Original Article

Impacts of a multi-trap line on benthic habitat containing emergent epifauna within the Mid-Atlantic Bight

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Alteration and degradation of benthic structure by fishing gear can impede efforts to manage fish stock sustainably. Although the impacts of mobile gear are well known, effects of passive gear (e.g. fish traps) upon structure have been little studied. We modified commercial traps for American lobster *Homarus americanus* and black sea bass *Centropristis striata* by attaching GoPro® cameras to ascertain the degree and nature of impacts to seafloor habitats. Customized traps were included within a line of 20 traps, deployed and retrieved according to standard commercial fishing practice. Less than 5% of traps landed directly on bedforms when deployed. However, during retrieval traps dragged along the ocean floor, increasing trap–habitat contact rate to 50%, and causing traps to collide with corals, bryozoans, and other epifauna. Drag time of traps depended on the position in the trap line. Experimentally extending the trap line reduced drag time during retrieval for traps near the distal end of the line. Our results show that impacts of commercial trap fishing can be substantial during trap retrieval, and that the impact depends on their location on a trap line. Fishing practices should be developed that minimize effects of trap retrieval on structural benthic habitat.

Keywords: benthos, corals, fishing, habitats, impacts, traps

Introduction

Hard-bottom habitats consisting of sponges, corals, and bryozoans increase structural complexity and provide critical habitat for juvenile and adult fish and invertebrates, many of which have high economic value (Lough *et al.*, 1989; Auster *et al.*, 1997; Kaiser *et al.*, 1999; Thrush *et al.*, 2002; Hixon and Jones, 2005; Scharf *et al.*, 2006; Gregor and Anderson, 2016). Complex habitats also enhance community structure and species richness (Kaiser *et al.*, 1999).

The disruption and alteration of highly complex bedforms containing biogenic epifauna could impair stock recovery and recruitment of exploited species (van der Knaap, 1993; Auster and Langton, 1999; Mangel, 2000; Peterson *et al.*, 2000; Lotze *et al.*, 2006; Scharf *et al.*, 2006; Rogers *et al.*, 2014). One of the greatest sources of disturbance to benthic habitat is fishing and the effects

of mobile gear, such as trawls and dredges (Watling and Norse, 1998; Freese *et al.*, 1999; Norse and Watling, 1999; Rumohr and Kujawski, 2000; Lindholm *et al.*, 2008; Buhl-Mortensen *et al.*, 2016; Pitcher *et al.*, 2016; Rijnsdorp *et al.*, 2016). Repeated use of mobile fishing gear can degrade habitat complexity by: (1) directly removing or damaging epifauna; (2) smoothing sedimentary bedforms and reducing rugosity; and (3) eliminating taxa that produce structure where fish aggregate (Auster and Langton, 1999). Disturbance to benthic habitat as a result of fishing with passive gear, such as traps, can create impacts similar in scope to that of mobile gear (Auster and Langton, 1999). Despite the widespread use of passive fishing gear, the impacts of fish traps on live-bottom habitat have not been studied well and have become a growing concern (Auster and Langton, 1999).

Most published research regarding trap impacts on benthic habitat focus on a single trap on a single buoy line (van der Knaap, 1993; Auster and Langton, 1999; Marshak *et al.*, 2008; Jenkins and Garrison, 2013; Stephenson *et al.*, 2017). These studies have largely been conducted within coral reef sites in shallow clear water. Eno *et al.* (2001) reported minimal impacts on sea pens by commercial crab traps consisting of a three-trap line in UK waters. In their study camera frames designed to mimic traps that were pulled by divers to simulate the retrieval process. Stephenson *et al.* (2017) reported similar findings and reported minimal disturbance of a single trap to rock reef habitats in UK waters. The authors held the assumption that a single trap is representative of a 10-trap line. In waters where whales migrate, the number of buoy lines that fishermen can deploy are restricted, requiring the use of trap lines consisting of 20 or more traps rather than a single trap (NOAA, 2015a, b). Unfortunately, there are currently no tests of the supposition that impacts caused by a single trap are representative of multi-trap lines.

The degree to which trap fishing can impact the benthos is dependent on the composition and physical structure of the benthos (Grabowski *et al.* 2014). There is little published research on benthic habitat within the Mid-Atlantic Bight (MAB) and there are no published data on the current condition of benthic habitats within this region. Recent surveys of benthic topography and communities have been conducted within wind energy development areas in the MAB, but not in areas with significant structural complexity that are commonly targeted by commercial fisheries (V. Guida, pers. comm.). Structure within the MAB is primarily comprised of highly fragmented natural bottom (i.e. rock-like clay bedforms and rocks; Steimle and Zetlin, 2000) and artificial reefs (e.g. shipwrecks, concrete pipes, cable cars). It is currently unknown how traps affect benthic communities and habitat comprised of rock-like bedforms, which are more susceptible to degradation than rock.

Previous studies to evaluate trap impacts on the benthos commonly relied on diver observations (Eno *et al.*, 2001; Marshak *et al.*, 2008; Stephenson *et al.*, 2017). However, in deeper, more turbid waters with limited visibility can make studying trap impacts more difficult for divers, so alternative methods are required. Video survey tools such as remotely operated vehicles (ROVs) and remote camera systems can be effective for observing habitats at depths greater than 30 m. These tools can collect important information regarding habitat characteristics, such as abundance, degree of fragmentation, colonizing organisms, and relative health or signs of degradation (Johnson *et al.*, 2003; Parry *et al.*, 2003). However, the utilization of ROVs can cost $1000–3500 per day and often require additional personnel who are experienced in controlling the ROV in turbid waters to find traps and observe any interactions with hard bottom habitat during the deployment and retrieval process *in situ*. Therefore, it is important to develop more reliable, cost-effective remote camera systems to allow *in situ* observations of impacts due to trap fishing.

The objectives of this research were: to determine the degree and nature of impacts to complex live-bottom habitats caused by commercial trap fishing utilizing a cost-effective method to make *in situ* observations of the deployment and retrieval of commercial multi-trap lines; to develop methods to quantify such interactions; to test alternative fishing practices designed to reduce adverse impacts; and, to estimate the current conditions of the fishing sites based on quantifying damage to gorgonian sea whips (*Leptogorgia* spp.) as indicator species.

Methods

Study sites and environmental data

Three study sites (1, 2, and 3) in the MAB were selected based on commercial fishing activity in 2015. Sites were located approximately 19–33 km off the coasts of Delaware, Maryland, and Virginia, ranging in area from 0.80 to 1.61 km^2 between the latitudes of 37° N and 38.5° N, at depths of 20 to 33 m (Figure 1).

Environmental data (i.e. mean wave height and period) were collected to determine if sea conditions and depth affected drag duration. Sea conditions were retrieved from NDBC buoy 44009, located off the Delaware Bay mouth at 38.461° N, 74.703° W (http://www.ndbc.noaa.gov/station_page.php?station=44009). Depth was recorded by sonar.

Remotely operated vehicle

A *Seabotix LBV 150* ROV was used to observe characteristics and conditions of complex live-bottom habitat within three commercially fished sites. Attempts use the ROV to view effects of trap deployment and retrieval on benthic habitat were prevented by sea conditions and current. Therefore, video surveys were conducted in a haphazard manner to assess habitat composition and condition, and to determine if there were any differences between sites. Video surveys could not be conducted randomly because the extent of each fishing area was unknown.

To assess habitat composition from the 15 min video surveys, 20 randomly selected frame images from each site were analysed for presence or absence of sea whips *Leptogorgia* spp., northern stone coral *Astrangia poculata*, sponge *Cliona* spp., hydroids *Bugula* spp., cobble, bedforms, and sand. A sample size of 20 frames was selected to maintain independence between frames and because it was on the asymptote of the rarefaction curve.

To obtain a baseline for habitat condition, sea whips were selected as a potential indicator species because gorgonians have a wide geographic range. Sea whips occur along the entire Atlantic coast of North America, and are abundant in heavily fished habitats (Gotelli, 1991; Steimle and Zetlin, 2000; Sánchez, 2007). Within the MAB, sea whips alone provide additional height to substrate and damage can be easily quantified. To assess their reliability as an indicator species, we analysed abundance and community composition data for the three fishing sites.

Sea whip images were analysed in ImageJ (version 1.6.0_65, NIH) to determine the proportion of damage of sea whip structure. To calculate the extent of damage or overgrowth, regions of interest were created around each sea whip in the selected images and the total area, and area of damage or epifaunal colonization was measured for each sea whip. The sites were given a damage index (DI) on a scale from 1 to 5 (Table 1) based on the mean proportional damage of sea whips at each site.

Commercial trap-line design, hauling, and deployment

The standard commercial trap line used in the MAB contains 20 traps and is approximately 384 m in length. Each end of the line is connected to two rectangular block concrete anchors (~34 kg) that are marked by buoys (Figure 2). Commercial traps (1.1 m L × 0.53 m W × 0.3 m H; 8.2 kg) used to capture black sea bass *Centropristis striata* and American lobster *Homarus americanus* are constructed by the fishermen with galvanized wire mesh connected to the trap line via clove hitch knots every 18 m via a 2.5 m "snood" line. Buoys are attached to the block anchors with floating line. The majority of the trap line consists of sinking line,
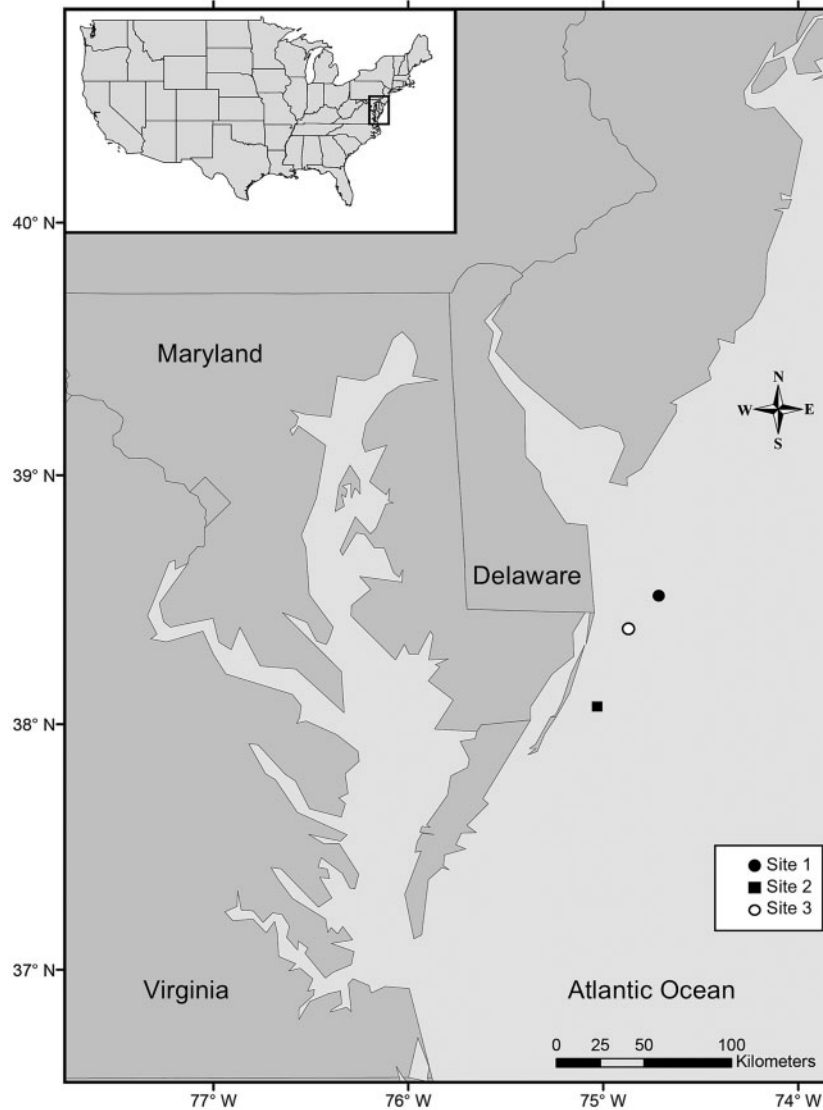
Figure 1. Locations of trap study sites off the coast of Delaware, Maryland, and Virginia in the MAB, United States. The map was created using ArcGIS 10.4.1 software.

Table 1. Criteria used to classify individual sea whip DI and habitat DI for images captured during the ROV survey.

DI	Damage	Description
1	Minimal	<0.05 damage
2	Minor	0.06–0.25 damage.
3	Moderate	0.26–0.50 damage.
4	Severe	0.51–0.75 damage.
5	Critical	>0.75 damage

Note: For individual sea whips, damage is defined as any visible tissue damage and fouling of epifauna.

except for the first and last nine m, which consists of floating line, to avoid abrasion. All rope lines are 9.5 mm diameter. Federal regulations require that the trap line and snood lines be comprised of sinking line (https://www.greateratlantic.fisheries.noaa. gov/protected/whaletrp/14alwtrpfrbulletin.pdf).

The hauling process begins with the retrieval of the buoy lines, which are led over a hydraulic pinch block that pulls the trap up at a variable rate. Simultaneously, the captain drives the boat, at various speeds, along the trap line moving toward the traps. When a trap is pulled on deck, the boat is temporarily idle while the trap contents are emptied. After the trap is emptied, the captain continues to haul the next trap in the same fashion while the deckhand prepares the empty trap for redeployment.

When traps are deployed, the buoy is deployed followed by the anchor block. Traps are set on the transom and are pulled into the water by the trap line, causing the trap line to become taut between traps. After the last trap is deployed, the second anchor block is dropped followed by the second buoy.

Camera trap design and preliminary data

One commercial trap was customized within a string of five traps to assess trap–habitat interactions. This trap line was similar to commercial rigs consisting of a trap line, with an anchor block and buoy at each end. Stabilizing weight (2.7 kg) and floats (PVC tubes) were added to the customized trap, neutralizing ~4 kg, to
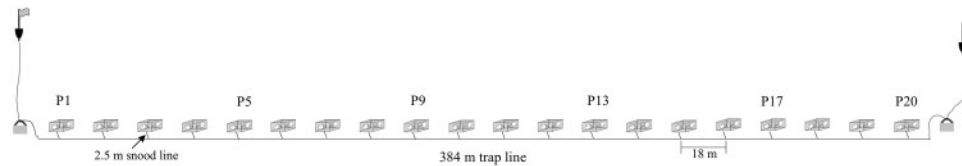
Figure 2. Diagram illustrating a common commercial trap line used off the coast of Delaware, Maryland, and Virginia. Marker buoys are tied to a rectangular concrete block anchor, which connects to a 384 m trap line containing 20 traps. Traps are connected to the trap line via a 2.5-m "snood" line every 18 m. Positions where camera traps were used are indicated by P1, P5, P9, P13, P17, and P20.
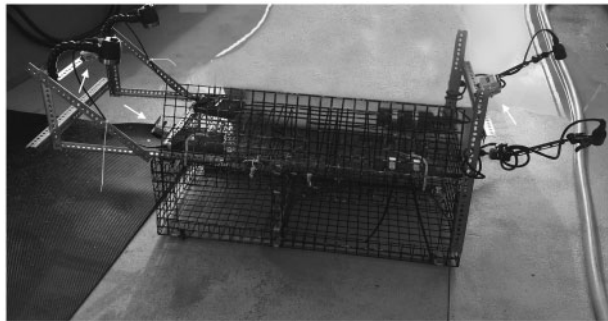

Figure 3. Customized trap consisting of a standard commercial fishing trap modified with dive lights and GoPro® *Hero* cameras. White arrows indicate placement of cameras.

Table 2. Summary of trap drops and mean drag time, by position and trap-line length.

Position	Trap-line Length	*n*	Drag duration (s)	
			Mean	**SE**
P1	Short	23	59.23	4.83
	Long	25	45.75	3.66
P5	Short	27	42.58	2.96
	Long	22	32.27	3.75
P9	Short	26	34.44	1.77
	Long	22	30.87	3.61
P13	Short	20	33.23	2.73
	Long	20	33.52	3.10
P17	Short	22	20.24	1.32
	Long	24	24.28	1.63
P20	Short	23	7.15	0.64
	Long	19	6.68	0.96

Note: Short = 384 m, long = 768 m; *n*, number of drops; SE, standard error.

help balance the additional weight of dive lights and to ensure that it landed upright. Three GoPro® *Hero* cameras were placed on the trap to give a view of the front base of the trap, the forward view, and rear view of the trap (Figure 3). First edition GoPro® cameras were updated and replaced with *Hero 3+* cameras because the latter did not require underwater lighting for visibility (mention of trade names does not imply endorsement by the University of Maryland Eastern Shore or NOAA). This allowed us to remove lights, weights, and floats as well, resulting in a less encumbered trap.

Preliminary sampling was conducted from June to September 2014 to determine how traps functioned during the retrieval process, and to experiment with camera setup. Video analysis of the retrieval process showed traps dragging along the ocean floor for varying durations. To determine if there was a relationship between trap position on the rig and drag duration, subsequent experiments utilized a trap string identical to that used by commercial fishermen.

Within a standard rig containing 20 traps, two commercial traps were replaced with our customized "camera traps". Customized traps were assigned a number based on their position on the trap line during deployment. Camera traps were rotated throughout the trap line to obtain recording from six positions along the trap line: P1, P5, P9, P13, P17, and P20. Position P1 was the first trap deployed whereas P20 was the last (20th) trap deployed; the remaining traps were the 5th, 9th, 13th, and 17th in the line. However, commercial fishermen deploy and retrieve trap lines in reverse order such that P1 was the last trap to be retrieved and P20 was the first.

The trap line was deployed and hauled three times before rotating our camera traps to the next positions. Starting positions for the two camera traps were randomly assigned each day. All positions were recorded during a sampling day. Traps were

hauled and deployed by commercial fishermen to ensure no changes were made to the standard fishing process.

To determine if there was a difference between drag durations between the first-generation camera traps (i.e. camera traps containing GoPro® *Hero*, dive lights, stabilization weights, and floats) and the second-generation traps (i.e. camera traps containing only GoPro® *Hero* 3+), drag durations of traps were compared within positions by *t*-test. No significant differences in drag time were detected (*t*-test, $p > 0.05$) between first- and second-generation traps; therefore, the data were pooled together.

Sampling took place in two series. The standard 384-m trap line was deployed within Series 1 from June to September 2015. Sampling Series 2 occurred from October to November 2015, during which the spacing between traps was doubled from 18 to 36 m, yielding a 768-m trap line. This was done to determine if increasing the distance between traps would reduce drag time.

Within Series 1, 148 drops of customized traps were completed, all of which were analysed for interactions with habitat and epifauna. Due to video complications or deployment problems (e.g. traps getting tangled), seven videos were not used for drag analysis. The final sample size analysed for Series 1 included 148 sets for habitat interactions and 141 for drag duration along the ocean floor (Table 2). During Series 2, 132 drops were completed successfully.

Trap impacts and dragging were assessed from the video recordings of the six positions. Since we had no accurate way of measuring the distance along which traps were being dragged on the bottom during retrieval, dragging was quantified using "drag time", defined as the time the trap was in motion while maintaining contact with the bottom during retrieval.

Video analysis consisted of three components. Initial review consisted of editing with Final Cut Pro 10.2.2©, and removing extended soak times and standby times on the boat. All footage of deployment and retrieval processes remained unedited. The second review consisted of assessing any interactions with complex live-bottom habitat or epifauna and defining the bottom type on which the trap landed. When a trap came into contact ("hit") with a patch of live-bottom habitat or epifauna, it was logged and counted as 1 hit. Hits were documented by trap position for the entire fishing process, including deployment and retrieval. A single trap could have multiple hits throughout the fishing process. The third review was used to determine drag time.

Statistical analysis

Data were analysed using R version 3.2.1 (R Core Team, 2016), GraphPad Prism version 6.0 and PRIMER v 6.1.16 (Clarke and Gorley, 2006). Prior to regression analysis, data were tested for normality using the Shapiro–Wilk test. Grubbs' test was used to detect and remove outliers within the drag time data (Grubbs, 1969). Drag times were normally distributed (Shapiro–Wilk test, $p = 0.27$), but three outliers were detected (in positions P1, P5, and P17) and removed from further analysis. Drag times were fitted with a weighted Poisson linear regression.

To test for depth and wave effects, depth and sea state were grouped as described and treated as factors by subdividing the data into two and three factor levels, respectively: shallow (<30.5 m) and deep (≥ 30.5 m); and low (<0.6 m), medium (0.6–1.0 m), and high (>1.0 m) seas. A multi-factor ANOVA and a linear mixed effects (LME) model were used to determine presence of depth or wave effects. Morisita's index of dispersion was used to analyse the distribution of hits throughout trap positions (Krebs, 1989). The effects of trap position and drag time on hit count, and all possible interactions, were analysed using a generalized linear model (GLM) using the Poisson family distribution. The five models included both position and drag time (g_1), position only (g_2), drag time only (g_3), position and drag time in a multiplicative model (g_4), and position, drag time, and depth (g_5).

Akaike Information Criterion (AIC) values from each model were used to calculate AIC_c, a second-order bias correction estimator. Δ_i values were used to rank the different models (g_i) against the model with the lowest AIC_c. Model probabilities (w_i) estimated the probability that a particular model g_i was the best model. Models with w_i less than 0.1 were eliminated. Goodness of fit of the selected model was examined using the Hosmer–Lemeshow goodness-of-fit test.

The relationship between drag time and position was compared between the standard commercial trap line (384 m) and the experimental extended trap line (768 m) using linear regression and analysis of covariance. Student's t-test was used to compare drag time between the standard and extended trap lines within each position, after determination of normality.

Sea whip proportional damage ratios were logit-transformed before being analysed with a one-way ANOVA. The mean and standard error across all three sites were calculated using a stratified design, weighted by sample size at each site. To determine if the sites observed by the ROV differed in community structure, habitat composition and richness were compared between sites using permutational multi-variate analysis of variance (PERMANOVA). *Post-hoc* comparisons were made between all three combinations of paired sites.

Results

Habitat composition and sea whip condition

Three fishing sites consisting of natural bottom were visited during the ROV video surveys, which allowed us to document differences in habitat composition and sea whip condition among sites.

Twenty-two sea whips were analysed from the fishing sites for proportional damage and sites assigned a DI as described previously (Table 1; Figure 4). There was no significant difference between the three commercially fished sites ($F = 0.38$; df = 18; $p = 0.68$). All three sites contained sea whips with various levels of degradation; the stratified mean value of proportional area damage was 0.371 ± 0.057 (SE) for all sites, which is indicative of moderate levels of degradation (Table 3).

Habitat composition differed significantly among the three sites (PERMANOVA, df = 2; pseudo-$F = 6.13$; $p = 0.001$) and between combinations of paired sites (Table 4). Site 1 consisted of more cobble, site 3 had more sandy bottom, and site 2 contained more sponge habitat (Figure 5). The three sites did not differ significantly in abundance of sea whips, hydroids, and stony corals (GLM, Poisson family distribution, $p > 0.3$ in all cases). Sea whips were common at all sites, having been observed in 35–55% of the 20 images per site. Similarly, hydroids occurred in 35–50% of images at each site, and stony corals in 30–50%. Consequently, sea whips were a reliable indicator species of habitat degradation across the three sites, and live habitat community structure did not differ significantly among sites except for abundance of sponges (GLM, Poisson family distribution, $p = 0.054$). Sponges were in 45% of the images at site 2, but in only 10% of the images at sites 1 and 2.

Trap–habitat interactions

Video analysis revealed that $<5\%$ (6/148) of the dropped commercial fishing traps landed on complex live-bottom habitat. However, 50% of traps (74/148) came into contact with live-bottom habitat during trap retrieval due to traps being dragged along the ocean floor. Interactions between traps and habitat, including epifauna, were observed from every recorded position. Interactions included damaging or breaking corals and running over epifauna such as sea stars, sponges, hydroids, bryozoans, and anemones (Figure 6).

During retrieval, traps were dragged along the ocean floor for various durations, such that there was a significant correlation between trap position and drag time. Drag time was longest for traps at position P1 (last trap retrieved) and declined monotonically through the trap line ($r^2 = 0.73$, $p < 0.0001$; Figure 7). For Series 1, depth and wave height did not contribute significantly to drag duration ($F = 0.67$; $F = 0.99$, respectively; df = 132; $p > 0.05$; Table 5).

A single trap could encounter habitat or epifauna more than once per drop and retrieval cycle (i.e. have multiple hits) if it dragged over multiple coral patches or other organisms. Morisita's index of dispersion (1.78) indicated that the distribution of hits was random ($X^2 = 196.6$, critical value = 227.3) and the data could therefore be represented by a Poisson distribution (Figure 8).

From the five models analysing trap and habitat interactions, g_3 was eliminated because the probability (w_i) was less than 0.05 and g_5 was eliminated because the addition of the depth variable increased the AIC_c by only 2.18 (Table 6). Among the remaining models, model g_4 was selected as the most reasonable, which was
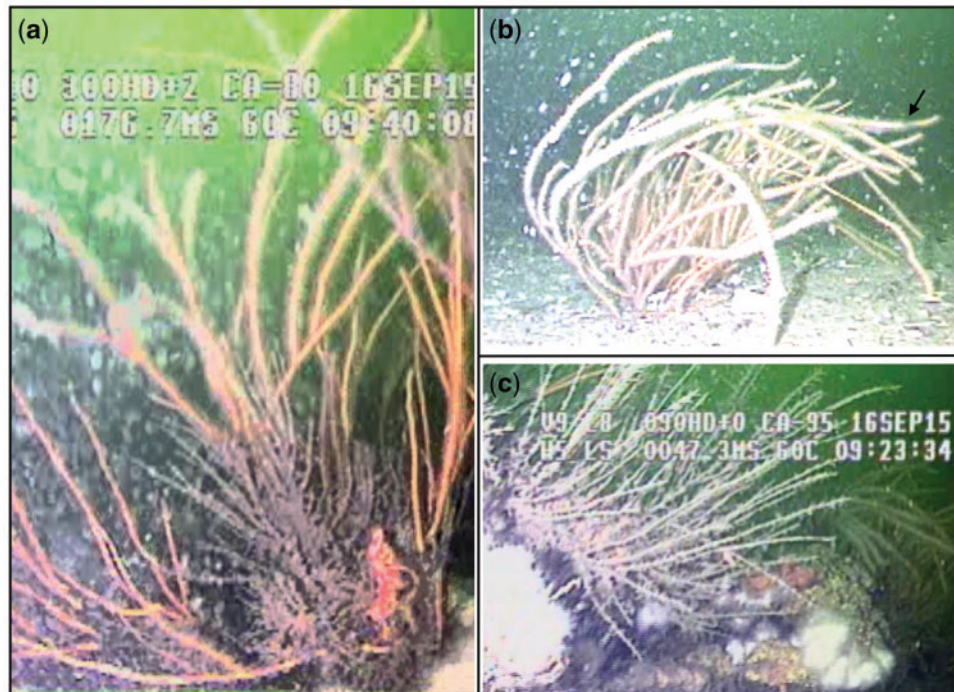
Figure 4. Three images from the ROV surveys taken from two different sites showing the spectrum of proportional damage. (a) Sea whip (site 2) showing the mean proportional damage of 0.40, which is representative of the mean proportional damage throughout the three sites. (b) A sea whip (site 1) exhibiting minimal damage ratio of 0.02 highlighted by the arrow. (c) A sea whip (site 2) exhibiting a damage ratio of 1.00.

Table 3. Summary of the mean sea whip proportional damage and habitat DI by site.

Site	n	Mean	SE	Min	Max	DI
1	5	0.31	0.12	0.02	0.63	3
2	11	0.40	0.07	0.15	1.00	3
3	6	0.37	0.13	0.09	0.86	3
Mean	22	0.371	0.057			3

Note: n, the number of sea whips analysed for proportional damage; mean, the mean proportional damage for the observed sea whips of that site; SE, standard error; min, smallest proportional damage observed; max, largest proportional damaged observed.

Table 4. Results of (PERMANOVA) pair-wise test comparing habitat composition between commercially fished sites 1, 2, and 3.

Sites	t Statistic	p value	Unique Permutations
2 vs. 3	2.19	<0.01	108
3 vs. 1	1.61	0.05	92
2 vs. 1	3.46	<0.01	78

Figure 5. Canonical analysis of principal coordinates (CAP) based on Euclidean distance from PERMANOVA analysis of habitat composition for three commercially fished sites: 1, 2, and 3. Composition analysis was based on presence or absence counts for sea whips (SW), northern stone coral (SC), sponge (SP), hydroids (HY), cobble (CB), bedforms (RK), and open bottom sand (SD). Site 3 showed higher presence of sand and sea whips. Site 1 showed a higher presence of cobble and bedforms. Site 2 showed higher presence of sponge and cobble.

significantly better than each of the other two models by the likelihood ratio X^2 test ($p = 0.02$), and passed the Hosmer–Lemeshow goodness of fit test ($X^2 = 16.32$, df = 8, $p > 0.05$). This suggests that there is an interaction effect between trap position and drag time, both of which had a significant effect on hit rate ($p < 0.001$ and $p < 0.05$, respectively). This interaction is illustrated by Figure 8; traps at P1 and P5 show a convex curve,
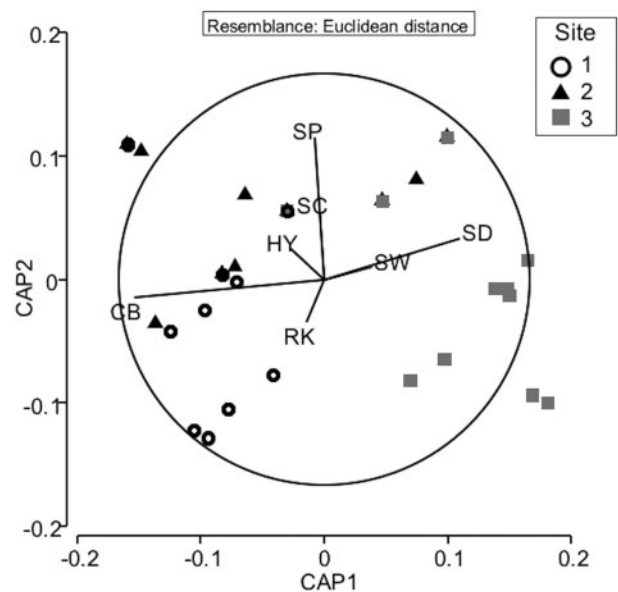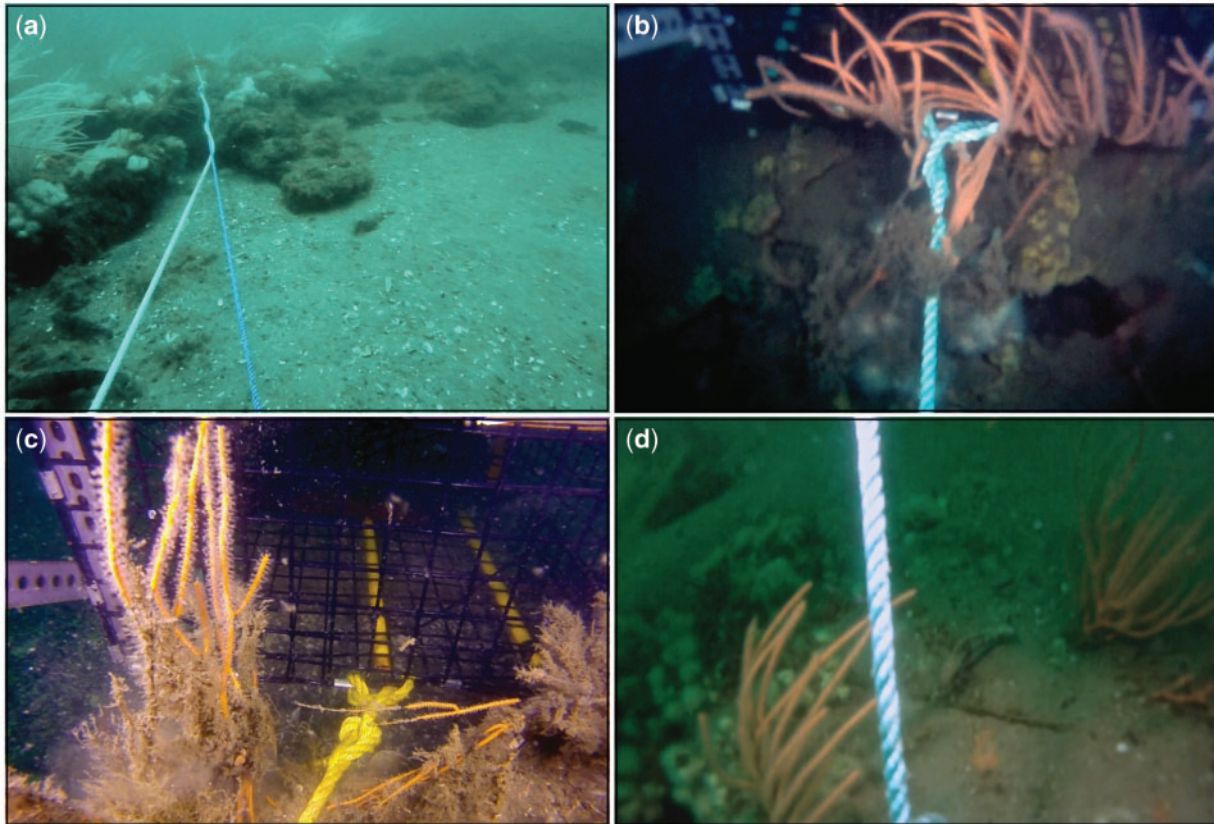
Figure 6. Images captured from the camera traps during trap retrieval. (a) Drop 144-position 17, showing the forward view of trap. The trap and snood line show the path along which the trap will be dragged. (b) Drop 30-position 5: a trap running over sea whip corals during retrieval. (c) Drop 60-position 5: a trap running over a patch of bryozoans, northern stone coral, and sea whips, which have some degree of damage and overgrowth of fouling organisms. (d) Drop 35-position 1: forward view showing the coral patch over which the trap is dragged during retrieval.
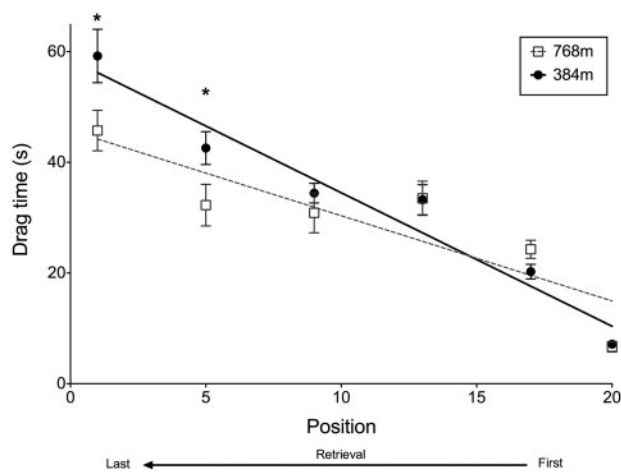

Figure 7. Duration of mean bottom drag times for traps at different positions on a 20-trap line of either standard (384 m, slope = −2.76; $r^2 = 0.73$; $p < 0.0001$) or extended length (768 m, slope = −1.5; $r^2 = 0.32$). The intercepts and slopes of the standard 384-m trap line and the extended 768 m trap differed significantly ($p = 0.001$). Extending the trap line resulted in a reduction in the rate of linear decrease in drag time with position number. Vertical bars = standard error.

whereas those at P9 and P13 show a declining trend, while P17 and P20 show a concave curve.

Extended trap line

Within each trap position, data were normally distributed ($p > 0.05$), but three outliers were detected [P1 ($n = 2$) and P13] and removed from analysis. Similar to the standard line, drag times for the extended line were significantly related to trap position ($r^2 = 0.32$; $p < 0.0001$). The slopes of the 384 m and the 768-m trap lines differed significantly (ANCOVA, $p = 0.001$; Figure 7). Contrary to our hypothesis, doubling the trap line only reduced the drag times of traps at positions P1 and P5 (t-test, $p = 0.024$; $p = 0.034$, respectively), and presumably those between them, indicating a trend of decreased drag time. The second series of sampling was conducted in rougher seas (wave height > 1.0 m) than Series 1 (t-test $p < 0.01$). As a result, and unlike the standard trap line, depth and wave height had an effect on drag time ($F = 17.85$; $F = 3.7$; df = 128; $p < 0.01$; $p = 0.02$ respectively; Table 5). Since the standard trap line was not tested during high seas it is uncertain if an extended or standard trap line is more effective at reducing drag times during rough weather. However, the data suggest that drag times would need to be reduced to <10 s to minimize habitat interactions.

Table 5. Results from analysis of covariance showing effects of trap position, wave height, depth, and line length on trap drag time.

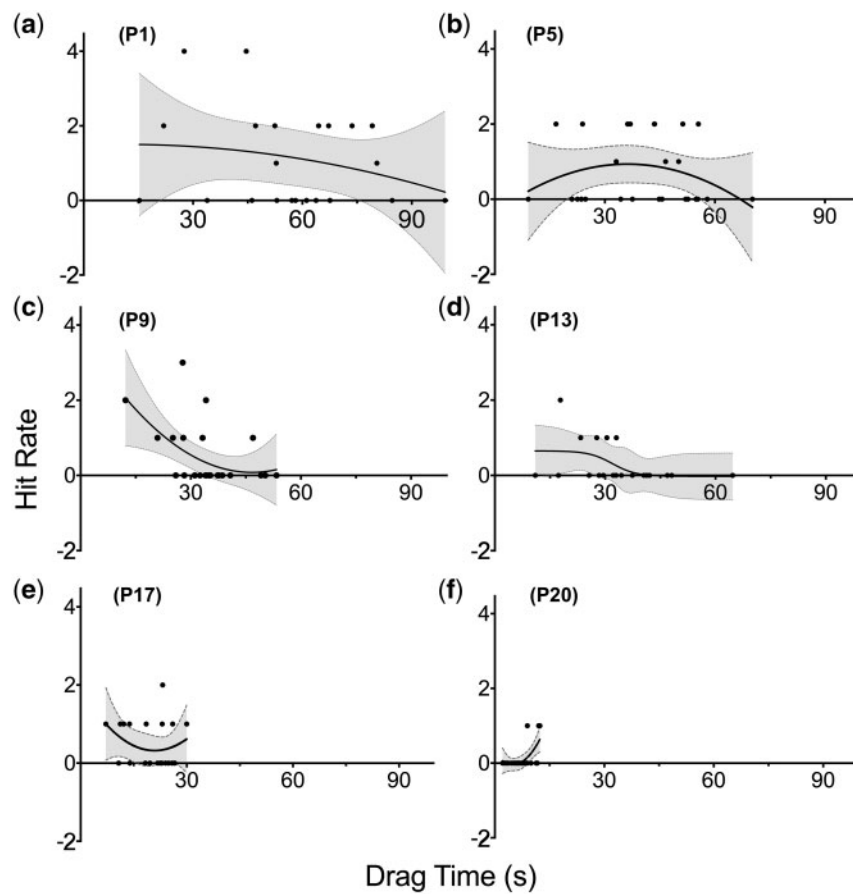|		DF	Sum of squares	Mean square	F-value	p-value
384 m	Position	1	26 437	26 437	149.98	<0.001
	Wave height	1	175.4	175.4	0.995	0.32
	Depth	1	118.9	118.9	0.674	0.41
	Residuals	132	232 67.1	176.3		
768 m	Position	1	13 948.4	13 948.4	71.03	<0.001
	Wave height	2	1444.8	722.4	3.679	0.03
	Depth	1	3504.6	3504.6	17.847	<0.001
	Residuals	128	25134.2	196.4		
384 m vs. 768 m	Trap line	1	2440	2440	18.463	<0.001
	Residuals	251	33175	132		

Figure 8. Relationships between drag time and hit rate by trap position on the line. Curves are second order polynomials with 95% *CI* shaded. (a) Position 1; (b) position 5; (c) position 9; (d) position 13; (e) position 17; (f) position 20.

Discussion

Habitat loss, reduced patch size and integrity, and fragmentation have been strongly associated with loss of biodiversity and decreased species persistence (Gibbs, 1998; Collingham and Huntley, 2000; Hovel and Lipcius, 2001; Thrush *et al.*, 2008). It is therefore important to quantify adverse impacts to benthic habitat by all types of fishing gear to thoroughly understand the consequences of such impacts for biodiversity, fish behaviour, and recruitment (Auster, 1998; Kaiser *et al.*, 2003; Scharf *et al.*, 2006). Trawls and dredges have been the focal point of most research on the impacts of fishing and ecological disturbance, whereas passive fishing gear (e.g. traps) have generally been overlooked, especially multi-trap lines. A better understanding of how passive fishing gear, such as the 20-trap lines in our study, impacts complex habitat can facilitate a shift toward ecosystem-based fisheries management (EBFM; Pikitch *et al.* 2004; Armstrong and Falk-Petersen, 2008; Möllmann *et al.*, 2014).

We attached GoPro® cameras onto commercial lobster and black sea bass traps, which proved to be an effective method to observe trap fishing *in situ* to determine the magnitude of trap–habitat impacts. We observed traps being dragged along the floor and colliding with and breaking clay bedforms, running over

Table 6. Comparison of models g_1–g_5 corresponding to the different hypotheses about habitat and trap interactions represented by the models g_i.

Model	k	Variables	AIC$_C$	Δ_i	w_i
g_1	8	Position (P) + drag time (DT)	270.65	1.71	0.26
g_2	7	P	273.66	4.71	0.06
g_3	3	DT	286.27	17.32	<0.01
g_4	13	P × DT	268.95	0.00	0.6
g_5	9	P + DT + depth	272.83	3.88	0.09

Note: k, number of parameters; AIC$_c$, corrected AIC value; Δ_i, difference between each model and the best model in the set; and w_i, model probability that a given model provided the best fit to the data. Abbreviations: P, position = placement of the camera trap on the line; DT, drag time = duration traps dragged along the ocean floor; depth = ocean depth at specific sites. Model g_4 was selected as the best model.

stone coral, breaking gorgonian corals, burying epifauna, and overturning cobble. Our findings are similar to impacts documented with mobile gear such as running over and burying epifauna, removing, damaging, and displacing structure-forming species, and overturning, moving and burying cobbles and rocks (Watling and Norse 1998), but on a smaller spatial scale. However, the frequency with which traps are deployed, and come into contact with benthic habitats, can be much greater than for trawls in some areas (Auster and Langton, 1999). Trap rigs within the MAB are retrieved every 7–10 days equating to 36–52 times per year, resulting in a chronic disturbance.

Previous research investigating trap impacts on benthic habitat suggested that traps created minimal disturbance (Eno *et al.*, 2001; Stephenson *et al.*, 2017). However, these studies investigated the impacts of single trap or three-trap lines. In our study, the first trap retrieved had the shortest drag time and minimal disturbance to benthic structure (<5%), which is complimentary to previous studies. However, the first trap in the line is not an accurate representation of the entire trap line. Traps farther down the line had significantly longer drag times, which caused 50% of the traps to collide with bedforms, corals, sea whips, and other epifauna.

Despite the correlation between drag time and habitat interactions overall, such interactions were inconsistent between trap positions (Figure 8). This variability was likely due to disparities in the process of trap retrieval for traps at different locations along the trap line. For instance, a trap retrieved later in the process might be moving over habitat that has already been cleared by previous traps, resulting in a decline in hits for longer drag time as observed for trap locations P1, P5, P9, and P13. However, when traps had considerably shorter drag times, which was commonly observed for P20 (first trap retrieved in the line), habitat interactions were minimal throughout all research sites. This is consistent with other research where observations consisted of a single trap and three-trap lines. More research is needed to develop ways to significantly reduce trap drag time for all traps during retrieval. Our research suggests that a drag time of less than 10 s would have minimal impacts on benthic habitat, as observed for the first few retrieved traps.

In an attempt to reduce drag time and therefore trap–habitat impacts, we extended the trap line to 768 m to counteract depth and to allow more time for the fishing vessel to manoeuvre toward the trap. The hypothesis was that this line extension would mimic retrieval for trap P20 for all traps allowing the boat

more time to counter act dragging when moving toward the trap line. However, we were not successful in reducing trap drag time to 10 s, and therefore do not recommend this alteration. More research is needed to determine the factors contributing to drag time and to explain why drag time was shorter for the first few retrieved traps.

In this study, we limited our observations to direct impacts of traps on benthic structure. This study did not quantify possible impacts imposed by the trap line itself, trap movement during soak time, or how sea conditions affect trap retrieval. During deployment, traps are pulled off the stern by the weight of the trap line and anchor block, resulting in a tight line between traps (Figure 6a). Trap lines may not move between soak times, with the exception of lines affected by severe weather events (e.g. hurricanes) or by getting entangled with other fishing gear. Our observations for the standard trap line were limited to fair sea conditions during spring and summer and not in the rougher conditions commonly experienced in fall. More research is needed to determine ways to quantify impacts of traps and line during severe weather events, and how other environmental factors such as waves, current, and wind can affect drag time.

ROV surveys were conducted to obtain a baseline of the habitat condition at the three sites. We quantified damage on sea whip coral as an indicator species to determine a baseline for site condition. Previous research has reported sea whip (*Halipteris willemoesi*) damage (Lindholm *et al.*, 2008) and loss of sea whip density (Malecha and Stone, 2009) in regions of higher fishing pressure as well as lower densities of commercially important fish within regions of low sea whip abundance (Stone *et al.*, 2005). However, sea whips (*Leptogorgia* spp.) within the MAB possess a degree of branch complexity (Figure 4) which may affect sensitivity to fishing gear and pressure. We observed that sea whips were relatively abundant at similar levels at all sites, as were stony corals, and therefore could be reliable indicator species. Future studies investigating correlations between fish abundance sea whip health and density are required for indicator species validation.

In addition, all three sites had moderate and equivalent levels of degradation such that we are confident that our study is representative of similar sites in the MAB. Unfortunately, there are no published studies on historical habitat conditions or composition within the MAB, which precludes assessment of damage to habitat relative to a pre-fishing baseline. Furthermore, social surveys would be required to obtain fishing history on sites to estimate fishing pressure, which was not part of this study. Since the ROV surveys were a snapshot of current conditions we cannot conclude that the observed damage was exclusively a result of fishing practices. Without comparable data from unfished areas we cannot rule out other potential sources of disturbance such as storms and changing water conditions. These data can, however, serve as a baseline for future monitoring studies to quantify habitat resilience or degradation over time.

This is the first study to utilize GoPro® action cameras to investigate how a multi-trap line impacts benthic structure. This proved to be a reliable and cost-effective method to observe traps *in situ*. The costs of purchasing an ROV can range from $40 000 to over $500 000. ROV rentals can exceed $1000 per day and require access to vessels equipped to handle the size and weight of larger ROVs. The use of ROVs can be restricted by sea conditions limiting their use and the success of finding traps. In comparison, GoPro® cameras cost only $200–$700 each and can be used in many environmental settings.

Previous research investigating impacts of trap fishing on benthic structure was limited to a single trap or three-trap lines (Eno *et al.*, 2001; Jenkins and Garrison, 2013; Stephenson *et al.* 2017), which were not representative of trap lines containing 10 or more traps. Dragging of traps along the ocean floor, specifically for single traps, has been documented since the late 1990s (Auster and Langton, 1999). Assumptions held by previous researchers and fishermen, with whom we worked, were that all traps had minimal drag before being lifted off the bottom, and that trap–habitat impacts were negligible (C. Townsend and M. Hawkins, pers. comm.). However, our findings indicate that all traps in the line are dragged along the bottom and damage living epifauna, suggesting that the real impacts of trap lines have been underestimated. The necessary extension of this study is to determine the cumulative impact of trap fishing in MAB benthic habitats by documenting the relative abundance of damaged and undamaged epifauna. Fish abundance and habitat quality are positively correlated in that habitat degradation reduces fish and fisheries production, such as in coral reef ecosystems (Lotze *et al.*, 2006; Newman *et al.*, 2015). Similar investigations should be the focus of future studies on the effects of habitat degradation on fish and fisheries production in temperate rock reef ecosystems such as the MAB.

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