NRC Research Press

ARTICLE

A seascape-scale habitat model to support management of fishing impacts on benthic ecosystems

T. Scott Smeltz, Bradley P. Harris, John V. Olson, and Suresh A. Sethi

Abstract: Minimizing fishing impacts on seafloor ecosystems is a growing focus of ocean management; however, few quantitative tools exist to guide seascape-scale habitat management. To meet these needs, we developed a model to assess benthic ecosystem impacts from fishing gear contact. The habitat impacts model is cast in discrete time and can accommodate overlapping fisheries as well as incorporate gear-specific contact dynamics. We implemented the model in the North Pacific using fishing data from 2003 to 2017, estimating that habitat in 3.1% of the 1.2 million km² study area was disturbed at the end of the simulation period. A marked decline in habitat disturbance was evident since 2010, attributable to a single regulatory gear change that lifted trawl gear components off the seafloor. Running scenarios without these gear modifications showed these policies might have contributed to a 24% reduction in habitat disturbance since their implementation. Ultimately, model outputs provide direct estimates of the spatial and temporal trends of habitat effects from fishing — a key component of regulatory policies for many of the world's fisheries.

Résumé : Si la minimisation des impacts de la pêche sur les écosystèmes du fond marin est un domaine d'intérêt croissant en gestion des océans, il existe toutefois peu d'outils quantitatifs pour guider la gestion des habitats à l'échelle du paysage marin. Pour répondre à ces besoins, nous avons développé un modèle pour évaluer les impacts sur les écosystèmes benthiques des contacts d'engins de pêche. Le modèle d'impacts sur l'habitat est configuré en temps discret et peut intégrer des pêches se chevauchant, ainsi que la dynamique de contact d'engins précis. Nous avons appliqué le modèle dans l'océan Pacifique Nord en utilisant des données de pêche recueillies de 2003 à 2017, et estimé que les habitats dans 3,1 % de la région étudiée de 1,2 million de km² étaient perturbés à la fin de la période de simulation. Une baisse marquée de la perturbation des habitats est observée depuis 2010, qui est attribuable à un changement aux engins prescrit par la réglementation qui fait en sorte que des composantes des chaluts ne reposent plus sur le fond marin. La simulation de scénarios sans ces modifications des engins montre que ces politiques pourraient avoir contribué à une baisse de 24 % de la perturbation les habitats depuis leur entrée en vigueur. En conclusion, les résultats du modèle fournissent des estimations directes des tendances spatiales et temporelles des effets de la pêche sur l'habitat, une composante clé des politiques réglementaires pour bon nombre de pêches du monde. [Traduit par la Rédaction]

Introduction

Sustainable fisheries management has progressed from a focus on individual stocks to ecosystem-based approaches that recognize that habitat integrity is a critical component of healthy fisheries (Pikitch et al. 2004; McLeod and Leslie 2009). A number of options are available to mitigate habitat impacts associated with fishing while minimizing socioeconomic impacts on the fishery, including (*i*) spatial closures to protect sensitive habitat, (*ii*) policies that reduce effort by increasing capture efficiency, and (*iii*) changes to gear to reduce bottom contact. To support habitat management, analysts require tools to understand the spatiotemporal distribution of fishing effort and associated habitat disturbances to weigh the relative benefits and costs associated with different policies to maintain benthic ecosystem integrity. Here we develop a seascape-scale habitat impacts model designed with features useful for the fish habitat management process (henceforth, the "Fishing Effects" model). Building from previous efforts, the Fishing Effects model balances impact and recovery processes using discrete-time dynamics, which can incorporate seasonality common to most fisheries. Furthermore, the model accommodates habitat disturbance from multiple fisheries and gear types with overlapping fishing effort — a reality common to most marine ecosystems. Developed as spatially explicit and with options to represent gear changes through bottom contact adjustments, the Fishing Effects model is generic enough to support analyses of a wide range of habitat management policies.

Seafloor habitat features create structural complexity, providing protection from predators and forage opportunities for benthic organisms. Structure-forming habitat features vary in function and composition and include biogenic structures such as coral or shell structures, epi- or infauna, and geological structures such as cobble piles or sand waves. To varying degrees, all benthic habitat features are vulnerable to disturbance associated with bottom-tending fishing activity (Jennings and Kaiser 1998; Thrush et al. 1998; Hall 1999). Biogenic structures and epifaunal and infaunal organisms may be damaged or completely removed when con-

Received 14 June 2018. Accepted 18 December 2018.

T.S. Smeltz. New York Cooperative Fish and Wildlife Research Unit, Department of Natural Resources, Cornell University, Ithaca, New York, USA; Fisheries, Aquatic Science, and Technology (FAST) Laboratory at Alaska Pacific University, Anchorage, Alaska, USA.
B.P. Harris. Fisheries, Aquatic Science, and Technology (FAST) Laboratory at Alaska Pacific University, Anchorage, Alaska, USA.
J.V. Olson. National Marine Fisheries Service, NOAA, Anchorage, Alaska, USA.
S.A. Sethi. Fisheries, Aquatic Science, and Technology (FAST) Laboratory at Alaska Pacific University, Anchorage, Alaska, USA; US Geological Survey, New York Cooperative Fish and Wildlife Research Unit, Cornell University, Ithaca, New York, USA.
Corresponding author: T. Scott Smeltz (email: tsmeltz@alaskapacific.edu).

tacted by fishing gear (Kenchington et al. 2006). Physical features such as cobble and boulder piles may be scattered or dispersed (Freese et al. 1999), and microstructures formed by soft sediment such as sand waves may be resuspended and homogenized (Thrush et al. 1998; Watling and Norse 1998; O'Neill and Ivanović 2016). Mobile fishing gear such as trawls or dredges are often the primary focus of seafloor impact studies, but fixed gears that contact the seafloor also have potential to disturb habitat features and are often deployed in steep and rocky vulnerable habitats where use of mobile gears is not possible (e.g., Chuenpagdee et al. 2003; Heifetz et al. 2009; Stone et al. 2015; Doherty et al. 2018).

A number of meta-analyses have been conducted to understand the vulnerability of habitats to fishing activities (Collie et al. 2000; Kaiser et al. 2006; Grabowski et al. 2014; Hiddink et al. 2017). However, to support ocean habitat management, we need tools that scale localized impacts up to the seascape to estimate the cumulative impacts of fishing. A small number of frameworks have been developed for this purpose (Fujioka 2006; Dichmont et al. 2008; NEFMC 2011; Ellis et al. 2014; henceforth referred to as cumulative fishing impacts models). By explicitly combining habitat impacts and recovery to produce an estimate of cumulative fishing effects, these models are differentiated from a number of other seascape habitat models that focus on impacts or habitat vulnerabilities separately (e.g., Hiddink et al. 2007; Halpern et al. 2008; Rijnsdorp et al. 2016). Integrating impacts and recovery into a cohesive framework allows for improved temporal resolution of net habitat effects.

Differences among existing cumulative fishing impacts models can be categorized based upon three structural characteristics: (*i*) use of continuous versus discrete time, (*ii*) the shape of the habitat recovery curve, and (*iii*) complexity of the model translating fishing effort into habitat impacts. Most adopt a continuous-time dynamic, casting the impact and recovery as a differential equation (Fujioka 2006; Dichmont et al. 2008; Ellis et al. 2014), with only the Swept Area Seabed Impacts model developed by the US New England Fishery Management Council (NEFMC 2011) utilizing a discrete-time dynamic. The benefit of a discrete-time model is that it facilitates habitat disturbance assessments over series of relevant time steps, whereas continuous-time models are typically used to estimate equilibrium levels of habitat disturbance under assumed constant rates of fishing impact and recovery. Conceptually, recovery is similar among models in that impacted habitats recover naturally over time based upon some prescribed functional shape ranging from logistic (Dichmont et al. 2008; Ellis et al. 2014), asymptotic (Fujioka 2006), or linear (NEFMC 2011).

The existing cumulative fishing impacts models differ more strongly in the way habitat disturbance is modeled as a function of fishing effort. A key process in relating fishing effort to habitat impacts is the treatment of overlapping fishing events in space, where repeated fishing events over already disturbed habitat may have different impacts than fishing events in undisturbed habitats. The Fujioka model (Fujioka 2006) translates summed fishing effort (in units of proportional area swept) to habitat impacts using a "habitat coefficient" scalar. The models constructed by Dichmont et al. (2008) and Ellis et al. (2014) similarly scale effort to habitat disturbance, but their models also include a process to account for overlapping fishing events, providing flexibility to model habitat disturbance under spatially disparate versus aggregated fishing. With a focus on testing gear modification scenarios, the Swept Area Seabed Impact model (NEFMC 2011) was developed with the most explicit approach to translating fishing effort to habitat impacts by including modules relating gear configuration to bottom contact as well as gear-specific habitat susceptibility parameters.

As fisheries management systems mature, many now explicitly include requirements to minimize impacts to fish habitats to the extent practicable (Nimick and Harris 2016). Spurred by needs identified by the US North Pacific Fishery Management Council (NPFMC) for additional model functionality to assess the current state of benthic habitat disturbance at seascape scales and to facilitate analysis of the trade-offs between different habitat impact reduction options, we developed the Fishing Effects model by combining features from a number of existing cumulative fishing impacts models. Specifically, we use a discrete-time framework that incorporates time-varying fishing effort among multiple fisheries. The model is implemented on monthly time scales, reducing problems of estimating impacts using annualized measures of fishing effort (van Denderen et al. 2015), and includes habitat susceptibility and recovery dynamics that are parameterized to accept information from empirical studies on fishing impacts. Finally, the Fishing Effects model includes a gear contact adjustment parameter useful for evaluating seascape-scale effects of fishing gear modifications that reduce bottom contact. Below, we first provide a mathematical description of the Fishing Effect model, defining the habitat state dynamics, and the impact and recovery processes. Subsequently, we describe a workflow to prepare data inputs necessary to implement the model. Finally, we outline three applications of the modeling framework that together exemplify the use of seascape-scale habitat impacts models to inform management using the North Pacific as a case study.

Table 1. Model parameters and indices.

	Description
Model parameters	
H	Undisturbed habitat
h	Disturbed habitat
$\tilde{I}$	Proportional impacts
$\tilde{\rho}$	Proportional recovery
$I_{i,t,s,j(g)}$	Impact from a fishing event
$f_{i,t,s,j(g)}$	Ground contact by a fishing event
$q_{s(g)}$	Susceptibility
$A_{i,t,s,j(g)}$	Nominal swept area by a fishing event
$c_{j(g)}$	Contact adjustment
$\bar{\tau}_s$	Mean time to recover
$\phi_{i,s}$	Proportion of habitat type s in cell i
Indexing variables	
i	Grid cell, for n total cells
t	Time step
s	Habitat types, for k total habitats
j	Fishing event, for m total events
g	Gear type

Methods

Model description

Habitat states

The Fishing Effects model assumes that a region may be composed of both disturbed habitat, h, and undisturbed habitat, H (see Table 1 for a list of all model parameters). Casting h and H as proportional areas within a region, the total amount of undisturbed or disturbed habitat sums to one: $H + h = 1$. The Fishing Effects model considers transitions between habitat states in discrete time steps, t. Let $\tilde{I}_t$ represent the proportion of undisturbed habitat that transitions to disturbed habitat by fishing impacts from one time step to the next and $\tilde{\rho}_t$ as the proportion of disturbed habitat that recovers to an undisturbed state over the same time step, leading to the discrete-time habitat state equation:

$$H_{t+1} = H_t(1 - \tilde{I}_t) + h_t\tilde{\rho}_t \quad (1)$$

Thus far, eq. 1 implies a single generic model spatial domain. In practice, the model is implemented on a spatially explicit grid indexed by both time, t, and cell, i, such that eq. 1 governs the dynamics within each grid cell.

A model grid cell can contain multiple habitat types, for example as defined by substrate class (mud, sand, bedrock, etc.), each of which may contain an assemblage of habitat features such as biogenic structures like sponge communities or physical structures like boulder piles or sand waves. As outlined below, the Fishing Effects model accounts for impacts and recovery by habitat type, representing disturbance to the habitat features associated with each habitat type. Subsequently, for the purposes of calculating the proportion of disturbed habitat within a given cell at a point in time, $H_{i,t}$ is calculated as a weighted mean over k habitat types (indexed by s) based on the proportion of each habitat type in the cell, $\phi_{i,s}$:

$$(2) \qquad H_{i,t} = \sum_{s=1}^{k} H_{i,t,s}\phi_{i,s}$$

Although habitat types may be spatially explicit regions within a grid cell, in practice such fine-resolution habitat information is usually not available. Thus, it is assumed that each habitat type is distributed uniformly throughout a grid cell and that the relative proportions of habitat types within cells remains fixed across time regardless of whether these habitats are in a disturbed or undisturbed state.

Impacts

The impacts process translates fishing activity into habitat disturbance outcomes. Impacts, I, represent the area of each habitat type in a grid cell that converts to the disturbed state in a time step by a single interaction (e.g., trawl pass, longline set) with fishing gear. Impacts are calculated as the proportionate area of a habitat type in a cell contacted by fishing gear, f, times a habitat susceptibility parameter, q, conceptually similar to the "catchability" of a habitat feature from fishing gear contact: $I = fq$. To provide downstream functionality to explore changes to fishing gear and to track the combined impacts of different fishing gears that overlap during the course of a fishing season, we tracked impacts by fishing events, j (e.g., a single trawl tow, a longline set, etc.), and habitat types, in addition to time and space. Both the amount of seafloor habitat contacted by fishing gear and the susceptibility of habitats to fishing gear contact vary across gear types. We use the notation "(g)" to indicate that a parameter value depends on the fishing gear type. Combining all the indexing variables for subsequent model runs, we have the following:

$$(3) \qquad I_{i,t,s,j(g)} = f_{i,t,s,j(g)}q_{s(g)}$$

The seafloor area contacted by fishing gear will typically be less than the nominal "area swept" because not all gear elements contact the seafloor during fishing (Winger et al. 2010). For example, trawl nets can be rigged with lifting elements to raise towing gear up off the seafloor (Rose et al. 2010; see the Results section below for an example of habitat impact outcomes associated with trawl sweep modifications). Thus, bottom contact, f, is calculated as the product of nominal area swept, A, and gear-specific contact adjustment, c:

$$(4) \qquad f_{i,t,s,j(g)} = A_{i,t,s,j(g)}c_{j(g)}$$

where A is measured as a proportion relative to the area of a grid cell, and c is a proportion relative to the nominal swept area. In addition to correcting gear bottom contact, the inclusion of a contact adjustment parameter provides functionality to model gear modifications that lift gear elements off the seafloor.

In the course of fishing, a grid cell may experience multiple fishing events with overlapping swept areas, particularly in high effort hotspots on fishing grounds. Thus, fishing impact models need to calculate habitat disturbance under potentially overlapping fishing impacts. To generate combined habitat impacts in the Fishing Effects model, we first sum impacts across m fishing events that occur within a cell in a single time step (the "•" subscript indicates summation across an index):

$$(5) \qquad I_{i,t,s,\bullet} = \sum_{j=1}^{m} I_{i,t,s,j}$$

Subsequently, sum of impacts, I, are translated into proportional impacts, which are constrained between zero and one, $\tilde{I}$, as utilized in eq. 1:

$$(6) \qquad \tilde{I}_{i,t,s} = 1 - e^{-I_{i,t,s,\bullet}}$$

While not obvious, the relationship in eq. 6 implies that fishing events are randomly distributed within a grid cell (Gerritsen et al. 2013; see online Supplemental Material[1] for derivation of this assumption). If fishing activity is more spatially aggregated than expected under random fishing, eq. 6 would produce an overestimation of proportional impacts (i.e., more overlap than expected). Conversely, uniformly distributed fishing activity would distribute contact more widely throughout the cell than expected under random fishing and result in an underestimation of impacts (Ellis et al. 2014). Note that the scale of the grid cell will affect this assumption. At a seascape scale, fishing activity is clearly aggregated, but at very small scales (e.g., an area smaller than the swept area of a single tow) fishing becomes uniformly distributed; the assumption of random fishing effort is likely to be met at some spatial grain between these two extremes.

Recovery

Most biogenic and physical benthic habitat features exhibit the ability to recover after disturbance (Grabowski et al. 2014; Hiddink et al. 2017). Habitat recovery, $\tilde{\rho}$, in eq. 1 is the proportion of disturbed habitat that transitions to undisturbed habitat from one time step to the next. The recovery process could take various functional forms, including density dependence, or include temporal or spatial dynamics (e.g., Pacific decadal oscillation regime shifts; Mantua and Hare 2002). Here, we implement a constant habitat recovery rate function whereby recovery dynamics follow an exponential asymptotic trajectory parameterized by a mean time to recovery, $\overline{\tau}$, specific to different habitat types (indexed by s):

$$(7) \qquad \tilde{\rho}_{i,t,s} = 1 - e^{(-1/\overline{\tau}_s)}$$

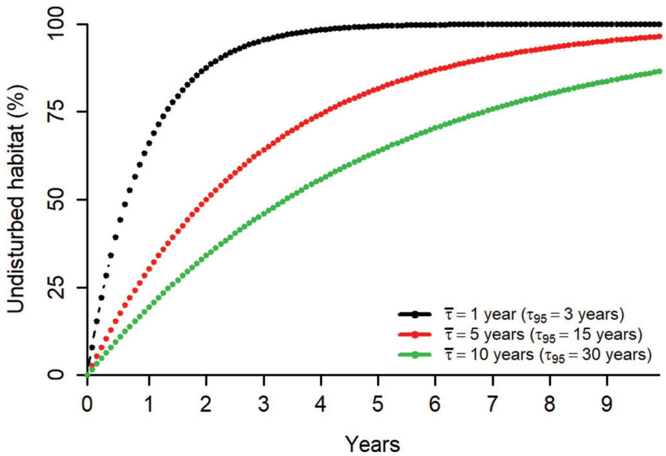
Conceptually, this model reflects the cumulative probability that an impacted habitat will transition to an undisturbed state as a function of a mean time to recovery (Fig. 1). To facilitate interpretation of recovery times, we present recovery parameters both in terms of the mean recovery time and the time at which cumulative probability of recovery is 95%, τ^*, where $\tau^* = -\overline{\tau}\ln(1 - 0.95)$.

Model implementation

Requirements to implement the Fishing Effects model include (i) a defined spatial domain with an appropriately sized grid over-

[1]Supplementary data are available with the article through the journal Web site at http://nrcresearchpress.com/doi/suppl/10.1139/cjfas-2018-0243.

Fig. 1. Examples of discrete monthly recovery trajectories. The points show percentage of undisturbed habitat in monthly time steps under $\bar{\tau}$ = 1-, 5-, and 10-year recovery scenarios, corresponding to τ_{95} = 3, 15, and 30 years, respectively. Undisturbed habitat is initially set at 0% (H_0 = 0; h_0 = 1) with no subsequent impacts. [Colour online.]



lay, (*ii*) the spatial distribution of habitat types, (*iii*) fishing event locations, most likely derived from electronic monitoring such as vessel monitoring system (VMS) data, (*iv*) nominal gear width and gear contact adjustments for each fishing event, and (*v*) habitat susceptibility and recovery parameters. Here, we describe each of these inputs using an implementation of the model in the North Pacific as an example.

Spatial domain and habitat distribution

The case study area is the North Pacific continental shelf within the United States Exclusive Economic Zone at depths less than 1000 m, resulting in a model domain of 1.2 million km^2. We split the domain into three subregions that align with management units of the NPFMC: Aleutian Islands, Bering Sea, and Gulf of Alaska. All three subregions contain multiple fisheries and gears with different bottom contact intensities (e.g., bottom trawl versus longline). The Aleutian Islands are characterized by steep, rocky, and overall more variable terrain than the other subregions. The Bering Sea is characterized by a large and flat continental shelf of homogeneous soft habitat, extending over 500 km off the coast. The Gulf of Alaska also sits on a large continental shelf but is incised by deep troughs and contains many rocky outcrops.

We implemented a 5 km × 5 km model grid, reflecting the spatial resolution of available fishing and habitat information within the North Pacific (NOAA 2017*b*). Detailed information on the spatial distribution of specific benthic habitat features was generally lacking; however, observations from sediment surveys in the North Pacific were more widely available. Thus, we used sediment-based habitat types (mud, sand, granule–pebble, cobble, and boulder) and developed a GIS workflow to map the sediment observations across the domain. Sediment observations (232 517 total points) were compiled from disparate surveys (Smith and McConnaughey 1999; NOS 2013; Zimmermann et al. 2013; Zimmermann and Prescott 2015) in a GIS and parsed using a text mining algorithm (Feinerer and Hornik 2017) to map 8861 different sediment labels onto the five primary sediment types. Subsequently, indicator kriging interpolation (Geospatial Analyst, ArcGIS version 10.4.1) was used to create a presence–absence surface for each sediment on a 2.5 km grid, with the presence of each sediment type indicated by a 1 (present) or 0 (absent). This resulted in four sediment subcells nested within each 5 km model grid cell. Subsequently, we calculated the proportions of the sediment type within a 5 km grid cell, $\phi_{i,s}$, by standardizing the sum of presences for a particular sediment type among subcells against the sum of presences for all sediment types in the cell:

$$(8) \qquad \phi_{i,s} = \frac{\sum_{z=1}^{4} \pi_{i,s,z}}{\sum_{z=1}^{4}\sum_{s=1}^{5} \pi_{i,s,z}}$$

where π is an indicator variable equal to 1 if a sediment type is present, or 0 if not, for subcell z.

Fishing gear and fishing effort

Spatially explicit fishing effort was obtained from the US National Marine Fisheries Service, consisting of VMS tracks of federally managed commercial vessels. Implementation of VMS has been ongoing since 2003 in the North Pacific, with full coverage of nearly all vessels using bottom-tendered fishing gear. Omissions from this data set are primarily small longline and pot vessels, which would have no discernable impact on regional model outputs. Gear-specific nominal widths and contact adjustment parameters were taken from a database of gear dimensions compiled by the NPFMC (2017; see Table S6[1], Supplemental Material). Unique nominal gear widths ranged from 50.0 to 259.0 m for 59 variants of trawls, 2.0 m for longlines, and 5.6 m for pot sets. For longlines and pots, the nominal widths were calculated based on the area potentially impacted from setting or retrieval of the gear, up to three times the footprint of the gear itself. The polylines of fishing activity were buffered by one-half the nominal widths and intersected with the 5 km grid (ArcGIS version 10.4.1). Contact adjustment was reported in the database as an estimated range (minimum – maximum) for each gear type. We ran the model using the mean of these values for each gear type (see Fig. S3[1], Supplemental Material for sensitivity of model outputs to these ranges). To reflect the implementation of required raised sweep gear modifications on flatfish-targeting vessels in 2011 for the Bering Sea fisheries (NOAA 2010) and in 2014 for Gulf of Alaska fisheries (NOAA 2014), we set contact adjustment parameters to unity (complete contact) prior to their implementation and ranged from 0.20 to 0.75 afterwards (i.e., 25%–80% reduction in seafloor contact; NPFMC 2017). The gear modifications generally consisted of 25 cm disks placed on the trawl sweeps. Experimental studies showed slight reductions on catch rates of some flatfish species with these modifications (Rose et al. 2010*b*), but caused substantially less damage to sessile invertebrate compared with conventional gears (Rose et al. 2010*a*). To test for random effort within a grid cell, we used a subset of the VMS data to evaluate this assumption, finding that while most grid cells the North Pacific do demonstrate relatively random effort, about 10% of the grid cell had substantially aggregated fishing effort. This may result in a small overestimate (~0.13 percentage points) in habitat disturbance from model outputs (Fig. S1[1], Supplemental Material).

Susceptibility and recovery

Habitat susceptibility and recovery parameters characterize benthic ecosystem vulnerability to fishing impacts, translating fishing events into habitat disturbance reflecting a balance of impact and recovery processes. Habitat disturbance may be measured in a variety of ways (e.g., damaged individuals, loss of biomass, decrease in species richness, etc.); thus, the parameterization of recovery and susceptibility will define how to interpret disturbance. For this implementation, we adapted susceptibility and recovery values from the Grabowski et al. (2014) review of habitat feature vulnerabilities to fishing gear, which framed recovery and susceptibility as habitat functionality to provide shelter for fish species. We used the Grabowski et al. (2014) study specifically because they accounted for contact adjustment in their estimation of susceptibility. These values were reported as ranges for structural seabed

Table 2. Susceptibility and recovery parameter means from aggregation of habitat features associated with each habitat type.

	Susceptibility			Recovery (years)	
Habitat type	Trawls	Longlines	Traps	Mean recovery ($\bar{\tau}$)	95% recovery (τ_{95})
Mud	0.30	0.13	0.16	2.2	6.7
Sand	0.32	0.12	0.13	2.4	7.3
Granule–pebble	0.28	0.13	0.14	2.9	8.7
Cobble	0.32	0.13	0.15	2.9	8.8
Boulder	0.30	0.14	0.15	3.1	9.2

Table 3. North Pacific and subregion areas, fishing effort, and sediment profiles.

			Sediment profile				
	Area (1000 km²)	Fishing effort (year⁻¹)[a]	Mud	Sand	Granule–pebble	Cobble	Boulder
North Pacific at large	1248	9.8%	32%	52%	6%	10%	<1%
Aleutian Islands	108	4.1%	1%	46%	17%	36%	<1%
Bering Sea	801	13.7%	35%	57%	3%	5%	0%
Gulf of Alaska	339	2.5%	33%	44%	11%	12%	<1%

[a]Nominal swept area as percentage of total area, summing overlapping effort.

features associated with each habitat type (see Tables S1–S5[1], Supplemental Material). In running the baseline model, we calculated a mean susceptibility value among seabed features for each habitat–gear interaction and a mean recovery for each habitat type. Mean susceptibility was similar across all sediment types for trawl gears ($q \approx 0.30$) and among fixed gears (longlines and pots, $q \approx 014$). Mean recovery ranged from $\bar{\tau} \approx 2.2$ years ($\tau_{95} \approx 6.7$ years) for mud to $\bar{\tau} \approx 3.1$ years ($\tau_{95} \approx 9.2$ years) for boulder habitats (Table 2).

Initial conditions

Options for initial habitat conditions for a model run, H_0, include starting from "pristine" undisturbed habitat, a case-specific set of initial conditions that match known habitat states, or equilibrium initial conditions based upon a "burn-in" period under simulated fishing effort. With insufficient data available to determine the spatial distribution of impacts prior to 2003, but operating on the assumption that impacts were present, we used a burn-in approach for the North Pacific. To calculate H_0, we first randomly selected a value for an initial H_0 from a uniform distribution (0.0 to 1.0) for all grid cells that had nonzero fishing effort from 2003 to 2017. We then ran the model using a 30 year burn-in simulated using the first 3 years of fishing data (2003–2005) repeated ten times, which provided ample time for the model to lose dependence on the initial H_0 and reach a stable habitat state. The terminal month of the burn-in period was then used as H_0 for the full 2003–2017 model run. Sensitivity analyses showed that choice of initial conditions were likely to have little influence on estimates of habitat disturbance by end of the model run (Fig. S3[1], Supplemental Material).

Model applications: management scenarios

Ultimately, the goal of a Fishing Effects model implementation is to inform management of benthic habitat disturbances associated with commercial fishing. Key information needs for habitat management are the spatial and temporal trends in habitat disturbance and evaluation of past or future policies. To demonstrate the utility of the Fishing Effects model for informing habitat management decisions, we considered two cases. First, we assessed the historical trajectory and current state of habitat disturbance across the North Pacific. Model outputs were aggregated over the total domain and subregions by summing disturbed area across grid cells and reported as a proportion to the respective region of analysis (equivalent to averaging proportional disturbance across grid cells). Second, we used the model as a retrospective tool to evaluate outcomes associated with past policy. For this case, we estimated the net reduction in habitat disturbance associated with the implementation of sweep modifications for demersal trawl fisheries implemented in 2011 for the Bering Sea fisheries and 2014 for Gulf of Alaska fisheries. We compared model runs with and without (i.e., $c = 1$ for 2011 and beyond) the sweep modifications to isolate the efficacy of the policy to reduce habitat disturbance. Policy effects were quantified as the percent reduction in total disturbed habitat under sweep modifications relative to the case of no sweep modifications.

Results

Model inputs for the North Pacific

The combined North Pacific (<1000 m depth) model domain is predominately sand (52%) and mud (32%), with lesser amount of cobble (10%), granule–pebble (6%), and boulder habitat (<1%; Table 3; see Fig. S2[1], Supplemental Material for sediment distribution maps). The Bering Sea and Gulf of Alaska subregions had similar habitat proportions, with slightly less rocky substrate (granule–pebble and cobble) in the Bering Sea (8%) than the Gulf of Alaska (23%). The Aleutian Islands had a substantially different sediment profile, with more cobble (36%) and less mud (1%) than the other two subregions.

Fishing effort was not uniformly distributed among sediment types. Proportionally, fishing occurred more on sand than on other sediment types, accounting for 64% of the nominal fishing effort — about 25% greater than the proportion of sand in the North Pacific at large. Mud sediment (31% of nominal effort) was fished relatively equal to its proportional coverage in the North Pacific. Granule–pebble sediments (4% of nominal fishing effort) were fished about 30% less. Cobble (2% of nominal fishing effort) and boulder (<1% of nominal fishing effort) were both fished about 80% less than their respective coverages.

Total yearly nominal swept area across all gears summing overlapping effort ranged between 106 000 km² (2010; 7.1% of the total area) and 146 000 km² (2016; 11.6% of the total area) with a mean 9.8% over the 2003–2017 study period (Fig. 2). Domain-wide, demersal otter trawls were the most common type of gear used, followed by longlines and pots. Fishing effort was proportionally greatest in the Bering Sea (mean annual nominal area swept 2003 to 2017 = 13.7%) compared with the Aleutian Islands (mean = 4.2%) and the Gulf of Alaska (mean = 2.5%).

Fig. 2. Time series of yearly nominal swept area for Aleutian Islands (AI, red diamonds), Bering Sea (BS, blue triangles), and Gulf of Alaska (GOA, green squares) subregions and North Pacific at large (black circles). The vertical axis is the sum of all swept area over a year from nominal gear widths (not contact-adjusted) and sums across overlapping efforts. Nominal swept area is represented as a percentage of the indicated spatial domain. [Colour online.]

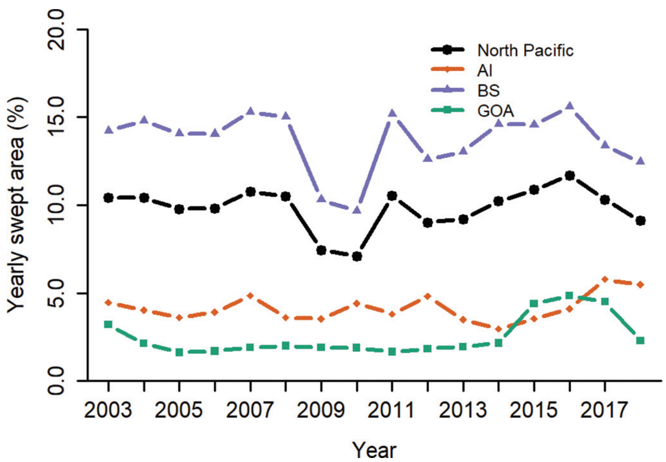


Fig. 3. Predicted habitat disturbance for Aleutian Islands (AI), Bering Sea (BS), and Gulf of Alaska (GOA) subregions and the North Pacific at large on monthly time steps from all commercial fishing activities for 2003–2017. The solid lines show the baseline models for each region. The dashed lines show scenarios with no gear modifications. [Colour online.]

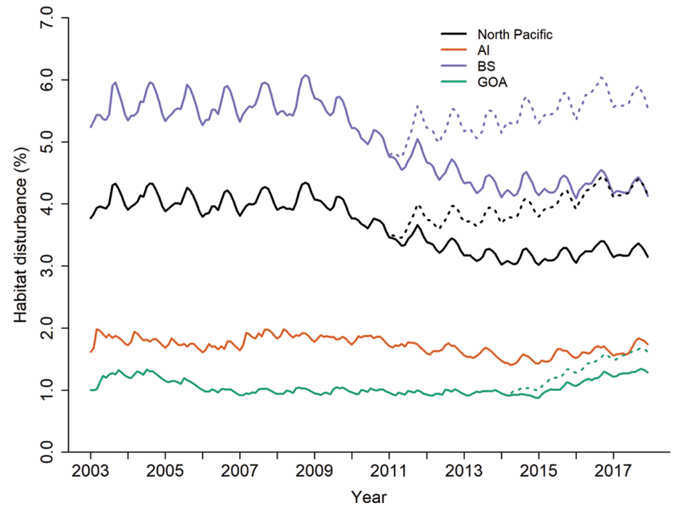
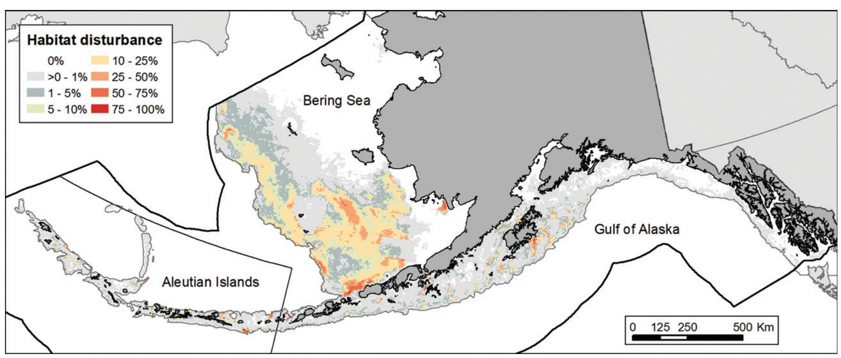


Fig. 4. Habitat disturbance estimates on a 5 km grid at the terminal month of the model run (December 2017). The spatial domain of the output is clipped to depths less than 1000 m, outlined by the thin grey line. The solid black lines indicate US Exclusive Economic Zone and subregion boundaries. Warmer colours indicate higher disturbance; cooler colours indicate lower disturbance. White indicates regions with no fishing activity. [Colour online.]



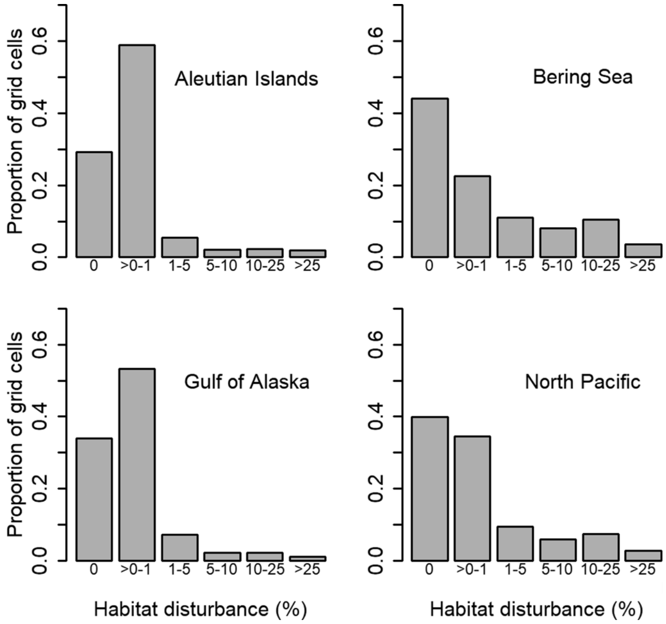
Model application: management scenarios

Using realized fishing data from 2003 to 2017, the Fishing Effects model estimates that 3.1% of habitat in the North Pacific model domain was in a disturbed state at the end of the assessment period (December 2017). Habitat disturbance in the Bering Sea (4.1%) was over twice that estimated for the Aleutian Islands (1.7%) or Gulf of Alaska (1.3%; Fig. 3). Domain-wide, overall habitat disturbance decreased after 2009 (Fig. 3), presumably as a result of reduced fishing effort in 2009 and 2010 (Fig. 2) and implementation of gear regulations requiring trawl sweep modifications in 2011 and 2014. This decline was most apparent the Bering Sea, while habitat disturbance in the Aleutian Islands and Gulf of Alaska remained relatively stable through all modeled years (Fig. 3). The Fishing Effects model captured seasonal fluctuations in habitat disturbance of up to ±0.25 percentage points (Fig. 3), reflecting the strong seasonality in fishing effort in the North Pacific. Habitat disturbance tended to reach the highest levels in the fall (August–October) and lowest in winter months (December–February).

Habitat disturbance was highly spatially aggregated throughout the North Pacific (Fig. 4). About 38% of grid cells experienced zero fishing activity (i.e., 0% habitat disturbance) since the start of the study period, and relatively high disturbance cells (>25%) were rare, making up approximately 3% of the domain (Fig. 5). Among subregions, the Bering Sea had a higher proportion of cells (44%) that were unfished compared with the Aleutian Islands (29% unfished) or Gulf of Alaska (34% unfished). However, the Bering Sea also had substantially higher proportion (10%) of moderate to heavily disturbed grid cells (defined here as >10% disturbance) compared with the Aleutian Islands (4%) or Gulf of Alaska (3%). In contrast, the Aleutian Islands and Gulf of Alaska had substantially higher proportion of grid cells (58% and 53%, respectively) with small, but nonzero disturbance (>0%–1% disturbance), compared with the Bering Sea (23%). This reflects the higher prevalence of longlines and pot fishing gears utilizing habitat in these subregions that is inaccessible to trawls (e.g., steep and rocky terrain) but that have a relatively small contact footprint.

Using the Fishing Effects model as a retrospective policy evaluation tool, we found that in the North Pacific at large, the current estimate of habitat disturbance is about 24% lower than if the trawl sweep gear modifications had never been implemented (Fig. 3). This effect was most pronounced in the Bering Sea (25% reduction in disturbance), which had a higher proportion of gear-modified trawl fisheries, as compared with the Gulf of Alaska (21% reduction). The Aleutian Islands, which had no required gear

Fig. 5. Frequency distributions of habitat disturbance among grid cells in December 2017 for Aleutian Islands, Bering Sea, and Gulf of Alaska subregions and the North Pacific at large.



modifications, had no corresponding reduction in habitat disturbance.

Discussion

The Fishing Effects model provides fisheries managers a quantitative tool to assess seascape-scale impacts to benthic habitats from fishing. The primary outputs are spatiotemporal estimates of habitat impacts, providing information about the current state of habitat, and revealing trends that may not easily be identified through conventional scientific survey efforts alone, especially at seascape scales. Casting the model in discrete time is a marked departure from models of similar scope (Ellis et al. 2014; Fujioka 2006; Dichmont et al. 2008), providing a framework for inclusion of time-varying impacts and recovery. Model output clearly demonstrates that habitat impacts vary both at seasonal and yearly scales, owing to the timing of fishery openings, fleet behavior, and adaptive management decisions. Though not implemented here, time-varying recovery dynamics that reflect seasonal or long-term climate cycles such as El Niño – La Niña (Thrush and Whitlatch 2001) or persistent climate warming (Perry et al. 2005; Harley et al. 2006) could also be implemented if information on these relationships were available. An additional advantage of a discrete-time model is that it enables analysts to project the habitat effects from management decisions in finite time horizons that align with the fishery management process.

We estimated current habitat disturbance to be 3.1% of the modelled North Pacific domain, with higher predicted disturbance in the Bering Sea compared with the Aleutian Islands or Gulf of Alaska. While proportionally a small area — mainly due to a combination of an ecosystem fishery harvest cap (2 million t), large spatial closures, unfishable areas, and regions too far from ports to be economically viable — 3.1% represents over 37 000 km^2 of disturbed habitat. Differences among the subregions generally followed the relative effort in each area, with the highest intensity of fishing effort in the Bering Sea. Aggregate habitat disturbance in the North Pacific demonstrated a declining trend, especially in the Bering Sea, beginning with decreased effort in 2009 and 2010 from reductions in walleye pollock (*Gadus chalcogrammus*) quotas (NOAA 2017*b*) and continuing beyond 2010 from implementation of gear modifications in 2011 and 2014. Because of the time scales required for recovery, habitat disturbance continued to decline beyond 2014, even though the cumulative amount of impacted habitat remained relatively constant.

As a means to evaluate the efficacy of past policies, we used the Fishing Effects model to assess the positive effect gear modifications may have on habitat disturbance. By comparing scenarios with and without gear modifications, we demonstrated trawl gear modifications are associated with an estimated 24% habitat impact reduction by the end of the model run time period (December 2017). We note that the difference in the proportions of estimated domain-wide disturbance is small between the baseline and no gear modification scenarios; however, this reflects the large spatial scale of the model domain relative to the scale of individual fisheries and includes regions closed to fishing.

Mapping habitat impacts at seascape scales carries a high information burden owing to the complexity of marine ecosystems and variable fishing practices. Of the data inputs used in this implementation, the spatial distribution of habitats is the least resolved — a common limitation throughout seascapes globally (Kaiser et al. 2016). While large swaths of the North Pacific study domain are characterized by relatively homogeneous soft bottom habitats, other benthic ecosystems have much greater habitat diversity, and incorporation of higher resolution habitat data could improve the specificity of model applications (e.g., Williams et al. 2011; Eno et al. 2013). While habitat information was difficult to find, model applications presented here benefitted from a nearly complete VMS database providing a high-quality data source for fishing effort. Implementing the model in other regions may prove challenging if VMS coverage is not as extensive or if the VMS data are confidential and not available to analysts (Hinz et al. 2013). Given that fishing effort and habitat information are the key inputs into habitat impacts models, investments into these data sources will be critical for monitoring and managing habitat impacts associated with fishing.

We further capitalized on a gear database developed for the North Pacific with industry and management partners (NOAA 2017*a*). The specificity of nominal gear widths and contact adjustments enabled a high degree of confidence when scaling VMS tracks to swept areas. Still, gear configurations may vary by fishing behavior or ocean conditions from one tow to the next (Somerton and Weinberg 2001). Improvements in the resolution of these gear parameters may be achieved with further collaboration with industry partners to collect tow-level data streams of gear mensuration often already utilized by vessel captains. For fixed gears (e.g., pot sets or longlines), their total swept area footprint is expected to be less than that for mobile gears, yet they may be deployed in steeper and rockier habitats that tend to be more vulnerable to impacts. Thus, better understanding of their footprint and associated impacts is warranted.

While site-specific empirical studies to evaluate benthic ecosystem habitat impacts from fishing are critical for understanding marine ecosystem ecology, scaling up impact and recovery processes to the basin-wide scales needed to monitor and manage fishing impacts has remained a challenge. The Fishing Effects model provides a framework to map the distribution and nature of fishing effort into habitat outcomes at seascape scales. By incorporating discrete-time dynamics and functionality to relate gear designs to habitat impact outcomes, the Fishing Effects model can be used to evaluate the effectiveness of habitat management strategies and to test hypotheses about the impact and recovery dynamics of benthic ecosystems under fishing stress.

Acknowledgements

Funding was provided by the Atkinson Center Academic Venture Fund, the Groundfish Forum, and NOAA's National Marine Fisheries Service Habitat Conservation Division. We thank C. Rose for his help on compiling the gear database and S. Lewis for his work on processing the VMS data. We thank V. vonBiela for comments that improved an earlier draft of this piece. We also thank members of the NPFMC Scientific and Statistical Committee for their suggestions during model development. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the US Government.

References

Chuenpagdee, R., Morgan, L.E., Maxwell, S.M., Norse, E.A., and Pauly, D. 2003. Shifting gears: assessing collateral impacts of fishing methods in US waters. Front. Ecol. Environ. **1**(10): 517–524. doi:10.1890/1540-9295(2003)001[0517:SGACIO]2.0.CO;2.

Collie, J.S., Hall, S.J., Kaiser, M.J., and Poiner, I.R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. J. Anim. Ecol. **69**: 785–798. doi:10.1046/j.1365-2656.2000.00434.x. PMID:29314001.

Dichmont, C.M., Deng, A., Punt, A.E., Ellis, N., Venables, W.N., Kompas, T., Ye, Y., Zhou, S., and Bishop, J. 2008. Beyond biological performance measures in management strategy evaluation: bringing in economics and the effects of trawling on the benthos. Fish. Res. **94**(3): 238–250. doi:10.1016/j.fishres.2008.05.007.

Doherty, B., Johnson, S.D.N., and Cox, S.P. 2018. Using autonomous video to estimate the bottom-contact area of longline trap gear and presence–absence of sensitive benthic habitat. Can. J. Fish. Aquat. Sci. **75**(5): 797–812. doi:10.1139/cjfas-2016-0483.

Ellis, N., Pantus, F., and Pitcher, C.R. 2014. Scaling up experimental trawl impact results to fishery management scales — a modelling approach for a "hot time". Can. J. Fish. Aquat. Sci. **71**(5): 733–746. doi:10.1139/cjfas-2013-0426.

Eno, N.C., Frid, C.L.J., Hall, K., Ramsay, K., Sharp, R.A.M., Brazier, D.P., Hearn, S., Dernie, K.M., Robinson, K.A., Paramor, O.A.L., and Robinson, L.A. 2013. Assessing the sensitivity of habitats to fishing: from seabed maps to sensitivity maps. J. Fish Biol. **83**(4): 826–846. doi:10.1111/jfb.12132. PMID:24090550.

Feinerer, I., and Hornik, K. 2017. tm: text mining package [online]. Available from https://cran.r-project.org/package=tm.

Freese, L., Auster, P.J., Heifetz, J., and Wing, B.L. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. Mar. Ecol. Prog. Ser. **182**: 119–126. doi:10.3354/meps182119.

Fujioka, J.T. 2006. A model for evaluating fishing impacts on habitat and comparing fishing closure strategies. Can. J. Fish. Aquat. Sci. **63**(10): 2330–2342. doi:10.1139/f06-120.

Gerritsen, H.D., Minto, C., and Lordan, C. 2013. How much of the seabed is impacted by mobile fishing gear? Absolute estimates from Vessel Monitoring System (VMS) point data. ICES J. Mar. Sci. **70**(3): 523–531. doi:10.1093/icesjms/fst017.

Grabowski, J.H., Bachman, M., Demarest, C., Eayrs, S., Harris, B.P., Malkoski, V., Packer, D., and Stevenson, D. 2014. Assessing the vulnerability of marine benthos to fishing gear impacts. Rev. Fish. Sci. Aquacult. **22**(2): 142–155. doi:10.1080/10641262.2013.846292.

Hall, S.J. 1999. The effect of fishing on marine ecosystems and communities. Blackwell, Oxford, UK.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., and Watson, R. 2008. A global map of human impact on marine ecosystems. Science, **319**(5865): 948–952. doi:10.1126/science.1149345. PMID:18276889.

Harley, C.D.G., Randall Hughes, A., Hultgren, K.M., Miner, B.G., Sorte, C.J.B., Thornber, C.S., Rodriguez, L.F., Tomanek, L., and Williams, S.L. 2006. The impacts of climate change in coastal marine systems. Ecol. Lett. **9**(2): 228–241. doi:10.1111/j.1461-0248.2005.00871.x. PMID:16958887.

Heifetz, J., Stone, R.P., and Shotwell, S.K. 2009. Damage and disturbance to coral and sponge habitat of the Aleutian archipelago. Mar. Ecol. Prog. Ser. **397**: 295–303. doi:10.3354/meps08304.

Hiddink, J.G., Jennings, S., and Kaiser, M.J. 2007. Assessing and predicting the relative ecological impacts of disturbance on habitats with different sensitivities. J. Appl. Ecol. **44**(2): 405–413. doi:10.1111/j.1365-2664.2007.01274.x.

Hiddink, J.G., Jennings, S., Sciberras, M., Szostek, C.L., Hughes, K.M., Ellis, N., Rijnsdorp, A.D., McConnaughey, R.A., Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso, R.O., Parma, A.M., Suuronen, P., and Kaiser, M.J. 2017. Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. Proc. Natl. Acad. Sci. **114**(31): 8301–8306. doi:10.1073/pnas.1618858114. PMID:28716926.

Hinz, H., Murray, L.G., Lambert, G.I., Hiddink, J.G., and Kaiser, M.J. 2013. Confidentiality over fishing effort data threatens science and management progress. Fish Fish. **14**(1): 110–117. doi:10.1111/j.1467-2979.2012.00475.x.

Jennings, S., and Kaiser, M.J. 1998. The Effects of fishing on marine ecosystems. *In* Advances in marine biology. *Edited by* J.H.S. Blaxter, A.J. Southward, and P.A. Tyler. Academic Press. pp. 201–352. doi:10.1016/S0065-2881(08)60212-6.

Kaiser, M.J., Clarke, K.R., Hinz, H., Austen, M.C.V., Somerfield, P.J., and Karakassis, I. 2006. Global analysis of response and recovery of benthic biota to fishing. Mar. Ecol. Prog. Ser. **311**: 1–14. doi:10.3354/meps311001.

Kaiser, M.J., Hilborn, R., Jennings, S., Amaroso, R., Andersen, M., Balliet, K., Barratt, E., Bergstad, O.A., Bishop, S., Bostrom, J.L., Boyd, C., Bruce, E.A., Burden, M., Carey, C., Clermont, J., Collie, J.S., Delahunty, A., Dixon, J., Eayrs, S., Edwards, N., Fujita, R., Gauvin, J., Gleason, M., Harris, B., He, P., Hiddink, J.G., Hughes, K.M., Inostroza, M., Kenny, A., Kritzer, J., Kuntzsch, V., Lasta, M., Lopez, I., Loveridge, C., Lynch, D., Masters, J., Mazor, T., McConnaughey, R.A., Moenne, M., Nimick, A.M., Olsen, A., Parker, D., Parma, A., Penney, C., Pierce, D., Pitcher, R., Pol, M., Richardson, E., Rijnsdorp, A.D., Rilatt, S., Rodmell, D.P., Rose, C., Sethi, S.A., Short, K., Suuronen, P., Taylor, E., Wallace, S., Webb, L., Wickham, E., Wilding, S.R., Wilson, A., Winger, P., and Sutherland, W.J. 2016. Prioritization of knowledge-needs to achieve best practices for bottom trawling in relation to seabed habitats. Fish Fish. **17**(3): 637–663. doi:10.1111/faf.12134.

Kenchington, E.L.R., Gilkinson, K.D., MacIsaac, K.G., Bourbonnais-Boyce, C., Kenchington, T.J., Smith, S.J., and Gordon, D.C. 2006. Effects of experimental otter trawling on benthic assemblages on Western Bank, northwest Atlantic Ocean. J. Sea Res. **56**(3): 249–270. doi:10.1016/j.seares.2006.03.010.

Mantua, N.J., and Hare, S.R. 2002. The Pacific Decadal Oscillation. J. Oceanogr. **58**(1): 35–44. doi:10.1023/A:1015820616384.

McLeod, K., and Leslie, H. (*Editors*). 2009. Ecosystem-based management of the oceans. 2nd ed. Island Press.

NEFMC. 2011. The Swept Area Seabed Impact (SASI) approach: a tool for analyzing the effects of fishing on essential fish habitat. New England Fishery Management Council, Newburyport, Mass.

Nimick, A.M., and Harris, B.P. 2016. Essential fish habitat regulation in the United States: lessons for high latitudes? *In* The yearbook of polar law. Vol. 8. *Edited by* G. Alfredsson and T. Koivurova. pp. 266–278.

NOAA. 2010. Fisheries of the Exclusive Economic Zone off Alaska; modified nonpelagic trawl gear and habitat conservation in the Bering Sea Subarea. National Oceanic and Atmospheric Administration. 50 CFR 679.

NOAA. 2014. Fisheries of the Exclusive Economic Zone off Alaska; Tanner Crab Area Closure in the Gulf of Alaska and gear modification requirements for the Gulf of Alaska and Bering Sea Groundfish Fisheries. National Oceanic and Atmospheric Administration. 50 CFR 679.

NOAA. 2017*a*. Essential Fish Habitat (EFH): 5-year review [online]. National Oceanic and Atmospheric Administration, Anchorage, Alaska. Available from https://alaskafisheries.noaa.gov/sites/default/files/2015efh_5yearreview.pdf.

NOAA. 2017*b*. Groundfish harvest specifications [online]. National Oceanic and Atmospheric Administration. Available from https://alaskafisheries.noaa.gov/harvest-specifications/field_harvest_spec_year/2016-2017-751 [accessed 25 September 2017].

NOS. 2013. NOAA/NOS and USCGS Seabed Descriptions from Hydrographic Surveys. National Ocean Service, NOAA National Centers for Environmental Information. doi:10.7289/V5BG2KWG.

NPFMC. 2017. Environmental Assessment for: Essential Fish Habitat (EFH) Omnibus Amendment: Appendix 7 [online]. North Pacific Fishery Management Council, Anchorage, Alaska. Available from http://npfmc.legistar.com/gateway.aspx?M=F&ID=f55791ad-2e93-4fa3-9300-e62bdaddd993.pdf.

O'Neill, F.G., and Ivanović, A. 2016. The physical impact of towed demersal fishing gears on soft sediments. ICES J. Mar. Sci. **73**: i5–i14. doi:10.1093/icesjms/fsv125.

Perry, A.L., Low, P.J., Ellis, J.R., and Reynolds, J.D. 2005. Climate change and distribution shifts in marine fishes. Science, **308**: 1912–1915. doi:10.1126/science.1111322. PMID:15890845.

Pikitch, E.K., Santora, C., Babcock, E.A., Bakun, A., Bonfil, R., Conover, D.O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E.D., Link, J., Livingston, P.A., Mangel, M., McAllister, M.K., Pope, J., and Sainsbury, K.J. 2004. Ecosystem-based fishery management. Science, **305**: 346–347. doi:10.1126/science.1098222. PMID:15256658.

Rijnsdorp, A.D., Bastardie, F., Bolam, S.G., Buhl-Mortensen, L., Eigaard, O.R., Hamon, K.G., Hiddink, J.G., Hintzen, N.T., Ivanović, A., Kenny, A., Laffargue, P., Nielsen, J.R., O'Neill, F.G., Piet, G.J., Polet, H., Sala, A., Smith, C., van Denderen, P.D., Van Kooten, T., and Zengin, M. 2016. Towards a framework for the quantitative assessment of trawling impact on the seabed and benthic ecosystem. ICES J. Mar. Sci. **73**: i127–i138. doi:10.1093/icesjms/fsv207.

Rose, C., Munk, E., Hammond, C., and Stoner, A. 2010*a*. Cooperative research to reduce the effects of Bering Sea flatfish trawling on seafloor habitat and crabs [online]. *In* AFSC Quarterly Report. Available from https://www.afsc.noaa.gov/Quarterly/jfm2010/jfm2010feature.pdf.

Rose, C.S., Gauvin, J.R., and Hammond, C.F. 2010*b*. Effective herding of flatfish by cables with minimal seafloor contact. Fish. Bull. **108**(2): 136–144.

Smith, K.R., and McConnaughey, R.A. 1999. Surficial sediments of the eastern Bering Sea continental shelf: EBSSED database documentation. US Dep. Commer., NOAA Tech. Memo. NMFSAFSC-104.

Somerton, D.A., and Weinberg, K.L. 2001. The affect of speed through the water on footrope contact of a survey trawl. Fish. Res. **53**(1): 17–24. doi:10.1016/S0165-7836(00)00272-1.

Stone, R.P., Masuda, M.M., and Karinen, J.F. 2015. Assessing the ecological importance of red tree coral thickets in the eastern Gulf of Alaska. ICES J. Mar. Sci. **72**(3): 900–915. doi:10.1093/icesjms/fsu190.

Thrush, S.F., and Whitlatch, R.B. 2001. Recovery dynamics in benthic communities: balancing detail with simplification. *In* Ecological comparisons of sedimentary shores. *Edited by* K. Reise. Springer-Verlag Berlin Heidelberg. pp. 297–316. doi:10.1007/978-3-642-56557-1_15.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., Dayton, P.K., Cryer, M., Turner, S.J., Funnell, G.A., Budd, R.G., Milburn, C.J., and Wilkinson, M.R. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecol. Appl. **8**(3): 866–879. doi:10.1890/1051-0761(1998)008[0866:DOTMBH]2.0.CO;2.

van Denderen, P.D., Hintzen, N.T., van Kooten, T., and Rijnsdorp, A.D. 2015. Temporal aggregation of bottom trawling and its implication for the impact on the benthic ecosystem. ICES J. Mar. Sci. **72**(3): 952–961. doi:10.1093/icesjms/fsu183.

Watling, L., and Norse, E.A. 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. Conserv. Biol. **12**(6): 1180–1197. doi:10.1046/j.1523-1739.1998.0120061180.x.

Williams, A., Dowdney, J., Smith, A.D.M., Hobday, A.J., and Fuller, M. 2011. Evaluating impacts of fishing on benthic habitats: a risk assessment framework applied to Australian fisheries. Fish. Res. **112**(3): 154–167. Elsevier B.V. doi:10.1016/j.fishres.2011.01.028.

Winger, P.D., Eayrs, S., and Glass, C.W. 2010. Fish behavior near bottom trawls. *In* Behavior of marine fishes: capture processes and conservation challenges. *Edited by* P. He. Wiley-Blackwell. pp. 67–95.

Zimmermann, M., and Prescott, M.M. 2015. Smooth sheet bathymetry of the central Gulf of Alaska. US Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-287. doi:10.7289/V5GT5K4F.

Zimmermann, M., Prescott, M.M., and Rooper, C.N. 2013. Smooth Sheet Bathymetry of the Aleutian Islands. US Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-250.