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Evaluating benthic impact of the Gulf of Maine lobster fishery using the swept area seabed impact (SASI) model

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Abstract

The Magnuson-Stevens Fishery Conservation and Management Act mandates U.S. fisheries minimize adverse effects of fishing on essential fish habitat (EFH). The Gulf of Maine (GoM) American lobster fishery is the most valuable U.S. fishery, and can deploy more than three million traps annually. To date, the impact of this fishery on benthic EFH has not been addressed quantitatively. To evaluate the impact of the GoM lobster fishery on EFH, lobster fishing effort was incorporated into a model linking habitat susceptibility and recovery to area impacted by fishing gear; the Swept Area Seabed Impact model. Impact to EFH was localized along the coast and highest along mid-coast Maine. Upwards of 13% of the benthos is in the process of recovery, but between 99.92 – 99.96% of initially affected habitat fully recovers. These estimates suggest that lobster fishing negligibly contributes to accumulation of EFH damage in the GoM due to the expansive area fished and the small footprint of each trap. Identifying areas of persistent impact is crucial in developing effective fisheries management for critical marine habitats.

INTRODUCTION

The Gulf of Maine (GoM) lobster fishery is the most valuable fishery in the United States (MDMR 2019a, NMFS 2019). Over the past three decades, the annual lobster landings in the GoM have rapidly increased, effectively multiplying historical landings five-fold (NMFS 2015a). Lobster population expansion has been attributed to relaxed top-down pressure (Jackson et al. 2001; McMahan et al. 2013; Wahle et al. 2013), herring bait subsidization (Saila et al. 2002; Grabowski et al. 2010), increased algal habitat for juveniles due to reductions in urchin populations (Bologna and Steneck 1993, Steneck et al 2004), and ocean warming shifting this species' range northward (Pinsky et al. 2013) and offshore (Tanaka and Chen, 2016; Mazur et al. 2020) due to increased habitat thermal suitability (LeBris et al. 2018; Goode et al. 2019). Alongside population increase, advancements in fishing technology (i.e. vessels and traps; ASMFC 1996) has increased fishing effort and gear abundance in the GoM (Steneck et al. 2017). However, increases in fishing activity may come with unintended consequences. Every type of fishing gear that interacts with the benthos, to some extent, can damage essential fish habitat (EFH) crucial to the reproduction, development, and protection of fish species (Grieve et al. 2014; Grieve et al. 2015). Impacts to EFH by fishing gear range from changes to sediment habitats as well as the damage and/or loss of emergent epiflora (Bridger 1972; Peterson et al. 1983; Currie and Parry 1996; Watling and Norse 1998; Watling et al. 2001). Such impacts by fishing gear can also greatly reduce benthic structural diversity and alter population productivity (Dayton et al. 1995; Watling and Norse 1998). Therefore, the increase in fishing effort by the GoM lobster fishery may be inadvertently increasing degradation of EFH.

The Magnuson-Stevens Fishery Conservation and Management Act 1996 (USA) mandates that EFH be protected, to the extent practicable, from fishing related impacts. The large spatial footprint and potential impact of the GoM lobster fishery necessitates a more comprehensive understanding of how the frequency and intensity of lobster fishing effort affects EFH in the GoM. To assess this impact, a quantitative framework examining habitat-gear interactions is necessary.

The Swept Area Seabed Impact (SASI) model is a method by which we can estimate the potential impact of fishing gear on the EFH of the benthos (NEFMC 2011; Grabowski et al. 2014). The SASI model was developed to assess the benthic impacts of the most common bottom fishing gears in New England (e.g., otter trawls, scallop dredges, hydraulic clam dredges, gillnets, longlines, and lobster traps). Briefly, the SASI model determines what percentage of the fishing gear's footprint functionally reduces the biological and/or geological EFH features of the benthos based on gear abundance, fishing frequency, and substrate classification. This modeling approach can help identify regions of high impact by fishing gear and, importantly, the degree to which these habitats are able to recover.

Here, we estimate the potential functional reduction of biological and geological EFH features by the GoM lobster fishery using the SASI model. Specifically, we simulate two different effort scenarios which represent the maximum and minimum potential impacts of the GoM lobster fishery on EFH. We also estimate the recovery potential of functionally reduced EFH and the accumulation of functionally reduced EFH over time. This application of the SASI model aims to quantify temporal dynamics and spatial variability in the functional reduction of EFH within the GoM and identify regions where persistent impacts occur. Delineating such locations will help fisheries management target their attention and/or develop actions that better protect regions that are more prone to damage.

METHODS

The Gulf of Maine lobster fishery

The American lobster in the United States' is regulated by both state and federal lobster management areas. Lobster fishing grounds are subdivided into seven federally regulated management areas that range from Cape Hatteras, North Carolina to the Maine-Canada border. The coastal GoM is federal nearshore management area 1 and is the domain over which we are conducting our study. This management region is further subdivided into state-level fishing management areas; Maine zones A-G, New Hampshire, and northern Massachusetts (Fig. 1). Across these zones, and distance from shore, lobster fishing regulations and practices vary significantly (McCarron and Tetreault, 2012; NMFS 2015a).

The Swept Area Seabed Impact (SASI) Model

Developed by the New England Fishery Management Council's Habitat Plan Development Team, the SASI model is a quantitative framework designed to assess the vulnerability of EFH to six of the most commonly fished bottom-tending gears in New England; trawls, scallop dredges, gillnets, longlines, traps, and hydraulic clam dredges (NEFMC 2011; Grabowski et al. 2014). This SASI team convened an expert panel of scientists, conducted an extensive literature review, and developed a framework to combine peer-reviewed habitat-specific susceptibility and recovery rates into a single quantitative assessment of fishing gear impacts to benthic EFH. Marine substrates were subdivided into five categories based upon substrate data availability and usefulness in regional resource and fisheries management; mud, sand, granule-pebble, cobble, and boulder (Table 1). Predominant biological and geological EFH features were identified and assigned to each substrate type (Table S1). Biological EFH features necessitated a greater depth of analysis. Commonly found marine species were assigned to various biological feature functional groups, and the relative importance of these species to each functional group was determined. A vulnerability assessment was developed to organize and generate quantitative estimates of susceptibility and recovery values for each biological and geological EFH feature group (Table S2, S3). Results from this assessment were combined to estimate substrate-specific susceptibility and recovery scores to then be utilized in estimating fishing gear-specific impacts to benthic EFH. Further information on the literature review and evidence considered can be found in NEFMC (2011) and Grabowski et al. (2014).

Applying SASI to the Lobster Fishery

Four parameters are required to evaluate potential habitat impact using the SASI model: (1) fishing gear abundance (number of traps), (2) fishing frequency (deployments day^{-1}), (3) seabed substrate composition and distribution, and (4) susceptibility of each substrate type to damage by fishing gear (% area functionally reduced).

(1) Fishing Gear Abundance

The GoM American lobster fishery is not subject to mandatory vessel trip reports or vessel monitoring systems, which are typical methods of gathering fishing effort data. In lieu of this, determination of high-resolution fishing effort data has been the topic of research from a multitude of organizations. Fishing effort has been estimated by the Island Institute Mapping Working Waters project (The Island Institute 2012, 2016), the State of Maine Department of Marine Resources (MDMR) Lobster Sea Sampling Program (MDMR, 2016), the Maine Lobsterman's Association dasymetric mapping effort (Brehme et al. 2015), and the National

Marine Fisheries Service (NMFS) Vertical Line Model (NMFS, 2014). We chose to use the NMFS Vertical Line Model in our analysis based on the utilization of federal- and state-level fishing activity data, monthly varying endline estimates, and consistent spatial resolution of gridded endline estimates.

Fishing effort was characterized by the number of vertical lines per 10x10 arcmin area from the Vertical Line Model. The number of vertical lines was assumed to be homogeneously distributed within each 10 x10 arcmin area. Each 10 x10 arcmin area was subdivided into four 5 x 5 arcmin areas to better characterize distance from shore and management zone. We used the lobster gear report summary from McCarron and Tetreault (2012) to allocate the number of traps fished per endline, and to vary the number of traps per endline as a function of lobster management zone and distance from shore.

We estimated two effort to capture the potential bounds in lobster fishing effort. The Atlantic Large Whale Take Reduction Plan management area requirements (NMFS 2015a) were used to assign maximum and minimum trap per trawl limits based on distance from shore. Applying these trap per trawl limits to the Vertical Line Model, variations in lobster gear configuration and fishing practices, and fishery regulations (Table 2), we estimated the maximum and minimum number of traps fished within the GoM.

(2) Fishing Frequency

The Maine Lobstermen's Association conducted a survey from northern Maine to Massachusetts to identify how lobster fishing practices vary along the coast, distance from shore, and throughout the year (McCarron and Tetreault, 2012). Although fishing practices vary at the individual level, the survey aimed to determine the most common gear configurations adopted by lobstermen and the timeframes over which most fishing occurs. The survey identified how frequently fishermen haul their traps, how this frequency changes during the year, and differences in these patterns among fishing zones. We used the values and timeframes identified from this survey to calculate the number of gear hauls by the GoM lobster fishery.

(3) Seabed Substrate Composition and Distribution

We gathered seabed sediment classification data from the United States Geological Survey (USGS) East Coast Sediment Texture Database (McMullen et al. 2014). This database characterizes sediments as one of fourteen potential classifications (Table 1) and has been used to investigating lobster habitat suitability (Tanaka and Chen 2016) and macrobenthos variability (McHenry et al. 2017). However, the SASI model simplifies marine substrates into five categories; mud, sand, granule-pebble, cobble, and boulder (NEFMC 2011; Grabowski et al. 2014). To convert the substrate classification scheme from the USGS to that used by the SASI model, we consolidated substrate class based on similarity in Shepard classification and Wentworth scale (e.g., sand dominated sand-silt-clay mixtures were categorized as sand according to the SASI model; Fig. S1; Wentworth, 1922; Shepard, 1954; Schlee and Webster, 1967; Poppe et al., 2004). Sediment classification was linearly interpolated over a 0.5-arcmin resolution grid. The relative percentage of each substrate type per 5 x 5 arcmin area was determined using the interpolated sediment classification grid. One-way ANOVA and Tukey-Kramer post-hoc tests were performed to identify significant differences in rocky substrate cover by lobster management zone. The variance of substrate percent cover was determined and standardized by the maximum possible variance using the USGS and SASI sediment classifications. Standardized variance values closer to one represent a more homogenous substrate distribution while values closer to zero represent a more heterogeneous substrate distribution per 5x5 arcmin area. We used a Wilcoxon signed rank non-parametric test to identify if the downscaling of sediment classification significantly changed the spatial complexity of substrate classification.

(4) Substrate susceptibility to damage

The SASI model vulnerability assessment estimated the susceptibility of EFH to damage by fishing gear interactions (NEFMC 2011; Grabowski et al. 2014). Susceptibility was defined as the percent reduction in functional value that any feature provides to a fish species (e.g. an EFH feature with a S score of 100% would have no functional value after being disturbed by fishing gear). Susceptibility scores are estimates of the reduction in functional value of substrate-associated EFH features after a single-pass fishing event. The size, fragility, and relative abundance of geological features and species present were considered when assigning susceptibility scores per substrate. Susceptibility scores were typically highest for more invasive, mobile gears compared to fixed gears (NEFMC 2011; Grabowski et al. 2014). Traps have susceptibility scores ranging from 5.3% to 17.5% depending on substrate and type of EFH feature (Table 1; NEFMC 2011; Grabowski et al. 2014).

Evaluating area of impact

Using fishing gear abundance and fishing frequency, we estimated the area of the seabed that is swept by lobster fishing gear. The area of the seabed swept by lobster fishing gear is influenced by the number of traps, trap size, length of groundline between traps, and how far the gear is dragged along the seabed (Fig. 2). Since dragging can substantially increase the area of the seabed interacted by lobster fishing gear, we assumed that the entire area of the seabed between the first and last trap on a lobster trawl is impacted during a hauling event (Fig. 2; Schweitzer et al., 2018; Stevens, 2020). This approach produces comparable estimates of interacted benthos to other studies (Schweitzer et al., 2018; Stevens, 2020). We calculated the area of the seabed swept by lobster fishing gear (*ASC*; m^2) at each location (*i*) and month (*m*) as:

$$ASC_{trap,i,m} = \left(\sum_{1}^{n_{i,m}} [d_{tn} \cdot l_{tn}] + \sum_{1}^{n_{i,m}-1} [d_{gn} \cdot l_{gn}] \right) * f_{i,m}$$

where $n_{i,m}$ is the number of traps, $n_{i,m}$-1 is the number of groundlines between traps, d_{tn} is the lateral distance the *n*th trap moves over the seabed, l_{tn} is the length of the *n*th trap, d_{gn} is the lateral distance of the *n*th groundline that moves over the seabed, l_{gn} is the length of the *n*th groundline, and $f_{i,m}$ is the haul frequency of the fishing gear (NEFMC, 2011). Consistent with NEFMC report (2011), we assumed that lobster trap length and the side-to-side dragging of traps were both one meter. Given the low probability that fishing gear falls on, and functionally damages, the same exact area of the benthos multiple times, we assumed that every fishing event interacts with a new area of the benthos. Since the SASI model considers repeated interactions on an EFH feature to not increase the area or degree of impact, this assumption likely produced an over-estimate of the area functionally damaged by fishing gear.

We then applied the seabed substrate composition and corresponding susceptibility scores to determine the area of the seabed functionally damaged. We calculated the area of the seabed functionally damaged by actively fished gear (*A*; m^2) at each 5x5 arcmin location (*i*) per month (*m*) as:

$$A_{i,m} = \sum_{1}^{h=5} [ASC_{i,m} * \ P_{h,i} * S_{h,i}]$$

where h is the habitat substrate type, $P_{h,i}$ is the proportion of the 5x5 arcmin area of each substrate type, and $S_{h,i}$ is the susceptibility of the benthos to functional reduction.

Recovery Assessments

We conducted two recovery potential assessments of functionally damaged EFH. The first assessment determined how much of the area functionally damaged in a year can fully recover. Full recovery occurs when the initially damaged area of the seafloor remains undisturbed by fishing activity long enough to recover its functional contribution as EFH (Table 1). To calculate this, we simulated the random overlap of continued fishing gear on already damaged habitats. The proportion of fishing overlap at each location ($P_{overlap,i}$) was the amount of initially damaged EFH that remains damaged via repeated interactions with fishing gear, and was calculated as:

$$P_{overlap,i} = \sum_{1}^{h=5} \left[\frac{A_{annual,i,h} * A_{month,i,h} * T_h}{B_i^2} \right]$$

where h is the habitat substrate type, $A_{annual,i,h}$ is the annual area of EFH functionally reduced, $A_{month,i,h}$ is the monthly average area of EFH functionally reduced, T_h is the time of recovery in months, and B_i is the available bottom area.

The second recovery assessment estimated the accumulation of EFH damage over time. Benthic community status following impact by fishing gear exists at an equilibrium state between depletion rate and recovery (Jennings et al. 2012; Pitcher et al. 2017). This theoretical framework can be extrapolated to fishing gear impacts over large spatial scales. Like benthic communities, the status of EFH is a balance between additional impact and rate of recovery (NEFMC 2011). Equilibrium occurs where added fishing impact balances the rate of habitat recovery, and the area of impacted benthos at which this occurs can be estimated. The accumulated area (A) of the seabed functionally damaged at each location (i) over time was calculated as follows:

$$A_{t+1,i} = A_{t,i} + A_{month,i} - A_{t,i} * \frac{1}{T_i}$$

where t is time in months, $A_{month,i}$ is the monthly average area of EFH functionally reduced, and T_i is the time of recovery in months. If we ignore the first two terms and set monthly added damage equal to the rate of recovery;

$$A_{month,i} = A_{t,i} * \frac{1}{T_i}$$

We can then solve for the area at which this balance occurs. We estimated the cumulative area of the seabed that is functionally reduced as:

$$A_{t,i} = A_{month,i} * T_i$$

This procedure was conducted for each location (i) accounting for variability in substrate (h) recovery times:

$$A_{cumulative,i} = \sum_{1}^{h=5} (A_{i,month,h} * T_h)$$

For both recovery assessments, we increased the time required for EFH to recover as depth increased. Baseline recovery rates were assigned to each EFH and substrate type (Table 1), and we increased the time of recovery by one standard error with each additional 50 m of depth, as suggested in Grabowski et al. (2014; Fig. S2).

Data Analysis and Visualization

The areas of EFH functional reduction were collated by distance from shore. Then we used a three-way ANOVA and Tukey-Kramer post-hoc test to determine differences in EFH damage as a function of distance from shore, between annual and cumulative damage, and between geological and biological EFH features. The annual area of EFH functional reduction was averaged over each lobster management zone and one-way ANOVAs and Tukey-Kramer post-hoc tests determined which zones experienced higher or lower areas of EFH damage. T-tests assuming unequal variances were conducted to determine if there were differences in the annual area of biological or geological EFH functional reduction and if there were differences in the remaining area of biological or geological EFH functionally reduced after recovery. All maps were generated using Matlab® version R2020a and the M_Map mapping software (Pawlowicz 2020).

RESULTS

Benthic habitat characterizations were consolidated before being used in the SASI model within the GoM. Of the areas in which fishing effort data were available, we associated 613 $5x5$ arcmin areas with a SASI sediment type (Fig. 1). Downscaling sediment classification from the USGS to SASI classification scheme retained large-scale patterns of substrate distribution (Fig. 3 A and B). For example, both classifications show that the percent cover of rocky substrate differed significantly by zone; $F_{8,533} = 10.6$, $p < 0.001$ (Fig. 3 C). Zones A, F, G, and NH had the highest percent cover of rocky substrate, while zones B, C, D, and MA had the lowest. Despite retaining large-scale patterns of substrate distribution, the consolidation of sediment classification significantly changed the substrate complexity per 5x5 arcmin area. Standardized SASI substrate variance (mean $\pm$ SE = 0.43 $\pm$ 0.01) was significantly higher than the USGS classification scheme (mean ± SE = 0.26 ± 0.01; *n* = 1120, Z = -20.3, p < 0.0001; Fig. 3 D). Thus, converting USGS sediment classifications to the SASI model shifts sediment complexity to be more homogeneous, a necessary simplification by the SASI model to relate substrate type to benthic community structure.

The Vertical Line Model estimate of lobster fishing effort is consistent with previous approaches (McCarron and Tetreault 2012) and produced total trap estimates comparable to the number of traps legally allowed to fish (MDMR 2019b). Highest fishing effort occurred along mid-coast and northern Maine (Fig. S3). Fishing activity gradually increased from January to the beginning of May-June, and gradually decreased beginning in September ending around December-January (Fig. S3, S4), consistent with Boenish and Chen (2018). This seasonality was also present in our estimates of total traps fished in the GoM, where the maximum number of traps fished occurred in August and the minimum number of traps fished occurred in March (Fig. 4). Using the two fishing effort scenarios, we estimated the seasonal minimum and maximum number of traps fished. We estimated the lowest number of traps fished during the year as 0.24 and 0.59 million traps, and the highest amount fished as 1.15 and 2.77 million traps, minimum and maximum trap per endline scenario, respectively (Fig. 4). The maximum trap per endline scenario of 2.77 million traps is comparable to the estimates of lobster traps fished by Auster and Langton (1998) and the average annual number of trap tags sold by the MDMR from 1998-2018 of 2.94 million tags (MDMR, 2019b). Therefore, we present the results of our analysis using the maximum trap per endline scenario.

Annual EFH functional reduction was unevenly distributed between lobster management zones and largely reflected the distribution of fishing gear. Specifically, annual EFH functional reduction ranged from 0.01 to 5.74% per 5x5 arcmin area and was significantly higher for biological features (mean ± S.E. = 1.17 ± 0.03%) than geological features (mean ± S.E. = 0.94 ± 0.03%) (Fig. 5; $t_{1038} = 5.63$, $p < 0.0001$). This significant difference was driven by higher susceptibility scores for biological compared to geological habitat features (Table 1). The majority of biological EFH impacts occurred between 0-3 nm from shore (Fig. S5; $F_{11,2336} = 111.8$, $p < 0.0001$) and in lobster management zones NH, F, E, D, A, and C, highest to lowest impact, respectively (Fig. 5; $F_{8,557} = 7.63$, $p < 0.0001$). Similarly, impacts to geological EFH features were greatest between 0-3 nm (Fig. S5; $F_{11,2336} = 111.8$, $p < 0.0001$) and in lobster management zones D, E, F, NH, C, and MA, highest to lowest impact, respectively (Fig. 5; $F_{8,557} = 8.19$, $p < 0.0001$).

The recovery potential of functionally reduced EFH varied significantly by type of EFH feature. After the necessary time to recover, we estimated the area that remained functionally reduced via continued gear interaction. These areas ranged between 0 to 15,670 m^2 or 0 to 0.025% of each 5x5 arcmin area. Significantly more biological EFH features (mean ± S.E. = 559 ± 47 m^2, 0.00096 ± 0.00001%) remained functionally reduced compared to geological features (mean ± S.E. = 222 ± 19 m^2, 0.00036 ± 0.00003%; Fig. 6; $t_{808} = 7.43$, $p < 0.0001$). These areas of continued impact correspond to approximately 0.08 and 0.04% of the initial area functionally reduced, biological and geological EFH features respectively. Thus, approximately 99.92 and 99.96% of the functionally reduced biological and geological EFH, respectively, fully recover within their assigned period of recovery.

While patterns in accumulated functional reduction of EFH over time was similar to annual functional reduction across zones, we did observe a significant shift in accumulated functional reduction offshore. Offshore habitats were more susceptible to accumulating functionally reduced EFH because the model attributes longer recovery times to deeper, offshore habitats. Recovery times of less than a year allow recovery to outpace additional impact and result in a cumulative area of EFH functional reduction that is less than the area reduced annually (Fig. 7). Conversely, recovery times greater than a year allow additional disturbances to outpace recovery such that the cumulative area of EFH functional reduction becomes greater than the area reduced annually (Fig. 7). Longer recovery times correspond to larger areas of cumulative EFH functional reduction and longer periods until added damage balances recovery. Thus, cumulative functional reduction to EFH was higher offshore compared to annual values (Fig. S5, $F_{11,2336} = 118.8$, $p < 0.0001$). However, the geological and biological habitat features are differentially affected. There was a larger percentage increase in the area of functional reduction for biological (mean ± S.E. = 1.51 ± 0.03%) than geological features (mean ± S.E. = 0.38 ± 0.06%; $t_{824} = 17.2$, $p < 0.0001$; Fig. S5). Cumulative EFH functional reduction ranged from 0 to 12.8% per 5x5 arcmin area and was significantly higher for biological (mean ± S.E. = 2.73 ± 0.07%) than geological features (mean ± S.E. = 1.07 ± 0.04%; Fig. 8; $t_{995} = 21.0$, $p < 0.0001$). Biological features were functionally reduced the greatest in lobster management zones F, E, NH, and A, highest to lowest respectively (Fig. 8; $F_{8,557} = 14.7$, $p < 0.0001$). Similarly, geological features were functionally reduced the greatest in lobster management zones F, E, and NH (Fig. 8; $F_{8,557} = 12.7$, $p < 0.0001$).

DISCUSSION

The Magnuson-Stevens Fishery Conservation and Management Act established that essential fish habitats be protected, to the extent practicable, from fishing-related impacts. The Gulf of Maine has supported some of the most iconic and valuable fisheries in North America. One such example is the American lobster fishery, which is currently the most valuable single-species fishery in the United States (MDMR 2019a, NMFS 2019) with some nearly three million traps fished annually (MDMR 2019b). To evaluate the impact of the GoM lobster fishery on benthic EFH, we used a quantitative framework that relates habitat-specific impacts from fishing gear on EFH, the Swept Area Seabed Impact model (NEFMC 2011; Grabowski et al. 2014).

All fishing gear can impact habitat; however, our results demonstrate that EFH features impacted by the GoM lobster fishery are capable of near full recovery. The capacity of EFH to recover is linked to variation in substrate-specific susceptibility and recovery scores in the SASI model. Biological EFH features are impacted to a greater extent due to their higher susceptibility and longer recovery times for the more abundant substrate types within the GoM (Table 1; Fig. 3; NEFMC 2011; Grabowski et al. 2014). Longer recovery times in deeper habitats (Fig. S2), increase the areas offshore in the process of recovery (Fig. 8; Fig. S5). However, the relatively large area fished dilutes the functional reduction of EFH annually and reduces the probability of multiple gear deployments on the same EFH features. The low probability of repeated gear interactions and low susceptibility of EFH to damage by lobster traps (NEFMC 2011; Grabowski et al. 2014; Grieve et al. 2014) result in > 99.9% functional recovery of EFH features (Fig. 6).

Our estimates of fishing effort by the GoM lobster fishery conform well to other gear abundance metrics. Our maximum traps per endline scenario yielded an estimated 2.77 million traps fished, a value comparable to the estimates of traps fished by Auster and Langton (1998) and the average number of trap tags sold by the MDMR from 1998-2018, 2.94 million tags (MDMR 2019b). Since it is difficult to determine what percentage of trap tags sold are fished, we cannot reliably determine the accuracy of our estimates to the actual number of traps fished. Nevertheless, our maximum trap per endline scenario produced estimates comparable to the only verifiable metric of lobster traps in the GoM, which also presents an upper limit to the possible number of traps fished.

Ongoing changes in the distribution of fishing effort may cause varying degrees of benthic disturbance that shift over time and affect the total amount of EFH that is damaged. For example, lobster fishing effort has tracked abundance shifts northeastward (Steneck and Wilson 2001; Kleisner et al. 2017), attributable to rapid warming (Pershing et al. 2015; Friedland et al. 2020) and increased habitat suitability (Tanaka and Chen 2016; Le Bris et al. 2018; Goode et al. 2019; Mazur et al. 2020). Additionally, limited entry to the GoM lobster fishery has shifted the age of license holders to 50-65 years old (Stoll et al. 2016; Stoll 2017), few younger potential lobster fishers are replacing those that exit the fishery (Fig. S6), and state requirements for number of licenses sold per existing licenses retired (MDMR 2020) are reducing the number of fishers and total fishing effort (MDMR 2019b). Declines in young-of-year lobster suggest uncertainty in future landings (Le Bris et al. 2018; Oppenheim et al. 2019), potentially affecting the size and distribution of the GoM lobster fishery. As this fishery continues to adapt to management regulation of the North Atlantic right whale (e.g., Record et al. 2019), a likely result will be an increase in the minimum required number of traps per endline. Such a change would increase the amount of groundline that interacts with benthic EFH in part because of the greater distance over which longer trap trawls are dragged (Schweitzer et al. 2018; Stevens 2020), and would change the areas in which these gear configurations can be fished. As this fishery continues to change, so too will the areas impacted by fishing gear and their potential to recover. Moreover, our application of the SASI model is flexible and could be used to evaluate how various management alternatives affect fishing effort, behavior, and potential impact to EFH over longer time scales.

The limited availability of high spatial and temporal resolution endline estimates of GoM lobster fishing effort is perhaps the largest information bottleneck for more accurate future projections of habitat impact (Boenish and Chen, 2018). For example, the relatively coarse spatial resolution (10x10 arcmin) of the NMFS Vertical Line Model limits the extent to which we can identify localized areas of persistent fishing disturbance. Although we can effectively compare effort across the GoM, smaller bedforms targeted by fishing cannot be distinguished. Inability to characterize targeted fishing effort may under-estimate impact and over-estimate recovery potential on heavily impacted bedforms which are predicted to have lower biodiversity (Sousa 1979; Sousa 1984) and lower resilience to community shifts (Holling 1973). Less targeted bedforms, however, may have reduced impacts and have a higher probability of fully recovering, possibly balancing out total estimates of impact. Implementation of common monitoring practices (e.g., 100% vessel trip reporting or vessel monitoring systems) would provide comprehensive, real-time estimates of fishing effort needed to address these issues.

Recovery time strongly influences the accumulation of damage and the recovery potential of EFH, highlighting the importance of accurate evaluation of regional, substrate, and fishing gear specific rates of recovery. Commensurate with other studies, the SASI model revealed that EFH features are more susceptible to, and take longer to recover from, mobile fishing gears (e.g. Auster and Langdon 1998; Kaiser et al. 2000). We demonstrated that substrates with even moderate recovery times (~5 years) can take over 10 years to reach a balance between new impacts and recovery (Fig. 7). Thus, even slight increases in recovery time can dramatically increase the area of recovering EFH. For example, sensitive deep-sea corals and sponges can take an estimated 20 and 30 years, respectively, to recover population biomass following damage by trawling (Rooper et al. 2011). The SASI model's ability to incorporate EFH features with multi-decade recovery rates may potentially be limited by the availability of long-term recovery studies (Grabowski et al. 2014). However, we contend that the recovery rates used by the SASI model are well informed based on the relative species composition within our study domain. Sensitive, long recovering deep-sea corals occur well outside Federal Lobster Management Area 1, and the relative abundance of other sensitive fauna (e.g. corals and sponges) is relatively low (Fig. S7; McHenry et al. 2017). The lower abundance and comparatively faster recovery (e.g. Henry et al. 2003) of these faunae dampen their impact on overall recovery rates and the sensitivity of our results to varying recovery times.

While SASI is a useful starting point to start quantitatively characterizing the complex interactions between habitat and fishing gear, there are assumptions and generalizations that can introduce limitations. The SASI model treats repeated gear encounters on EFH as independent events that do not increase the magnitude of impact on EFH (NEFMC 2011; Grabowski et al. 2014). This assumption does not account for how initial disturbances could be more or less impactful than subsequent impacts (e.g., Hall-Spencer and Moore 2000), how added damage to EFH may further decrease biodiversity (e.g., Sousa 1984), or how persistent disturbances may shift benthic assemblages to new stable states that provide less functional benefit to fish (Lewontin 1969). The SASI model’s simplification of substrate complexity and biotic assemblages underestimates the variability of the benthos and the distribution of EFH. Simplification of substrate classification (Fig. 3) results from the fact that *in situ* studies rarely investigate fishing gear-habitat interactions at the granularity of the USGS classification system. Additionally, substrate characteristics alone can only partially explain benthic biodiversity of the GoM (McHenry et al. 2017), despite the lower biodiversity compared to similar large marine ecosystems (Witman et al. 2004). While incorporation of other potential abiotic drivers (e.g.

temperature, salinity, and current structure; McHenry et al. 2017) could increase the realism and accuracy of the SASI model, the current assumptions are key to parameterizing the SASI model in a complex marine environment.

Indirect anthropogenic factors also alter the benthos, and determining their relative impact is important when assessing the health of EFH. The macrobenthos community has changed multiple times due to fishing impacts on keystone species and will continue to shift as the ocean experiences climate change (Harris and Tyrrell 2001). For example, overfishing of demersal groundfish relaxed top-down pressures on urchin populations which contributed to a benthic regime shift from macroalgal communities to urchin barrens (Steneck et al. 2002). Not long thereafter, targeted fishing on urchins resulted in a shift back towards macroalgal communities comprised of more invasive alga and devoid of large-bodied fish predators (Steneck et al. 2002). Each of these trophic cascades altered benthic communities without direct physical manipulation and made a lasting impact thought to have helped bolster the GoM lobster fishery (Steneck and Wahle 2013). Additionally, climate change continues to impact the health and distribution of native and non-native species. Thermally mediated range expansion of invasive and novel species (e.g. European green crab, Tepolt and Somero 2014; Asian shore crab, Stephenson et al. 2009; black seabass, McMahan et al. 2019) is facilitating displacement of native fauna via competition or direct predation (e.g. Race 1982; Brenchley and Carlton 1983; Eastwood et al. 2007). Ocean acidification exacerbates thermal stress, and acts to decrease native fauna resilience to change and disease (Lesser 2016; Harrington et al. 2020). So, while evaluating direct impact by fisheries on EFH is important, we contend that fishery impact must be contextualized in the scope of many processes affecting the environment.

While we have done our best to simulate lobster fishing effects on benthic habitats, additional work needs to be conducted to evaluate impacts on EFH more holistically over the coastal New England Shelf. We have, to the best of our ability, estimated the impact of lobster fishing on benthic EFH in Federal Lobster Management Area 1. This area, despite being the primary source of US lobster landings (NMFS 2019), is only a fraction of American lobster fishery, and separate analyses would be necessary for other management areas. While our study agrees with previous efforts that fixed gears have relatively low impacts, EFH is also impacted by several other, more invasive, fisheries (NEFMC 2011; Grabowski et al. 2014). Thus, a cumulative model including all fisheries an important next step. Fishery co-occurrence would almost certainly change the area of impact and recovery potential of EFH (e.g. NEFMC 2011). However, the presence of fixed gears may act to preclude more damaging fishing activities (e.g. trawling) and protect regions with sensitive EFH (Kaiser et al. 2000). Conversely, fishers typically avoid trawling in highly structured, more vulnerable habitats like cobble and boulder bottom due to the risk of gear hang-ups that result in damage and loss of gear, whereas fixed gears can be deployed across a wider range of bottom types. Understanding these multi-fishery dynamics and relative impacts would provide a valuable tool to assess the vulnerability and status of EFH. Lastly, while models play an important role in fisheries science, perhaps one of their most important functions is to identify information gaps. Application of the SASI model to the GoM lobster fishery has highlighted several needs in order to better characterize fishing gear impacts, Our specific recommendations for improved model estimations include (1) higher resolution fishing effort data, (2) ability to evaluate targeted bedforms, (3) the impact of multiple gear disturbances on the same patch of bottom, (4) better characterization of benthic assemblages and their association with abiotic factors (e.g. depth, substrate, salinity), and (5) the sensitivity of our results to highly susceptible, long recovering species such as deep sea corals and emergent sponges.

A relatively unknown aspect of North America's largest fishery, the GoM lobster fishery, is to what extent does fishing effort impact the EFH of the GoM (see NEFMC 2011). We employed a quantitative framework that relates habitat-specific impacts from fishing gear on EFH, the Swept Area Seabed Impact model, to estimate the area of EFH functionally reduced by the GoM lobster fishery. Trap estimates were generated using current management regulations and the most comprehensive estimates of lobster fishing effort in the GoM. Annual estimates of functionally reduced EFH average less than 2% of the total available area while the accumulation of functionally reduced EFH over time average less than 3%. We found that between 99.92–99.96% of annually functionally reduced EFH features can fully recover despite 13% of some areas being in the process of recovery. Our results suggest that the GoM lobster fishery has minimal impacts to EFH features of the GoM. Our analysis was ultimately limited by the granularity of the input data with respect to benthic geological and biological habitat type and the spatial and temporal variability in effort. Nevertheless, we present a baseline impact analysis and a flexible approach that will undoubtedly be improved as new information becomes available. Our model also provides a valuable resource for management strategy evaluation. The flexibility of our application of the SASI model has the potential to compare the impacts of various lobster fishery management scenarios and evaluate the lasting impacts to EFH.

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Table 1. Sediment class identification and corresponding susceptibility and recovery values using the SASI model. Values are mean and standard error.

Table 2. Model assumption of zones in the Gulf of Maine.

Figure 1. Geographic boundaries of the Gulf of Maine used in our application of the SASI model. Letters denote lobster management zones. Map was created in Matlab® using M_Map base layers. Boundary data were sourced from the NOAA Data Discovery Portal (NOAA 2019).

Figure 2. Visual representation of variables used to determine the area swept clear by lobster trap trawls. The grey area is the total area swept clear.

Figure 3. Sediment classification of the Gulf of Maine. **A:** Sediment classification using the USGS East Coast Sediment Texture Database. **B:** Sediment classification using the SASI model. **C:** Zonal distribution of rocky substrate cover. Values are mean and standard error. Letters denote statistical similarity. **D:** Normalized substrate variance per 5x5 arcmin area between substrate classification schemes. Values are mean and standard error. 1 = homogenous, 0 = heterogeneous. **** denotes a significant difference at the $p < 0.0001$ level. Map was created in Matlab® using M_Map base layers. Sediment data were sourced from USGS (McMullen et al. 2014).

Figure 4. Estimated traps fished in the Gulf of Maine. Black circles are the maximum trap per endline scenario. Grey squares are the minimum trap per endline scenario.

Figure 5. Annual area (%) of functionally reduced EFH in the Gulf of Maine. Inset: Zonal distribution of annual EFH functional reduction. Values are mean and standard error. Letters denote statistical similarity. Map was created in Matlab® using M_Map base layers.

Figure 6. Recovery potential of initially functionally reduced EFH. The area (m^2 and %) that remains functionally reduced per 5x5 arcmin area via continued gear interaction. Values are mean and standard error. **** denotes a significant difference at the $p < 0.0001$ level.

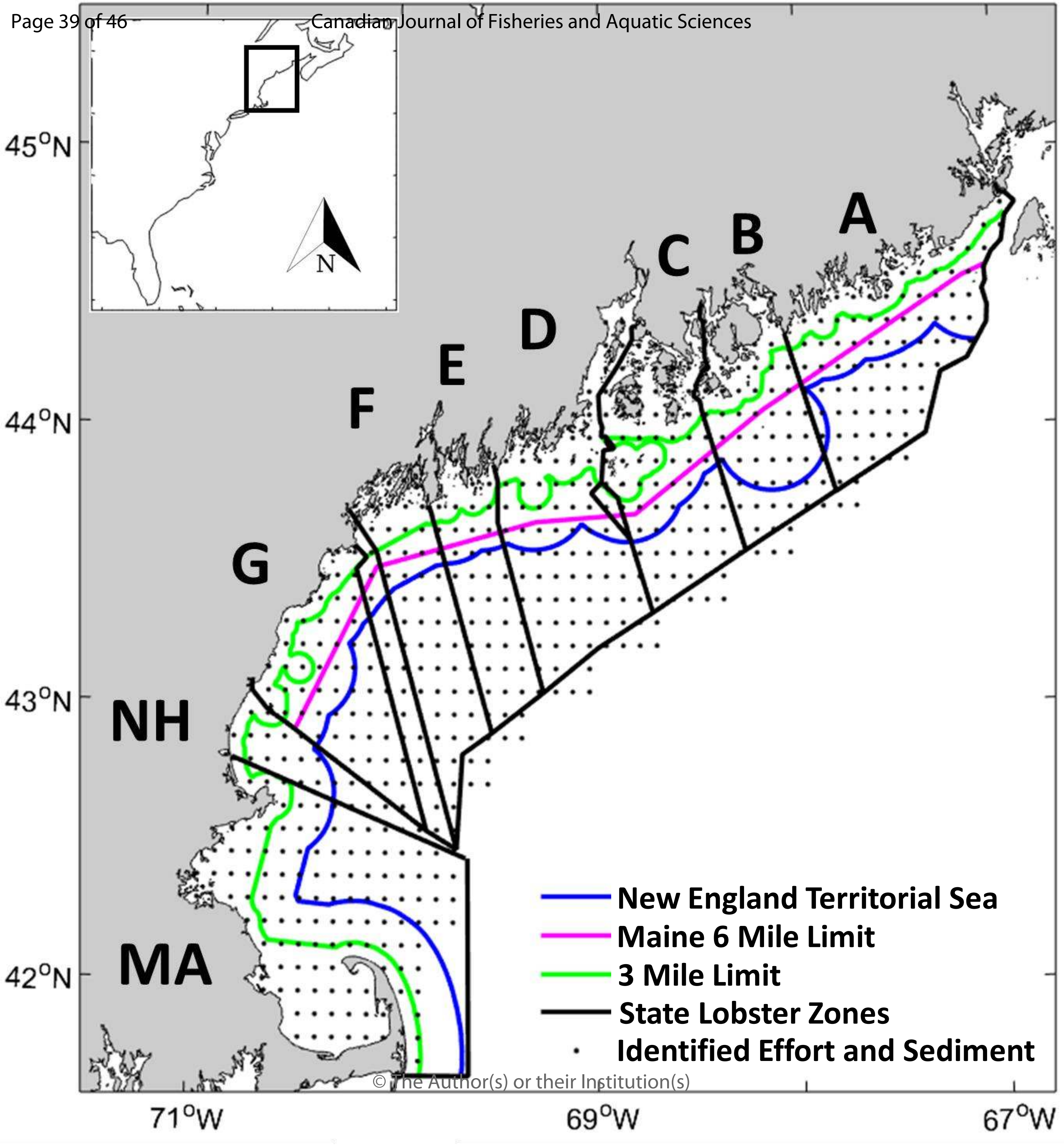
Figure 7. Example of accumulation of EFH area (%) functionally reduced on substrates with varying recovery times.

Figure 8. Cumulative area (%) of functionally reduced EFH. Inset: Zonal distribution of cumulative EFH functional reduction. Values are mean and standard error. Letters denote statistical similarity. Map was created in Matlab® using M_Map base layers.

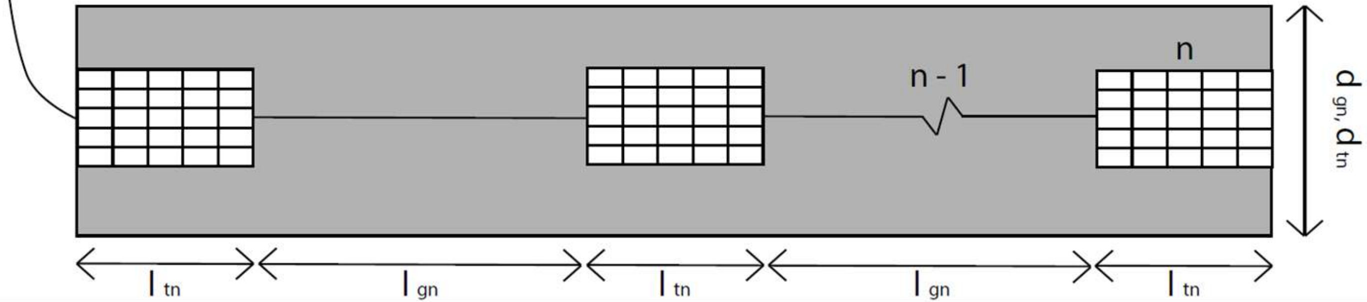
		Biological Features		Geological Features	
U.S.G.S. Sediment Classification	**SASI Sediment Classification**	**Susceptibility (% reduced)**	**Recovery Time (years)**	**Susceptibility (% reduced)**	**Recovery Time (years)**
Bedrock	Boulder	16.8 ± 0.6	2.0 ± 0.4	5.3 ± 0.6	2.7 ± 2.3
Boulders	Boulder				
Gravel	Cobble	16.1 ± 1.5	1.9 ± 0.3	10.2 ± 4.7	2.4 ± 2.0
Gravelly Sediment	Granule-Pebble	16.1 ± 1.3	2.0 ± 0.3	5.3 ± 0.6	0.5 ± 0.1
Sand	Sand	13.0 ± 2.5	1.6 ± 0.4	13.7 ± 2.8	0.5 ± 0.1
Silty Sand	Sand				
Clayey Sand	Sand				
Sand/Silt/Clay	Sand				
Sandy Silt	Mud	13.3 ± 2.8	1.4 ± 0.6	17.5 ± 0.6	0.5 ± 0.1
Silt	Mud				
Clayey Silt	Mud				
Sandy Clay	Mud				
Silty Clay	Mud				
Clay	Mud				

Zone	Season	Days Between Hauls (Inseason)	Days Between Hauls (Offseason)	Min Traps Per Line Scenario				Max Traps Per Line Scenario			
				0-3 nm	3-6 nm	6-12 nm	>12 nm	0-3 nm	3-6 nm	6-12 nm	>12 nm
A	Apr-Dec	3	7	1	3	(5)	(7.5)	3	(5)	(10)	(20)
B	May-Oct	3	10.5	1	3	(5)	(7.5)	3	(5)	(10)	(20)
C	Mar-Dec	4	10.5	1	3	(5)	(7.5)	3	(5)	(10)	(20)
D	Mar-Dec	4	12	1	3	(5)	(7.5)	3	(5)	(10)	(20)
E	Apr-Dec	3	9.5	1	3	(5)	(7.5)	3	(5)	(10)	(20)
F	Apr-Dec	2.5	7	1	3	(5)	(7.5)	3	(5)	(10)	(20)
G	Apr-Nov	4.5	8.5	1	3	(5)	(7.5)	3	(5)	(10)	(20)
C/D Overlap	Mar-Dec	4	11.25	NA	3	(5)	NA	NA	(5)	(10)	NA
F/G Overlap	Apr-Dec	3.5	7.75	1	3	(5)	(7.5)	3	(5)	(10)	(20)
NH	Apr-Jan	3.4	9	1	(5)	(5)	(7.5)	3	(10)	(10)	(20)
MA	Apr-Dec	3	7	1	(5)	(5)	(7.5)	3	(10)	(10)	(20)
Outside	Apr-Nov	3.4	9	NA	NA	NA	(7.5)	NA	NA	NA	(20)

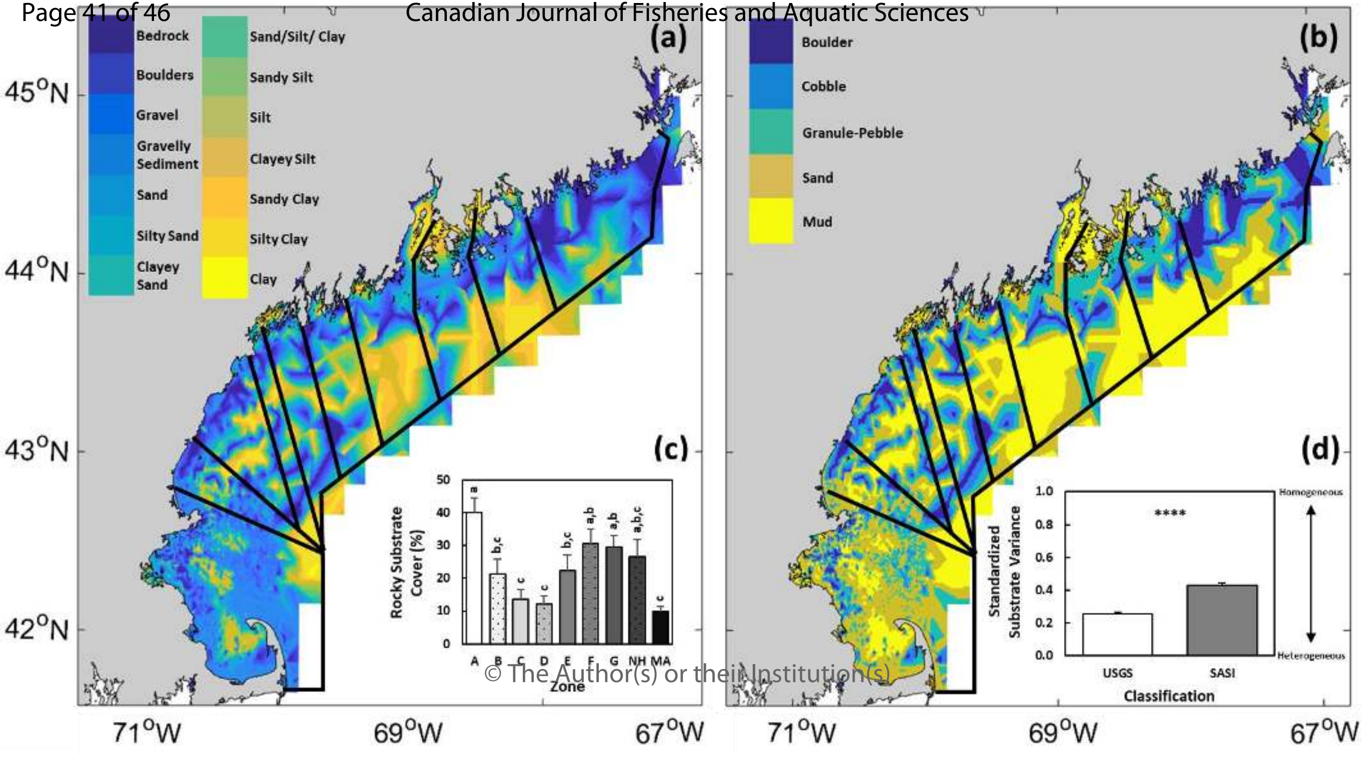
Note: Bracketed values assume two endlines per trawl.

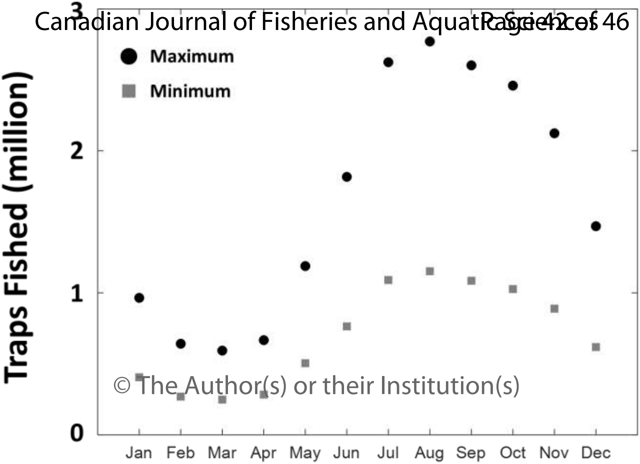

N
A
B
C
D
E
F
G
NH
MA
45°N
44°N
43°N
42°N
71°W
69°W
67°W
New England Territorial Sea
Maine 6 Mile Limit
3 Mile Limit
State Lobster Zones
Identified Effort and Sediment

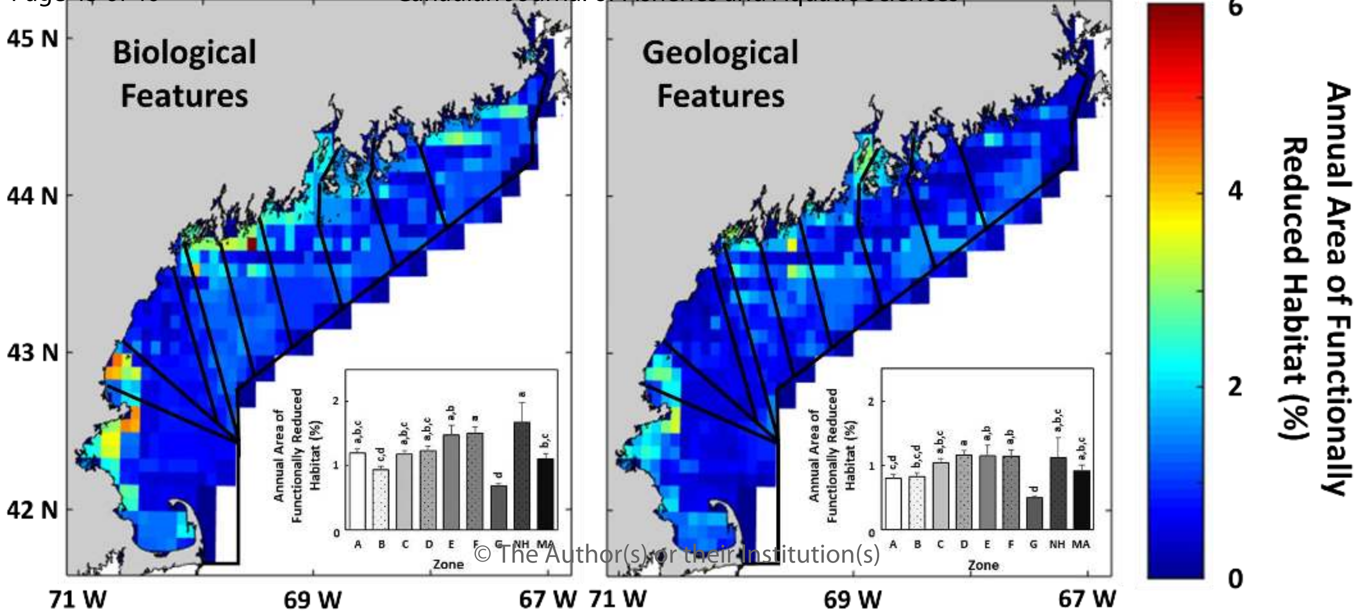
$$A_{trap} = (n \cdot d_{tn} \cdot l_{tn}) + ([n-1] \cdot d_{gn} \cdot l_{gn})$$

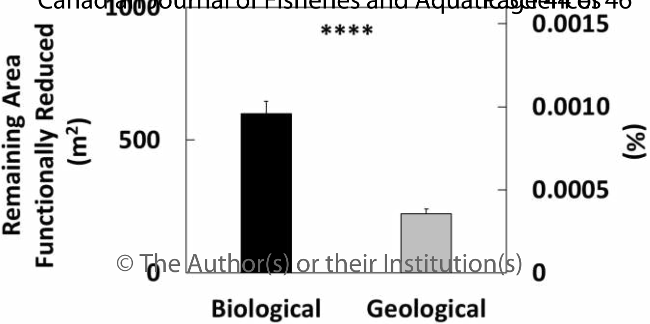


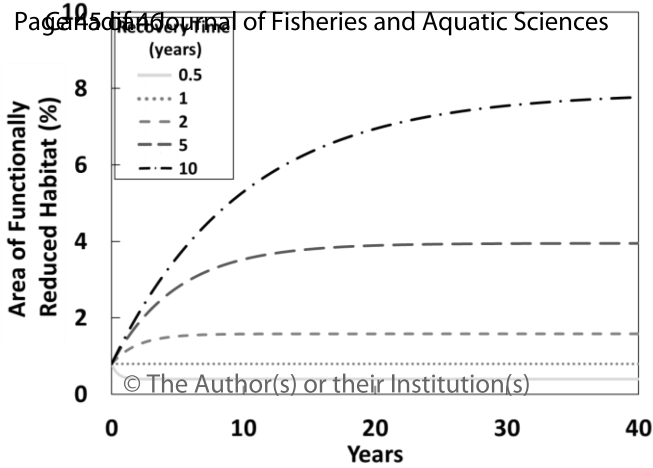
Model Variable	Description
A_{trap}	Area Swept Clear
n	Number of Traps
n - 1	Number of Groundlines
d_{tn}	Lateral Drag Distance of a Trap (m)
d_{gn}	Lateral Drag Distance of a Groundline (m)
l_{tn}	Trap Length (m)
l_{gn}	Groundline Length (m)

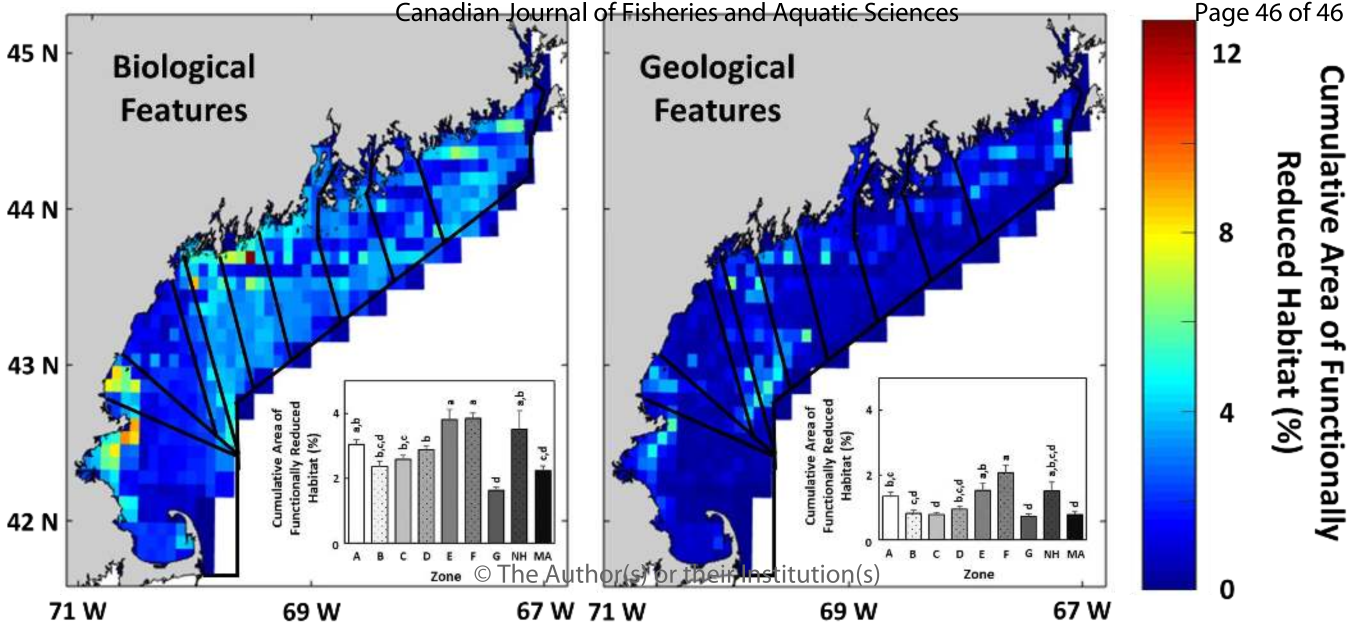

(a)
Bedrock
Boulders
Gravel
Gravelly Sediment
Sand
Silty Sand
Clayey Sand
Sand/Silt/ Clay
Sandy Silt
Silt
Clayey Silt
Sandy Clay
Silty Clay
Clay
(b)
Boulder
Cobble
Granule-Pebble
Sand
Mud
(c)
Rocky Substrate Cover (%)
50
40
30
20
10
0
a
b,c
c
c
b,c
a,b
a,b
a,b,c
c
A
B
C
D
E
F
G
NH
MA
Zone
(d)
Standardized Substrate Variance
1.0
0.8
0.6
0.4
0.2
0.0
****
Homogeneous
Heterogeneous
USGS
SASI
Classification
45°N
44°N
43°N
42°N
71°W
69°W
67°W


Traps Fished (million)
0
1
2
3
Maximum
Minimum
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec


Biological Features
Geological Features
45 N
44 N
43 N
42 N
71 W
69 W
67 W
Annual Area of Functionally Reduced Habitat (%)
0
2
4
6
Annual Area of Functionally Reduced Habitat (%)
Zone
A
B
C
D
E
F
G
NH
MA
a,b,c
c,d
a,b,c
a,b,c
a,b
a
d
a
b,c
c,d
b,c,d
a,b,c
a
a,b
a,b
d
a,b,c
a,b,c
0
1
2


Remaining Area
Functionally Reduced
(m²)
1000
500
0
****
0.0015
0.0010
0.0005
0
(%)
Biological
Geological


Recovery Time (years)
0.5
1
2
5
10
Area of Functionally Reduced Habitat (%)
10
8
6
4
2
0
0
10
20
30
40
Years


Biological Features
Geological Features
45 N
44 N
43 N
42 N
71 W
69 W
67 W
71 W
69 W
67 W
Cumulative Area of Functionally Reduced Habitat (%)
12
8
4
0
Cumulative Area of Functionally Reduced Habitat (%)
Zone
A
B
C
D
E
F
G
NH
MA
0
2
4
a,b
b,c,d
b,c
b
a
a
d
a,b
c,d
Cumulative Area of Functionally Reduced Habitat (%)
Zone
A
B
C
D
E
F
G
NH
MA
0
2
4
b,c
c,d
d
b,c,d
a,b
a
d
a,b,c,d
d