# Impact of derelict fish traps in Caribbean waters: an experimental approach 

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#### Abstract

Fish traps are a widely used fishing gear throughout the Caribbean Sea, but become marine debris when lost or abandoned, with uncertain impacts to coral reef ecosystems. Derelict fish traps are thought to cause fish mortality through unintentional fishing or ghost fishing; yet no scientific studies have been conducted to quantify the threat from ghost fishing in Caribbean waters. Using an experimental approach, 12 unbaited traps were deployed at nearshore and offshore locations in the US Virgin Islands to simulate derelict traps. Six traps were set with escape panels closed (fishing) and six had escape panels open. Frequent underwater visual surveys of fishes inside and surrounding traps were conducted to quantify fish assemblage composition, body size, fish behavior, condition, and mortality over 6 mo. Twenty mortalities ( $2 \%$ of trapped fish observed) were documented. All but one mortality occurred in closed traps with an estimated total market value of US $\$ 160$ over 6 mo or US $\$ 52$ per trap annually. Skin abrasions were observed on 34 fishes. A permutational multivariate analysis of variance revealed a significant effect of time on trapped fish assemblages regardless of the location. The results demonstrate that ghost fishing does occur, but that simple modifications to fishing gear such as correctly functioning escape panels will significantly reduce mortality from ghost fishing. Further reduction of impact from derelict traps can be achieved through implementation of land-based trap disposal programs.


Traps are an effective and economically important multi-species fishing gear used widely for harvesting crustaceans and finfish around the world (Recksiek et al. 1991). Throughout the Caribbean region, traps are commonly used to catch lobster and coral reef-associated fish, typically composing a large proportion of fishery landings (Gobert 1998, Hawkins et al. 2007). On many islands, the trap fishery
has an important socioeconomic impact and cultural heritage value (Mahon 1993, Appeldoorn 2008).

Like many marine industries, trap fisheries contribute to marine debris through accidental gear loss or abandonment. Traps may become lost or derelict as a result of several processes including: intentional abandonment, loss of trap markers, fouling on the benthos (i.e., traps and ropes entangling with rocky substrate), ship strikes, human error, and inclement weather (Laist 1995, Clark et al. 2012). Derelict traps are widely thought to result in mortality because of "ghost fishing," a term used to describe the process by which derelict fishing gear continues to trap organisms and induce mortality in an uncontrolled manner (Matsuoka et al. 2005). Given the extent of trap fishing throughout the Caribbean region, the phenomenon of ghost fishing is a concern to fisheries managers and the fishing community interested in long term sustainability of the trap fishery.
Numerous studies have investigated the function, selectivity, and catch rates of actively used fish traps in the Caribbean region (High and Beardsley 1970, Munro et al. 1971, Wolff et al. 1999, Robichaud et al. 2000, Garrison et al. 2004). Studies that have investigated the impacts of ghost fishing have mostly focused on trap fisheries in temperate waters (Pecci et al. 1978, Smolowitz et al. 1978, Guillory 1993, Bullimore et al. 2001) and tropical waters outside the Caribbean (Al-Masroori et al. 2004); or related to other fishery gear types such as gill or trammel nets (Carr et al. 1992, Kaiser et al. 1996, Tschernij et al. 2003, Baeta et al. 2009). No experimental evidence exists for ghost fishing mortality caused by derelict fish traps in the Caribbean fishery resulting in a major knowledge gap impeding our ability to assess the potential threat to coral reef ecosystems.

To address this problem, we designed an underwater experiment to characterize fish species at risk from derelict traps and to quantify mortality from ghost fishing in the US Virgin Islands. The US Virgin Islands trap fishery is a multi-species, reefbased fishery centered in the waters around St. Thomas and St. John. The trap fishery accounts for $83 \%$ of the reported commercial landings for those islands [D Olsen, St. Thomas Fishermen's Association, unpubl data (http://www.stfavi.org/Landings. html)] providing critical resources for local consumption and livelihoods (Agar et al. 2008). It is comprised of approximately 3700 fishes and 2500 lobster traps (Renchen et al. 2012). The impetus for the present study emerged from the need for a greater scientific understanding of the potential threat from derelict fishing gear as identified by: (1) the commercial trap fishermen of the St. Thomas and St. John who were interested in evaluating present fishing practices to ensure viability of a long-term trap fishery, and (2) federal marine management agencies interested in reduction of marine debris and enhanced marine stewardship for ecosystem health and sustainable livelihoods. We used simulated derelict fish traps to test the following hypotheses: (1) derelict traps with escape panels open result in lower mortality than traps with escape panels closed, and (2) trapped fish species composition is significantly different over time between nearshore and offshore locations.

## Materials and Methods

Trap Design.-Two fish trap designs (Fig. 1) were used for experimental fishing: (1) the chevron or arrowhead trap $(n=3)$, and (2) the more common rectangular trap $(n=3)$. All traps were constructed by a local St. Thomas fisherman, with a steel


Figure 1. Diagrams of the chevron or arrowhead and rectangular traps used in the experimental derelict trap study. Illustrations created by C Jeffrey, NOAA Biogeography Branch, 2011.
rebar frame, horseneck funnel (with an initial upward bend followed by a downward bend to inhibit escape), and $5.08 \mathrm{~cm}(2 \mathrm{in})$ vinyl coated square mesh. As regulations require, each trap was fitted with an escape panel that was tied shut with a biodegradable material, specifically $0.32 \mathrm{~cm}(1 / 8 \mathrm{in})$ diameter jute twine.

Study Sites and Sampling Design.-In total, 12 experimental traps were deployed in near-reef habitats at a nearshore and offshore location. Sites were selected to represent deeper offshore waters targeted by commercial trap fishermen and nearshore waters by mostly unlicensed subsistence fishermen. Benthic habitat strata were defined using the National Oceanic and Atmospheric Administration's (NOAA) benthic habitat maps (Kendall et al. 2001) followed by underwater reconnaissance. Deployment sites were selected randomly within each stratum and traps were deployed in groups of three at the selected sites. The nearshore location, Perseverance Bay ( $\left.18^{\circ} 20^{\prime} 22.96^{\prime \prime} \mathrm{N}, 64^{\circ} 59^{\prime} 34.73^{\prime \prime} \mathrm{W}\right)$, is an open bay on the southern shore of St. Thomas, US Virgin Islands, while the offshore location, Flat Cay ( $18^{\circ} 19^{\prime} 00.86^{\prime \prime} \mathrm{N}$, $64^{\circ} 59^{\prime} 22.96^{\prime \prime} \mathrm{W}$ ), is off a small, uninhabited island approximately 2 km south of Perseverance Bay (Fig. 2). Nearshore traps were deployed at depths of approximately 7 m , while offshore traps were deployed at about 18 m . Both locations were actively fished as evidenced by sightings of active and derelict fish traps. Six of the 12 traps were randomly selected and deployed with the escape panels closed. Closed traps were deployed to simulate an actively fishing trap that had become lost. The escape panels on the sides of the traps were tied closed with 0.32 cm diameter untreated jute twine (typically referred to as biodegradable rot cord), as required by local fishing


Figure 2. Map of the study area on the south shore of St. Thomas, US Virgin Islands showing nearshore and offshore trap locations. The distribution of benthic habitat types is depicted by NOAA's benthic habitat map.
regulations. The remaining six traps were deployed with escape panels open to determine if open panels allowed fishes to escape, thereby expected to reduce the incidence of ghost fishing.
Traps were surveyed using scuba $3 \mathrm{~d} \mathrm{wk}^{-1}$ from January 2010 to July 2010 for a total of 56 surveys. Field observations included photographs and visual surveys to record the trap contents including the abundance of fish species, estimated total length to the nearest centimeter, bodily injuries (abrasions), and mortality. The trap mesh size was used as a reference in estimating fish total lengths. Surveys lasted approximately 15 min per group of three traps. Fishes were deemed trapped if they were too large to escape through the trap mesh. That is, they only could enter and exit the trap through the funnel. To evaluate the impact of ghost fishing, all traps were left untouched for the duration of the study and trapped species were neither removed nor tagged to limit the amount of human influence on trap-fauna interactions. Any fishes observed within 1 m above and to the sides of the traps were also recorded and their body length estimated to quantify the fish aggregating function of derelict traps.
The economic loss associated with ghost fishing mortality was calculated based on the mortalities observed in the closed traps and the 2011 local fish market prices. The economic value (\$USD) for each individual closed trap was averaged for the 6-mo study period. This average value was extrapolated to develop an estimate of the economic loss per trap per year. The importance of specific fish families to the US Virgin

Islands trap fishery was determined from data collected on local fishing effort (Olsen et al. 2008) and verified by observations of catch at local fish markets.

Data Analysis.-Permutational multivariate analysis of variance (PERMANOVA+) in the Primer v6 software package was used to test the null hypothesis of no difference in species composition in nearshore and offshore waters that were consistent on temporal scales. PERMANOVA has substantial advantages over conventional multivariate analysis of variance; it is robust to data sets with many zeros and unbalanced designs and makes no assumptions about underlying data distributions (Anderson 2001, Walters and Coen 2006). As we were primarily interested in evaluating ghost fishing, only closed traps were used in the multivariate analyses. Analyses were based on untransformed data and Bray Curtis similarity matrices, with a dummy variable of 1 added due to sparseness of data at some sites (Clarke and Warwick 2001). The design used 9999 permutations of residuals under a reduced model. A repeated measures design was used with the random factor Site [ $\mathrm{Si}(\mathrm{Lo})$, nested in location] and fixed factors Location (Lo) and Time (Ti). The highest level interaction $\mathrm{Si}(\mathrm{Lo}) \times \mathrm{Ti}$ was excluded as only one sample existed per site at each time (Anderson 2001). Non-metric multidimensional scaling (nMDS) was used to visualize the dissimilarity matrices (Clarke and Warwick 2001).

## Results

Fish Abundance.-During the 6-mo study period, 1183 individual fishes of 34 species from 16 families were observed within the experimental traps (Table 1). Fish assemblages were dominated by fishes from the surgeonfish and snapper families (Fig. 3). The mean total length of fishes for nearshore traps was 24.5 (SD 12.5) cm, and 25.1 (SD 19.7) cm for offshore traps. Fish abundance varied through time, with low abundance in week $1(n=3.39)$, a peak at week $11(n=7.44)$, and abundance decreasing again to week $19(n=1.75)$. The lowest abundance was seen in week $6(n=1.28)$. Peaks in fish abundance were driven by trapped schools of blue tang, Acanthurus coeruleus (see Online Appendix 1 for species authorities), offshore and saucereye porgy, Calamus calamus, and schoolmaster snapper, Lutjanus apodus, nearshore. Species richness also varied, peaking in week $12(n=23)$ and reaching its lowest in weeks 2 and 15 ( $n=8$ and $n=7$, respectively).
Approximately 12,400 fishes were observed within 1 m of the open traps. Densities of schooling fishes were highest where traps were placed in nearshore locations with abundant seagrass. For instance, after 7 wks, as many as 500 juvenile fishes were observed schooling in mixed species groups within 1 m of traps in seagrass. These fishes assemblages were dominated by juvenile fishes from the Haemulidae and Lutjanidae families. Approximately $85 \%$ of fish within 1 m of the closed traps were $\leq 10 \mathrm{~cm}$ total length.

Assemblage Composition in Closed Traps.-A significant effect of Time ( $P=$ 0.03 ) on assemblage composition was identified, but no significant Location $\times$ Time interaction or main Location effect was identified (Table 2). Significant variation was also identified between Sites nested in Location, but the strength of this test was low due to the small number of degrees of freedom (1, 4). The lack of a Location $\times$ Time interaction and main Location effect meant that these were not investigated further. A three-dimensional nMDS ordination illustrating the effect of Time on assemblage

Table 1. List of species observed in the experimental derelict fish traps at nearshore and offshore locations on the southern shore of St. Thomas, US Virgin Islands. The number of fish observed was summed over all months of the study. The mean biomass $(g)$. Biomass was calculated by converting fish total length to weight in grams using the relationship $W=a L^{b}$. Values for the constants $a$ and $b$ were obtained for each species from FishBase (Froese and Pauly 2010).

| Species | Offshore |  | Nearshore |  | Species total |
| :---: | :---: | :---: | :---: | :---: | :---: |
|  | $\begin{aligned} & \text { No. of } \\ & \text { fish } \end{aligned}$ | Mean biomass (SE) | $\begin{aligned} & \text { No. of } \\ & \text { fish } \end{aligned}$ | $\begin{gathered} \text { Mean } \\ \text { biomass (SE) } \end{gathered}$ |  |
| Acanthuridae |  |  |  |  |  |
| Acanthurus bahianus | 11 | 56.3 (11.9) | 7 | 52.9 (15.1) | 18 |
| Acanthurus chirurgus | 64 | 123.8 (6.5) | 48 | 87.3 (4.3) | 112 |
| Acanthurus coeruleus | 122 | 56.3 (2.8) | 9 | 114.8 (21.0) | 131 |
| Aulostomidae |  |  |  |  |  |
| Aluterus scriptus | 5 | 279.0 (12.4) | 0 | 0.0 | 5 |
| Balistidae |  |  |  |  |  |
| Balistes vetula | 7 | 832.8 (114.1) | 0 | 0.0 | 7 |
| Carangidae |  |  |  |  |  |
| Caranx crysos | 0 | 0.0 | 1 | 124.7 (0.0) | 1 |
| Caranx ruber | 0 | 0.0 | 4 | 550.3 (214.5) | 4 |
| Chaetodontidae |  |  |  |  |  |
| Chaetodon ocellatus | 20 | 55.8 (5.7) | 0 | 0.0 | 20 |
| Chaetodon sedentarius | 2 | 68.3 (15.9) | 0 | 0.0 | 2 |
| Chaetodon striatus | 13 | 24.6 (2.2) | 0 | 24.6 (2.2) | 13 |
| Diodontidae |  |  |  |  |  |
| Diodon hystrix | 0 | 0.0 | 15 | 533.2 (6.1) | 15 |
| Gerreidae |  |  |  |  |  |
| Gerres cinereus | 0 | 0.0 | 2 | 595.5 (0.0) | 1 |
| Haemulidae |  |  |  |  |  |
| Haemulon plumierii | 19 | 475.9 (49.7) | 13 | 545.6 (57.9) | 32 |
| Haemulon sciurus | 10 | 1,048.0 (140.5) | 49 | 1,062.5 (56.1) | 59 |
| Holocentridae |  |  |  |  |  |
| Holocentrus adscensionis | 0 | 0.0 | 8 | 66.5 (6.4) | 8 |
| Lutjanidae |  |  |  |  |  |
| Lutjanus analis | 0 | 0.0 | 15 | 552.9 (50.4) | 15 |
| Lutjanus apodus | 47 | 626.0 (24.2) | 87 | 524.5 (22.3) | 134 |
| Lutjanus griseus | 0 | 0.0 | 2 | 686.8 (0.0) | 2 |
| Lutjanus jocu | 26 | 2,882.1 (55.3) | 0 | 0.0 | 26 |
| Lutjanus mahogoni | 6 | 485.8 (0.0) | 0 | 0.0 | 6 |
| Lutjanus synagris | 1 | 3,597.0 (0.0) | 12 | 711.2 (65.1) | 13 |
| Ocyurus chrysurus | 3 | 2,989.4 (1111.3) | 11 | 3,453.0 (679.7) | 14 |
| Ostraciidae |  |  |  |  |  |
| Acanthostracion quadricornis | 8 | 123.23 (1.97) | 5 | 281.0 (22.5) | 13 |
| Lactophrys bicaudalis | 3 | 93.0 (6.2) | 1 | 171.5 (0.0) | 4 |
| Lactophrys triqueter | 4 | 55.6 (11.1) | 77 | 114.9 (9.3) | 81 |
| Pomacanthidae |  |  |  |  |  |
| Holacanthus ciliaris | 13 | 243.8 (22.5) | 0 | 0.0 | 13 |
| Pomacanthus arcuatus | 130 | 230.6 (11.4) | 2 | 300.7 (245.6) | 132 |

Table 1. Continued.

| Species | Offshore |  | Nearshore |  | Species total |
| :---: | :---: | :---: | :---: | :---: | :---: |
|  | No. of fish | $\begin{gathered} \text { Mean } \\ \text { biomass (SE) } \end{gathered}$ | $\begin{aligned} & \text { No. of } \\ & \text { fish } \end{aligned}$ | $\begin{gathered} \text { Mean } \\ \text { biomass (SE) } \end{gathered}$ |  |
| Rhincodontidae |  |  |  |  |  |
| Ginglymostoma cirratum | 21 | 3,593.5 (247.0) | 31 | 5,102.5 (274.1) | 52 |
| Scaridae |  |  |  |  |  |
| Scarus iserti | 2 | 49.3 (20.4) | 0 | 0.0 | 2 |
| Sparisoma rubripinne | 1 | 202.4 (0.0) | 2 | 472.0 (0.0) | 3 |
| Sparisoma viride | 21 | 683.2 (147.0) | 7 | 259.1 (68.9) | 28 |
| Serranidae |  |  |  |  |  |
| Cephalopholis fulva | 1 | 125.6 (0.0) | 0 | 0.0 | 1 |
| Epinephelus guttatus | 22 | 302.3 (18.6) | 0 | 0.0 | 22 |
| Epinephelus striatus | 0 | 0.0 | 1 | 1,425.9 (421.1) | 2 |
| Total no. fish observed | 594 |  | 589 |  | 1,183 |



Figure 3. Proportion of the total number of fishes caught in experimental derelict fish traps grouped by fish family for a 6-mo study period on the south shore of St. Thomas, US Virgin Islands. Fish family importance to the St. Thomas and St. John trap fishery is indicated by bar color: important (black) or unimportant (gray).

Table 2. PERMANOVA results for the relative distribution of the species assemblage composition, in response to fixed factors Location (Lo) and Time (Ti), random factor Site (Si), and their interactions. Analyses were conducted using Bray Curtis similarities on untransformed data with a dummy variable of 1 . Asterisks denote a significant result ( $P<0.05$ ).

| Source | df | SS | MS | F | $P$ |
| :--- | ---: | ---: | ---: | ---: | :---: |
| Lo | 1 | 53,248 | $53,248.0$ | 1.8380 | 0.2008 |
| Ti | 55 | 106,480 | $1,936.1$ | 1.1708 | $0.0265^{*}$ |
| $\mathrm{Si}(\mathrm{Lo})$ | 4 | 115,880 | $28,970.0$ | 17.5190 | $0.0001^{*}$ |
| Lo $\times \mathrm{Ti}$ | 55 | 100,670 | $1,830.4$ | 1.1069 | 0.0989 |
| Res | 220 | 363,800 | $1,653.6$ |  |  |
| Total | 335 | 740,090 |  |  |  |



Figure 4. Non-metric multidimensional scaling plot illustrating similarities in assemblage composition over time and presented by week. The 3D representation stress $=0.11$, but 2D representation is presented for clarity. Arrows show the trajectory through time.
composition (stress $=0.11$ ) clearly shows a progression through time, despite some variation. For clarity, two dimensions are presented in Figure 4.

Ghost Fishing Mortality and Injury.-Nineteen fishes were unable to escape and subsequently died in traps with escape panels closed (Table 3), with mortality attributed to ghost fishing. Based on the total number of observed fish, approximately $2 \%$ of trapped fishes expired in the traps. In contrast, only one mortality (a barracuda, Sphyraena barracuda) was observed in a trap with the escape panel open. Dead fish, their skeletal and fleshy remains, were observed in all of the traps that incurred known mortalities. The temporal distribution of mortality ranged from one to six mortalities per month. Mortalities peaked during months $1(n=5)$ and $4(n=$ 6 ), while the lowest number of mortalities occurred in the last month ( $n=1$ ). Fish species with a short or deep body shape (e.g., C. calamus) accounted for the highest number of mortalities ( $n=11$ ). The fusiform body shape (e.g., yellowtail snapper, Ocyurus chrysurus) accounted for six mortalities while elongated (S. barracuda), laterally compressed (gray angelfish, Pomacanthus arcuatusn), and depressed (nurse shark, Ginglymostoma cirratum) each accounted for one mortality.
Using the price per pound of weight for each species, the 20 known mortalities, equivalent to 23.13 kg ), represented an economic loss of US $\$ 160$ over the 6 -mo study period. Extrapolation of this estimate is equivalent to an average annual loss of approximately US\$52 per derelict trap.
Thirty-four trapped fishes were documented with skin wounds or abrasions. Although the actual cause of mortality was not determined, nine of those species that expired had abrasions on their snouts and foreheads likely due to repeated collisions with the mesh while trying to escape. This behavior was observed frequently in samples and has been documented in other work (Luckhurst and Ward 1987, Renchen et al. 2012). Species that were observed on multiple occasions with injuries included

Table 3. Fish mortality in the experimental derelict fish traps during the 6-mo ghost fishing study. The mean biomass (g) and market price are presented in grams for consistency, however market prices in the US Virgin Islands are typically reported as US\$/lb. *Species considered by fishermen as potentially unmarketable due to risk of ciguatera poisoning. $M=$ mortality.

| Family | Species |  | Fish observed <br> per species | M with <br> injuries | Mean M <br> biomass (SE) | Market price <br> $(\mathrm{US} \$ / 100 \mathrm{~g})$ |
| :--- | :--- | :--- | :---: | :--- | :---: | :---: |
| Ostraciidae | Acanthostracion quadricornis | 1 | 13 | - | 303.5 | 0.7 |
| Monacanthidae | Aluterus scriptus | 2 | 5 | 1 | $284.1(35.7)$ | Bycatch |
| Sparidae | Calamus calamus | 6 | 192 | 1 | $407.4(263.1)$ | 1.1 |
| Serranidae | Epinephelus guttatus | 1 | 22 | 1 | 229.9 | 1.3 |
| Rhincodontidae | Ginglymostoma cirratum | 1 | 52 | - | $4,708.2$ | Bycatch |
| Ostraciidae | Lactophrys triqueter | 2 | 81 | - | $132.7(112.1)$ | 0.7 |
| Lutjanidae | Lutjanus apodus | 1 | 134 | 1 | 215.9 | Bycatch* |
| Lutjanidae | Lutjanus jocu | 2 | 26 | 1 | $2,423.7(658.3)$ | Bycatch* |
| Lutjanidae | Ocyurus chrysurus | 2 | 14 | 2 | $4,625.3(2,903.0)$ | 1.3 |
| Pomacanthidae | Pomacanthus arcuatus | 1 | 132 | - | 42.5 | 1.3 |
| Sphyraenidae | Sphyraena barracuda | 1 | 1 | 1 | 434.6 | Bycatch* |

Sparisoma viride, C. calamus, O. chryrurus, Lactophrys triqueter, Epinephelus guttatus, and Acanthurus chirurgus.

## Discussion

The results of the present study demonstrate that derelict fish traps are capable of ghost fishing and may experience changes in catch over time. The fish assemblage composition documented in our experimental derelict traps was similar to catch compositions documented in other US Virgin Islands trap studies (Garrison et al. 2004, Trumble et al. 2006, Olsen 2008). The comparison of fish assemblages between nearshore and offshore locations did not reveal a significant effect; however, there was a significant effect of time on the assemblage composition regardless of the location. Trap catches are dependent upon fishes encountering the traps, therefore behavioral characteristics (Renchen et al. 2012), migration, and movement pathways between habitats and season (Galzin 1987) may all influence whether and when fishes become caught in traps. An acoustic tracking study showed that for some species of grunt and snapper, nocturnal activity spaces are larger than day time activity spaces due to nocturnal foraging behavior (Hitt et al. 2011). A peak in fish abundance occurred in the present study approximately half-way through the experiment and was driven by capture of large numbers of $A$. coeruleus, L. apodus, and C. calamus. All three species are highly mobile; therefore, they may have been more likely to encounter the traps and subsequently escape than less mobile species (Robichaud et al. 2000). Wolf and Chislett (1974) found that trap catches in multiple Caribbean locations were reduced in May, June, and July corresponding with warming water temperatures. Although the duration of our study was not long enough to investigate seasonal changes in fish assemblages, the seasonal temporal component is known to influence fish abundance (Mateo and Tobias 2004) and subsequent trap catches in the US Virgin Islands (D Olsen, St. Thomas Fishermen's Association, unpubl data).
The abundance of trapped fishes was high initially, then decreased and leveled off by the fifth week. After the increase in abundance in week 11, abundances again
tapered off, but never reached zero. Although we cannot account for fishes that: (1) entered and escaped without being observed, (2) were consumed by predators, or (3) were replaced by conspecifics, our results suggest that catch efficiency in derelict traps is highly variable (Angelsen and Olsen 1987). In a shorter duration study, Munro et al. (1971) documented $50 \%$ escapement after traps had soaked for 14 d . Munro (1974) also suggested that gradually more fishes escape during each successive soak day, but that conspecifics may attract fishes into traps. We observed fishes of the family Acanthuridae, Sparidae, Ostraciidae, and Haemulidae interacting with conspecifics in traps, particularly when fishes inside traps exhibited schooling behavior. Similar conspecific behavior was also extensively examined by Luckhurst and Ward (1987).
Overall, the number of observed fish mortalities was unexpectedly low, with most fish able to leave the traps. Nevertheless, our experimental derelict traps did result in fish mortality, providing direct evidence that ghost fishing does occur within intact derelict traps where escape panels remain closed. In contrast, when escape doors were open, prolonged entrapment and subsequent mortality was very rare. To more comprehensively assess economic impact would require estimates of the number of traps on the seafloor and their structural condition as degraded traps with large openings would be unlikely to result in mortality associated with ghost fishing. The percentage of mortality observed in our study was 4.5 times lower than estimates from a trap bycatch study conducted by the St. Thomas Fishermen's Association (Olsen 2008). The bycatch study, however, examined mortality associated with actively fished traps and the rapid pulling of traps from greater depths introduces an additional mortality factor that was not simulated in the present study.
Not unexpectedly, the experimental traps deployed in our study functioned as fish aggregating devices within days of deployment, attracting hundreds of juvenile fishes and macro-invertebrates. The fish attraction function was particularly enhanced in areas of low structural complexity, such as traps in nearshore seagrass habitats. Studies of artificial reefs have documented that fish densities and biomass can be higher at artificial reefs than at sand or natural reef habitats, but are highly variable (Bohnsack et al. 1994, 1997). During the passage of Hurricane Earl (after the 6 -mo study period), three of the nearshore traps were lost and subsequently the aggregations of juvenile fishes diminished in that area. Derelict traps also functioned as artificial settlement substratum for coral and gorgonian communities which were observed colonizing and growing on the traps during the study (Renchen and Pittman 2012).
The present study has provided baseline quantitative data on the potential impacts of ghost fishing in derelict traps in the US Virgin Islands. We were not able to effectively evaluate the influence of habitat or trap type on ghost fishing. Habitats will likely influence trap catches (Wolff et al. 1999, Garrison et al. 2004), but previous work on trap type has suggested that chevron and rectangular traps produce similar catches (Garrison et al. 1998). Although limited in duration and replication, the study provides the first quantitative estimates of mortality directly caused by ghost fishing in Caribbean waters. Further, studies elsewhere in the Caribbean region are needed to assess the generality of our results. We demonstrated that risk of ghost fishing will be influenced by trap condition and can be reduced with the use of functioning escape panels. Federal and territorial regulations in the US Virgin Islands require that fishermen have at least one escape panel that is tied shut with a
biodegradable rot cord. Compliance with these regulations and disposal of traps on land could potentially reduce mortality from ghost fishing and reduce marine debris. Despite relatively low levels of mortality documented in our study, such unintentional losses are undesirable for the fishing community and from a management perspective. Disposal of old traps at sea is common practice in Caribbean trap fisheries and preliminary surveys off St. Thomas and St. John with autonomous underwater vehicles revealed several test areas with higher than expected derelict trap densities (Clark et al. 2012). With a fish trap population of approximately 4000 units for the St. Thomas-St. John fishery and an estimated trap life of 5 yrs for commercial fishing (Agar et al. 2005), disposal of all traps at sea would result in an accumulation of approximately 16,000 traps on the seafloor over a $20-y r$ period. This estimate, however, has obvious uncertainty due to a lack of reliable data on trap use, dereliction rates, and disposal behavior and limited data on the quantity of unregistered traps in the US Virgin Islands fishery.

Simple modifications to fishing practices, gear design, and the implementation of a trap disposal program should be examined and evaluated by both the fishing community and supported by marine management agencies. Introduction of escape gaps to fish traps has been documented to decrease bycatch by up to $80 \%$ in traps using a smaller mesh size $(2.54 \mathrm{~cm})$ (Johnson 2010). Although this estimate is not directly comparable to the US Virgin Islands trap fishery due to the difference in mesh sizes, escape gaps could further reduce the impact of ghost fishing, particularly on species with short, laterally compressed body shapes, such as surgeonfish and angelfish, and should therefore be fully evaluated. Due to the variability in derelict trap condition and the inherent spatial heterogeneity in fish assemblages, threat to marine life and economic loss from ghost fishing needs to be carefully evaluated.

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