacificus*), Lingcod (*Ophiodon elongatus*), and Widow and Yellowtail Rockfishes (*Sebastes entomelas* and *Sebastes flavidus*).

We obtained parameters regarding life history (growth, maturity, and natural mortality), fishery selectivity, and estimated exploitation rate from the most recent stock assessments available for each (Table 1; Adams et al., 2019; Kapur et al., 2021; Stephens & Taylor, 2017; Taylor et al., 2021; Wetzel & Berger, 2021). In these species, estimated adult home range diameters range from 4 m (Lingcod) to 2.5 km (Sablefish). Similarly, there is a wide range in natural mortality rate ($M$) and lifespan; from $M = 0.42$ year$^{-1}$ in Lingcod (implying a life expectancy of 2.39 years; maximum age observed between 13 and 18 years old) to $M = 0.1$ year$^{-1}$ in Widow Rockfish (life expectancy of 10 years; maximum age observed 50 years) (Hannah et al., 2008). The expectation is that less mobile species may benefit more from spatial protection within OWF closures, though closed areas might also contribute recruits to fished areas through larval transport. Additionally, shorter lived species are expected to accumulate biomass more rapidly within closed areas and boost larval supply to fished areas earlier in the process (Barceló et al., 2021; White et al., 2011, 2014).

## Spatial domain

We implemented a spatially explicit model along an idealized linear coastline with a simulated environment of 100 distinct 1-km patches, following the example of similar models used to develop general guidelines for MPAs (Botsford et al., 2001; Moffitt et al., 2013; White, Botsford, Moffitt, & Fischer, 2010). To avoid artifacts from the edge of the habitat domain, we rendered the coastline as effectively infinite, with larval and adult movements looping around the domain edges (as in White, Botsford, Moffitt, & Fischer, 2010). We assumed that the OWF would be placed in the first 10 patches on the "south" end of the spatial domain; this assumption can also be interpreted as placing the OWF in the middle of the domain, as we looped movement between the northern and southern edges. As mentioned before, we did not vary the size of the closure because the effects of closure size are well known (i.e., general closures that are larger relative to the scales of adult movement and larval transport will lead to greater increases in fish biomass inside the protected area but cause greater reduction in fishery yields if the fishery was not overharvested; White, Botsford, Moffitt, & Fischer, 2010). That amount of closed area represents 10% of the coastline.

## Process model

We created age-structured models of fish populations using parameter values from the most recent stock assessments for each species (Table 1). Fishing mortality had age-dependent selectivity, which was based on the trawl fishery for each species (Appendix S1: Figure S1). We assumed that fishing effort was set by a harvest control rule that specified a target spawning biomass per recruit (SPR), which we translated into an instantaneous

**TABLE 1** Parameter values providing a basis for archetypal species used in model simulations, and literature references for values.

| Parameter | Definition | Dover sole | Lingcod | Sablefish | Widow Rockfish | Yellowtail Rockfish |
|---|---|---|---|---|---|---|
| $M$ | Natural mortality (year$^{-1}$) | 0.117 | 0.418 | 0.07 | 0.14 | 0.0843 |
| $A_{max}$ | Maximum age (years) | 50 | 25 | 70 | 54 | 64 |
| $L_\infty$ | Asymptotic maximum length (cm) | 48.5 | 103.645 | 57 | 50.34 | 52.2 |
| $k$ | von Bertalanffy growth (year$^{-1}$) | 0.117 | 0.193 | 0.41 | 0.15 | 0.17 |
| $c$ | Mass at length coefficient $\left(\text{kg} \times \text{cm}^{-d}\right)$ | $2.97 \times 10^{-6}$ | $2.802 \times 10^{-6}$ | $3.315 \times 10^{-6}$ | $1.7355 \times 10^{-5}$ | $2.87 \times 10^{-2}$ |
| $d$ | Mass at length exponent | 3.33 | 3.2766 | 3.27264 | 2.9617 | 2.822 |
| $\sigma_h$ | Home range diameter (km) | 0.0175 | 0.00425 | 2.5 | 2.5 | 0.4625 |

*Note*: Sources for home range diameter are Dover Sole, Hagerman (1952); Lingcod, Tolimieri et al. (2009); Sablefish, Ehresmann et al., 2018; Widow Rockfish, Ressler et al. (2009); and Yellowtail Rockfish, Pearcy (1992). Sources for all other parameters are: Dover Sole, Wetzel and Berger (2021); Lingcod, Taylor et al. (2021); Sablefish, Kapur et al. (2021); Widow Rockfish, Adams et al. (2019); Yellowtail Rockfish, Stephens and Taylor (2017).

fishing mortality rate ($F$, year$^{-1}$) for each model species. SPR is calculated as the average fecundity of a recruit over its lifetime under a particular fishing harvest rate, relative to the average fecundity of a recruit over its lifetime in the absence of fishing. This approach derives from the principle that sustainable harvest requires a conservative level of demographic replacement (Botsford et al., 2019). The usual target SPR in U.S. Pacific coast groundfish management is 0.5, and we used that as our baseline SPR target in simulations (Pacific Fisheries Management Council, 2024).

Once a target SPR and the corresponding $F$ were determined, we assumed that was the average $F$ over the spatial domain, but the level of fishing in any given spatial cell could vary with fleet movements. We assumed that fishing effort was redistributed among patches using a gravity model, in which fishing effort redistributes each year based on the spatial distribution of catch per unit effort (CPUE) in the prior year, until at equilibrium CPUE is equal at all locations (Walters & Bonfil, 1999). This tends to lead to "fishing the line" effects at the boundaries of protected areas, such that fishing intensity is higher close to the boundary of the protected area where species density is higher (Cabral et al., 2017; Kellner et al., 2007).

In each annual time step of the model, the local population in each habitat patch spawns larvae (based on an age-specific fecundity relationship); those larval propagules disperse along the coastline according to a dispersal kernel (see next paragraph) and then settled into adult habitat where they experience intra-cohort density-dependent mortality following the Beverton–Holt relationship. We assumed that given the density of settling larvae in patch $i$ at time $t$, $S_{i,t}$, the density of surviving recruits, $R_{i,t}$, depends on the density-independent survival term $\alpha$ and the maximum recruit density, $\beta_i$. We represented variation in habitat quality along the spatial domain by letting $\beta_i$ be proportional to habitat density in patch $i$, essentially setting the maximum fish density in each patch (as in White et al., 2014). The settler-recruit relationship is then

$$R_{i,t} = \frac{\alpha S_{i,t}}{\left(1 + \frac{\alpha S_{i,t}}{\beta_i}\right)}. \tag{1}$$

The recruits, $R_{i,t}$ then become the first age class in the population in that patch and time step: $N_{1,i,t} = R_{i,t}$.

We assumed that larval dispersal followed a Gaussian dispersal kernel (Siegel et al., 2003), but given the assumed widespread dispersal in all of these species (i.e., the stock assessments assume they are well-mixed populations), we used kernels with a mean of zero and a SD of 10, which represents a scenario with relatively widespread dispersal and no possibility of self-persistence within a protected area (Botsford et al., 2019; White, Botsford, Hastings, & Largier, 2010). Adult movement was simulated as in Moffitt et al. (2013), which assumes adult fish assigned to each model cell (after having settled there following the larval/juvenile dispersal period) use that cell as the center of their home range but spend time in neighboring cells (and experience the fishing pressure there) in a home range that follows a normal distribution with a SD chosen to approximate literature reports of the diameter of species' home ranges (Botsford et al., 2019; Rassweiler et al., 2012; Thomas et al., 2014; Vandeperre et al., 2011; White, Botsford, Moffitt, & Fischer, 2010).

To examine the possibility that OWFs could be placed irrespective of key fishery habitats, we simulated three different scenarios in which the distribution of habitat quality varied over space, relative to the OWF location. By "habitat quality" in this context we mean the proportion of a spatial cell which is suitable habitat for the species (e.g., for the rockfish species this could be the proportion of that cell that is hard rocky reef instead of soft sediment substrate). There are many ways this type of scenario could be imagined, so we simulated conditions in which the distribution of suitable habitat followed a normal distribution along the two-dimensional coastline, with the maximum habitat quality (the mean of the normal distribution) falling at 5, 30, or 55 km from the southern edge of the spatial domain, with a SD of 10 km (Figure 2). This leads to three different OWF placement scenarios: one in which the best habitat is inside, one in which the OWF is near to the best habitat, and one in which the OWF is far from the best habitat. In each scenario, we scaled the $\beta_i$ term (the maximum recruit density in the settler-recruit relationship, Equation 1) so that it had a value of 1 at the

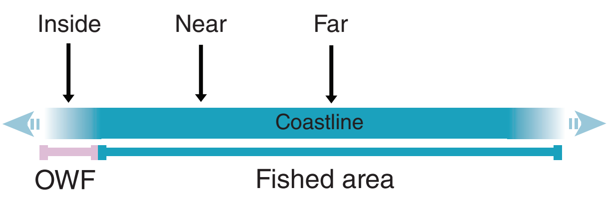


**FIGURE 2** Diagram illustrating the spatial domain of the model, representing an idealized linear coastline with an offshore wind farm (OWF). Ten percent of the coastline is closed to fishing due to the OWF (pink) and fishing is permitted in the remainder (blue). The coastline is effectively infinite because both larval and adult fish movement wrap around the edge of the coastline to the other end of the domain. Vertical black arrows indicate the center of the three habitat distribution scenarios (i.e., the location of the patch with the highest habitat density).

peak of the normal probability density function, so that maximum recruit density ranged between zero and one.

## Observation model

Our analysis included simulating a fishery-independent survey to assess trends in biomass to inform the harvest control rule. We simulated a scenario in which the population was sampled annually at 10 locations spaced every 10 km starting at 5 km from the edge (i.e., at the 5, 15, 25, ... 95 km positions); the biomass density estimates at each location are averaged to estimate a mean stock biomass density for the entire spatial domain. After the installation of the OWF, one of the 10 survey locations was lost. Many modern fishery-independent surveys employ more complex sampling designs, such as a depth- or habitat-based stratified random design (e.g., Keller et al., 2017), and it is also possible to account for spatial variation in habitat quality when aggregating survey data (e.g., Bell et al., 2022; Switzer et al., 2023). Therefore, our approach was intended to approximate a simpler, more general case in which a proportion of the survey domain is excluded from sampling, and there is no attempt to reweight or correct later survey data to account for that exclusion; this allows us to gauge the cost of failing to take those additional steps.

We assumed that biomass sampling included measurement error that followed a normal distribution with a CV of 0.2 (we explored other values for the CV and found that they changed the results in predictable ways, so we did not explore the sensitivity of results to that value).

## Management model

We used a management model that had a simple control rule based on the fishery-independent survey estimates of biomass, with a goal of keeping biomass above a pre-specified biological reference point. Following the current practice for groundfish management in the US West Coast, we implemented a "40–10" fishery control rule (Pacific Fisheries Management Council, 2024). This rule sets a target SPR of 0.5 when estimated biomass is above 40% of the unfished value ($B_0$) and then ramps down the target SPR linearly from 0.5 to 0 when biomass is between 40% and 10% of $B_0$ and completely suspends fishing when biomass is below 10% of $B_0$. In all simulations, we initialized models so that they were at a sustainably harvested steady state at the target 50% SPR prior to OWF implementation. We simulated a management model in which assessment of target biomass happened every 4 years, and the harvest control rule was applied following that assessment. This is clearly a simplified representation of how fishery stocks are assessed; we are not simulating a full data-integrated assessment process, and we recognize that fishery-independent surveys are usually used to create an index to estimate trends in abundance rather than absolute biomass. Nonetheless, this is a simple and straightforward way to capture the effect of lost data on management decisions.

The model was implemented in R version 4.3.2 (R Core Team, 2023), and the code is available in Campbell (2025).

# RESULTS

To simplify the presentation of results, we first present the analysis using the parameter values for an archetypal species like Sablefish here in the main text, which reveal the general patterns that extend to other life histories in predictable ways. We then describe the results for the simulations using the life history parameters for the other archetypal species in terms of their comparison to the archetypal Sablefish results.

## Fishery biomass distributions before and after OWF implementation

At the steady state prior to OWF implementation, the distribution of fish biomass reflected the distribution of favorable habitat (Figure 3a). Note in Figure 3, the inside scenario, the distribution of suitable habitat is constrained by the southern boundary of the model domain, so in order to maintain the same integral abundance of habitat, the maximum abundance is somewhat higher at the center of the habitat distribution than in the other two scenarios.

In general, the effect of implementing an OWF was an increase in biomass in that closure area, but the degree of increase was dependent on the OWF's proximity to high-density habitat (Figure 3b). In the inside and near scenarios, fish biomass increased within the OWF (the left-most 10 km of the model domain) and was slightly depressed immediately outside the OWF. In the far scenario, there was no noticeable biomass change within the OWF location after 20 years. How those different outcomes arose is best understood by examining the temporal population dynamics of fishing activities following OWF implementation, as we do in the next section.

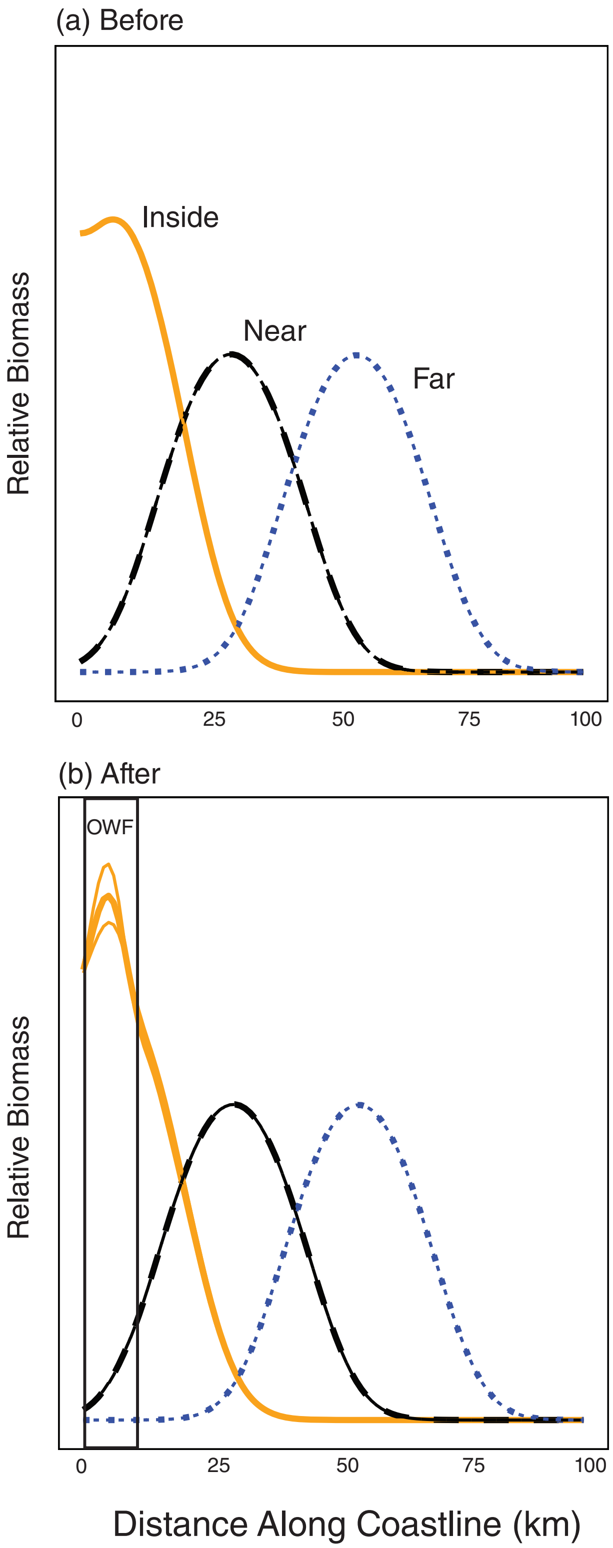


**FIGURE 3** Distribution of the archetypal Sablefish species biomass along the coastline (a) before (at a steady state) and (b) 20 model years after offshore wind farm (OWF) implementation, in scenarios with good habitat centered inside the OWF, near, and far from the OWF (indicated by color, as labeled in panel a). Thinner lines (barely visible in some cases) indicate the upper quartile (0.75) and lower quartile (0.25) of model simulation results.

## Fishery population trajectories over time

In the inside scenario, in which the OWF was placed in an area of high-density habitat, the implementation led to an immediate but gradual increase in fish biomass due to the removal of harvest in that area (Figure 4a). However, this increase was tempered by the displacement of fishing effort into the remaining fished areas, which reduced biomass there. While overall biomass increased, the estimated mean stock biomass sampled in the observation model of the fishery-independent survey decreased due to the loss of survey data in the highest quality habitat patches, where fish biomass was also greatest (note the declining purple curve in Figure 4a). The resulting perceived reduction in biomass ($B/B_0$) led to the control rule reducing the target SPR and thus $F$ (Figure 4d), which caused an abrupt reduction in fishery yields (Figure 4g). The sampled biomass then gradually increased slightly, mirroring the overall increase in actual biomass over the subsequent years. After 20 years, biomass had risen by 25% (Figure 4a), due both to the increase in biomass inside the OWF and the reduction in fishing effort by the control rule. During that 20-year post-OWF period, the sampled biomass remained lower than actual biomass, causing the control rule to keep fishing effort lower than pre-OWF levels. Consequently, yield remained low over the 20-year period, though it gradually increased from the initial drop, reflecting spillover from the increased reproductive biomass in the OWF (Figure 4g). Notice that the staggered trends in the harvest rate in Figure 4d reflect the 4-year assessment cycle in the management model.

In the near and far scenarios in which the OWF was placed outside the region of highest habitat density, the effects of the OWF on fish biomass and fishery yield were quite minimal (Figure 4). In these scenarios, the overall increases in fish biomass over 20 years were imperceptible in the simulations using the Sablefish life history parameters. In both cases, the loss of sampling from regions of relatively lower biomass density resulted in an upward bias in the observed biomass (Figure 4b,c), so the control rule allowed SPR and the harvest rate to remain at the pre-OWF target even after the closure. While the target SPR did not change in these scenarios, there was a small reduction in yield due to the loss of fishing grounds (Figure 4h,i).

## Summary across archetypal species

The model results for the simulations based on the other four archetypal species were similar to those for the archetypal Sablefish but varied with life history and

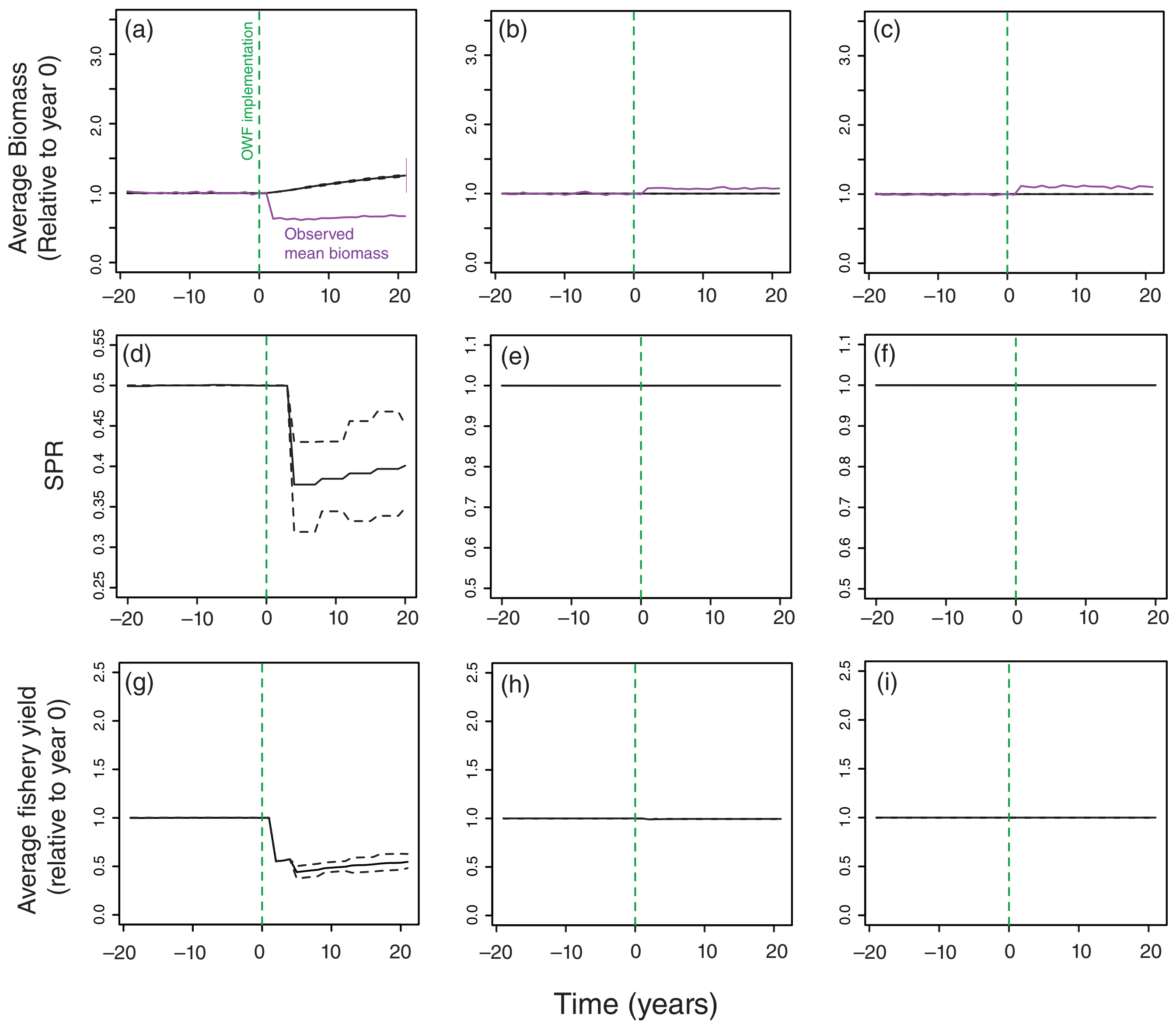


**FIGURE 4** Temporal trends in archetypal Sablefish species (a–c) relative mean biomass along the coastline (solid black curves) and observed biomass from the simulated fishery-independent survey (purple curves), (d–f) target spawners per recruit (SPR), and (g–i) relative fishery yield. Results are shown for three different habitat distribution scenarios: (a, d, g) the highest habitat density inside the offshore wind farm (OWF), (b, e, h) the highest habitat density near the OWF, and (c, f, i) the highest habitat density far from the OWF. Dashed lines give the 5% and 95% quantiles of the results (based on 100 simulations), the green vertical dashed line indicates the time at which the OWF was established, and the purple line indicates the biomass estimated from fishery-independent sampling. Values of biomass and yield are expressed relative to the value immediately prior to OWF implementation, and time is expressed relative to the OWF placement (year 0).

mobility in predicted ways primarily related to the species' lifespan and spatial scale of adult movement (Appendix S1: Figures S2–S9). In Figure 5, we depict the relative changes in biomass, harvest rate, and fishery yield, averaged over 20 model years following OWF implementation. In that figure, we have ordered the archetypal species in decreasing size of the adult home range. The simulations using parameters for archetypal Dover sole, intended to represent species with small home range diameter (i.e., less time spent in the fished area) and a low natural mortality rate relative to the archetypal Lingcod species, exhibited the greatest increase in biomass due to OWF placement (Figure 5a). Differences in the change in harvest rate among species reflected the degree to which biomass was unequally distributed over space, and thus how much the loss of sampling in the OWF area biased the survey estimate of mean biomass, as well as what the relative biomass was at the baseline target SPR of 0.5 (Figure 5b). For example, the simulations using archetypal Dover sole parameters

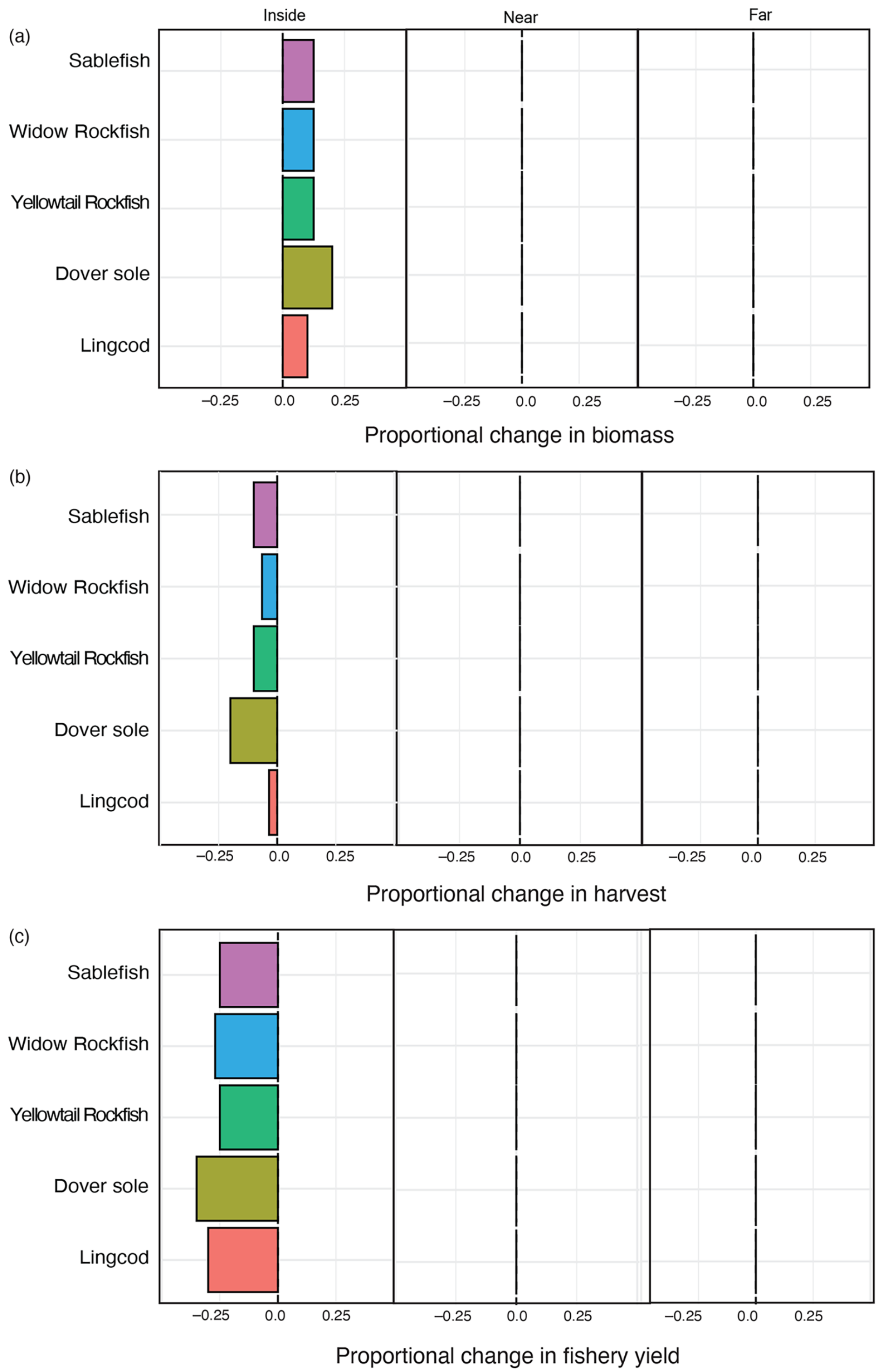


**FIGURE 5** Summary of proportional change in (a) biomass, (b) target spawners per recruit, and (c) yield in each scenario (high-density habitat centered in, near, or far from the offshore wind farm [OWF]), for each archetypal species. Values are averaged over the 20 years post-OWF implementation. The changes are expressed as a proportion of the value of each variable immediately prior to OWF placement.

had more biomass concentrated in the high-density habitat (due to the aforementioned combination of small home range and low natural mortality rate), so the loss of survey data in the OWF biased the estimated biomass downwards considerably when the OWF contained the highest density habitat (Figure 5b). Alternatively, the long-lived archetypal Yellowtail Rockfish had very high biomass relative to $B_0$ even when fished at SPR 0.5, so the loss of survey data was not sufficient to trigger the harvest control rule (Appendix S1: Figure S3). With respect to fishery yields, the trend across species essentially corresponded to home range size, with species with larger home ranges exhibiting smaller relative decreases in yield from the OWF, because the portion of the population inside the OWF was still partially accessible to the fishery. Across all species, the magnitude of biomass increases and fishery yield decreases was greatest when the OWF was centered on the highest quality habitat, and negligible when the OWF was far from high-quality habitat (compare across columns in Figure 5a).

## DISCUSSION

The rapid expansion of OWFs along the US coastlines has ignited a need for comprehensive research into their potential impacts on fisheries and fisheries management. Unlike traditional MPAs designed specifically for conservation purposes, OWFs have the potential to promote conservation and be classified as OECMs. Work has been done within the United States showing that exclusion zones for coastal military bases and NASA installations provide conservation benefits (Esgro et al., 2020; Sullivan-Stack et al., 2022). Other nontraditional areas that can have unintended conservation outcomes include oil and gas rigs, aquaculture net pens, maritime transportation lanes, or around private property (Rogers-Bennett et al., 2013; Sullivan-Stack et al., 2022). However, our results highlight a crucial aspect of OECMs for fishery species: the potential OECM will only have a conservation benefit (in terms of an overall increase in fish biomass) when it closes high-quality habitat. This is an intuitive result that mirrors results from the MPA literature: closing an area that includes a greater density of essential habitat will produce a greater benefit for population biomass. However, that benefit is accompanied by a loss in fishery yields. In our analysis, the loss of yield arose both because of the loss of fishing grounds as well as the loss of fishery-independent data, which led to greater restrictions on harvest.

Our research delved into the pressing issue of potential loss in sampling areas and its implications for fisheries management. Using simulations based on archetypal groundfish species, we highlighted the importance of addressing both the loss of sampling data for stock assessments and the placement of OWFs in relation to favorable habitats. We found that when the stock was fished at the original, sustainable harvest rate of 50% SPR, the initial loss in yield due to OWF closure was comparable to the proportion of the stock that was found in the excluded area. Thus, the initial loss in yield was greatest when the OWF happened to be sited on the highest density habitat region of the coastline, and little change was seen if the OWF was far from high-density habitat. That initial loss in yield was gradually offset by larval spillover, as fish inside the OWF lived longer and increased their lifetime reproductive output, then exporting larvae to fished area (Hopf et al., 2023). However, the fishery loss was compounded by the loss of fishery-independent survey data within the OWF, which led to both biased and less precise estimates of stock biomass. That effect was also greatest when the OWF coincided with the highest density habitat areas, because those areas supported the greatest fish biomass. Losing those data led to (incorrect) lower estimates of stock status and then reductions in the harvest rate when the pre-specified harvest control rule was applied. The magnitude of these effects varied among fish species, with long-lived species with small adult home ranges experiencing both the greatest increase in population biomass due to the OWF closure and larger reductions in fishery yield.

The outcomes of the OWF simulation we conducted echoed the findings in MPA literature, which has shown that when fishery harvest rates are sustainable (i.e., not overfishing), spatial management produces a trade-off between conservation outcomes (maximizing fish biomass), and fishery outcomes (maximizing yield; Holland & Brazee, 1996; White, Botsford, Hastings, & Largier, 2010). The MPA literature suggests that if the pre-OWF harvest rate were unsustainable then the closure could be expected to lead to increases in both stock biomass and yield, with the level of increase depending on whether fishing effort was removed or reallocated following the closure (White, Botsford, Moffitt, & Fischer, 2010). However, focusing on the case of sustainable harvest rates in our analysis, the biomass-yield trade-off was heightened by the loss of fishery-independent data, particularly when those data would have sampled the highest biomass density portion of the stock. The potential for this type of data loss and its effect on fishery management has been suggested previously (Field et al., 2006) but, to our knowledge, not directly considered in models of management using spatial fishery closures. Our analysis indicated that species with small ranges of adult movement had somewhat greater declines in expected fishery yield due to the OWF than more mobile species, despite increases in biomass inside the OWF area for all species. The slow increases in fish biomass inside the OWF and even slower increase in yield due to larval spillover is consistent with findings from the MPA literature; the maximum level of larval spillover to fished areas would not be reached for years after closure, because of the time lag associated with new larval recruits to the closed area reaching reproductive maturity, and then time lag for the larvae they spawn that export to fished areas to reach the age at first capture (Barceló et al., 2021). The 20-year time horizon in our simulations is a long time from the point of view of management but short relative to those demographic lags in these groundfish species.

Our study offers valuable insights into the potential impacts of OWFs on groundfish populations, although it is crucial to recognize the limitations and assumptions inherent in our modeling approach. A notable constraint is our reliance on a simplified habitat, omitting considerations for factors such as depth and substrate type (Juan-Jordá et al., 2009). Additionally, our model operates within a single-species framework, overlooking the influence of interspecies interactions on population dynamics in the real-world context (Baskett & Barnett, 2015; Pikitch et al., 2004). An embedded assumption in our model is the assumed constancy of habitat suitability for each species over time. Yet, real-world habitat quality can fluctuate due to natural factors like variation in environmental conditions and predation. Furthermore, our model overlooks potential fish behavioral responses to OWFs, such as avoidance or attraction, which could significantly impact the distribution of groundfish populations and their interactions with OWFs (Bray et al., 2016). Floating OWFs have the potential to be fish aggregation devices which would create a biomass build up within this closure area; however, it is unknown how they will affect different species of fish. For example, the floating OWFs that could be built on the West Coast may have limited influence on groundfish species that are living hundreds of meters below the primary OWF structures, but pelagic species that inhabit surface waters may be attracted to the structures (Wilhelmsson et al., 2006). Another limitation arises from our model's failure to explicitly represent the economic feedback dynamics of the fishing fleet. We implemented a harvest control rule that assumed a proportional relationship between fishing effort and biomass; however, it disregarded real-world influences like economic conditions (Haynie & Pfeiffer, 2012). For example, Dover sole is currently not fully exploited because of market forces and co-occurrence with other species with restrictive catch limits (Wetzel & Berger, 2021). These external factors can consequently affect the observed impacts of OWFs on yield and target spawners per recruit. Future research to develop more spatially realistic models that specifically evaluate the species-specific effects of OWFs will be informative. Despite these acknowledged limitations, our research serves as a valuable starting point for understanding the potential fishery costs and trade-offs associated with OWF implementation. It can inform decision-making about OWF development and guide further research efforts in areas identified as needing more comprehensive investigation.

In particular, our work highlights the need to mitigate the potential loss of fishery-independent survey data in large-scale OWF implementation. There are several ways to approach that mitigation; for example, in some regions, large-scale trawling surveys are excluded from OWFs but smaller vessels are not, so sampling conducted by those vessels (e.g., hook-and-line or video survey) could be calibrated with trawling surveys prior to OWF implementation to provide a substitute data stream (Methratta et al., 2023). Another alternative would be to include information on the spatial distribution of habitat features (e.g., in the form of a species distribution model) when translating the survey data into an index of abundance, so that the index could be reweighted to account for lost sampling areas (Bell et al., 2022; Switzer et al., 2023). However, any of those solutions would need to be implemented prior to the enforcement of area closures.

As we approach clean energy targets and the 30 by 30 goal to protect 30% of the world's oceans by 2030, OWFs will continue to play a significant role in marine spatial planning. Our modeling study highlights the conservation impact of closing areas for OWF development due to fishing restrictions, suggesting that OWFs could be classified as OECMs. However, they would only tend to have conservation benefits (if benefits are quantified in terms of increasing fish biomass) if they were placed in areas that were previously fished. That said, the guidance for identifying OECMs varies; for example, Jonas et al. (2023) provide a list of possible biodiversity benefits that do not really map onto the single-species population dynamic results we have produced.

Ongoing efforts nationwide aim to comprehensively evaluate all the impacts of OWFs, including the economic impact and involving stakeholders in this process (e.g., Brunbauer et al., 2023; Chaji & Werner, 2023). Management strategies may need to address potential solutions to information loss, such as relocating sampling locations or adopting innovative techniques. The implications of OWF expansion extend to common pool resource problems, demanding creative solutions and in-depth research. This timely research topic contributes significantly to the scientific literature, assisting decision-makers in making forward-looking, climate-conscious choices.

## AUTHOR CONTRIBUTIONS

M. Campbell and J. W. White conceived the study. All authors designed the analysis. M. Campbell wrote model code and performed the analysis. All authors interpreted the results. M. Campbell wrote the manuscript with contributions from J. W. White and J. F. Samhouri.

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## CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

## DATA AVAILABILITY STATEMENT

Model code (Campbell, 2025) is available from Zenodo: https://doi.org/10.5281/zenodo.15083888.

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## REFERENCES

Adams, G. D., M. S. Kapur, K. McQuaw, S. Thurner, O. S. Hamel, A. Stephens, and C. R. Wetzel. 2019. *Stock Assessment Update: Status of Widow Rockfish (Sebastes entomelas) along the U.S. West Coast in 2019*. Portland, OR: Pacific Fishery Management Council.

Agardy, T., G. N. di Sciara, and P. Christie. 2011. "Mind the Gap: Addressing the Shortcomings of Marine Protected Areas through Large Scale Marine Spatial Planning." *Marine Policy* 35: 226–232.

Allen-Jacobson, L. M., A. W. Jones, A. J. Mercer, S. X. Cadrin, B. Galuardi, D. Christel, A. Silva, A. Lipsky, and J. B. Haugen. 2023. "Evaluating Potential Impacts of Offshore Wind Development on Fishing Operations by Comparing Fine- and Coarse-Scale Fishery-Dependent Data." *Marine and Coastal Fisheries* 15: e10233.

Barceló, C., J. W. White, L. W. Botsford, and A. Hastings. 2021. "Projecting the Timescale of Initial Increase in Fishery Yield after Implementation of Marine Protected Areas." *ICES Journal of Marine Science* 78: 1860–71.

Baskett, M. L., and L. A. K. Barnett. 2015. "The Ecological and Evolutionary Consequences of Marine Reserves." *Annual Review of Ecology, Evolution, and Systematics* 46: 49–73.

Beger, M., S. Linke, M. Watts, E. Game, E. Treml, I. Ball, and H. P. Possingham. 2010. "Incorporating Asymmetric Connectivity into Spatial Decision Making for Conservation." *Conservation Letters* 3: 359–368.

Bell, R. J., B. Grieve, M. Ribera, J. Manderson, and D. Richardson. 2022. "Climate-Induced Habitat Changes in Commercial Fish Stocks." *ICES Journal of Marine Science* 79: 2247–64. https://doi.org/10.1093/icesjms/fsac154.

Bonsu, P., J. Letschert, K. Yates, J. Svendsen, J. Berkenhagen, M. Rozemeijer, T. Kerkhove, J. Rehren, and V. Stelzenmüller. 2024. "Co-Location of Fisheries and Offshore Wind Farms: Current Practices and Enabling Conditions in the North Sea." *Marine Policy* 159: 105941. https://doi.org/10.1016/j.marpol.2023.105941.

Botsford, L. W., A. Hastings, and S. D. Gaines. 2001. "Dependence of Sustainability on the Configuration of Marine Reserves and Larval Dispersal Distance." *Ecology Letters* 4: 144–150.

Botsford, L. W., J. W. White, and A. Hastings. 2019. *Population Dynamics for Conservation*. Oxford: Oxford University Press.

Bray, L., S. Reizopoulou, E. Voukouvalas, T. Soukissian, C. Alomar, M. Vázquez-Luis, S. Deudero, M. J. Attrill, and J. M. Hall. 2016. "Expected Effects of Offshore Wind Farms on Mediterranean Marine Life." *Journal of Marine Science and Engineering* 4: 18.

Breton, S.-P., and G. Moe. 2009. "Status, Plans and Technologies for Offshore Wind Turbines in Europe and North America." *Renewable Energy* 34: 646–654.

Brunbauer, M., K. McClellan Press, K. A. Williams, B. K. Dresser, J. Gulka, and G. Lampman. 2023. "Effective Stakeholder Engagement for Offshore Wind Energy Development: The State of New York's Fisheries and Environmental Technical Working Groups." *Marine and Coastal Fisheries* 15: e10236.

Burkhard, B., and K. Gee. 2012. "Establishing the Resilience of a Coastal-Marine Social-Ecological System to the Installation of Offshore Wind Farms." *Ecology and Society* 17: 32.

Cabral, R. B., S. D. Gaines, B. A. Johnson, T. W. Bell, and C. White. 2017. "Drivers of Redistribution of Fishing and Non-Fishing Effort after the Implementation of a Marine Protected Area Network." *Ecological Applications* 27: 416–428.

Campbell, M. 2025. "campmarg/Model_1: Coastline Model Code V1." Zenodo. https://doi.org/10.5281/zenodo.15083887.

Chaji, M., and S. Werner. 2023. "Economic Impacts of Offshore Wind Farms on Fishing Industries: Perspectives, Methods, and Knowledge Gaps." *Marine and Coastal Fisheries* 15: e10237.

Ehresmann, R. K., A. H. Beaudreau, and K. M. Green. 2018. "Movement Patterns of Juvenile Sablefish within a Nursery Area in Southeast Alaska." *Transactions of the American Fisheries Society* 147: 1052–66. https://doi.org/10.1002/tafs.10099.

Esgro, M. W., J. Lindholm, K. J. Nickols, and J. Bredvik. 2020. "Early Conservation Benefits of a de Facto Marine Protected Area at San Clemente Island, California." *PLoS One* 15: e0224060.

European Commission. 2019. "Offshore Wind and Commercial Fisheries." European Marine Spatial Planning Platform Conflict Fiche 5. https://www.msp-platform.eu/sector-information/offshore-wind-and-fisheries.

Farr, H., B. Ruttenberg, R. K. Walter, Y.-H. Wang, and C. White. 2021. "Potential Environmental Effects of Deepwater Floating Offshore Wind Energy Facilities." *Ocean and Coastal Management* 207: 105611.

Field, J., A. Punt, R. Methot, and C. Thomson. 2006. "Does MPA Mean 'Major Problem for Assessments'? Considering the Consequences of Place-Based Management Systems." *Fish and Fisheries* 7: 284–302.

Game, E. T., E. McDonald-Madden, M. L. Puotinen, and H. P. Possingham. 2008. "Should We Protect the Strong or the Weak? Risk, Resilience, and the Selection of Marine Protected Areas." *Conservation Biology* 22: 1619–29.

Garcia, S. M., J. Rice, A. Himes-Cornell, K. J. Friedman, A. Charles, D. Diz, J. Appiott, and M. J. Kaiser. 2022. "OECMs in Marine Capture Fisheries: Key Implementation Issues of Governance, Management, and Biodiversity." *Frontiers in Marine Science* 9: 920051.

Grafton, R. Q., and T. Kompas. 2005. "Uncertainty and the Active Adaptive Management of Marine Reserves." *Marine Policy* 29: 471–79.

Grüss, A., D. M. Kaplan, S. Guénette, C. M. Roberts, and L. W. Botsford. 2011. "Consequences of Adult and Juvenile Movement for Marine Protected Areas." *Biological Conservation* 144: 692–702.

Hagerman, F. B. 1952. *The Biology of the Dover Sole, Microstomus pacificus (Lockington)*. Fish Bulletin No. 85. Sacramento, CA: State of California Department of Fish and Game.

Hannah, R. W., T. V. Buell, and M. T. O. Blume. 2008. *Reducing Bycatch in Oregon's Recreational Groundfish Fishery: Experimental Results*. Information Reports Number 2008–03. Newport, OR: Oregon Department of Fish and Wildlife.

Haynie, A. C., and L. Pfeiffer. 2012. "Why Economics Matters for Understanding the Effects of Climate Change on Fisheries." *ICES Journal of Marine Science* 69: 1160–67.

Holland, D. S., and R. J. Brazee. 1996. "Marine Reserves for Fisheries Management." *Marine Resource Economics* 11: 157–171.

Hopf, J. K., G. P. Jones, D. H. Williamson, and S. R. Connolly. 2023. "Fishery Consequences of Marine Reserves: Short-Term Pain for Longer-Term Gain." *Ecological Applications* 26: 818–829.

Jonas, H. D., K. MacKinnon, D. Marnewick, and P. Wood. 2023. *Site-Level Tool for Identifying Other Effective Area-Based Conservation Measures (OECMs)*, 1st ed. IUCN WCPA Technical Report Series No. 6. Gland: IUCN.

Juan-Jordá, M. J., J. A. Barth, M. E. Clarke, and W. W. Wakefield. 2009. "Groundfish Species Associations with Distinct Oceanographic Habitats in the Northern California Current." *Fisheries Oceanography* 18: 1–19.

Kapur, M. S., Q. Lee, G. M. Correa, M. A. Haltuch, V. Gertseva, and O. S. Hamel. 2021. *Draft Status of Sablefish (Anoplopoma fimbria) along the US West coast in 2021*. Portland, OR: Pacific Fishery Management Council.

Keller, A. A., J. R. Wallace, and R. D. Methot. 2017. *The Northwest Fisheries Science Center's West Coast Groundfish Bottom Trawl Survey: History, Design, and Description*. Seattle, WA: U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-136. https://doi.org/10.7289/V5/TM-NWFSC-136.

Kellner, J. B., I. Tetreault, S. D. Gaines, and R. M. Nisbet. 2007. "Fishing the Line near Marine Reserves in Single and Multispecies Fisheries." *Ecological Applications* 17: 1039–54.

Leslie, H. M. 2005. "A Synthesis of Marine Conservation Planning Approaches." *Conservation Biology* 19: 1701–13.

Methratta, E., A. Hawkins, B. Hooker, A. Lipsky, and J. Hare. 2020. "Offshore Wind Development in the Northeast US Shelf Large Marine Ecosystem: Ecological, Human, and Fishery Management Dimensions." *Oceanography* 33: 16–27.

Methratta, E., A. Lipsky, and J. Boucher. 2023. "Offshore Wind Project-Level Monitoring in the Northeast U.S. Continental Shelf Ecosystem: Evaluating the Potential to Mitigate Impacts to Long-Term Scientific Surveys." *Frontiers in Marine Science* 10: 1214949. https://doi.org/10.3389/fmars.2023.1214949.

Moffitt, E., J. White, and L. Botsford. 2013. "Accurate Assessment of Marine Protected Area Success Depends on Metric and Spatiotemporal Scale of Monitoring." *Marine Ecology Progress Series* 489: 17–28.

Pacific Fisheries Management Council. 2024. "Pacific Coast Groundfish Fishery Management Plan." https://www.pcouncil.org/managed_fishery/groundfish.

Pearcy, W. G. 1992. "Movements of Acoustically-Tagged Yellowtail Rockfish (*Sebastes f1avidus*) on Heceta Bank, Oregon." *Fishery Bulletin* 90: 726–735.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, et al. 2004. "Ecosystem-Based Fishery Management." *Science* 305: 346–47.

Pınarbaşı, K., I. Galparsoro, D. Depellegrin, J. Bald, G. Pérez-Morán, and Á. Borja. 2019. "A Modelling Approach for Offshore Wind Farm Feasibility with Respect to Ecosystem-Based Marine Spatial Planning." *Science of the Total Environment* 667: 306–317.

Punt, A. E., D. S. Butterworth, C. L. de Moor, J. A. A. De, Oliveira, and M. Haddon. 2016. "Management Strategy Evaluation: Best Practices." *Fish and Fisheries* 17: 303–334.

Püts, M., A. Kempf, C. Möllmann, and M. Taylor. 2023. "Trade-Offs between Fisheries, Offshore Wind Farms and Marine Protected Areas in the Southern North Sea – Winners, Losers and Effective Spatial Management." *Marine Policy* 152: 105574.

R Core Team. 2023. *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Rassweiler, A., C. Costello, and D. A. Siegel. 2012. "Marine Protected Areas and the Value of Spatially Optimized Fishery Management." *Proceedings of the National Academy of Sciences of the United States of America* 109: 11884–89.

Ressler, P. H., G. W. Fleischer, V. G. Wespestad, and J. Harms. 2009. "Developing a Commercial-Vessel-Based Stock Assessment Survey Methodology for Monitoring the U.S. West Coast Widow Rockfish (*Sebastes entomelas*) Stock." *Fisheries Research* 99: 63–73.

Rogers-Bennett, L., K. E. Hubbard, and C. I. Juhasz. 2013. "Dramatic Declines in Red Abalone Populations after Opening a "De Facto" Marine Reserve to Fishing: Testing Temporal Reserves." *Biological Conservation* 157: 423–431.

Siegel, D., B. Kinlan, B. Gaylord, and S. Gaines. 2003. "Lagrangian Descriptions of Marine Larval Dispersion." *Marine Ecology Progress Series* 260: 83–96.

Stephens, A., and I. G. Taylor. 2017. *Status of Yellowtail Rockfish (Sebastes flavidus) along the U.S. Pacific Coast in 2017*. Portland, OR: Pacific Fishery Management Council.

Sullivan-Stack, J., O. Aburto-Oropeza, C. M. Brooks, R. B. Cabral, J. E. Caselle, F. Chan, J. E. Duffy, et al. 2022. "A Scientific Synthesis of Marine Protected Areas in the United States: Status and Recommendations." *Frontiers in Marine Science* 9: 849927.

Switzer, T. S., S. F. Keenan, K. A. Thompson, C. P. Shea, A. R. Knapp, M. D. Campbell, B. Noble, C. Gardner, and M. C. Christman. 2023. "Integrating Assemblage Structure and Habitat Mapping Data Into the Design of a Multispecies Reef Fish Survey." *Marine and Coastal Fisheries* 15: e210245.

Taylor, I. G., K. F. Johnson, B. J. Langseth, A. Stephens, L. S. Lam, M. H. Monk, A. D. Whitman, and M. A. Haltuch. 2021. *Status* *of Lingcod (Ophiodon elongatus) along the Northern U.S. West Coast in 2021*. Portland, OR: Pacific Fishery Management Council.

Thomas, C. J., J. Lambrechts, E. Wolanski, V. A. Traag, V. D. Blondel, E. Deleersnijder, and E. Hanert. 2014. "Numerical Modelling and Graph Theory Tools to Study Ecological Connectivity in the Great Barrier Reef." *Ecological Modelling* 272: 160–174.

Tolimieri, N., K. Andrews, G. Williams, S. Katz, and P. Levin. 2009. "Home Range Size and Patterns of Space Use by Lingcod, Copper Rockfish and Quillback Rockfish in Relation to Diel and Tidal Cycles." *Marine Ecology Progress Series* 380: 229–243.

UNFCCC. 2022. "New International Biodiversity Agreement Strengthens Climate Action." https://unfccc.int/news/new-international-biodiversity-agreement-strengthens-climate-action.

Vandeperre, F., R. M. Higgins, J. Sánchez-Meca, F. Maynou, R. Goñi, P. Martín-Sosa, A. Pérez-Ruzafa, et al. 2011. "Effects of No-Take Area Size and Age of Marine Protected Areas on Fisheries Yields: A Meta-Analytical Approach." *Fish and Fisheries* 12: 412–426.

Walters, C. J., and R. Bonfil. 1999. "Multispecies Spatial Assessment Models for the British Columbia Groundfish Trawl Fishery." *Canadian Journal of Fisheries and Aquatic Sciences* 56: 601–628.

Walters, C. J., and R. Hilborn. 1978. "Ecological Optimization and Adaptive Management." *Annual Review of Ecology and Systematics* 9: 157–188.

Warlick, A. J., O. R. Liu, C. J. Harvey, J. F. Samhouri, E. L. Hazen, and K. S. Andrews. 2025. "A Framework to Evaluate Dynamic Social and Ecological Interactions between Offshore Wind Energy Development and Commercial Fisheries in a Changing Climate: A US West Coast Perspective." *PLOS Climate* 4: e0000526.

Wetzel, C. R., and A. M. Berger. 2021. *Status of Dover Sole (Microstomus pacificus) along the U.S. West Coast in 2021*. Portland, OR: Pacific Fishery Management Council.

White, J. W., J. K. Hopf, D. P. Kilduff, A. Hastings, and L. W. Botsford. 2025. "The Roles of Population Dynamics Theory in the Design and Assessment of Marine Reserves." *Theoretical Ecology* 18: 15.

White, J. W., J. Schroeger, P. T. Drake, and C. A. Edwards. 2014. "The Value of Larval Connectivity Information in the Static Optimization of Marine Reserve Design." *Conservation Letters* 7: 533–544.

White, J. W., L. Botsford, A. Hastings, and J. Largier. 2010. "Population Persistence in Marine Reserve Networks: Incorporating Spatial Heterogeneities in Larval Dispersal." *Marine Ecology Progress Series* 398: 49–67.

White, J. W., L. W. Botsford, A. Hastings, M. L. Baskett, D. M. Kaplan, and L. A. K. Barnett. 2014. "Transient Responses of Fished Populations to Marine Reserve Establishment: Transient Dynamics in Marine Reserves." *Conservation Letters* 6: 180–191.

White, J. W., L. W. Botsford, E. A. Moffitt, and D. T. Fischer. 2010. "Decision Analysis for Designing Marine Protected Areas for Multiple Species with Uncertain Fishery Status." *Ecological Applications* 20: 1523–41.

White, J. W., L. W. Botsford, M. L. Baskett, L. A. Barnett, R. J. Barr, and A. Hastings. 2011. "Linking Models with Monitoring Data for Assessing Performance of No-Take Marine Reserves." *Frontiers in Ecology and the Environment* 9: 390–99.

Wilhelmsson, D., T. Malm, and M. C. Öhman. 2006. "The Influence of Offshore Windpower on Demersal Fish." *ICES Journal of Marine Science* 63: 775–784.

Yu, Q., K. Liu, A. P. Teixeira, and C. G. Soares. 2020. "Assessment of the Influence of Offshore Wind Farms on Ship Traffic Flow Based on AIS Data." *Journal of Navigation* 73: 131–148.

## SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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