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Spatial population dynamics models estimate minimal effects on groundfish biomass and fishery landings from proposed offshore wind farm development off Oregon, USA

M. Campbell[1✉], O. R. Liu[2,3], J. F. Samhouri[2] & J. W. White[1]

With an international push to develop and reach renewable energy targets, there is increased interest in creating floating offshore wind farms (OWFs). The installation of OWF infrastructure is expected to limit access and movement of most vessels, effectively creating permanent spatial closure for certain fisheries. Here we explore the implications of proposed OWFs off the coast of Oregon, USA, for two trawl-caught groundfish species. We created a spatially explicit population model based on both fishery independent survey data and logbook records. We simulated the short-term impacts of the proposed OWFs on two stocks, assuming the fishery either maintained constant effort or constant yield outside of the OWF area after implementation. Our analysis predicts that OWF closures would lead to minimal changes in fishing yield and that biomass build up within OWF areas that spilled over into areas open to fishing would not produce a substantial increase in fishery yield. This research advances understanding of the potential impacts of offshore energy development on fisheries and conservation outcomes.

Keywords Groundfish, Oregon, Offshore wind farms, Other effective conservation area-based measure (OECM), Marine spatial planning, Population dynamics model

Globally, demand is growing for the development of offshore renewable energy, including wind and wave energy[1,2]. One type of infrastructure that has been proposed but has not yet seen widespread implementation is floating offshore wind farms (OWFs) on the continental shelf[3]. Implementing and operating OWF infrastructure is expected to require closing affected areas of the shelf to vessel traffic or creating operational constraints for fishing vessels, preventing access to any previously used fishing grounds in that area. While some of the effects of OWF infrastructure on fish biology have been investigated, such as the potential for turbine structures to act as artificial reefs or fish aggregation devices[4–6], the possible impacts of the spatial closures on fisheries are less well studied.

Most studies of OWF effects on marine ecosystems have been conducted in Europe where over 2,000 turbines are deployed across 69 sites; those are all fixed-bottom platforms, and there is a lack of data on floating platforms[7]. European studies on fixed-bottom wind farms offer valuable insights into the need for baseline datasets, comprehensive impact studies, and stakeholder involvement during all project stages[8,9]. However, deployments off the United States Pacific coast would be at much greater depths and further offshore on the continental shelf than these existing European studies; there have been proposals to create OWFs in deep (> 100 m) offshore regions of the continental shelf[10]. Importantly, the closed areas associated with OWFs could

[1]Department of Fisheries, Wildlife, and Conservation Sciences, Coastal Oregon Marine Experiment Station, Oregon State University, Newport, Oregon 97365, USA. [2]Conservation Biology Division, Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, 2725 Montlake Boulevard East, Seattle, Washington 98112, USA. [3]Under Contract to the Northwest Fisheries Science Center, National Oceanic and Atmospheric Administration, Ocean Associates, Inc, 2725 Montlake Boulevard East, Seattle, Washington 98112, USA. ✉email: mccampbell2000@gmail.com

function similarly to other types of spatial fishery closures, such as marine protected areas. Yet, OWFs are not designed with the same goals and planning processes as marine protected areas and so may or may not have the same outcomes for fisheries and conservation[11]. Here we explore the implications of potential OWFs in Oregon for the short-term dynamics of specific groundfish fisheries. We focused on groundfish fisheries because of their economic importance to commercial fisheries in Oregon and the expected overlap between OWF areas, OWF infrastructure (moorings and transmission cables) and groundfish habitat.

In the United States, the federal agency overseeing offshore wind development is the Bureau of Ocean Energy Management (BOEM), which follows an authorization process containing multiple phases before proposed OWF areas can be leased and developed. The first stage is the planning and analysis phase. In 2021, BOEM identified two potential 'call areas' for energy development off the coast of the state of Oregon, near the ports of Coos Bay and Brookings (Fig. 1). In response, local fishers expressed concern about the loss of fishing grounds and potential impacts to various groundfish species in the Coos Bay[12]. BOEM conducted a preliminary environmental impact assessment for those two call areas in 2022, primarily focused on potential impacts to crab harvesting, seabird populations, and marine mammal displacement from offshore wind energy development[10]. This assessment was based on geospatial data layers, including data on fishery landings in the area[10]. Based on that assessment and other constraints, such as avoiding areas used by the U.S. military, BOEM identified two draft 'Wind Energy Areas' within the two call areas for potential wind development leases. At the time of

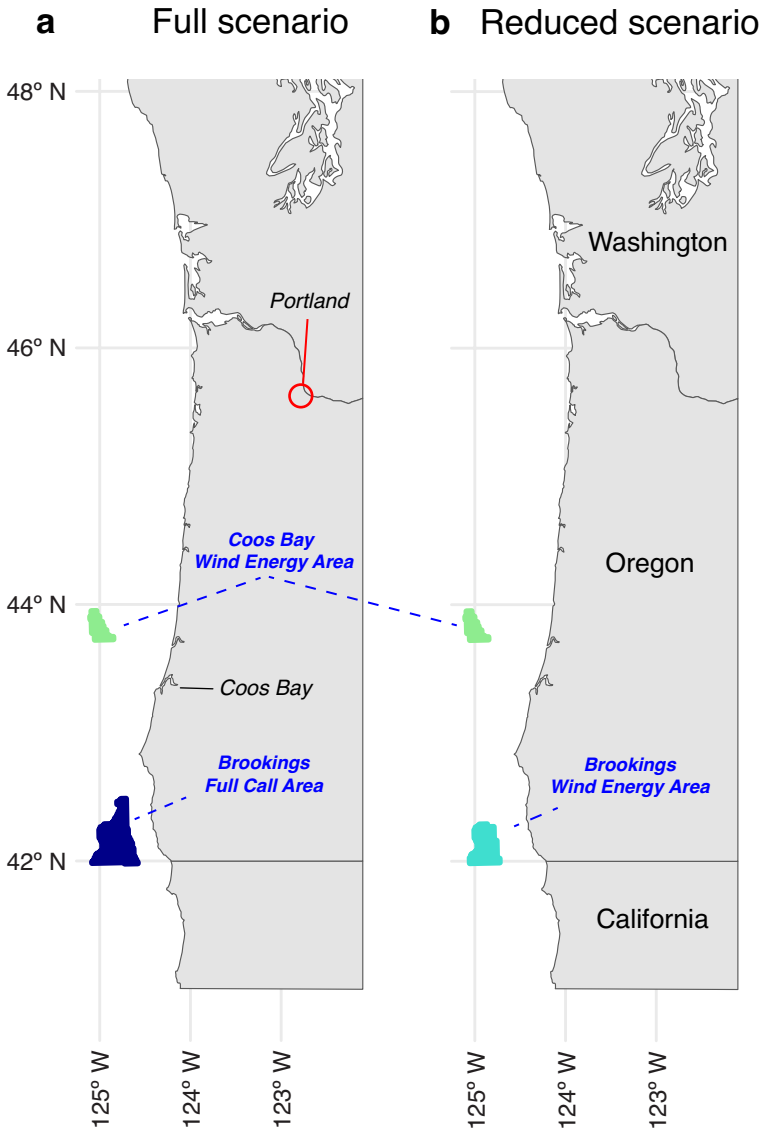

Fig. 1. Map depicted the two scenarios for possible wind energy area areas off the coast of Oregon. The (**a**) Full and (**b**) Reduced scenarios differ in the size of the 'Brookings' area. The location of the city of Portland, OR, and Coos Bay are shown for reference. Figure was created using package 'sf'[13] and 'rnaturalearth'[14] in R version 4.5.1[15] with GIS layers provided by Oregon Department of Fish and Wildlife.

writing, the leasing process has been paused by the government due to lack of interest from developers and the change to a new U.S. presidential administration with different energy development priorities. Had the projects moved forward, the subsequent steps for leasing, site assessment, and construction would have been expected to last until approximately 2032 before operations begin. Our analysis addresses the hypothetical impacts of wind energy development in this region had it moved forward in 2025, based on data available to date; it will also be informative for any future development plans, presuming there are not large-scale shifts in groundfish fisheries between now and then. The BOEM analyses of fishery impacts of OWF development in Oregon waters[10] have been static, effectively assuming that any fishery landings that would have been taken in the proposed OWF areas will be lost, with economic costs proportional to that lost fishery yield. However, much like when a no-take marine protected area causes fishing grounds to be closed to fishing, one would expect that OWF development would cause the fish populations to respond to the change in fishing effort, and a dynamic response of the fishing fleet to redistribute harvesting effort to locations outside of the closure areas. These issues have not been well explored in the context of OWF development, but both static and dynamic impact analyses have been applied in the context of marine protected area planning, and they can produce different results. For example, White et al.[16]compared static and dynamic assessments of the fishery consequences of proposed marine protected areas in California, and found that when one accounts for the potential for biomass buildup and reallocation of fishing effort in protected areas, there could either be net gains to both fishery yield and overall fish population biomass from the closures (if the fishery is being overharvested) or there could be a tradeoff between fishery yield and fish biomass (if harvesting is sustainable). However, even the dynamic analysis by White et al.[16]was based on long-term equilibrium outcomes, and short-term transient responses of fished populations (and fishery yields) to changes in management can differ sharply from long-term equilibrium conditions[17,18].

To move beyond static assessments of the projected fishery impacts of the potential OWF areas on the Oregon coast, we developed a dynamic population model to forecast the transient changes in fish populations and their fisheries following OWF designation. From the extensive body of literature related to marine protected area design and assessment, we can develop basic expectations for the consequences of closure area size, larval connectivity, adult fish movement, and fishery management on fish population dynamics within these protected areas over both long-term (when systems have reached dynamic equilibrium, e.g. centuries) and short-term (years to decades) time scales[19–21]. Over short time scales (relative to the life histories of the species), we would expect the transient population dynamics to reflect the initial cessation of fishing within a closed area, leading to a gradual increase in biomass there. The effects on fishery yields, in turn, would depend on the response of the fleet to the closure; if overall harvesting effort remains constant but is displaced to the rest of the fishing grounds, past results suggest that there would be an initial reduction fishery yields, followed by an increase in fish biomass at a time lag set by fish life history, and potential spillover of adult or larval fish into the fished area[17,21–23]. In that scenario, the magnitude and direction of the expected response is expected to depend on the level of fishing prior to the closure. For instance, if the stock was being overharvested, we would expect the closure to eventually increase both fish biomass and fishery yield[22]. Alternatively, the fleet may increase effort to make up for the loss in fishing grounds in order to maintain consistent yields after the closure; that scenario has not been theoretically explored in the marine protected area literature, but presumably it would lead to very little effect of the closure on fishery yield, but would reduce stock biomass.

We expected those general results from the marine protected area literature to hold in our analysis of the effects of Oregon OWF closures on groundfish fisheries. In a recent simulation analysis using spatially explicit age-structured models of five different groundfish species archetypes along an idealized coastline, we found that the effect of OWF closures depended greatly on the placement of the closure area relative to the location of high-quality habitat for the species archetype of interest[11]. If the OWF fell within high-quality habitat (because OWF sites may not be selected with regard to fish habitat per se but instead based on where wind conditions and other logistical considerations are best), our analysis predicted the most substantial increases in fish biomass and the greatest losses to the fishery. We also showed that shorter-lived and more sedentary species were able to build up biomass within OWF areas faster than those that are longer lived and more mobile[11].

Here we build on the analyses of species archetypes by Campbell et al.[11], by developing a spatially explicit model of groundfish population dynamics on the continental shelf offshore of Oregon in order to quantify the expected effects of the two potential OWF sites on fish populations and fisheries. We specifically considered the scenario in which fishing effort would be spatially reallocated following OWF implementation, and we considered two possibilities for how the fishing fleet would respond to the closure: maintaining the same level of effort, or increasing effort in an attempt to achieve yields equivalent to the allowable catch. The central question we asked was whether the eventual impact on fishery yields would be greater or less than simply the loss of the proportion of landings within the offshore wind farm area prior to implementation (i.e., would the dynamic response differ from the static one?). The two groundfish species, Dover sole (*Microstomus pacificus*, Pleuronectidae) and Lingcod (*Ophiodon elongatus*, Hexagrammidae), we modeled are commonly caught in Oregon, but currently have estimated exploitation rates below their management targets, and relatively small spatial scales of home range movement (Table 1). Therefore, in the case in which overall fishing effort was not increased, we expected the models to predict an overall small reduction of fishery yield due to OWF closures and a gradual, modest increase in biomass within the closure areas following the time delay associated with new recruits entering the fishery.

Methods

Our goal was to predict the short-term groundfish fishery impacts of OWFs off the coast of Oregon. We used a spatially explicit, age-structured population dynamics model that was informed by fishery-independent biological survey data and fishery logbook data to project the dynamics of fishing effort, fish biomass, and

Parameter	Definition	Dover Sole	Lingcod
M	Natural mortality ($year^{-1}$)	0.117 (Ref.[25], p. 20)	0.418 (Ref.[24], p. 105, 128)
H	Steepness	0.8 (Ref.[25], p)	0.8 (Ref.[24], p)
A_{max}	Maximum age (years)	50 (Ref.[25], p. 17)	25 (Ref.[24], p. 36)
A_{mat}	Age at maturity (years)	7.5 (Ref.[25], back-calculated from maturity-at-length)	3.23 (Ref.[24], p. 37)
L_{∞}	Asymptotic maximum length (cm)	48.5 (Ref.[25], p. 20)	103.645 (Ref.[24], p. 39)
k	von Bertalanffy growth (year -1)	0.117 (Ref.[25], p. 20)	0.193 (Ref.[24], p. 135)
A_{sel}	Age at 50% fishery selectivity (years)	14.5 (Ref.[25])	7.5 (Ref.[24], p. 195)
c	Weight at length coefficient (kg/cm^d)	2.97×10^{-6} (Ref.[25], p. 19)	2.802×10^{-6} (Ref.[24], p. 38)
D	Weight at length exponent	3.33 (Ref.[25], p. 19)	3.2766 (Ref.[24], p. 38)
σ_h	Home range diameter (km)	1.25 (Ref.[26], based on annual movement range)	0.00425 (Ref.[27])

Table 1. Parameter values of species which have been sourced from the latest stock assessments for each of the modeled species. Values derived from refs.[24] and [25], except for home range estimates taken from Refs.[26,27].

fishery yield stemming from the loss of fishing area. Full details and equations are provided in the Supplemental Materials.

Study location

The policy process for designating OWF sites offshore of Oregon consisted of multiple steps. Initially two "call areas" (locations identified for public comment for potential commercial wind energy leases in the area) were designated offshore of Coos Bay site and Brookings (Fig. 1)[28]. Both sites were situated on the continental shelf outside of Oregon territorial waters (3 nautical miles) but inside the US Exclusive Economic zone (EEZ) (200 km)[28]. Subsequently, smaller draft "wind energy areas" (WEAs) were identified within the call areas as the potential initial leasing sites for OWF development (though no development is currently being considered at the time of writing)[28] In our analysis, we examined two different potential closure scenarios: a 'Full Scenario' that consisted of the Full call area for Brookings (1,159 km^2) and the Coos Bay WEA and a 'Reduced Scenario' that consisted only of the WEAs for both Brookings (541.5 km^2) and Coos Bay (247.7 km^2) (Fig. 1). We chose not to include the full Coos Bay call area because after it was proposed it became clear that external constraints (e.g., U.S. Department of Defense uses and the presence of submarine cables) would prevent the full area from ever being developed for OWF, regardless of political considerations.

Study species and model spatial domain

We modeled the population dynamics of Lingcod (*Ophiodon elongatus*) and Dover sole (*Microstomus pacificus*), two groundfish species that are actively fished along the US Pacific coast, have recently undergone full stock assessments, and whose spatially-explicit habitat preferences are well-characterized by recent work in this region. We obtained life history and fishery selectivity parameters from the most recent stock assessments available for each species (Table 1)[24,25]. Dover sole has a longer lifespan (50 years) and an older age at maturity (~7.5 years) relative to Lingcod (25 years and ~3.2 years). Both species are exploited but as of the most recent stock assessments for each, they are not currently overfished, undergoing overfishing, or rebuilding[24,25]. Lingcod biomass has recently trended downward but stock biomass is estimated to be above management targets and the exploitation rate has been below catch limits, while Dover sole is underexploited with landings well below the catch limits because of market forces and catch limits on other species[24,25]. Based on observations of adult movement, Lingcod have adult home ranges (i.e., the typical movement space within a year, other than during spawning migrations) that have diameters on the order of a few meters in scale, while Dover sole exhibit adult movement patterns on a scale an order of magnitude greater[26,27].

To represent the spatial dynamics of the two species, we required estimates of the spatial distribution of some aspect of habitat quality and the spatial distribution of fishing effort. To characterize the habitat domain, we used species distribution models developed by Liu et al.[29,30]. These models were based on eight years (2003-2010) of fishery-independent, catch per unit effort (CPUE) data from an annual trawl survey, the West Coast Bottom Trawl Survey, operated by the National Oceanographic and Atmospheric Administration[31] and environmental covariates (bottom temperature and bottom dissolved oxygen) that were extracted from a California Current implementation of the Regional Ocean Modeling System (ROMS)[29,30]. The ROMS analysis included a hindcast for 1980-2010 that represented historical conditions, and three alternative climate projections for the period 1980-2100 based on downscaled global climate models[29, 30]. The species distribution models created by Liu et al.[29,30] were generated by fitting the 2003-2010 trawl survey data to the spatial distributions of environmental covariates from the ROMS hindcast. They then predicted the expected trawl catch per unit effort (CPUE) for the period 1980-2022 by projecting the model onto the annual bottom temperature and bottom oxygen fields extracted from the three projected ROMS models for that time period. In the current study, we assumed this fishery-independent CPUE was a reasonable first-order approximation for the relative biomass that could be supported in each location, so we calculated the mean projected CPUE from 1980-2022 across the three climate projections and across those years for each species as our metric of spatial habitat quality. We discretized the

species distribution model output (Fig. 2) into a grid of 10 × 10 km grid cells, a size chosen because it was the smallest possible resolution that was computationally efficient for the spatial model and also large enough that trawls in the logbook data (see next paragraph) tended to start and stop entirely within one cell.

To characterize the spatial distribution of fishing effort prior to OWF implementation, we obtained anonymized logbook data for 2012-2022 from the Oregon Department of Fisheries and Wildlife (ODFW) for commercial groundfish fleets. ODFW performed quality assurance/quality control filtering on the logbook data prior to sharing it. We did a second quality assurance/quality control and filtered the logbook data to extract Oregon landings for the two species and used the ODFW adjusted groundfish pounds for each species. We averaged these 10 years of data for each species at the same grid scale used for the species distribution model. We acknowledge there is a mismatch in the time period used in the CPUE forecast from the species distribution model (1980-2022) and the logbook data (2012-2022); this mismatch simply reflects the constraints of the data products available to us at the time of the analysis.

Based on the allocation of logbook landings to specific ports, we judged that the fishing effort in the OWF areas was based in ports distributed along the entire coast, so assuming a statewide fleet was reasonable (see Supplementary Fig. S1). In the dynamic model, we specified the relative fishing effort in each model cell for each of the species as being equal to the proportion of trips –over the most recent 10 years of logbook data– in which that species was landed. We assigned each trip to the grid cell where a given trawl ended, though assigning it instead to the starting locations did not meaningfully change the result (<1%) because the spatial model cells are large relative to the scale of single fishing events (the Lingcod fishery includes both trawl and fixed gear fleets, and we summed effort across the two for this purpose; in the past decade the landings from trawl have been 4-5x those from fixed gear[24], so the pattern in the data is driven primarily by the trawl fishery).

We used the BOEM coordinates of the OWF call area polygons to determine the proportion of each model grid cell was inside one of the OWF areas. The habitat domain for Lingcod consisted of 942 grid cells; of these cells, the Full Scenario had 1.7% of the habitat domain closed within OWF areas, and the Reduced Scenario had 1.1% closed within OWF areas. For Dover sole the habitat domain had 947 grid cells; within these, the Full Scenario had 1.9% of the habitat domain closed within OWF areas and the Reduced Scenario had 1.3% of the habitat domain closed within OWF areas (the values differ between the two species because their habitat

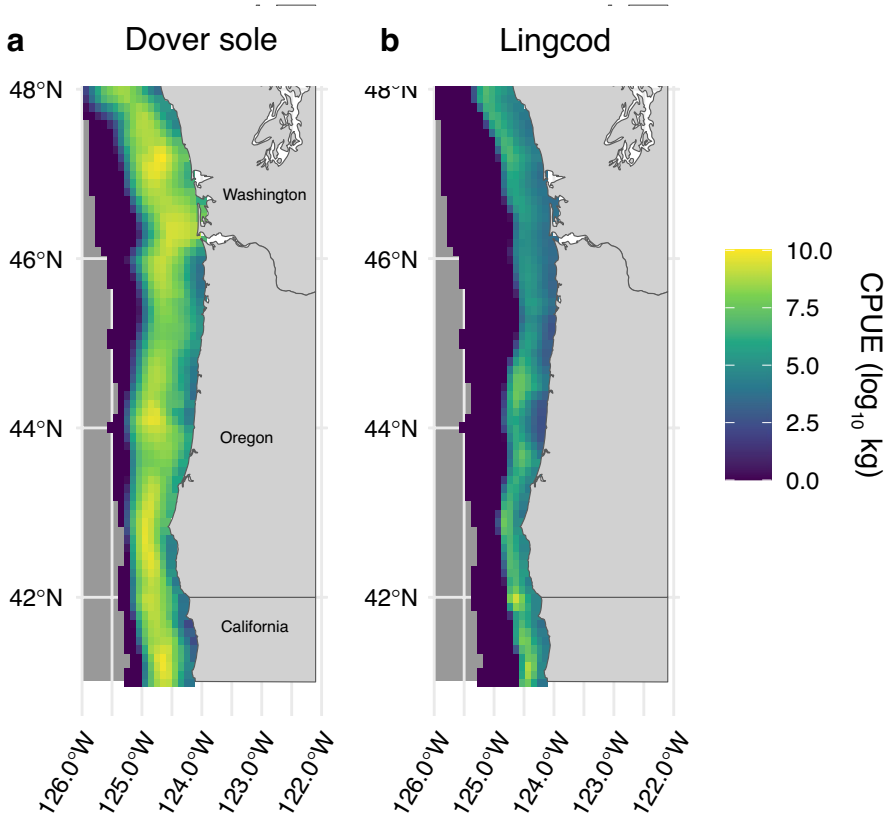

Fig. 2. Maps show the projected spatial distribution of relative abundance (measured as catch per unit effort [CPUE] in a scientific trawl survey) for (**a**) Dover sole and (**b**) Lingcod. Projections were derived from a species distribution model based on fishery independent scientific bottom trawl surveys and predicted present-day bottom temperature and oxygen conditions extracted from a coupled biogeochemical-ocean circulation model of the region[29,30]. Additional details are given in *Methods: Study species and model spatial domain*. Figure was created using package 'sf'[13] and 'rnaturalearth'[14] in R version 4.5.1[15] with GIS layers provided by Liu et. al.[29,30].

domains differed slightly). The grid cells differ slightly by species because of the data we used to create the habitat domain and the two datasets occupied different amount of cells. Other spatial closures, such as federal Rockfish Conservation Areas and nearshore marine reserves, were not explicitly modeled, because they primarily protect habitat not targeted by the fisheries for our two model species.

Population model

We created discrete-time, spatially explicit, age-structured population models for each of the two species. The model had the same general structure used by Campbell et al.[11]; the full set of equations is given in the Supplemental Material and uses parameter values from the most recent stock assessments for each species[24,25]. Briefly, the model had density-dependent recruitment into the first age class (see next paragraph), and density-independent natural mortality and fishing mortality in older age classes. Fishing mortality had age-dependent selectivity, based on the fleet-specific length-based selectivity curves estimated in each stock assessment (Fig. S2). We translated length-based selectivity into the age-structured model using the mean age-at-length from the von Bertalanffy relationship. Length-at-age and biomass-at-age were assumed to follow the von Bertalanffy relationship and an allometric length-biomass relationship, respectively. Reproductive output was proportional to biomass-at-age, for ages greater than the age at 50% maturity, which was also translated from the length-specific maturity in the stock assessments using the von Bertalanffy relationship.

We assumed that larval dispersal followed a larval pool model, consistent with the assumptions made in the respective stock assessments that both populations are well-mixed at the scale of the Oregon shelf, so larvae spawned in each habitat grid cell in each time step had an equal probability of settling in any cell[24,25]. Recruitment was density-dependent within each cell following the Beverton-Holt function[32]. We set the slope parameter of the Beverton-Holt function so that it had the same steepness as the spawner-recruit curve used in the stock assessments[24,25] following the procedure used by White[33]. The second parameter of the Beverton-Holt function sets the maximum possible density of recruits, which we assumed was proportional to the habitat quality in each cell, which we explain in the next paragraph.

In our analysis we were interested primarily in calculating the expected relative magnitude of changes in fish biomass and fishery yield because of OWF implementation, given the proposed scale of the OWF closures. We were not attempting to create forecast estimates of absolute fish biomass or tons of yield, in part because such estimates would require a level of precision in specifying the initial conditions of the system (i.e., the spatial distribution of biomass at the time of OWF implementation, and the spatial distribution of age/size structure of the stock) that are simply not available. Therefore we focused on the results of a deterministic analysis, starting at a steady-state distribution of the stock. Those deterministic results should be understood to be averaging across a range of uncertainties in parameter values, as well as the inherent stochastic variability of marine fisheries. To reflect those uncertainties we also performed simulations that had realistic levels of stochastic variability in recruitment, so that the magnitude of potential OWF effects could be compared to the magnitude of interannual variability in the stocks.

To conduct our spatially explicit analysis we required a way to represent the relative habitat quality of each grid cell; i.e., how much fish biomass can it support, relative to other grid cells. To calculate a measure of habitat quality for each grid cell, we took the time-averaged CPUE from each species distribution model for each grid cell and scaled it by the maximum time-averaged CPUE in the model domain, thus creating a species-specific spatial index of habitat quality ranging from 0-1. We set the maximum density term in the Beverton-Holt relationship (see Supplemental Information for equations) to be equal to that scaled CPUE value in each grid cell.

We assumed that adult fish in the model moved within approximately circular home ranges centered somewhere within the grid cell in which each fish settled from the larval stage. Because the habitat grid cells in the model are large (100 km^2) relative to the home range areas reported for our study species in the literature (Table 1), we were only concerned with the home range movements of individuals with home ranges centered at the edge of each cell, because those fish will experience the harvest rate of the other cell for part of their time. Therefore, for a given cell i, we calculated the proportion of the grid cell area that would be covered by home ranges centered on the border with a neighboring cell j (i.e., p = [1/2 of the area of a circle with the specified homerange radius]/100 km^2). We then assumed that, given the harvest rates F_i and F_j in the two cells, the population in cell i would experience an overall harvest rate of $(1 - p)F_i + pF_j$. We performed this calculation for each of the cells bordering each given cell, so that each cell experienced an effective harvest rate that depended partially on the value of F_j in each neighboring cell.

We initialized the population models by applying a spatial distribution of fishing effort that corresponded to historic landings data from 2012-2022 for each species. We assumed that the overall spatially averaged fishing mortality rate for the stock should be the value of F estimated from the stock assessment, so the fishing mortality rate F_i in each spatial cell i was made proportional to the proportion of total trips in each cell, while constraining the average F over the entire domain to remain the same as the stock assessment.

We considered two different possible responses of the fishing fleets to the OWF closures. Both species are managed with a 'total allowable catch' quota system, but landings for both have been well below the allowable limits in recent years. Therefore we considered two options that bracket the range of possible fleet responses. First, we assumed that total fishing effort would remain constant after OWF implementation, but would redistribute over space. This would reflect multispecies fisheries that are responding to multiple objectives other than maintaining landings of our particular focal species. Alternatively, we assumed that the fleet would increase fishing effort to maintain constant fishery yields after the closure. In both cases we modeled the spatial distribution of fishing efforts such that when the OWF was implemented in the model we set $F_i = 0$ for the model cells that were fully inside the OWF areas; model cells that were only partially inside an OWF were assigned a value of F_i proportional to the fraction of the cell not within the OWF. In the constant-effort scenario, the sum of F_i in the OWF cells was redistributed into the non-OWF areas, proportional to the relative number of trips

made there in the pre-OWF years. In the constant-yield scenario, we estimated the new spatial mean value of F that would be required to maintain pre-OWF yields, and then distributed that level of fishing effort over space, also proportional to the relative number of trips in pre-OWF years. In both cases we assumed that spatial fishing preferences outside of the OWF areas stayed the same.

For model simulations we started the population spatial and age structure at an arbitrary initial condition then iterated the model for 200 annual time steps under the logbook-based distribution of fishing effort; given our model formulation this approach ensured that we obtained a stable initial population distribution. We then implemented the OWF and iterated the model for 20 additional time steps to simulate the short-term effects of the closure areas. The response variables we recorded were the spatial distribution (over grid cells) of harvest rates (F), biomass (kg), and fishery yield (kg). We also calculated the 'spillover' contribution of the OWF areas to the fished portion of the population, in two ways. We calculated larval spillover as the proportion of larvae settling in the fished habitat that were spawned in the OWF areas, and we calculated adult spillover as the biomass of fish that had settled in OWF areas but were harvested in the fishery because home range movements took them into fished grid cells. The model was implemented in R version 4.5.1 (2025-06-13)[15]. Code can be found at https://doi.org/10.5281/zenodo.22015803

Results

Overall, our simulation of the fishery effects of establishing offshore wind farms off the Oregon coast matched our expectation: the model predicted that when total fishing effort was held constant, there would be increases in fish biomass inside the OWF and decreases in fishery yield commensurate with the proportional loss of fishing grounds, while the change in fishery yield would be minimal if effort was increased to obtain constant yield following the OWF closure; in either case there would be transient fluctuations in both biomass and yield responses over the 20-year simulation.

When harvesting effort was kept constant but redistributed following the simulated OWF closure, the harvest rate in other spatial cells increased slightly, with a maximum single-cell increase of only 2.5%. Fishery yield increased by approximately the same proportion in those cells, while yield was reduced to zero in the closed area. Given the small magnitude of these projected changes in individual cells, the overall patterns are difficult to discern visually in a map projection, even when zoomed in near the OWF areas (e.g., Fig. 3 shows results for Lingcod in the Full Scenario, which exhibited the most dramatic changes; results for Dover sole results and the other scenarios are in the Supplemental Information, Fig. S3, Fig. S4, Fig. S5, Fig. S6, Fig. S7 Fig. S8, Fig. S9), so we focus instead on non-spatial coast-wide average values.

Following OWF implementation with fishing effort held constant, overall fishery yield for both species decreased immediately in the Full Scenario, with greater predicted decreases for Lingcod than Dover sole (Fig. 4a, c). However, the magnitude of those changes was small; in the Full Scenario, yield of Lingcod was reduced by < 2% in the 20 years following the OWF implementation (Fig. 5a), and <1% for Dover sole (Fig. 5c). In the Reduced Scenario there was essentially no loss of yield, and an average change in yield barely greater than zero over the 20 year period (Fig. 4b,d, Fig. 5b, d). Those reductions were all less than the proportions of yield that were landed inside the OWF areas in the model prior to the simulated implementation of the OWF closures (Fig. 5). We note that these values (based on running our model to a steady state pre-OWF) are slightly higher than the proportion of landings actually reported in the logbook data for those areas (in the Full Scenario 1.1% (Lingcod) and 3.7% (Dover sole) of the yield was in OWF areas; in the Reduced Scenario those values were 1.0% (Lingcod) and 0.5% (Dover sole)). This discrepancy arises because the model used the same proportional harvesting effort as reported in the logbook, but the logbook landings also reflect historical variation in recruitment and stock abundance that we did not attempt to recreate in the pre-OWF period.

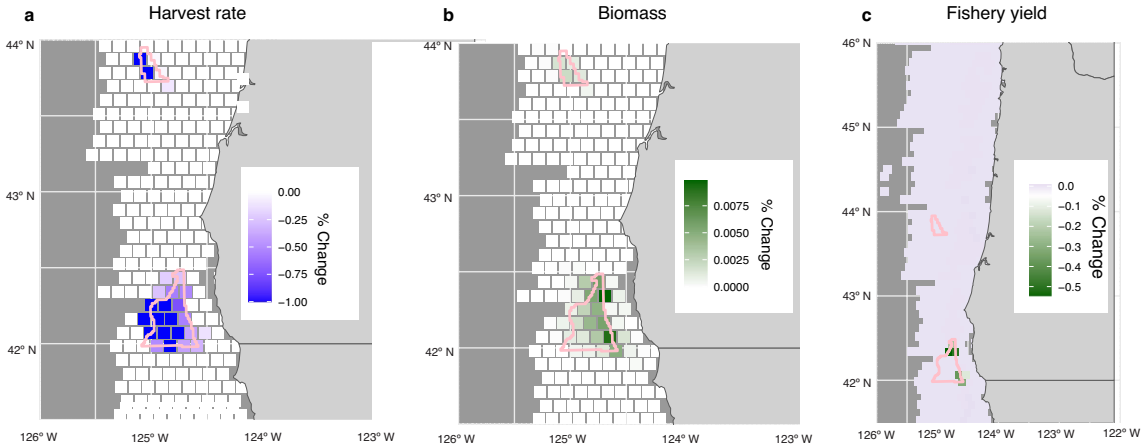

Fig. 3. Spatial distribution of proportional changes in (**a**) harvest effort, (**b**) fishery yield, and (**c**) biomass for Lingcod in the Full Scenario. The shading of each cell reflects the proportional change in each respective variable, calculated as the percentage difference between the pre-OWF conditions ($t = 0$) and the end of the twenty-year simulation ($t = 20$). Pink outlines denote the OWF areas.

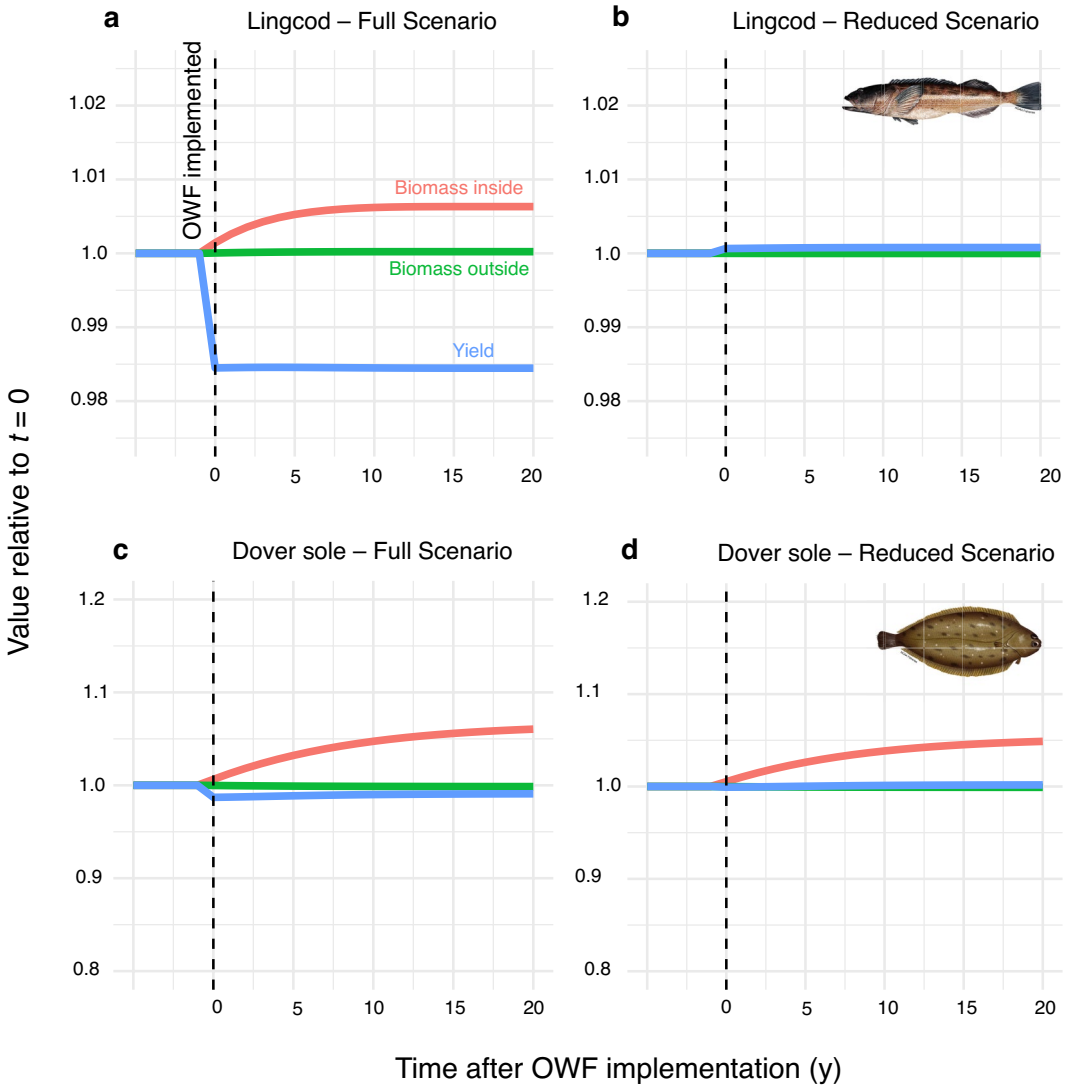

Fig. 4. Temporal trajectories of biomass (red) and yield (blue) for Lingcod and Dover sole in the two OWF scenarios, when fishing effort is held constant and redistributed after OWF implementation. Both variables are the average value over the entire model domain, expressed relative to the value before OWF implementation. The dashed line indicates $t = 0$, when the OWF was implemented in the simulations.

Excluding harvest from the OWF areas for both scenarios and species led to an increase in biomass both within the OWF areas and adjacent model cells, and thus an overall increase in stock biomass (Figs. 3, 4). The magnitude of biomass increase was greater in the Full Scenario than in the Reduced Scenario because of the larger closed area (Fig. 5 a,c). There was also a greater proportional increase in Lingcod biomass than Dover sole biomass within the OWF areas in the Full Scenario, but a slightly greater biomass for Dover sole than Lingcod in the Reduced Scenario (Fig. 5). The increases in biomass were gradual (as expected in deterministic simulations) and reached an asymptotic maximum approximately 10-20 model years after OWF implementation (Fig. 4).

As biomass began to build up within the OWF areas, the reproductive output of those cells also increased and increased larval supply to the entire population, including cells outside of the OWF area (Fig. 5). Larval spillover was greater than adult spillover for both species, although the relative magnitude varied somewhat among scenarios, consistent with the relative patterns of increase in biomass within the OWF for the two different species in the two scenarios (Fig. 4). While Dover sole were modeled with a larger adult home range than lingcod, the former species was predicted to have slightly less adult spillover than Lingcod in the full scenario; this likely reflects the slower increase in Dover sole biomass inside the OWF; i.e., there was simply less biomass accumulating to spillover (Fig. 5).

When we explored an alternative fishing fleet behavior, in which fishing effort was increased following the OWF closure in order to maintain the same level of pre-OWF yield, there was (as expected) essentially no change to fishing yield following the OWF closure (Fig. S10). Under those conditions there was an initial <1% drop in yields in the Full Scenario for Lingcod, but no change for the other scenario or for Dover sole; in all scenarios,

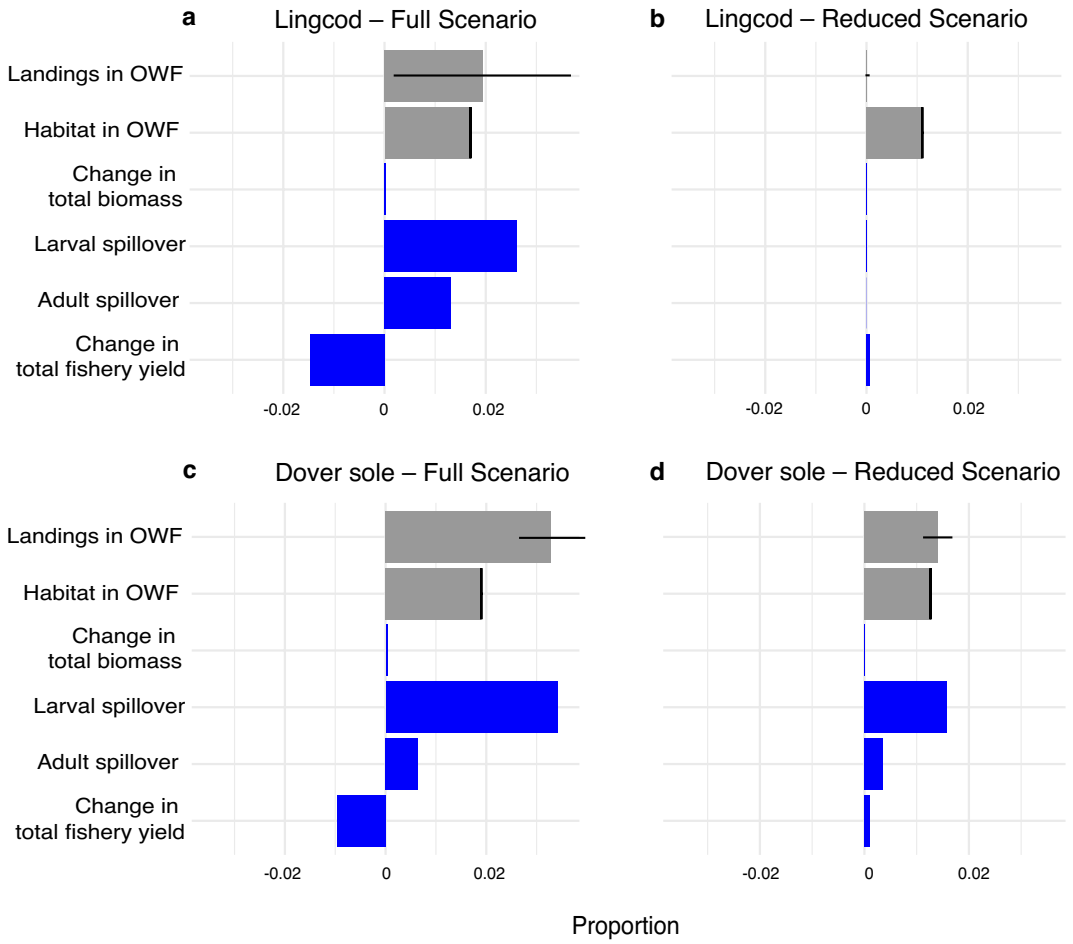

Fig. 5. Proportion of landings and habitat in OWF area before OWF implementation as well as post OWF implementation change in total biomass, larval spillover, adult spillover, and change in total fishery yield, in the scenario in which fishing effort remains constant after OWF implementation. Gray bars denote static quantities before OWF implementation, while blue denote outcomes of the dynamic population model, averaged over 20 years post-OWF implementation. See Methods: Population model for details on the calculation of spillover values. The error bar for landings indicates ± 1 standard deviation.

biomass increased slightly inside the OWF areas and was essentially unchanged outside of the OWF (Fig. S10, Fig. S11).

Finally, in simulations in which we added realistic levels of stochastic variation in larval recruitment, representing process uncertainty in the system, the distributions of trajectories of fish biomass and fishery yield spanned zero for both species in both the Full and Reduced Scenarios (we only considered the constant-effort fleet model in these simulations; Fig. S12, Fig. S13). That level of spread in the stochastic model outcomes suggests that the magnitude of the deterministic trends (e.g., Fig. 4) is small relative to the amount of interannual variability in population dynamics of these two species.

Discussion

In this study, we developed a spatially explicit simulation of groundfish population dynamics off the Oregon coast to estimate the extent and timing of potential impacts of closures due to offshore wind farms (OWF) on two groundfish fisheries. While these two OWF areas are no longer being considered for leasing as of 2025, these results could inform the planning of other potential offshore energy development projects in the future. Using historical spatial patterns of fishing effort for the two species as a guide for where fishing effort removed from the closed areas might be redistributed along the continental shelf, we found that closing areas of the ocean for OWFs could be expected to produce a modest increase in biomass of those groundfish species within the closure area, while causing minimal to no reductions in fishery yield, depending on the response of the fishing fleet. From a conservation perspective, these findings are consistent with the notion of placing OWFs in higher quality habitat[11]. However, we did not observe the expected declines in fishery yield[11] associated with designating OWFs in higher quality habitat.

Our population-dynamic assessment of the effects of OWF implementation has some advantages over the typical static assessment approach. The latter would assume that fishery landings previously taken within the

OWF area are completely lost, but our analysis included the dynamic response of reallocated or increased fishing effort and larval and adult spillover. The spillover effects in the two fisheries we modeled were relatively minor (on the same order of magnitude as the proportional area closed), and neither the closure nor the spillover had an effect of more than a 1-2% change when measured at the statewide scale, even in the most conservative (constant effort), deterministic case. However, we recognize that impacts at the statewide scale will not necessarily be distributed homogeneously on individual fishers, especially those that habitually use the fishing grounds directly affected by OWF implementation and encourage future work to explore the potential for such distributional impacts. When we included stochastic variability in our simulations to reflect the inherent process variability in groundfish population dynamics, it was not possible to detect any effect of the OWF closures from natural population fluctuations. Together these results a) speak to the importance of including population dynamics in assessments such as this, rather than assuming the fishery will not respond dynamically, and b) suggest that closures of a small percentage of the fishing grounds for sustainably managed, underexploited fisheries would not be expected to have a large effect on landings.

We considered Lingcod and Dover sole, two federally managed groundfishes, in our analysis because of the availability of logbook data and species distribution models to conduct our analysis. However, these species are often caught in a multispecies fishery using relatively non-selective trawl gear. Because there are nearly 100 species of federally-managed groundfish[34], the cumulative effects on the groundfish fishery of changes in yield due to OWF implementation could be greater than the relatively minor impacts on individual species observed in this study. Furthermore, many other federally-managed and state-managed fisheries operational on the Oregon shelf could be impacted by OWFs due to logistical and maneuverability constraints in and amongst wind turbines, especially in the trawl fisheries. Another limitation is using fishery dependent data may contain inconsistencies or missing entries in logbook records. Additionally for the purposes of this study, we combined trawl and fixed gear datasets for Lingcod. Our approach illustrates the type of data that would be needed to conduct a similar analysis for other stocks.

Our model study offers valuable insights into the potential impacts of OWFs on groundfish populations off the coast of Oregon, although it is crucial to recognize the limitations and assumptions inherent in our approach. Our model operated within a single-species framework, overlooking the influence of species interactions on population dynamics in the real-world context. Lingcod are predators of other groundfishes, so increases or decreases in their populations can have effects on other fishery yields[35–37]. There are also details of the species' distributions with respect to bathymetry that may not be fully captured with the species distribution model layers we used to represent the habitat domain. For example, outside of the reproductive season, lingcod tend to segregate by sex, with females occupying deeper waters and males remaining shallower[35], a detail not captured by our single-sex model. However, in general Lingcod – and Lingcod landings – are distributed closer to shore than Dover sole, which may explain why Lingcod were more responsive to the simulated fisheries closures. In Dover sole, there are ontogenetic bathymetry associations, in which fish move to deeper water as they grow larger, though that habitat shift is less pronounced on the Oregon shelf than in California, and most fish remain inshore of 600 m[38]. In both cases, it is possible that our model could have underestimated the potential loss of fishery access to larger fish in the deep-water OWF areas; yet it remains the case that the potential OWFs represent small fractions of the overall habitat available to the two stocks we modeled, so the proportional change in results associated with sex- or size-related bathymetric associations is low.

Additionally, our model overlooked potential behavioral responses to OWFs, such as avoidance or attraction, which could significantly impact the distribution of groundfish populations and their interactions with OWFs[39,40]. If our model species had a behavioral attraction to OWF infrastructure, our model analysis would have underestimated both the biomass accumulating within the OWF and the loss of fishery yields, though the reproductive output from within the OWF would also be proportionally greater. Alternatively, any negative impacts from habitat disturbances during OWF installation would dampen the biomass increase inside the closure[41,42], although the floating OWF infrastructure proposed for use in Oregon is substantially different from earlier technologies so the potential effects are relatively unknown. Additionally, behavioral interactions with OWF infrastructure could also be different for non-groundfish or highly migratory species (tunas, sharks, etc.). On the other hand, there are some other types of fish movements that we did not attempt to resolve, such as possible seasonal spawning migrations (e.g.[42]); the effect of that type of movement would be for the populations to become more well-mixed in general, reducing the protective effect of the OWF. It is also likely that the species we modeled will be shifting their geographical ranges due to rising seawater temperatures during the time period in which we modeled their response to the OWF. In the California Current, the general trend is for those shifts to be into deeper water (rather than poleward[29,43]), which could also bring more shallow shelf species into the deeper OWF areas.

In our analysis, we had to make assumptions about how the fishing fleet would respond to OWF closures, and did not factor in potential external influences such as changing market conditions. We attempted to bracket the range of possible effects by modeling both a constant-effort scenario (in which the fleet would maintain the same level of costs in fuel and crew, etc.) and a constant-catch scenario (in which the fleet would increase costs to maintain the same level of yield relative to allowable catch limits), although ultimately the magnitude of projected changes under either assumption was small. We assumed the fleet would respond to closures by increasing effort in the most historically popular locations, but actual fleet behavioral responses to OWF implementation are uncertain, including the possibilities of shifting fishing grounds, switching gear or target species, or leaving the fishery[9,44]. Future work to explore the potential economic effects of OWF implementation that more explicitly treat fisher behavior and welfare issues could address these issues. Additionally, the loss of fishery-independent survey data formerly collected in the OWF areas could lead to increased uncertainty about stock status and require further decreases in effort[11]. We did not consider any possible scenarios in which management adjusts catch limits in response to the OWF. Despite these acknowledged limitations, our research serves as a valuable

starting point for understanding the potential impacts of OWFs. This is a first-order estimate of short-term dynamic changes for two relatively data-rich and sustainably-managed species, and similar analyses could be conducted for others. It would be particularly valuable to estimate OWF effects on species that are caught together in multispecies fisheries (like Lingcod and Dover sole), as well as those estimated to be overharvested or rebuilding.

Moving forward, continued research and monitoring efforts are essential to evaluate the long-term effects of OWFs on marine ecosystems and fishing activities, while addressing climate change mitigation technology. Incorporating stakeholder knowledge and perspectives, and reliance on adaptive management strategies will be crucial in mitigating potential conflicts and maximizing the benefits of offshore wind energy development, especially given uncertainty around how species will respond to new structures in the ocean (e.g. potential positive artificial reef effects)[45,46]. Here we have shown how population dynamics models could also be integrated into the planning process. As we navigate the transition towards a renewable energy future, quantitative methods are needed to estimate the likely costs and consequences of spatial planning choices to illustrate tradeoffs.

Data availability

All data and model code used in this paper is publicly available at https://github.com/campmarg/Model_2.git.

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Author contributions

MC, JFS, and JWW conceived the study. ORL provided data. MC performed the analyses. All authors interpreted the results. MC drafted the manuscript; ORL, JFS, and JWW provided editorial suggestions.

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Declarations

Competing interest

The authors declare no competing interests.

Additional information

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Correspondence and requests for materials should be addressed to M.C.

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